Lower Duwamish Waterway Group

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

Final Feasibility Study

Lower Duwamish Waterway

Seattle, Washington

Volume I - Main Text, Tables, and Figures

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1 Introduction

This report presents the feasibility study (FS) for the Lower Duwamish Waterway (LDW) Superfund Site in Seattle, Washington (Figure 1-1). This report has been prepared on behalf of the Lower Duwamish Waterway Group (LDWG), consisting of the City of Seattle, King County, the Port of Seattle, and The Boeing Company. LDWG signed an Administrative Order on Consent (AOC)[1] in December 2000 with the U.S. Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology) to conduct a remedial investigation/feasibility study (RI/FS) for the LDW (EPA, Ecology, and LDWG 2000). The LDW was subsequently added to EPA's National Priorities List (also known as Superfund) on September 13, 2001.[2] The LDW was added to Ecology's Hazardous Sites List on February 26, 2002.[3]

In 2003, a Phase 1 RI was prepared based on previously existing information (Windward 2003a). The Phase 1 RI included scoping-phase human health and ecological risk assessments. The Phase 1 RI also facilitated the identification of early action areas (EAAs) and data gaps to be filled during subsequent data collection efforts. In the following years, additional data were collected, as outlined in the Phase 2 Work Plan (Windward 2004) and various project quality assurance project plans (QAPPs) and data reports. Using the additional data that were collected, baseline human health and ecological risk assessments were completed (Windward 2007a, 2007b) and included as part of the RI (Windward 2010).

The Superfund and Model Toxics Control Act (MTCA) cleanup of the LDW includes three components: early cleanup actions, source control, and cleanup of the remainder of the LDW. This FS addresses the third component. Other previously released studies, including engineering evaluation/cost analyses (EE/CA), remedial designs, permitting, and construction/post-construction monitoring have been conducted for the early cleanup actions for smaller areas within and adjacent to the LDW. These documents are relevant to this FS but focus only on discrete areas of the LDW; this FS focuses on five miles of the LDW, extending from just south of Harbor Island (river mile [RM] 0 for the FS) to upstream of the Upper Turning Basin (RM 5.0, Figure 1-1).

The study area evaluated for remedial action in this FS focuses on the sediment and surface water of the LDW (RM 0 to RM 5.0), sometimes referred to as the "site" in this FS for convenience. The terms site, LDW-wide, and site-wide are sometimes used interchangeably in this FS, but generally refer only to the sediment and surface water of

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1 The AOC for the LDW, including Attachment A, the *Lower Duwamish Waterway Remedial Investigation/Feasibility Study Statement of Work* (LDWG 2000) (EPA Docket No. CERCLA 10-2001-055 and Ecology Docket No. 00TCPNR-1895).

2 Comprehensive Environmental Response, Compensation, and Liability Information System No. WA0002329803.

3 FS ID 4297743.

the LDW, not to the upland portions of the LDW Superfund Site. The final LDW Superfund Site boundaries, including upland areas that contributed contamination to the LDW, will be determined by EPA and Ecology in future decision documents.

Investigations and cleanups of facilities, storm drains, and combined sewer overflows (CSOs) within the LDW drainage basin are being conducted to address ongoing sources of contamination to the LDW. Ecology has issued several reports to document the source control strategy (Ecology 2004) for the LDW Superfund Site and the progress to date in addressing ongoing sources of contamination. The RI (Windward 2010) summarized the source control work completed as of July 2010, and more detailed information is available in Ecology's Source Control Status Reports (Ecology 2011b, http://www.ecy.wa.gov/programs/tcp/sites).

The RI/FS work required by the AOC is being conducted under both the federal Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and MTCA (Washington State hazardous waste law similar to CERCLA). Any response actions identified in the FS must comply with both CERCLA and MTCA. The specific documents that define the overall FS process for the LDW site include the following:

- Clarification of Feasibility Study Requirements (LDWG 2003), a clarification letter from LDWG to EPA and Ecology dated December 4, 2003
- The *Feasibility Study Work Plan* for the LDW (RETEC 2007a).

This FS is consistent with the following statutes and regulations:

- CERCLA, as amended (42 United States Code [U.S.C.] 9601 et seq.), and its regulations, the National Oil and Hazardous Substances Pollution Contingency Plan (40 Code of Federal Regulations [CFR] Part 300), commonly referred to as the National Contingency Plan (NCP)
- MTCA, Revised Code of Washington (RCW) Chapter 70.105D and its regulations, Washington Administrative Code (WAC; Chapters 173-340 and 173-204, the latter also called the Washington Sediment Management Standards [SMS]).

In addition, the following guidance documents were considered in developing this FS:

- *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988)
- *Clarification of the Role of Applicable or Relevant and Appropriate Requirements in Establishing Preliminary Remediation Goals under CERCLA* (EPA 1997a)
- *Rules of Thumb for Superfund Remedy Selection* (EPA 1997b)

- *Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites* (EPA 2002b)
- *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites* (EPA 2005b)
- *A Guide to Developing and Documenting Cost Estimates during the Feasibility Study* (EPA 2000a)
- *Sediment Cleanup Standards User Manual* (Ecology 1991).

1.1 Purpose of the Feasibility Study

The purpose of this FS is to develop, screen, and evaluate LDW-wide remedial alternatives to address the risks posed by contaminants of concern (COCs) within the LDW. This FS is based on the results of the RI (Windward 2010), which included the baseline human health and ecological risk assessments (Windward 2007a, 2007b).

The RI assembled data to identify the nature and extent of contamination in the LDW, evaluated sediment transport processes, and assessed current conditions within the LDW, including risks to people and animals that use the LDW. The FS uses the results of the RI and the baseline risk assessments to identify remedial action objectives (RAOs), develop preliminary remediation goals (PRGs) and cleanup objectives, and develop and evaluate LDW-wide remedial alternatives. The FS lays the groundwork for selecting a cleanup alternative that best manages risks to both human health and the environment.

1.2 The FS Process

The road map through the FS process includes several steps outlined in CERCLA guidance (EPA 1988), as well as additional considerations outlined in *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites* (EPA 2005b). These general steps and considerations include:

- Summarizing and synthesizing the results of the RI, the baseline human health and ecological risk assessments, and related documents, as well as refining the physical conceptual site model for the LDW.
- Developing RAOs specifying the COCs, exposure pathways, and PRGs that permit an evaluation of a range of remedial alternatives and consider state and local objectives for the LDW.
- Identifying applicable or relevant and appropriate requirements (ARARs) to comply with both state and federal regulations.
- Identifying general response actions for the LDW, including removal, disposal, containment, treatment, enhanced natural recovery, and monitored natural recovery.

- Estimating the sediment volumes or areas of sediments to which the general response actions could be applied.
- Identifying and screening remedial technology types and specific process options best suited to achieve cleanup objectives for the RAOs.
- Assembling the technology types and process options into LDW-wide remedial alternatives.
- Completing a detailed evaluation and comparative analysis of the remedial alternatives consistent with CERCLA and MTCA requirements.
- Evaluating how each alternative would achieve the cleanup objectives for the identified risk drivers as well as how each alternative would address the other COCs.

1.2.1 Integration of CERCLA and MTCA

As stated previously, the RI/FS is being conducted under both CERCLA and MTCA authorities. MTCA regulations also incorporate the Washington SMS regulations by reference.

Table 1-1 compares the major requirements used to select a remedial action under CERCLA with the corresponding requirements under MTCA. Although many CERCLA requirements have MTCA counterparts, there are some important differences. These differences are discussed below.

First, both CERCLA and MTCA have threshold requirements that must be achieved by a remedial action—namely, a remedial action must be protective of human health and the environment and comply with ARARs (generally defined by CERCLA as all federal and more stringent state environmental laws and regulations). In addition to these shared threshold requirements, MTCA requires a specific demonstration that the proposed remedy provides for compliance monitoring. Compliance monitoring is also required for remedial actions under CERCLA when hazardous substances remain on-site at concentrations that do not allow unrestricted use or unrestricted exposure at the site upon completion of the remedial action. Compliance monitoring is required to ensure either that areas are not recontaminated or to evaluate trends over time, such as changes in site-wide spatially-weighted average concentrations (SWACs). The implementing regulations for MTCA require that the nature of the compliance monitoring be discussed specifically.

Second, CERCLA and MTCA share similar balancing criteria for evaluating remedial actions, with very similar frameworks for considering those criteria. For instance, CERCLA prescribes five criteria that are to be balanced in making a remedial decision: long-term effectiveness and permanence; reduction of toxicity, mobility, or volume through treatment; short-term effectiveness; implementability; and cost. CERCLA also requires that EPA "select a remedial action that is protective of human health and the environment, that is cost effective, and that utilizes permanent solutions and alternative treatment technologies or resource recovery technologies to the maximum extent practicable" (CERCLA § 121(b)(1)). Similarly, MTCA requires that Ecology "give preference to permanent solutions to the maximum extent practicable" (RCW 70.105D.030(b)). In determining whether a remedial action uses permanent solutions to the maximum extent practicable under MTCA, a "disproportionate cost analysis" is applied; the analysis takes into account criteria that are essentially equivalent to the five CERCLA balancing criteria. MTCA also requires that restoration be completed within a reasonable time frame and include a long-term monitoring plan. This is similar to the balancing criterion of short-term effectiveness under CERCLA (with the exception concerning monitoring discussed above).

Finally, CERCLA contains two modifying criteria: state and tribal acceptance, and community acceptance. MTCA provides for consideration of local, state, federal, tribal, and community acceptance as part of the disproportionate cost analysis.

Because of the somewhat different CERCLA and MTCA criteria, separate analyses of the remedial alternatives are presented in this FS.

1.2.2 Selecting a Final Remedy

Under CERCLA, the FS presents, evaluates, and compares the remedial alternatives for a site. After review of the FS, the lead agency proposes a final cleanup remedy in a document called the Proposed Plan; this plan is then provided to the public for comment. After public comments on the Proposed Plan are received and evaluated, the lead agency documents the final remedy in a decision document. For CERCLA, this document is called a Record of Decision (ROD). For MTCA, the decision document is the Cleanup Action Plan (CAP), which is functionally equivalent to the CERCLA ROD. The MTCA CAP includes the requirements of the SMS cleanup study report. EPA and Ecology have determined that the cleanup decision document for the LDW will be a CERCLA ROD. The ROD will be issued by EPA with concurrence from Ecology.

The lead agencies for the LDW are EPA and Ecology, and these agencies will ultimately select the final remedy, including the final RAOs and cleanup levels. To this end, the agencies' selection of the final remedy will likely involve weighing the outcomes of evaluations that are conducted under a number of criteria, including:

- The nine CERCLA criteria provided in the NCP for evaluation of remedial alternatives
- The statutory determination requirements in the NCP for selected remedies (40 CFR 300.430(f)(5)(ii))
- Cleanup action requirements under MTCA (WAC 173-340-360) and the SMS (WAC 173-204)

- Risk management principles for sediment sites, as outlined in EPA guidance (EPA 2005b)
- Source control analyses, as described in Ecology's publication *Lower Duwamish Waterway Source Control Strategy* (Ecology 2004).

1.3 Definitions for the Feasibility Study

Definitions of regulatory terms, contaminant concentrations, various spatial areas, and time frames used in the FS are provided below. Some of these terms have site-specific definitions, but most are drawn directly from CERCLA or MTCA regulations or guidance documents. In the case of new definitions, similar terms are referenced when applicable.

1.3.1 Regulatory Terms

Area background, a term specific to MTCA, represents the concentrations of hazardous substances that are consistently present in the environment in the vicinity of the site as a result of human activities unrelated to releases from the site (WAC 173-340-200). When cleanup levels are less than area background concentrations, MTCA recognizes that area background concentrations can result in recontamination of a site to levels that exceed cleanup levels. In such cases, MTCA allows that portion of the cleanup action to be delayed until off-site sources of hazardous substances are controlled. CERCLA uses the term **anthropogenic (man-made) background** (EPA 1997b), and EPA's sediment remediation guidance (EPA 2005b) states that cleanup levels will normally not be set below natural or anthropogenic background concentrations. However, neither area nor anthropogenic background concentrations have been quantified in this FS. Instead, this FS references the upstream datasets for evaluating incoming, ambient concentrations to the LDW from external sources that may be influenced by urbanization.

Cleanup level under MTCA and CERCLA means the concentration of a hazardous substance in an environmental medium that is determined to be protective of human health and the environment under specified exposure conditions. CERCLA and MTCA provide similar processes for defining and selecting cleanup levels, but some of the terms in the two regulatory programs have slightly different meanings. Cleanup levels are proposed in the FS but are not finalized until the ROD.

Cleanup objective in this FS is used to mean the PRG or as close as practicable to the PRG where the PRG is not predicted to be achievable. This FS uses long-term model-predicted concentrations as estimates of "as close as practicable" to PRGs.

Contaminants of potential concern (COPCs)/Contaminants of concern (COCs) are two related terms used in the baseline risk assessments. The COPCs were initially identified through a conservative risk-based screening process. In this process, contaminant concentrations in sediment, water, and aquatic biota were compared to conservative risk-based screening levels or effects standards. Those contaminants present in any samples from the LDW at concentrations above the screening levels were identified as "contaminants of potential concern," which then underwent further analysis in the baseline risk assessments. The COCs represent a defined subset of the COPCs that were quantitatively evaluated in the baseline risk assessments, considering their distributions in all of the media, and were found to exceed threshold risk levels.

Natural background represents the concentrations of hazardous substances that are consistently present in an environment that has not been influenced by localized human activities (WAC 173-340-200). The MTCA definition includes both substances such as metals that are found naturally in bedrock, soils, and sediments, as well as persistent organic compounds such as polychlorinated biphenyls (PCBs) that can be found in soil and sediments throughout the state as a result of global distribution of these contaminants.

Point of compliance is defined by MTCA as the point or points where cleanup levels shall be achieved (WAC 173-340-200).

Practical quantitation limit (PQL) is defined by MTCA as the "lowest concentration that can be reliably measured within specified limits of precision, accuracy, representativeness, completeness, and comparability during routine laboratory operating conditions, using department approved methods" (WAC 173-340-200). MTCA includes consideration of the PQL in establishing cleanup levels (WAC 173-340-705(6)). Similarly, the NCP (40 CFR 300.430(e)(2)(i)(A)(3)) allows that cleanup levels can be modified based on "factors related to technical limitations such as detection/quantitation limits for contaminants." The term PQL is synonymous with quantitation limit and reporting limit.

Preliminary remediation goals (PRGs) are specific desired contaminant endpoint concentrations or risk levels for each exposure pathway that are believed to provide adequate protection of human health and the environment based on available site information (EPA 1997b). For the FS, PRGs are expressed as sediment concentrations for the contaminants that present the principal risks (i.e., the risk drivers). PRGs are based on consideration of the following factors:

- ARARs.
- Risk-based threshold concentrations (RBTCs) developed in the risk assessments.
- For final cleanups under MTCA, natural background concentrations are used to develop PRGs if protective RBTCs are below background concentrations.
- Analytical PQLs if protective RBTCs are below concentrations that can be quantified by chemical analysis.

PRGs are presented in the FS as the proposed cleanup levels and standards and will be finalized (as defined above) by EPA and Ecology in the ROD.

Remedial action objectives (RAOs) describe what the proposed remedial action is expected to accomplish (EPA 1999b). They are narrative statements of the medium-specific or area-specific goals for protecting human health and the environment. RAOs are used to help focus development and evaluation of remedial alternatives. RAOs are derived from the baseline risk assessments and are based on the exposure pathways, receptors, and the identified COCs. Narrative RAOs form the basis for establishing PRGs (defined above). RAO is a common CERCLA term. There is no comparable term under MTCA.

Remedial action levels (RALs) are contaminant-specific sediment concentrations that trigger the need for active remediation (e.g., dredging, capping, enhanced natural recovery). This term is used in the FS and has the same meaning as ***remediation level*** under MTCA, which is defined as "a concentration (or other method of identification) of a hazardous substance in soil, water, air, or sediment above which a particular cleanup action component will be required as part of a cleanup action at a site" (WAC 173-340-200). Remediation levels or RALs are not the same as cleanup levels or PRGs. Remediation levels may be used at sites where a combination of cleanup actions is used to achieve cleanup levels at the point of compliance (WAC 73-340-355 (1)). Remediation levels, by definition, exceed cleanup levels. For the purposes of this FS, the ranges of RALs developed for risk drivers consider the magnitude of risk reduction achieved, the rate of natural recovery, and the different types of remedial actions, such as dredging or enhanced natural recovery.

Risk drivers are used in the FS to indicate the subset of COCs identified in the baseline risk assessments that present the principal risks.[4] Risk drivers, as used in this FS, are synonymous with the MTCA term indicator hazardous substances, defined as the subset of hazardous substances present at a site selected for monitoring and analysis or for establishing cleanup requirements (WAC 173-340-200). This FS uses the term risk drivers.

Other COCs not designated as risk drivers will be discussed in the FS by estimating the potential for risk reduction following remedial actions. In addition, COCs may be assessed as part of the five-year review that is conducted once a CERCLA cleanup is completed, and they may be included in the post-cleanup monitoring program.

Total excess cancer risk is defined by MTCA as "the upper bound on the estimated excess cancer risk associated with exposure to multiple hazardous substances and multiple exposure pathways." In the LDW Human Health Risk Assessment (Windward

[4] This approach has been used in several RODs, including the Anaconda, MT Superfund site, Operable Unit 4 (EPA 1998c); Wyckoff Co./Eagle Harbor, WA (EPA 2000b); and Puget Sound Naval Shipyard Complex, WA (EPA 2000c).

2007b) and this FS, total excess cancer risk is defined as the sum of all cancer risks for multiple contaminants and pathways for an exposure scenario. For example, total excess cancer risks for the child beach play scenario include the dermal exposure pathway and the incidental ingestion pathway. The term "total risk" also applies to the sum of risks for multiple contaminants under a single exposure scenario. For example, the cumulative sum of cancer risks for PCBs, arsenic, carcinogenic polycyclic aromatic hydrocarbons (cPAHs), and dioxins/furans for direct contact netfishing exposure is also called total excess cancer risk in this FS.

1.3.2 Sediment Concentrations

Sediment concentrations are expressed and evaluated in the FS in two ways: as individual point concentrations or as SWACs. Risk-based threshold concentrations were developed in the RI and may be expressed as either point concentrations or SWACs (all defined below).

Point concentrations are contaminant concentrations in sediments at a given sampling location, where each value is given equal weight. Point concentrations are typically applied to small exposure areas (e.g., for benthic organisms with small home ranges). Point concentrations usually pertain to smaller-scale management areas for the protection of benthic communities under the SMS.

Risk-based threshold concentrations (RBTCs) are the calculated sediment and tissue concentrations estimated to be protective of a particular receptor for a given exposure pathway and target risk level. RBTCs are based on the baseline risk assessments and were derived in the RI. Sediment RBTCs are used along with other site information to set PRGs (defined above) in the FS.

Spatially-weighted average concentrations (SWACs) are similar to a simple arithmetic average of point concentrations over a defined area, except that each individual concentration value is weighted in proportion to the sediment area it represents. SWACs are widely used in sediment management and are integral to the determination of sediment cleanup levels. The selected area over which a SWAC would be applied may be adjusted for a specific receptor or activity. For example, LDW-wide SWACs may be appropriate for estimating human health risks associated with consumption of resident seafood, but not for direct contact risks from the collection of clams (which may be harvested only in certain areas), or for risks from direct contact with sediments during beach play (which represents a smaller exposure area). In this manner, site-wide or area-wide SWACs are intended to provide meaningful estimates of exposure point concentrations for either human or wildlife receptors.

SWAC calculations have been used at several large Superfund sediment sites to evaluate risks and cleanup levels (e.g., Fox River, Hudson River, Housatonic River, and Willamette River). For example, the Lower Fox River ROD selected a total PCB remedial action level of 1 milligram per kilogram (mg/kg) dry weight (dw) to achieve a site-wide SWAC of 250 micrograms per kilogram (µg/kg dw) over time.

95% upper confidence limit (UCL95) on the mean is a statistically derived quantity associated with a representative sample from a population (e.g., sediment or tissue chemistry results from a water body) such that 95% of the time, the true average of the population from which the sample was taken will be less than the quantity statistically derived from the sample dataset (e.g., 95% of the time, the true average sediment contaminant concentration for the water body will be less than the UCL95 based on sediment chemistry sample results). The UCL95 is used to account for uncertainty in contaminant concentrations and to ensure that contaminant concentrations are not underestimated.

1.3.3 Terms for Spatial Areas

Definitions of relevant spatial areas used previously in the LDW RI/FS process are provided below, along with definitions that are used in this FS. These definitions describe areas likely to require remediation.

Early action areas (EAAs) are areas identified for management actions (to be completed prior to starting construction of the selected remedy for the LDW) to reduce unacceptable risks in surface sediments. These areas are under some formal process that commits individual parties to conduct sediment cleanup. In 2003, LDWG proposed seven areas as candidates for early cleanup (Windward 2003b). Of the seven initially proposed, five areas are referred to as the EAAs in this FS (Figure 1-2):

- Duwamish/Diagonal
- Slip 4
- Terminal 117
- Boeing Plant 2/Jorgensen Forge
- Norfolk Area.

Early action cleanups have been completed in all or portions of three EAAs by King County, The Boeing Company, and the City of Seattle, and remedy decisions have been issued by EPA for the other two.[5] Sediment cleanups were conducted in the vicinity of

[5] The EAA boundaries are represented in this FS based on best available information as of October 2010 as documented in design documents and final cleanup reports used to delineate EAA boundaries. The Duwamish/Diagonal EAA boundary has been revised from the version used in the RI (Windward 2010) by removing the thin-layer placement area from the EAA footprint. The boundaries of the other EAAs used in this FS match those in the RI, but may differ from the final cleanup boundaries presented in the respective removal design documents or subject to the implemented actions. The Slip 4 and Terminal 117 EAA boundaries used in this FS represent those in the EPA-approved project (EE/CAs; Integral 2006 and Windward et al. 2010). The Boeing Plant 2 boundary was defined in 2008 with EPA approval of the Horizontal Boundary Technical Memorandum (Geomatrix and FSI 2008) and the subsequent Final Decision and Response to

the Norfolk combined sewer overflow/storm drain (CSO/SD) at the Norfolk EAA in 1999 and in the vicinity of the Duwamish/Diagonal CSO/SD in 2004/2005 by King County under a 1991 Natural Resource Damage Consent Decree. A much smaller sediment cleanup was conducted at the Norfolk EAA in 2003 by The Boeing Company in the vicinity of the Boeing Developmental Center's south storm drain under Ecology's voluntary cleanup program. In 2012, active cleanup was completed in Slip 4 by the City of Seattle under a formal cleanup Settlement Agreement, also known as an Order, with EPA. The two other EAAs (Boeing Plant 2/Jorgensen Forge and Terminal 117) are in various stages of remedial planning and implementation under Orders with EPA. Together, these five EAAs cover 29 acres, representing some of the highest levels of sediment contamination in the LDW (refer to Section 2 for additional details).

The EAAs are discussed in this FS because they are an integral part of the overall cleanup effort for the site. The EAAs are not included in the cost estimates for remedial alternatives. However, the areal extent and cleanup costs for these EAAs are provided in Section 8 for informational purposes. Remedial alternatives for the EAAs were evaluated in design reports, EE/CA reports, corrective measures studies, or similar documents (e.g., Integral 2006 and 2007; King County 1996, 2000, and 2003; MCS Environmental and Floyd|Snider 2006; RETEC 2006; Windward et al. 2010; Project Performance Corporation 2003).

Areas of potential concern (AOPC) represent the areal extent of sediments that present unacceptable risks and will likely require active or passive remedial technologies to be applied (e.g., dredging, capping, or future monitoring). The AOPC footprints are delineated using sediment PRGs (either on a point basis or by selecting points where remediation would yield a SWAC that achieves a PRG) and other applicable risk information (e.g., current or future exposure pathways). Sediment management method(s) considered within the AOPCs will be compatible with the physical, chemical, biological, and engineering factors present (EPA 1988, Ecology 1991).

Recovery categories have been delineated to represent areas of the LDW with differing potential for natural recovery based on physical characteristics and chemical trends observed in sediment samples. These categories are defined in detail in Section 6.3.

Site, as noted in the beginning of this section, would typically refer to the entire Superfund Site, as defined by EPA or Ecology. The term "site" is frequently used in this FS to refer to just the sediment and surface water portions of the LDW Superfund Site (RM 0.0 to RM 5.0), and generally not to the upland portions.

Comments for Boeing Plant 2 Sediments (EPA 2011d). The Jorgensen Forge boundary was defined in 2008, and EPA has approved the final EE/CA (Anchor QEA 2011). These boundaries differ from those identified in the 2003 Identification of Candidate Sites for Early Action (Windward 2003b). The two remaining areas proposed as candidates for cleanup were not carried forward as EAAs and are included in the area being considered for remediation in this FS.

1.3.4 Terms Related to Time Frames

The remedial alternatives refer to different time frames when describing different aspects of the remedy, such as the number of years to design or implement a remedy, or the number of years to achieve the cleanup objectives for the RAOs. For clarity, the terms related to time frames used in the FS are defined below.

Construction period. The time assumed necessary to construct the remedial alternatives. This period is assumed to begin 5 years following issuance of the ROD. During this 5-year period, the EAAs will be completed (i.e., Alternative 1); priority source control actions, negotiation of orders or consent decrees, initial remedial design/planning, baseline monitoring, and verification monitoring will also be conducted.

MTCA restoration time frame. The time between the start of construction and achievement of the cleanup objectives for the RAOs, either individually or comprehensively. This is discussed in the context of the MTCA evaluation in Section 11 and is the same as the term "time to achieve cleanup objectives" used for CERCLA.

Monitored natural recovery (MNR) period. The time during which the MNR-specific level of monitoring is needed in areas designated for this passive remedial technology. Monitoring conducted during the MNR period will assess whether sufficient progress is being made toward achieving cleanup objectives, or, alternatively, whether contingency actions are warranted to meet the project goals (e.g., the SMS). This FS makes an important distinction between "MNR" and "natural recovery." "Natural recovery" is a term used to describe the condition where natural recovery processes are expected to continue reducing surface sediment concentrations but no contingency actions are anticipated if cleanup objectives are not achieved.

Time to achieve cleanup objectives. The time from the start of remedial construction to when cleanup objectives (see Section 1.3.1) are achieved.

1.4 Document Organization

The remainder of this document is organized as follows:

- Section 2 (Site Setting, RI Summary, and Current Conditions) builds on the key findings of the RI and focuses on the site characteristics that affect the development of AOPCs, selection of representative technologies, and assembly of alternatives. The FS dataset, which includes additional chemistry data not included in the RI baseline dataset and additional physical data needed for engineering considerations, is summarized in this section.

- Section 3 (Risk Assessment Summary) presents the results of the baseline human health and ecological risk assessments (Windward 2007b and 2007a) and the RBTCs for risk drivers.

- Section 4 (Remedial Action Objectives and Preliminary Remediation Goals) presents the recommended RAOs, ARARs, and identifies PRGs for the FS.
- Section 5 (Evaluation of Sediment Movement and Recovery Potential) presents the framework and analysis of sediment movement in the LDW (through the sediment transport model and the bed composition model), describes the methods for predicting changes in sediment chemistry, and reviews the chemical trends for LDW surface sediments.
- Section 6 (Areas of Potential Concern, Remedial Action Levels, and Recovery Potential) presents the AOPC footprints and the array of RALs that may be applied within the AOPCs, and presents the recovery categories that delineate the potential for natural recovery within the LDW.
- Section 7 (Identification and Screening of Remedial Technologies) screens a broad array of remedial approaches and identifies representative technologies that may be applied to the AOPCs.
- Section 8 (Development of Remedial Alternatives) describes LDW-wide remedial alternatives designed to achieve the RAOs, based on the AOPC footprints and representative technologies.
- Section 9 (Detailed Analysis of Individual Remedial Alternatives) screens the remedial alternatives individually using CERCLA guidance. The risk reduction achieved by each remedy is also discussed.
- Section 10 (CERCLA Comparative Analysis) compares the remedial alternatives on the basis of CERCLA evaluation criteria.
- Section 11 (Detailed Evaluation of MTCA Requirements for Cleanup Actions) evaluates the remedial alternatives on the basis of MTCA requirements. This section also presents the disproportionate cost analysis that evaluates the benefits of each remedial alternative in proportion to its cost.
- Section 12 (Conclusions) summarizes the key findings of the FS and presents a general remedial approach for cleaning up the LDW.
- Section 13 (References) provides publication details for the references cited throughout the text.

Tables and figures appear at the end of the section in which they are first discussed. Details that support various analyses in the FS are presented in the appendices.

Table 1-1 Comparison of CERCLA and MTCA Cleanup Requirements

Criteria	CERCLA Requirements (Federal)	MTCA Requirements (State)
CERCLA and MTCA Threshold Criteria	**Overall protection of human health and the environment**	
	40 CFR 300.430(e)(9)(iii)(A) • How alternative provides human health and environmental protection	The first threshold requirement under MTCA is to protect human health and the environment (WAC 173-340-360(2)(a)(i)); also a component of setting cleanup levels (WAC 173-340-700(2)). MTCA's second threshold requirement is compliance with cleanup standards (WAC 173-340-360(2)(a)(ii)).
	Compliance with ARARs	
	40 CFR 400.430(e)(9)(iii)(B) • Substantive requirements from all federal environmental laws and more stringent state environmental and facility siting laws	MTCA's third threshold requirement is compliance with state and federal laws (WAC 173-340-360(2)(a)(ii)-(iii)). For sediment cleanups, MTCA requires compliance with SMS (WAC 173-340-760).[a]
	Compliance monitoring	
	• Compliance monitoring is not a specific component of CERCLA's selection criteria, but generally required under CERCLA's provisions regarding operation and maintenance of the remedy	MTCA's fourth threshold requirement is to provide for compliance monitoring (WAC 173-340-360(2)(a)(iv)) including protection monitoring, performance monitoring, and confirmational monitoring (WAC 173-340-410).
CERCLA Balancing Criteria and MTCA Minimum Requirements	**Long-term effectiveness and permanence**	
	40 CFR 300.430(e)(9)(C) • Magnitude of residual risk • Adequacy and reliability of controls	MTCA requires use of permanent solutions to the maximum extent practicable (WAC 173-340-260(2)(b)(1)). Practicality is determined using a disproportionate cost analysis (WAC 173-340-360(3)(e)). Part of the disproportionate cost analysis is evaluating "effectiveness over the long term," which includes the same criteria for CERCLA to evaluate long-term effectiveness and permanence (WAC 173-340-360(3)(f)(iv)). MTCA also requires a reasonable restoration time frame (WAC 173-340-360(2)(b)(ii)), institutional controls and financial assurances where necessary, control of present and future releases and migration of hazardous substances (WAC 173-340-360(2)(e) & (f)). MTCA does not allow cleanup to rely primarily on dilution and dispersion (WAC 173-340-360(2)(g)).

Table 1-1 Comparison of CERCLA and MTCA Cleanup Requirements (continued)

Criteria	CERCLA Requirements (Federal)	MTCA Requirements (State)
CERCLA Balancing Criteria and MTCA Minimum Requirements (continued)	**Reduction in toxicity, mobility, or volume through treatment**	
	40 CFR 300.430(e)(9)(D) • Treatment process used and materials treated • Amount of hazardous materials destroyed or treated • Degree of expected reductions in toxicity, mobility, and volume • Degree to which treatment is irreversible • Type and quantity of residuals remaining after treatment • Degree to which treatment reduces the risks from principal threats	The corresponding criterion under MTCA is the evaluation of the permanence of an alternative in the disproportionate cost analysis (WAC 173-340-360(3)(f)(i)). MTCA's individual criteria in evaluating permanence correspond to CERCLA's criterion for evaluating the reduction of toxicity, mobility, or volume.
	Short-term effectiveness	
	40 CFR 300.430(e)(9)(E)(1)-(3) • Protection of community during remedial actions • Protection of workers during remedial actions • Environmental impacts • Time until remedial action objectives are achieved	Short-term risks are evaluated as part of the disproportionate cost analysis under MTCA. MTCA's language is a bit broader, but compliance with CERCLA's requirements would satisfy MTCA's as well (WAC 173-340-360(3)(f)(i)).[b]
	Implementability (technical feasibility, administrative feasibility, availability of services and materials)	
	40 CFR 300.430(3)(9)(F)(1)-(3) • Ability to construct and operate the technology • Reliability of the technology • Ease of undertaking additional remedial actions, if necessary • Ability to monitor effectiveness of remedy • Ability to obtain approvals from and coordination with other agencies • Availability of off-site treatment, storage, and disposal services and capacity • Availability of necessary equipment and specialists • Availability of prospective technologies	Technical and administrative implementability is part of the disproportionate cost analysis and includes a very similar assessment of administrative issues and availability of services and materials (WAC 173-340-360(3)(f)(vi)).[c]
	Cost	
	40 CFR 300.430 (e)(9)(G)(1)-(2) • Capital costs, direct and indirect • O&M costs • Net present value of capital and O&M cost	MTCA includes similar cost considerations in the disproportionate cost analysis.[d]

Table 1-1 Comparison of CERCLE and MTCA Cleanup Requirements (continued)

Criteria	CERCLA Requirements (Federal)	MTCA Requirements (State)
CERCLA Modifying Criteria	**Community acceptance**	
	40 CFR 300.430(e)(9)(I) • Completed after the public comment period on the Proposed Plan.	MTCA requires consideration of public concerns solicited throughout the cleanup process pursuant to WAC 173-340-600 and community acceptance (including concerns of individuals, community groups, local governments, tribes, and federal and state agencies) is one of the factors to be weighed in performing a disproportionate cost analysis (WAC 173-340-360(3)(f)(vii)).
	State and tribal acceptance	
	40 CFR 300.430(e)(9)(H) • Completed after the public comment period on the Proposed Plan.	Same as for Community Acceptance

Sources: EPA 1988. *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA. Interim Final*. EPA/540/G-89/004. October 1988.
Ecology 2001. Model Toxics Control Act. Title 173, Washington Administrative Code, Chapter 173-340. Amended February 12, 2001.

Notes:

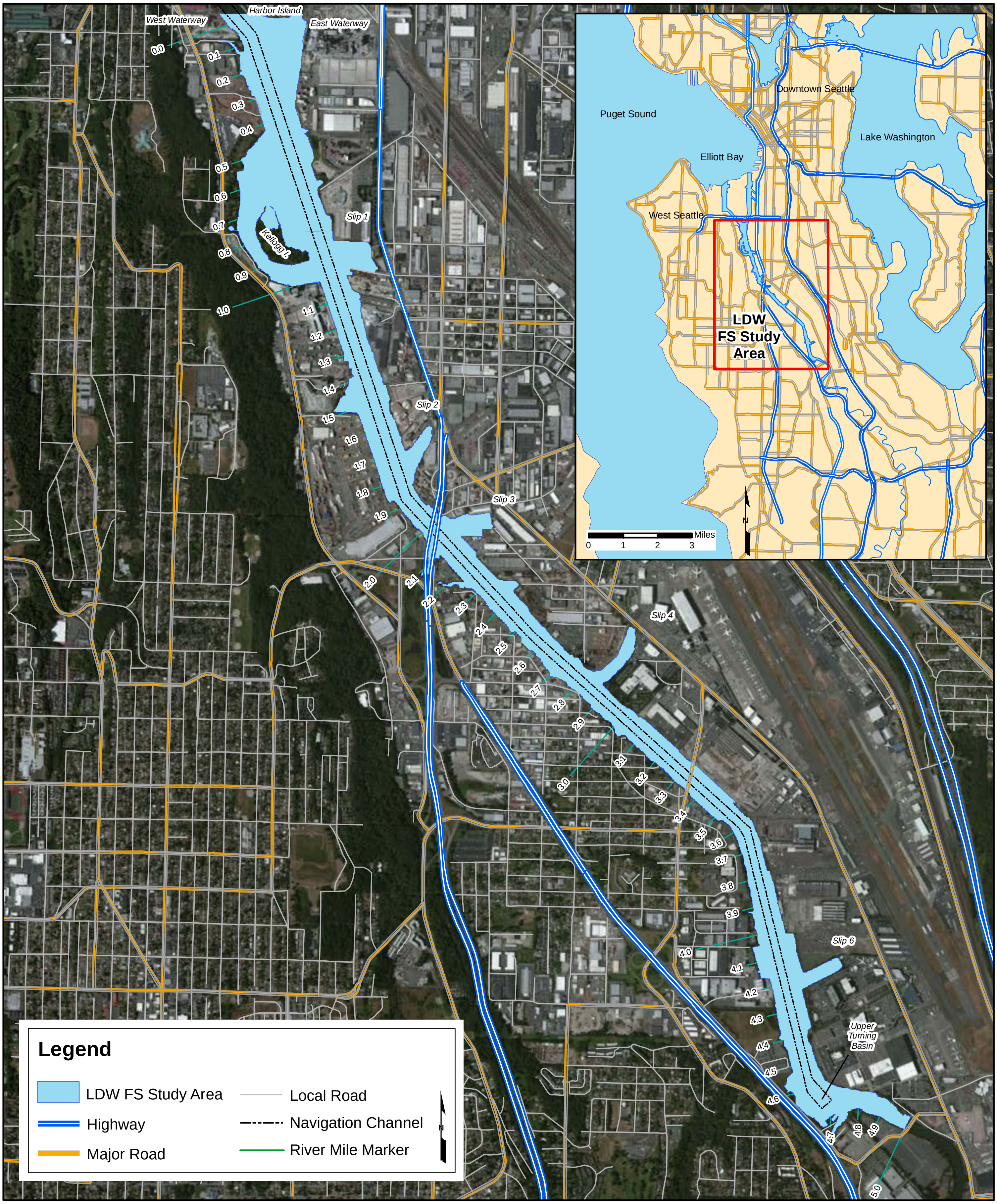
a. SMS requirements are a part of and are consistent with MTCA. SMS *numerical* criteria address risk to the benthic community and apply only to RAO 3 in this FS. SMS *narrative* criteria for protection of human health and biological resources are consistent with MTCA and CERCLA, which define the approach for addressing RAOs 1, 2, 3, and 4 in this FS.

b. The SMS generally requires that cleanup actions meet a "minimum cleanup level" defined as "the maximum allowed chemical concentration and level of biological effects permissible at the cleanup site to be achieved by year ten after completion of the active cleanup action" (WAC 173-204-570(3)). However, where it is not practicable to achieve minimum cleanup levels, Ecology may authorize longer cleanup time frames. (WAC 173-204-580(3)(b)).

c. See also SMS requirements at WAC 173-204-560(4)(g).

d. The final evaluation of cleanup alternatives under the SMS requires consideration of cost, including consideration of present and future direct and indirect capital, operation, and maintenance costs and other foreseeable costs. WAC 173-204-560(4)(h).

ARARs = applicable or relevant and appropriate requirements; CERCLA = Comprehensive Environmental Response, Compensation, and Liability Act; CFR = Code of Federal Regulations; O&M = Operation and Maintenance; MTCA = Model Toxics Control Act; RAO = remedial action objective; SEPA = State Environmental Policy Act; SMS = Sediment Management Standards; WAC = Washington Administrative Code



AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.33			Lower Duwamish Waterway FS Study Area
DATE: 10/31/12	DWRN:MVI/Sea	Revision: 0	FIGURE 1-1

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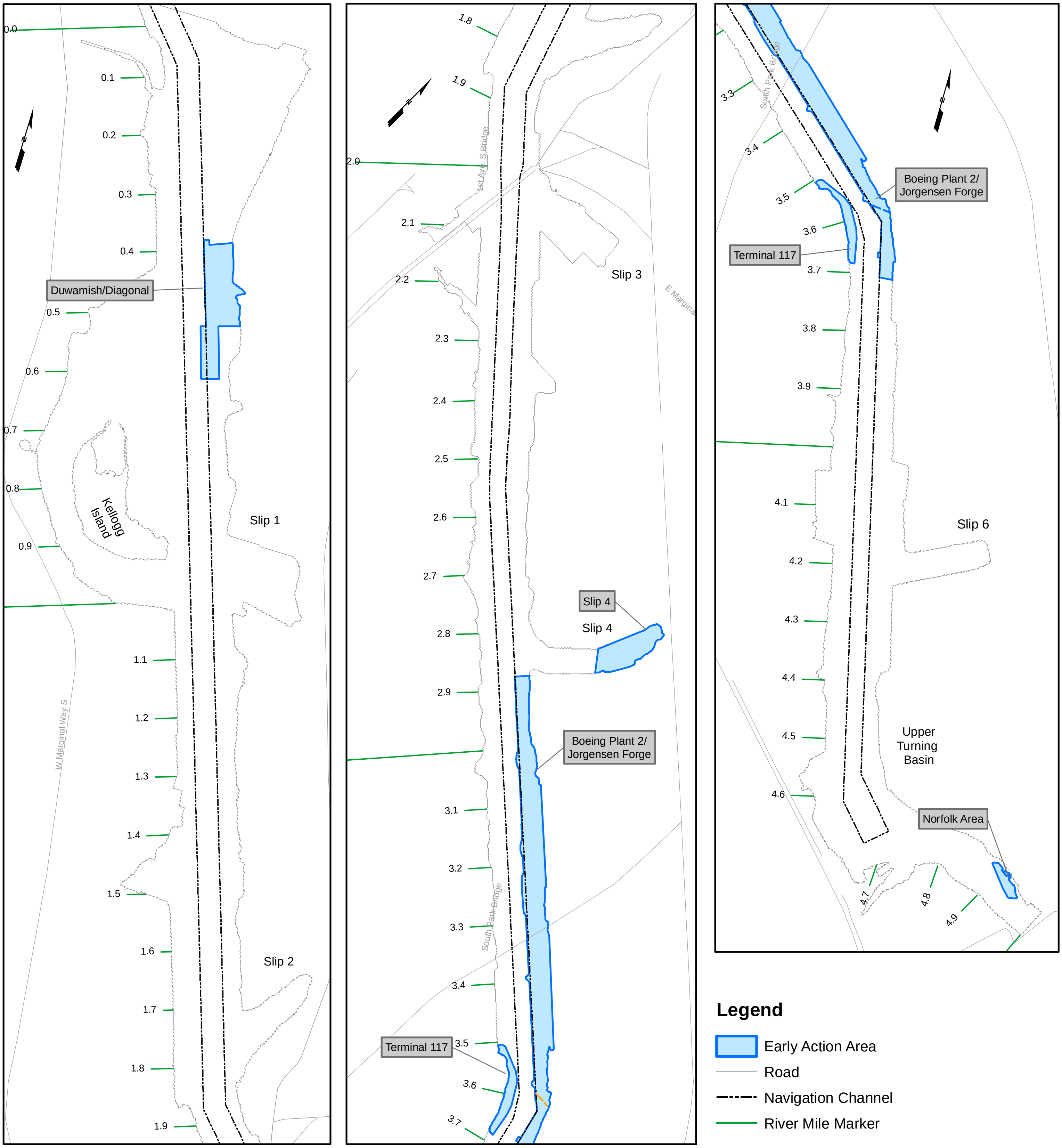

Notes:
1. The EAAs consist of 29 acres.
2. The Norfolk Area, Duwamish/Diagonal, and Slip 4 EAAs have been the subject of cleanup actions.
3. The Duwamish/Diagonal EAA boundary shown on this map has been revised from the version used in the RI (Windward 2010) by removing the thin-layer placement area from the EAA footprint. The boundaries of the other EAAs used in this FS match those in the RI, but may differ from the final cleanup boundaries presented in the respective remedial design documents or subject to the implemented actions. The Slip 4 and Terminal 117 EAA boundaries used in this FS represent those in the EPA-approved project EE/CAs (Integral 2006 and Windward et al. 2010). In 2012, active cleanup at Slip 4 was completed and cleanup at Terminal 117 was scheduled for completion in 2013/2014. The Boeing Plant 2 boundary was defined in 2008 with EPA approval of the Horizontal Boundary Technical Memorandum (Geomatrix and FSI 2008) and the subsequent Final Decision and Response to Comments for Boeing Plant 2 Sediments (EPA 2011d). The Jorgensen Forge boundary was defined in 2008, and EPA has approved the final EE/CA (Anchor QEA 2011).
4. Dashed line denotes the ownership boundary between Boeing Plant 2 and Jorgensen Forge properties within this EAA.

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AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.33	Early Action Areas
DATE: 10/31/12 DWRN:MVI/sea Revision: 1	FIGURE 1-2

2 Site Setting, RI Summary, and Current Conditions

This section summarizes the portions of the remedial investigation (RI; Windward 2010) relevant to the feasibility study (FS). It also introduces more recent data made available since finalization of the RI baseline dataset and analyses conducted for engineering purposes.

2.1 Environmental Setting

The Duwamish River originates at the confluence of the Green and Black Rivers near Tukwila, Washington, and flows northwest for approximately 12 miles, splitting at the southern end of Harbor Island to form the East and West Waterways, prior to discharging into Elliott Bay, in Puget Sound, Seattle, Washington.

In the early years of the twentieth century, the last six miles of the Duwamish River were straightened and channelized into a commercial corridor for ship traffic, officially designated as the Lower Duwamish Waterway (LDW) and the East and West Waterways (located near the river mouth). A federally authorized navigation channel runs down the center of the LDW and is 200 ft wide in the downstream reaches and 150 ft wide in the upstream reaches, where it terminates in the Upper Turning Basin at river mile (RM) 4.6 to 4.65. This channel is maintained at depths between -30 ft mean lower low water (MLLW) in the downstream reaches and -15 ft MLLW in the upstream reach.

The LDW Superfund/Model Toxics Control Act (MTCA) study area encompasses 441 acres, is about 5 miles long and approximately 400 feet (ft) wide (with many variations in width where slips and Kellogg Island occur), and consists of the downstream portion of the Duwamish River, excluding the East and West Waterways, which are part of the Harbor Island Superfund site. The LDW study area includes 4.65 miles of the navigation channel and a small portion of the river upstream of the Upper Turning Basin (Figure 1-1).

Outside of the navigation channel, the benches are comprised of sloped subtidal embankments created by the navigation channel deepening, shallow subtidal and intertidal areas (including five slips along the eastern shoreline, and three embayments along the western shoreline), and an island, Kellogg Island, at the downstream end on the western side of the navigation channel. In addition, a comparatively deep area (up to -45 ft MLLW) is present outside the navigation channel between RM 0.0 and 0.4.

The Upper Turning Basin serves as a trap for most of the bed load sediment carried downstream by the Green/Duwamish River. The Upper Turning Basin and portions of the navigation channel just downstream of the Upper Turning Basin are dredged periodically to remove accumulated sediment, reduce sediment transport into the lower reaches of the LDW, and maintain appropriate navigation depths.

The Green/Duwamish River and LDW flow through an industrial and mixed-use residential area in the City of Tukwila, unincorporated King County, and the southern portion of the City of Seattle. The LDW corridor is one of Seattle's primary industrial areas. Two Seattle neighborhoods, South Park and Georgetown, are also adjacent to the LDW to the west and east, respectively. These neighborhoods support a mixture of residential, recreational, commercial, and industrial uses.

The LDW is used for vessel traffic, primarily bulk carriers, tugs, barges, and small container ships, and, to a lesser extent, recreational vessels (refer to Section 2.6.6 for a discussion of vessel traffic). The LDW supports considerable commercial navigation, but is also used for various recreational activities such as boating, kayaking, fishing, and beach play. The LDW, which connects Puget Sound to the Green River, is also an important migratory pathway for salmon.

The LDW is frequently used by Native American tribes as a resource and for cultural purposes. The Muckleshoot Indian Tribe and Suquamish Tribe are both federally recognized tribes and are natural resource trustees for the Duwamish River. The Muckleshoot Indian Tribe currently conducts seasonal commercial, ceremonial, and subsistence netfishing operations in the LDW. The Suquamish Tribe actively manages resources north (downstream) of the Spokane Street Bridge, located just north of the LDW study area.

2.1.1 Site History

The LDW is an estuary that has been extensively modified over the past 100 years by the diversion of two major rivers (the White River and the Cedar River) and by dredging and other modifications.

In 1906, the White River was diverted from the Green River to the Puyallup River to help control flooding.[1] In 1916, the Cedar River was diverted to Lake Washington to provide water for the Lake Washington Ship Canal, a portion of which connects Lake Washington to Lake Union, and resulted in a drop in the elevation of Lake Washington. This caused the Black River, which had been fed by the Cedar River before it flowed into the Duwamish River, to be reduced to a minor stream. The point where this former tributary once joined the Duwamish River is where the Green River becomes the Duwamish River. The Green River is now the primary headwater of the Duwamish River.

These events reduced the flow volume and area of the Duwamish River watershed by about 70%, thus altering the transport of sediment into and within the system. In addition, the Howard Hanson Dam was constructed in 1961 approximately 65 miles

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[1] The White River had been a tributary to the Puyallup River approximately 5,700 years earlier, before a mudflow from Mount Rainier diverted it to the Green River (Booth and Herman 1998).

upstream of the LDW. Construction of the dam effectively decreased peak river flows, which now rarely exceed 12,000 cubic ft per second (cfs). Previously, large flood events (15,000 to 30,000 cfs) occurred. These changes to the river system's hydrology make the dynamics considered in the FS different from those of a natural river of similar size. Sediment dynamics in the LDW are discussed in Section 5.

Between the late 1800s and the mid-1900s, the Duwamish estuary and Elliott Bay underwent massive modifications as the navigation channel and Harbor Island were constructed to support Seattle's early industrial development (Table 2-1). A 1905 U.S. Army Corps of Engineers (USACE) bathymetric survey revealed a meandering river with most of the recorded mudline elevations along the channel being at 0 ft relative to the extreme low water line of 1897. Maximum depths along this channel extended to -10 ft in this datum (Pope 1905). Creation of the East, West, and Lower Duwamish Waterways involved dredging navigation channels, filling marshes and tideflats, and armoring shorelines with levees, bulkheads, slope protection, and other structures. This development resulted in the replacement of about 9.3 miles of meandering river with 5.3 miles of straightened channel by 1916 (Battelle 2001).

Many of the natural curves of the estuary were eliminated when construction of the navigation channel began in 1901 (Figure 2-1). The slips on the east side of the LDW are remnants of those meanders, and the shoreline on the western side of Kellogg Island, a wildlife refuge, reflects the original estuary configuration. Harbor Island, the terminus of the LDW, is a man-made island in an area once occupied by extensive tideflats.

Dredging conducted between 1903 and 1905 created the East and West Waterways, and dredged material from the river was used to create Harbor Island (Weston 1993). As industrial development continued through the 1900s, the East, West, and Lower Duwamish Waterways were deepened and widened to provide vessel access to various industries. Together, the three waterways currently provide over seven miles of inland navigation accessible from Elliott Bay, Puget Sound, and the Pacific Ocean (Battelle 2001).

Kellogg Island is highly altered from its historical size, shape, and function as the result of creating the LDW and the later dredging and diking for dredged material filling that occurred from the late 1940s or early 1950s through the 1970s. These activities greatly altered the island's interior (Canning et al. 1979).

Today, the slips on the east side of the LDW, originally old meander remnants, do not retain their natural character, having armored shorelines that have been filled to steep bank slopes. The shorelines of the slips are dominated by berthing areas and overwater structures. Approximately 3.7 miles of exposed bank are currently present in the LDW, of the approximate 18 miles of combined shoreline and dock face. Very little of this exposed bank is in the location of the original natural meandering riverbank.

2.1.2 Ownership History

Prior to 1920, the LDW was created by King County Commercial Waterway District No. 1 after it had acquired the property necessary for the relocation of the Duwamish River into a commercial waterway. The Waterway District initially created and then maintained navigation depths for a width of 250 feet on either side of the centerline in the LDW. When the Rivers and Harbors Acts of March 1925 and July 1930 authorized the Seattle Harbor Federal Navigation Project and maintenance dredging program, the USACE became responsible for maintaining the navigation channel (USACE 2006).

In 1963, the state legislature authorized port districts to assume all of the assets, liabilities, and functions of the commercial waterway districts. By resolution dated August 13, 1963, the Port of Seattle did so for King County Commercial Waterway District No. 1. Figure 2-2 illustrates the ownership within the LDW.

2.1.3 Hydrogeology, Sediment Stratigraphy, and Surface Water Hydrology

The hydrogeology and sediment properties of the LDW have been influenced both by natural events over geologic time (e.g., earthquakes and lahars, which are mudflows of volcanic material that flow down a river valley) and by anthropogenic events (e.g., channel straightening, dredging, and filling). The Osceola Mudflow and subsequent lahars from Mount Rainier (which occurred approximately 5,700 to 1,100 years ago), cumulatively extended the Duwamish Valley seaward by approximately 30 miles to its current extent (Collins and Sheikh 2005). Lahar events are recorded in the near-surface alluvial deposits of the Duwamish Valley, which extend to depths of roughly 200 ft below ground surface (bgs). These deposits are located within a trough bounded and underlain by either the bedrock unit or dense glacial deposits and non-glacial sedimentary deposits. The geologic history of this valley suggests that the alluvial deposit sequences include estuarine deposits, typically fine sands and silts (often including shell fragments), which progress upward into more complex, interbedded, river-dominated sequences of sand, silt, and gravel. These layers of alluvial deposits delineate the areas of advancing river delta sedimentation that increase in thickness from south to north (Booth and Herman 1998).

On a regional scale, the fill and alluvial deposits can be separated into various generalized units. These units show evidence of the portions of the LDW that used to be meandering river and that were originally upland. They are also used to identify the subsurface depths exhibiting natural properties and those that represent anthropogenic influences.

Based on information derived from upland borings (which can characterize the stratigraphic units of the historical Duwamish River and its floodplain prior to channelization) and LDW sediment cores, these soil and sediment units in the LDW (from younger to older) are:

- **Fill –** The lower Duwamish River was straightened in the early 1900s into a navigation channel, using fill materials derived mostly from local sources. Much of the fill placed in the old river channels during the period of straightening was material dredged to form the straightened channel (USACE 1919), and is similar in hydraulic conductivity to the native younger alluvium. In the vicinity of the LDW, various depths of fill are present, ranging in thickness from 3 to 20 ft. Locally, the shallowest aquifer occurs within the lower portion of this fill material, especially in the northern sections of the LDW where upland areas were created during the last century. The depth of fill varies greatly and generally consists of sand and silty sand in the saturated zone.

- **Younger Alluvium (Qyal) –** Younger alluvium deposits are composed predominantly of sand, silt, gravel, and cobbles deposited by streams and running water (USGS 2005). Younger alluvium has been identified at the bottom of filled Duwamish River channels (USGS 2005). In the central Duwamish Valley, roughly between RM 2.0 and RM 5.2 (with RM 0 being the southern end of Harbor Island and RM 5.2 being just upstream of the study area), younger alluvial deposits are of relatively constant thickness and depth, generally within 5 to 10 ft of present-day mean sea level. These deposits are thicker in the upstream portions of the LDW, with the thickest deposits estimated at a depth of roughly 100 ft bgs. The younger alluvium includes abundant natural organic material, and is often distinguished from the overlying fill by abundant fibrous organic material typical of tidal marsh deposits (USGS 2005). The younger alluvium may also have some gravelly layers.

- **Older Alluvium (Qoal) –** The older alluvium is characterized by estuarine deposits, often including shells at lower depths, and is composed of silts and clays with sandy interbeds (USGS 2005). The older alluvium is commonly identified between 50 and 100 ft bgs in the central Duwamish Valley, increasing in depth toward the mouth of the LDW to a range of 150 to 200 ft bgs. The older alluvium has been best characterized between RM 3.0 and 3.5 (Reach 2) in the central valley, where the older alluvium becomes finer-grained with increasing depth. In this area, the upper two-thirds of the older alluvium typically consist of sand and silty sand, and the lower third consists of sandy silt (Booth and Herman 1998). The older alluvium also becomes significantly finer at the downstream end, with the sand almost completely absent near the mouth of the LDW. Near this downstream location, the older alluvium is composed almost entirely of silt and clay, representing the farthest extent of the delta deposits into the marine waters and displaying the finest-grained material of the Duwamish Valley alluvial sequence.

Based on field observations from the 2006 RI cores and review of core logs from historical reports identified for the RI (Windward 2010) or downloaded from the GeoNW database (ESS 2007), the LDW younger alluvium sediments were grouped into three stratigraphic units. These units were delineated primarily based on unity of density, color, sediment type, texture, gross appearance, and distinct horizon changes:

- Recent material dominated mostly by unconsolidated organic silt
- Interbedded silt and sand with woody debris and shell fragments often present
- Dense non-silty brown sand with silty layers (prechannelization).

Other information (including the presence of debris, depth of unit relative to the units in surrounding cores, and available information on historical dredging events) was also considered. The delineation of these stratigraphic units is important for evaluating remedial alternatives in the FS. Figure 2-3 provides a longitudinal cross section through the LDW navigation channel, and shows the approximate difference in elevations and thicknesses of these units between upstream and downstream areas of the LDW.

The hydrology of the LDW is also affected by the salt wedge, where freshwater from the upstream Green/Duwamish River overlies denser saltwater from Elliott Bay. Water circulation within the LDW, a well-stratified estuary, is driven by tidal actions and river flow; the relative influence of each is highly dependent on seasonal river discharge volumes. Freshwater flowing from the Green/Duwamish River system enters the headwaters of the LDW, and saltwater from Puget Sound enters the lower reaches of the LDW from its mouth. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth. As the freshwater flows over the deeper saltwater wedge, only limited mixing occurs between these freshwater and saltwater lenses, resulting in a lens of freshwater overlying the salt wedge over a significant portion of the LDW a significant portion of the time. The salinity of the surface water varies with river flow and tidal conditions; during times of high river flow, the salinity in the surface water is low, whereas during low-flow conditions, the surface water salinity is higher. Santos and Stoner (1972) characterized the circulation patterns within the tidally influenced water (or salt wedge) area of the LDW, which typically extends from Harbor Island to near the head of the navigation channel. When freshwater inflow is greater than 1,000 cfs, the saltwater wedge does not extend upstream beyond the East Marginal Way South Bridge (RM 6.3; upstream of the study area), regardless of the tide height. During high-tide stages and periods of low freshwater inflow, the saltwater wedge has been documented as extending as far upstream as the Foster Bridge (RM 8.7) (Santos and Stoner 1972). At the river's mouth at the northern end of Harbor Island, a salinity of 25 parts per thousand (ppt) is typical for the entire water column; salinity decreases toward the upriver portion of the estuary. The thickness of the freshwater layer increases throughout the LDW as the river flow rate increases.

Dye studies indicate that downward vertical mixing over the length of the saltwater wedge is almost non-existent (Schock et al. 1998). Santos and Stoner (1972) described how the upstream location or "toe" of the saltwater wedge, typically located between Slip 4 and the head of the navigation channel, is determined by both tidal elevation and freshwater inflow. Fluctuations in tidal elevation also influence flow in the upper freshwater layer, which varies over the tidal cycle.

The U.S. Geological Survey (USGS) measured the average net upstream transport of saltwater below the Spokane Street Bridge and reported it to be approximately 190 cfs (Clemens 2007). This average net upstream flow is about 12% of the average downstream flow measured at the Tukwila gauging station.[2] During seasonal low-flow conditions, saltwater inputs from the West Waterway were more than one-third of the total discharge from the LDW (Harper-Owes 1983).

2.1.4 Seismic Conditions

The Puget Sound region is vulnerable to earthquakes originating primarily from three sources: 1) the subducting Juan de Fuca plate (intraplate), 2) between the colliding Juan de Fuca and North American plates (subduction zone), and 3) faults within the overriding North American plate (shallow crustal) (EERI and WMDEMD 2005). Earthquakes have the potential, depending on epicenter, magnitude, and type of ground motion, to change the vertical and lateral distribution of contaminated sediments in the LDW and soils in the Duwamish drainage basin. This potential is considered during the development and evaluation of remedial alternatives in this FS and will be refined during the remedial design phase.

[2] The USGS Green River gauging station #12113350 is located at RM 12.4.

The following are examples of regional earthquakes by source, estimated probability of occurrence in any given 50-year interval, type and date of events that have historically occurred, and their magnitude (Moment Magnitude Scale [M]),[3] (EERI and WMDEMD 2005):

- Intraplate (84% probability):
  - Nisqually 2001, M6.8
  - Seattle-Tacoma 1965, M6.5
  - Olympia 1949, M6.8
- Subduction Zone (10-14% probability):
  - January 1700, M9 (estimated)
- Shallow Crustal (5% probability):
  - Seattle Fault (approximately 1,100 years ago), M6.5 or greater.

Of particular concern to regional planners is a large earthquake on the Seattle Fault, similar to the one that occurred approximately 1,100 years ago and caused a fault displacement of the bottom of Puget Sound by several feet. The geologic record shows that this earthquake caused a 22-ft uplift of the marine terrace on southern Bainbridge Island, numerous landslides in Lake Washington, and landslides in the Olympic Mountains (Bucknam et al. 1992). Upland sand deposits at West Point, north of Elliott Bay, and at Cultus Bay on the southern end of Whidbey Island (Atwater and Moore 1992) suggest that that earthquake produced a tsunami that deposited up to 10 ft of material in some upland areas.

The Seattle Fault is believed to be capable of generating another major earthquake of M7 or greater (Pratt et al. 1997, Johnson et al. 1996, Brocher et al. 2000). EERI and WMDEMD (2005) developed a hypothetical Seattle Fault earthquake scenario for guiding regional preparation and responses to such a foreseeable event. The earthquake in this scenario was of magnitude M6.7, which has an estimated 5% probability of occurrence in any given 50-year period (once in approximately 1,000 years). This scenario is approximately equal in magnitude to the 1,100-year old Seattle Fault event. This scenario is based upon a shallow epicenter with a surface fault rupture (as opposed

[3] The moment magnitude scale (abbreviated as M) is used by the United States Geological Survey to measure the size of large earthquakes in terms of the energy released. This logarithmic scale was developed in the 1970s to succeed the Richter magnitude scale. It provides a continuum of magnitude values; moderate events have magnitudes of >5.0 and major earthquakes have magnitudes of >7.0. Great earthquakes have magnitudes of 8.0 or higher. Moment magnitude considers the area of rupture of a fault, the average amount of relative displacement of adjacent points along the fault, and the force required to overcome the frictional resistance of the materials in the fault surface and cause shearing.

to the deeper epicenters with other recent events such as Nisqually [2001], Seattle-Tacoma [1965], and Olympia [1949]). The Seattle Fault scenario would have major consequences for liquefaction-induced ground movements that could damage in-water and upland infrastructure in the Duwamish River Valley and lower Green River Valley. Damage to chemical and fuel storage tanks could result in releases. Under the scenario, ground deformation could be up to 3 ft, which would impact seawalls and release upland soils into the LDW. An earthquake of this magnitude would also likely cause widespread disruption of essential services.

Tsunamis could also affect the vertical and horizontal distribution of sediment contamination remaining in the LDW following cleanup and could contribute additional contaminants derived from other sources. Titov et al. (2003) modeled a M7.3 earthquake at the Seattle Fault and the resulting tsunami bore was modeled southward to approximately RM 1.5 on the LDW. The modeled tsunami would inundate Harbor Island, the South of Downtown District, and uplands along that portion of the LDW. The model also predicts some locally high velocities over the bench areas as the bore moves through the lower reach of the LDW.

Palmer et al. (2004) classify the soils in the bottom lands of the Duwamish and Lower Green River valleys as being susceptible to liquefaction, which would tend to magnify earthquake-induced motion. Surficial deposits of clean, dark, fine to medium sand from prehistoric liquefaction-induced ground failure dikes have been observed along the LDW at and near Kellogg Island. These deposits appear to be extrusions of deeper sediments into tidal-marsh deposits that were deposited after the Seattle Fault uplift approximately 1,100 years ago. The largest of the dikes is as much as 18 centimeters (cm) wide and 6 meters long. Kayen et al. (2007) concluded:

> *"Analysis of the stability of the Holocene deltaic deposits using field penetration test data indicates that extensive soil liquefaction and ground failure of native deltaic deposits are likely during moderate to large earthquake events."*

Section 8.1.3.2 includes information about how seismicity has been integrated into other feasibility studies and remedial designs for other projects in the LDW and the adjacent Elliott Bay. In addition, Section 8 discusses post-event responses of monitoring, detection, and repair following an earthquake as integral features of remedial alternatives.

2.1.5 Ecological Habitats and Biological Communities

Ecological habitats of the LDW have been modified extensively since the late 1800s as the result of hydraulic changes, channel dredging, filling of surrounding floodplains, and construction of overwater and bank stabilization structures. The only evidence of the river's original, winding course is present in the remnants of some of the natural meanders along the LDW (several of which are now used as slips) and the area around

Kellogg Island. Remnants of habitat also remain in the LDW, and portions of intertidal habitat are the focus of recent restoration efforts.

Several habitat restoration projects (some including the construction of new public parks) have already been completed. Habitat restoration areas to date in the LDW and immediately upstream of the study area include (Figure 2-4; Windward 2010):

- Port of Seattle/Coastal America at T-105 where a side channel slough was created at a former industrial property at RM 0.1W
- T-107 Public Access Site/Herring's House Park, at RM 0.3W to RM 0.7W near Kellogg Island, where intertidal habitat has been restored at the site of a former lumberyard and habitat restoration has been conducted at the mouth of Puget Creek
- Diagonal Avenue S/T-108 restoration area at RM 0.6E
- General Services Administration marsh restoration area at RM 0.8E
- First Avenue Bridge boat ramp (public access) at RM 2.0E
- Derelict barge removal at RM 2.0W and the construction by the Washington State Department of Transportation of a fish and wildlife habitat restoration channel that connects to an emergent vegetation area at the south landfall of the First Avenue Bridge
- Gateway North/8th Avenue South street end restoration area at RM 2.7E
- South Portland street end park at RM 2.8W
- Hamm Creek restoration area at RM 4.3W, where 1 acre of emergent salt marsh, 2 acres of freshwater wetlands, and nearly 2,000 ft of the Hamm Creek stream bed have been restored
- Muckleshoot Tribe restoration area at Kenco Marine near the Upper Turning Basin at RM 4.6W
- Upper Turning Basin at RM 4.7W, where four restoration projects, including several derelict vessel removals, a Coastal America project, and expansion of intertidal marsh for project-specific mitigations have led to a total of 5 acres of restored intertidal habitat
- South 112th Street mitigation site at RM 5.7E
- King County's Cecil B. Moses Park at RM 5.7W.

2.1.5.1 Habitat Types

The dominant natural habitat types in the LDW are intertidal mudflats, tidal marshes, and subtidal areas. About 98% of the approximately 1,270 acres of tidal marsh and 1,450 acres of mudflats and shallows, as well as all of approximately 1,230 acres of tidal wetland historically present in the historical Duwamish estuary, have either been filled or dredged. Areas of remnant tidal marshes account for only 5 acres of the LDW, while mudflats account for only 54 acres (Leon 1980).

Intertidal habitats are dispersed in relatively small patches downstream of RM 3.0, with the exception of the area around Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the LDW. In these intertidal habitat areas, birds and mammals can be exposed to contaminants either through direct contact with sediment or through consumption of fish or shellfish. However, these areas also provide wildlife habitat in an otherwise industrial waterway.

Kellogg Island is currently designated as a wildlife refuge. Habitat associated with the island encompasses high and low marshes, intertidal mudflats, and filled uplands. A mixture of introduced and native plant and tree species has colonized this 17.3-acre island.

2.1.5.2 Biological Communities

Based on research conducted for the RI, the LDW is home to diverse communities of fish, birds, mammals, and invertebrate species. Typical of estuarine environments, the benthic invertebrate community is dominated by annelid worms, mollusks, and crustaceans. Crustaceans are the most diverse of these three groups in the LDW, including more than 250 taxa. The most abundant large epibenthic invertebrates include slender crabs, crangon shrimp, and coonstripe shrimp. Dungeness crabs are also common, although their distribution is generally limited to the portions of the LDW with higher salinity. Mollusks include various bivalves and snails. Although the vast majority of benthic invertebrate species in the LDW are typical inhabitants of estuarine environments, a few organisms more typical of freshwater environments were found. For example, during the sampling events conducted for the RI, one chironomid larva was collected in intertidal habitat at RM 0.6, two chironomid larvae were collected in intertidal habitat at RM 1.4, and one chironomid larva was collected in the subtidal habitat at RM 1.6 (Windward 2010).

The LDW is inhabited by numerous anadromous and resident fish species. During sampling events conducted for the RI, 53 resident and non-resident fish species were captured in the LDW. Up to 33 resident and non-resident species of fish had been recorded in the LDW in prior sampling events (Windward 2010). As summarized in the baseline ecological risk assessment (ERA; Windward 2007a), shiner surfperch, snake prickleback, Pacific sandlance, Pacific staghorn sculpin, longfin smelt, English sole, juvenile Pacific tomcod, pile perch, rock sole, surf smelt, three-spine stickleback, Pacific herring, and starry flounder were identified as abundant at the time of the sampling events, as were chinook, chum, and coho salmon. Fish abundance in the LDW is greatest in late summer to early fall and is generally lowest in winter.

The Green and Duwamish rivers support eight species of salmonids: coho, chinook, chum, sockeye, and pink salmon, plus cutthroat trout, both winter- and summer-run steelhead, and bull trout. Coho, chinook, and steelhead runs consist of a combination of hatchery-bred and natural stocks, defined as naturally spawning fish that are descended from both wild and hatchery fish (Pentec 2003). Pink and sockeye salmon and bull trout stocks breed in the wild and are of unknown origin (Kerwin and Nelson 2000). Juvenile chinook and chum salmon are highly dependent on estuarine habitats.

Of the salmonid species, chinook salmon have been studied the most extensively in the Green/Duwamish system. Puget Sound chinook salmon were listed as threatened under the federal Endangered Species Act (ESA) on March 24, 1999. The decline of chinook salmon has been attributed primarily to habitat degradation and fragmentation, blockage of migratory corridors, impact from hatchery fish, and commercial and local harvesting practices (Myers et al. 1998).

Other species listed as threatened under the ESA include the coastal Puget Sound bull trout, the Puget Sound steelhead, and the bald eagle, the latter of which is currently under review for delisting (Myers et al. 1998).

Salmonid residence time in the LDW is species-specific. Juvenile chinook and chum salmon have been shown to be present from several days to two months within the LDW, whereas coho salmon pass through the LDW in a few days. Sockeye salmon are rare in the LDW. Salmon found in the LDW spawn mainly in the middle reaches of the Green River and its tributaries. The juvenile outmigration of all five species generally commences during the high-flow months of March to June. Outmigration usually lasts through mid-July to early August (Nelson et al. 2004, Warner and Fritz 1995). During these months, salmonids use the estuary to feed and begin their physiological adaptation to higher salinity waters. As a result, the regulatory agencies have established "fish windows," which generally restrict in-water marine work to the period from October through February.

The aquatic and semi-aquatic habitats of the LDW support a diversity of wildlife species. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds and 6 species of mammals use the LDW at least part of the year (Windward 2010).

2.1.6 Historical and Current Land Uses

Prior to the 1850s, the Duwamish River area was occupied by Native American tribal communities that used the area for fishing, hunting, gathering, and some limited farming. Settlers of European origin began to inhabit the area around the 1850s, clearing the Duwamish shorelines and draining wetlands to accommodate logging and agriculture.

Prior to the 20th century, flooding was a common occurrence in the Green/Duwamish river valley. In the early 1900s, continued issues with flooding led to the installation of levees and dams and subsequent channelization of the river (Table 2-1). The Howard Hanson Dam was constructed in 1961 for flood control and low flow augmentation to preserve fish life when river flows were naturally low (Sato 1997).

After channelization of the LDW in the early 1900s, most of the upland areas adjacent to the LDW have been and are still used for industrial purposes that include cargo handling and storage, marine construction, boat manufacturing, marinas, concrete manufacturing, paper and metals fabrication, food processing, and airplane parts manufacturing (Wilma 2001). The upland areas along the upstream portions of the LDW and along the Green/Duwamish River were used for farming. The LDW continues to be used by the Muckleshoot Tribe as part of their Usual and Accustomed Fishing area, and the Suquamish Tribe fishes the area north of the Spokane Street Bridge, immediately north of the LDW.

Industrial development increased as the mudflats were filled with soil from the regrading of Seattle's former hills. In 1928, Seattle's first municipal airport, Boeing Field, was opened. Seven years later, Boeing opened its Plant 2 on the west side of Boeing Field (Wilma 2001).

Although the area surrounding the LDW is largely regarded as an industrial corridor, the Duwamish estuary subwatershed (extending from RM 11.0 to Elliott Bay) of the Green/Duwamish watershed has more residential land use (36%) than industrial and commercial land use combined (29% combined; 18% and 11%, respectively). Eighteen percent of the subwatershed is used for right-of-way areas (including roads and highways); while 17% is open/undeveloped land and parks (Schmoyer, personal communication, 2011a).

The combined (storm and sanitary) sewer service area and separated storm drainage basin (i.e., the upland areas over which source control investigations/activities are occurring) are 19,800 acres and 8,936 acres, respectively. However, the combined and separated areas overlap in many places; the total area discharging to the LDW is 20,400 acres. Within the 19,800-acre combined sewer service area, land uses are: 36% residential, 15% industrial, 10% commercial, 26% right-of-way, and 13% open space. Within the 8,936-acre separated storm drainage basin, land uses are: 23% residential, 29% industrial, 8% commercial, 26% right-of-way, and 15% open space (Schmoyer, personal communication, 2011a).

Two mixed residential/commercial neighborhoods, South Park and Georgetown, are located adjacent to the LDW. The South Park neighborhood, within and adjacent to the southern edge of the Seattle city limit, borders the west bank of the LDW and includes approximately 984 ft of residential shoreline. The Georgetown neighborhood, located east of the LDW and E Marginal Way S, is separated from the LDW by several commercial facilities, although access to the LDW on foot from this neighborhood is possible. The U.S. Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology) believe there to be potential environmental justice concerns in accordance with Executive Order 12898, *Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations*, for those affected by the LDW site and cleanup. In response, EPA is developing an Environmental Justice Analysis for the LDW Superfund Cleanup, to be published as an appendix to the Proposed Plan.

Four marinas are located in the LDW, and several other access points allow the public to enter the LDW for recreational purposes. In a human access survey conducted along the LDW shoreline as part of the Human Health Risk Assessment (HHRA; Windward 2007b), owners/operators of 93 commercial/industrial, residential, and public properties were surveyed to determine their potential for public access and use. The survey identified 17 locations (in addition to the 4 marinas) used by the public to launch or haul out hand-powered boats or motorboats. In addition, 8 sites along the LDW have been used for swimming, and 10 have been used for picnicking (Figure 2-4). In addition, two public parks (Terminal 107/Herring's House and Duwamish Waterway Park) exist along the LDW shoreline (Figure 2-4). Although recreational use may increase at some point in the future, this area is anticipated to remain primarily commercial, industrial, and residential in use.

2.2 FS Datasets

Between 1990 and 2004, approximately 1,200 surface sediment samples, 340 subsurface sediment cores, and 90 fish and shellfish tissue samples were collected from the LDW by parties other than the Lower Duwamish Waterway Group (LDWG), the entity responsible for performing the RI/FS. These samples and cores were analyzed for metals and organic compounds and the data became part of the RI baseline dataset. Additional data were collected from 2004 to 2006 for the RI/FS to characterize contamination by hazardous substances and physical properties of the LDW. These data included approximately 900 samples of the following media: fish, clam, crab, and benthic invertebrate tissue; seep water (water seeping from banks along the river); surface sediment (the top 10 cm); subsurface sediment (below the top 10 cm); and porewater (water in spaces between sediment particles). Collectively, all of these data represent the baseline dataset used in the RI to characterize the nature and extent of contamination. The RI included data that were available as of October 2006.

Additional data have been collected since the finalization of the RI baseline dataset (i.e., since October 2006). The baseline dataset used in this FS (called the "FS baseline dataset") includes those data newer than October 2006 as well as older data that were not previously included in the RI baseline dataset (Table 2-2). The FS baseline dataset does not include data collected after April 2010. Windward prepared a technical memorandum, *Summary of Chemistry Datasets to be Used in the RI/FS – Addendum 3*, which discusses the data quality for each of these events (Windward 2012, review in progress). Additionally, Appendix N presents the new data included in the FS baseline dataset.

As shown in Table 2-2, data for 174 surface sediment locations and for 509 subsurface sediment samples were added to the RI baseline dataset to create the FS baseline dataset. The percentage of new surface sediment locations in the FS baseline dataset relative to those in the RI baseline dataset varies by analyte; for total polychlorinated biphenyls (PCBs) 7% (101 of 1,392) of the locations in the FS baseline dataset were not in the RI baseline dataset. The RI describes the methods for developing the FS baseline dataset, and Appendix N presents data tables (updated from those in the RI Appendix E).

Additionally, several other datasets, such as those for tissue and water and those for samples collected outside of the LDW, were used in the FS. These other datasets are discussed in Section 2.2.3.

2.2.1 FS Baseline Surface Sediment Data

The sample count for each of the hazardous substances that are human health risk drivers (as described in Section 3) is provided in Table 2-3. This dataset follows the same rules used to establish the RI baseline dataset (Section 4.1.2.1; Windward 2010). Within the early action areas (EAAs) where sediment removal actions have been conducted since the LDW RI/FS Administrative Order on Consent (AOC), (i.e., Duwamish/Diagonal and Boeing Developmental Center south storm drain), the preremedy data are used to characterize baseline conditions.[4] However, because the sediment removal action in the vicinity of the Norfolk combined sewer overflow/storm drain (CSO/SD) was conducted in 1999 prior to the LDW AOC, post-remedy monitoring data from the Norfolk CSO/SD cap are used to represent baseline conditions.

The FS baseline surface sediment dataset includes the baseline dataset used in the RI and the following additional data, which are summarized in Table 2-2. Most of these

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[4] For these areas that have post-remedy monitoring stations with repeated sampling over time, time trend data are used to evaluate the success of remedial technologies (Section 7); for the Duwamish/ Diagonal EAA, the most recent ENR and perimeter data were used in the assembly of remedial alternatives (Section 8).

events were conducted to characterize specific locations; however, two site-wide events were also conducted, as described below and shown in Figures 2-5 and 2-6a through 2-6i:

- Data were collected around the perimeter of and upstream of the Boeing Plant 2/Jorgensen Forge EAA to characterize the boundary of this EAA.
- Surface sediment post-remedy monitoring data were collected around the perimeter of the Duwamish/Diagonal EAA (2005 to 2009) as part of King County's annual monitoring of the cleanup action taken in this area.
- Surface and subsurface sediment data were collected around the perimeter of the Slip 4 EAA for the design report.
- Surface sediment data were collected around the perimeter of the Terminal 117 EAA and analyzed only for total PCBs and dioxins/furans to characterize the boundary of this EAA.
- Data were collected by individual parties at the 8801 E. Marginal (RM 3.9 to 4.0E) and Industrial Container Services (RM 2.2E) facilities. These two facilities are currently under MTCA cleanup orders (Ecology Agreed Order Nos. 6060 and DE 6720, respectively).
- Data were collected by the Port of Seattle in the intertidal area of Terminal 115 (RM 1.8W) prior to 2009 dredging to characterize the intertidal slope shoreward of the dredging prism.
- Data were collected by Ecology to characterize surface sediment upstream of the LDW. Five of these sample locations are at RMs 4.9 and 5.0. The other locations are upstream of the study area, and are thus not a part of the FS baseline dataset. All locations sampled for this event are shown in Figure 2-7. Summary statistics for these data are presented in Table 2-4 and are discussed in Appendix C. A table of all human health risk-driver data from this event is included in Appendix C, Part 3.
- Data from a sediment profile imaging (SPI) study conducted by Ecology were used to examine the feasibility of correlating metrics from sediment profile images with chemical, toxicity, and benthic community data (Gries 2007). This study generated surface sediment chemistry and toxicity data for 30 stations in the LDW from the mouth to Slip 4.
- Historical dioxin/furan data from four EPA 1998 site investigation (SI) surface sediment stations had been removed from the RI baseline dataset in accordance with the RI data trumping rules, which excluded all data for any old location within 10 ft of a newer location. The trumping exercise has

been refined for the FS in that each trumped location was reviewed on an individual contaminant basis. Only the contaminant data for which newer data are available were replaced. Therefore, an older location remains in the dataset when its co-located newer sample was not analyzed for the same suite of analytes as the older location (only the data for the contaminants that were not analyzed in the newer sample are retained from the older sample). Although the data for the trumped contaminants were removed from the FS baseline dataset, they were still used in the FS to evaluate time trends (see Appendix F).

- Data were collected by LDWG in 2009 and 2010. This sampling and analysis effort was conducted to increase the dioxin/furan dataset, which had contained 54 samples in the RI baseline dataset. A second objective of the 2009/2010 LDWG sampling event was to further characterize the beach play areas identified in the HHRA. This event included 41 discrete sediment samples analyzed for dioxins/furans, eight of which were also analyzed for PCBs, arsenic, and carcinogenic polycyclic aromatic hydrocarbons (cPAHs). One grab sample was also analyzed for the full suite of Washington State Sediment Management Standards (SMS) contaminants. Additionally, six composite sediment samples were collected from beaches. However the composite samples were not used in the FS baseline dataset for mapping baseline conditions because only individual grab samples are contained in this dataset. Although not in the FS baseline dataset, the composite samples were used to calculate baseline direct contact risks in beach areas (Section 3) and to evaluate technology assignments in the beach areas (Section 8). These composite data are provided in the project database.

2.2.2 FS Baseline Subsurface Sediment Data

Data from cores collected by six parties since the finalization of the RI baseline dataset were added to the subsurface sediment table in the FS baseline dataset. These parties include both public agencies and private companies:

- The Boeing Company collected 355 samples in 2008 and 2009 along the western boundary of the Boeing Plant 2/Jorgensen Forge EAA and under the historical overwater Plant 2 building to further characterize this EAA.
- PACCAR collected 25 samples in 2008 at RM 3.9 to 4.0E (8801 East Marginal Way) in support of its work under a MTCA cleanup order.
- The City of Seattle collected 38 samples in 2006 and 2008 in the Slip 4 EAA as part of its design work for this EAA.

- The Port of Seattle collected 11 samples in 2008 at Terminal 115 (RM 1.8W) for dredged material characterization to support berth modifications.
- USACE collected 32 samples in 1990, 1991, and 1996 to characterize material to be dredged from the navigation channel and collected 44 samples in 2008 and 2009 to support 2010 dredging. The data from the 1990s events were not included in the RI baseline dataset but have been added to the FS baseline dataset because they were used as lines of evidence for the bed composition model (BCM) upstream input parameters (see Section 5 and Appendix C).
- Delta Marine collected 4 samples at RM 4.2W in 2007 to support dredged material characterization for maintenance and deepening of the berthing area.

Table 2-2 describes each of these sampling events. These events resulted in 174 surface sediment and 509 subsurface sediment samples being added to the FS baseline dataset. Because some of these newer data replaced older data (on an individual contaminant basis), the surface sediment sample count for each one is not 174 greater than that for each contaminant in the RI baseline dataset. Table 2-3 provides the sample counts for each of the human health risk drivers.

These data were collectively used to refine the understanding of the nature and extent of contamination. These refinements were the basis for defining the areas of potential concern (AOPCs; Section 6) and for developing the remedial alternatives (Section 8). The newer data filled some data gaps but did not result in significant changes to the CSM.

2.2.3 Other Datasets Used in the FS

The FS baseline surface and subsurface sediment datasets described above were used to map the nature and extent of contamination in the LDW, to evaluate the remedial alternatives, and to estimate dredging volumes. Those datasets are included in several tables in a Microsoft Access file (FS project database) that accompanies this FS. Each table and dataset included in the FS project database has undergone rigorous quality control checks, as documented in technical memoranda (the most recent being Addendum 3; Windward 2012, review in progress).

Other datasets are also included in the project files, but have not been formatted into the standardized set of fields included in the project database. These files are provided in Microsoft Excel format (often maintained in the same format in which they were received) and have not undergone the same level of quality control checks as the database files. The FS project database and all accompanying Excel files are available on http://www.ldwg.org in one zip file. The zip file also contains an index describing each dataset and its file location. Table 2-5 lists the other datasets that were used as part of the FS.

In 2009 and 2010 (after the RI was finalized), LDWG collected composite surface sediment samples from each of six beach play areas. Because these data did not represent individual locations, they were not used in the FS baseline surface sediment interpolations. However, these data were used in risk calculations described in Section 3 and Appendix B. These data were also used to identify beach play areas potentially subject to active remediation. Appendix B provides maps showing the locations where these samples were collected. These data underwent rigorous quality control checks and are included in the FS project database.

The BCM, discussed in Section 5, was used to evaluate the potential for surface sediment to recovery naturally. Several datasets were used to characterize the contaminant concentrations associated with inputs from lateral and upstream sources. The datasets used to characterize lateral sources included CSO whole-water samples and storm drain solids samples[5] collected within the LDW drainage basin. The datasets used to characterize the contaminant concentrations associated with upstream inputs included dredged material characterization cores collected in the most upstream portion of the LDW navigation channel, surface sediment samples and solids from centrifuged water samples collected upstream of the LDW (many collected by Ecology), and whole-water samples collected by King County upstream of the LDW. All of these datasets are discussed in Appendix C, Part 3, and all except the upstream data collected by Ecology were presented in the RI (Windward 2010). Depending on the nature and source of each dataset, some of these datasets underwent independent quality control checks while others did not. All of these data are included in the FS project data files. An index that accompanies the data submittal (FS project database and accompanying Excel files) indicates where each dataset can be found.

Natural background concentrations of certain contaminants were estimated from a statistical evaluation of surface sediment data collected from Puget Sound. The DMMP agencies collected these data in 2008 during the Puget Sound sediment PCB and dioxin survey (*OSV Bold* Survey; EPA 2008b and DMMP 2009b). These data are discussed in Section 4 of the FS for the development of preliminary remediation goals and are included in a project Excel file, as part of the FS data submittal.

The recontamination potential of remediated sediments is evaluated in Appendix J using sediment time trend data collected within the LDW (Norfolk Area EAA, Duwamish/Diagonal EAA and adjacent enhanced natural recovery [ENR] area, and

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[5] Some of these solid samples were collected from drain lines that contain both CSO and separated stormwater inputs.

post-maintenance dredging surface sediment data from the FS baseline dataset) and from surface sediment data collected in the greater Puget Sound area (urban water body data, Dredged Material Management Program [DMMP] characterization of dioxins/furans, and RI samples collected offshore of greater Seattle area outfalls). These datasets are described in Appendix J. All of these datasets, except the urban water body and DMMP data, are in the FS project database. The other two datasets are in project Excel files.

Long-term surface sediment monitoring data from the perimeter of the Duwamish/Diagonal EAA were also used in this FS to evaluate time trends (recovery potential, see Appendix F). Because the perimeter monitoring locations are outside of the EAA, the most recent data are in the FS baseline dataset. The older resampled data are contained within a separate table in the FS project database. Long-term monitoring data (through 2009) from the Duwamish/Diagonal EAA, adjacent ENR area, and perimeter are also discussed in Section 7 to provide case-study information of remedial technologies used in the LDW.

Resampled locations that were removed from the FS baseline dataset as a result of data trumping were often used to evaluate the potential for natural recovery. Data for the paired locations (older and newer data) are provided in the FS project database, in a separate table. This table has a different format than other tables containing sediment data because the table pairs data from older and newer samples at each re-occupied location.

LDW tissue data were used for seafood consumption risk estimates in the HHRA (Windward 2007b) and have undergone quality checks. These data are included in both the RI and FS project databases. In addition, LDW tissue data collected in 2006 and 2007 are included in the project databases. Tissue data collected from Puget Sound and used for background calculations are discussed in Appendix B. These data are provided in a separate Excel table in the FS data files.

Seep and porewater data collected by LDWG and presented in the RI were compared to water quality criteria in Section 4 of this FS. These data are also discussed in Appendix N of this FS and are provided in both the RI and the FS project databases (Access files).

2.3 Conceptual Site Model (CSM)

The CSM for the LDW describes the physical and chemical conditions of the study area. The physical CSM describes the LDW in terms of three reaches: Reach 1 in the downstream portion of the LDW, Reach 2 in the middle, and Reach 3 in the upstream portion. Each reach has three distinct segments: a shallow (intertidal) bench area, a deep (subtidal) bench area, and the navigation channel. The three reaches were determined based on geomorphology and sediment dynamics, as described in Section 2.3.1.

The chemical CSM, which is discussed in Section 2.3.2, describes the distribution of contaminants of concern (COCs), specifically the risk drivers, in sediment. Sediment with the highest concentrations of risk drivers is not distributed uniformly across the LDW, but rather occurs in concentrated areas (e.g., EAAs). In depositional areas, higher contaminant concentrations are buried in the subsurface sediment by lower-concentration surface sediment originating from the upstream Green/Duwamish River. This aspect of the chemical CSM, along with a few notable exceptions, is discussed further in Section 5.

The CSM also identifies the potential sources of contaminants and the pathways by which contaminants may reach the LDW surface sediments and interact with receptors. A CSM generally incorporates information about sources, transport pathways, exposure pathways, and receptors and can be a valuable tool for evaluating the potential effectiveness of cleanup alternatives. The sources and transport pathways are discussed in Section 2.3.3. The exposure pathways and receptors are discussed in Section 3.

2.3.1 Physical CSM (Sediment Dynamics)

Sediment dynamics have been quantified through two sequential sediment transport models, with results published in the *Sediment Transport Analysis Report* (STAR; Windward and QEA 2008) and the *Sediment Transport Modeling Report* (STM; QEA 2008). The STAR, which documents the hydrodynamics related to water flow, identified three CSM reaches in the LDW, taking into consideration the geomorphology, extent of the saltwater wedge, and relative scour potential. The STM, which documents the movement of sediment (related to scour, deposition, and transport patterns), was then used to refine the CSM.

The STM (QEA 2008) built on the results of the hydrodynamic model and quantified sediment loading from different sources to each grid cell of the model domain (and from grid cell to grid cell) over time. Upstream river flow data spanning a 21-year period (1960 to 1980) were used to calibrate the STM. These data were used to establish the boundary conditions (i.e., upstream sediment load, hydrograph flow events, net sedimentation, and scour) used in model simulations (see QEA 2008, Appendix B). The movement of suspended and bed load sediment into the LDW from upstream and through the LDW was modeled over a 30-year (1960 to 1989) period. Average river flows were estimated to be 1,340 cfs, while river flows during the 100-year high-flow events are about 12,000 cfs (QEA 2008). Estimates of lateral inflows to the LDW from storm drains, CSOs, and streams were based on recent data collected by the City of Seattle and King County (QEA 2008, SPU 2008).

Results of the hydrodynamic and sediment transport modeling indicate that the LDW can be broadly separated into three reaches during high-flow conditions (shown in Figures 2-8a through 2-8c):

- Reach 1 is downstream (north) of RM 2.2 and is occupied by the saltwater wedge during all flow and tidal conditions. Sedimentation rates are variable; although this reach is net depositional in both the navigation channel and the adjacent bench areas. In the navigation channel, sedimentation rates vary from intermediate to high, with a small area near RM 0.8 to RM 0.9 having lower deposition rates. Net sedimentation rates on the benches are also intermediate to high, with two small areas having lower deposition. Empirical data show that the intertidal areas have relatively low net sedimentation rates, on the order of 0.5 cm/year. This reach is not likely to be subject to scour during the 100-year, spring-tide, high-flow event except in a few localized areas.

- Reach 2 extends from approximately RM 2.2 to RM 4.0 and includes the toe of the saltwater wedge during high-flow events; the saltwater wedge extends even farther upstream during average-flow conditions. The toe of the saltwater wedge is pushed downstream of this reach (to RM 1.8) only during extreme flow events (100-year, high-flow event and greater). Reach 2 is subject to some scour during high-flow events but is net depositional on annual time scales. Net deposition rates are spatially variable within this reach.

- Reach 3 extends from RM 4.0 upstream to RM 5.0. Flow in portions of this reach is characteristic of a freshwater tidal river during high-flow events. This reach is occupied by the saltwater wedge only during low- and average-flow conditions. This reach is also net depositional on annual time scales. Both the model and empirical data indicate that the navigation channel and Upper Turning Basin located in Reach 3 have higher net sedimentation rates than other areas of the LDW. Greater episodic erosion may occur in this reach than in the other reaches during high-flow events.

The STM (QEA 2008) also evaluated three physical processes significant for the FS: 1) bed stability related to scour potential from high-flow events and passing ship traffic, 2) net sedimentation rates, and 3) solids loading into and out of each model grid cell in the LDW. The sediments within each model grid cell are the result of these processes, and represent contributions from upstream sources, from within the LDW, and from lateral sources, collectively defined as bed composition. These processes are discussed in the following subsections.

2.3.1.1 Sediment Bed Stability and Scour Potential

Scour of bed sediment materials can be caused on a reach-wide scale by river discharge during high-flow events (i.e., high-flow-induced scour, see Figure 2-9) and by vessel traffic moving along the navigation channel. On localized scales, scour can occur as a result of vessel maneuvers in berthing areas (Figure 2-10). These three types of scour are discussed below.

High-Flow-Induced Scour

Scour of surface sediment as a result of high-flow events is a quantifiable disturbance. Based on historical data, high-flow periods are more tempered now than before construction of the Howard Hanson Dam. However, high-flow-induced scour events still occur when upstream inflow increases.

For the STAR (Windward and QEA 2008), field-derived erosion property data were collected from near-surface sediment within the LDW, and an analysis of natural erosion events was performed. The analysis focused on bed stability during episodic 2-, 10-, and 100-year high-flow events, which correspond to flows of 8,400, 10,800, and 12,000 cfs, respectively. In contrast, average flows are estimated to be 1,340 cfs.

Erosion rates as a function of shear stress and depth in the sediment bed were assessed in a laboratory using sediment cores collected from the LDW. Erosion rate tests were conducted using Sedflume, a device that gauges gross erosion rates over a range of shear stresses at various depths in a sediment core. These tests were used to predict erosion rates and critical shear stresses necessary to result in resuspension under various flow conditions. The relationship between shear stress and erosion rate was used to identify areas in the LDW that could potentially experience erosion under Green/Duwamish River discharge conditions ranging from average flow to the 100-year high-flow event. The general findings identified by the STAR (Windward and QEA 2008) and updated in the STM (QEA 2008) are summarized below:

- During all flow conditions, bed shear stress tends to be higher in the navigation channel than in the bench areas.
- During high-flow events in Reach 1, negligible bed scour occurs in most of the area downstream of RM 1.8. The denser saltwater wedge acts as a layer of protection against the high-flow velocities occurring above the salt wedge.
- During high-flow events in Reaches 2 and 3 (i.e., upstream of the saltwater wedge):
    - Generally, higher excess shear stresses occur in the navigation channel than on the benches for a given high-flow event and tidal condition.

- Minor differences exist in the general spatial pattern of excess shear stress during ebb and flood tides. Bed shear stresses are higher during spring tides than during neap tides.[6]
- Within the portions of the bench areas where erosion was predicted to occur, the potential for erosion tends to be highest near the navigation channel and tends to decrease toward the shoreline.
- Reach 3 tends to have higher excess shear stress values than the other reaches, but it also has higher sedimentation rates.

Overall, the maximum net erosion depth during a 100-year high-flow event is 22 cm, occurring in and just west of the navigation channel at RM 3.1 (Figure 2-11). Areas with high-flow scour exceeding 10 cm occur in scattered locations upstream of RM 2.9. See Section 5 for a discussion of model uncertainty related to the STM.

Ship-Induced Bed Scour from Passing Vessels Transiting the Navigation Channel

Propeller wash from vessels can produce increased bottom shear stress and, as a result, localized scour in some cases. The depth to which the erosion will occur varies with the velocity of the vessel, sediment type, and duration and frequency of the event. Propeller wash effects are generally proportional to the size, draft, and power of vessels; larger, deeper, and more powerful vessels exhibit propeller wash effects to greater depths. Propeller wash effects are most evident where navigation activity is concentrated, and where water depths are shallow and matched to the size of the vessels using the channels and berths.

The STAR (Windward and QEA 2008) reported the predicted results of a screening-level evaluation of transiting vessels in the navigation channel and their ship-induced bed scour, using parameters from two active, representative tugboats in the LDW, the *J.T. Quigg* and the *Sea Valiant*.[7] The results from the STAR and STM (Windward and QEA 2008, QEA 2008) are summarized as follows:

- Within the navigation channel, ship movement at the speed limit of 5 knots causes an average bed scour depth of less than 1 cm (and a maximum depth of 1 cm) per ship passage in Reach 1 and an average bed scour depth of less than 0.1 cm (and a maximum depth of 0.3 cm) per ship passage in

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[6] Spring tides occur during full- and no-moon phases, and the difference between higher high tide and lower low tide is maximum. Neap tides occur during the first and third quarters of the lunar cycle, and the difference between tide heights is minimal. Tides also vary with the solar cycle, with the amplitude being greatest (highest highs and the lowest lows) during the summer and winter solstices.

[7] These vessels are representative of those working in the LDW. Each ship has an open wheel propeller. The *J.T. Quigg* is a 100-ft long, 3,000-horsepower vessel. The *Sea Valiant* is a 128-ft long, 5,750-horsepower vessel.

Reaches 2 and 3. Within the bench areas, each ship movement at the speed limit of 5 knots can cause an average bed scour of about 1 to 2 cm in Reach 1 and less than 1 cm in Reaches 2 and 3.

- Reducing ship speed from the LDW speed limit of 5 knots to 2.5 knots significantly reduces bed scour, with predicted bed scour of less than 1 cm throughout the LDW for all conditions. Doubling the applied ship power has minimal effect on predicted scour depth. The typical vessel speed in the LDW is 2 to 3 knots (Riley, personal communication, 2006; Takasaki, personal communication, 2006).
- The reworked (i.e., mixed) sediment layer is equated with the depth of gross bed scour, based on the assumption that the same layer is continually reworked. The upper-bound estimate is less than a 10-cm depth. The most-downstream reach (Reach 1) was estimated to have an upper-bound average scour thickness of less than about 1 cm in the navigation channel and about 1 to 2 cm in bench areas. In the middle and upstream reaches (Reaches 2 and 3), the reworked sediment layer was estimated to have an upper-bound average thickness of less than 0.1 cm in the navigation channel and less than 1 cm in bench areas. The frequency of mixing is about 100 to 250 events per year.
- Bed scour by passing vessels does not have a significant effect on the erosion rate properties at particular locations in the bench areas or navigation channel of the LDW. These areas are conceptually displayed in a series of CSM figures (Figures 2-8a through 2-8c).

The effects of ship-induced bed scour are incorporated into the present structure of the LDW sediment bed because ship movement has been occurring for at least the past 40 years (Windward and QEA 2008). Ship-induced bed scour is viewed as an impulsive erosion-deposition process that tends to behave like an ongoing, small-scale, shallow mixing process for surficial bed sediment. Scour by transiting ships is not a significant sediment transport mechanism because it's estimated to occur in few grid cells, and where scour is estimated, the depth is shallow (less than 1 cm per ship passage in Reach 1 [RM 0 to RM 2.2], and less than 0.1 cm per ship passage in Reach 2 [RM 2.2 to RM 4.0]). The estimated scour depth is within the top 10 cm active mixing layer, and is therefore merely another mixing process within that zone. It is not a significant transport mechanism relative to the other active mixing processes. This analysis reviewed only transiting vessels, not vessels maneuvering at berthing areas (see below for maneuvering vessels).

Ship-Induced Bed Scour from Maneuvering Vessels

Ship-induced bed scour from vessel maneuvers near berthing areas was primarily evaluated on a spatial basis by examining sun-illumination-manipulated bathymetry maps (presented here) and was also evaluated by modeling (presented in Appendix C Part 7). Multi-beam bathymetric soundings were recorded for the RI in 2003 by David Evans and Associates (DEA) (Windward and DEA 2004). The soundings were converted into a digital terrain model of the 3-dimensional mudline elevation in ft MLLW. Sun-illumination (or hillshade) maps were then generated from the processed bathymetry file. Highlighting or shading emphasizes fine-scale features that would otherwise be missed using standard digitizing methods. This process, often referred to as hillshading, is a hypothetical illumination of a surface according to a specified azimuth and altitude for the sun. This creates exaggerated vertical scales and allows for better visualization of vertical relief features in the sediment bed. Where features are identified visually, a geographic information system (GIS) can be used to estimate the vertical scale (e.g., depth of a scour feature) by displaying the values of adjacent bathymetric readings.

By applying hillshading techniques to the bathymetric data, various bed forms are evident in and near the berthing areas. These bed forms include V-shaped, symmetrical, and asymmetrical depressions oriented in various directions (Figure 2-10). The sun-illumination maps for the LDW were visually inspected to identify areas with steep gradients or ridges and furrows, interpreted as ship-induced scour. In some cases, the bottom features show depressions where barges have been resting in the mud during low tide and mounds where barges have been secured/moved by lowering steel rods or "spuds" into the mud.

The entire LDW was reviewed for scour, but mapping of this layer was generally restricted to areas where active berthing (vessels and overwater structures as documented in 2002 Port Series No. 36 publication [USACE 2002]) was observed. Active berthing was described as higher-traffic areas based on the presence of a pier/wharf face (discussed in Section 2.6.3), documented maintenance dredging events, aerial photographs showing moored barges or other vessels, adequate water depths, and/or operator interviews indicating that the area supports frequent vessel traffic. Vessels maneuvering into these areas may be causing scour. Vessel traffic patterns are discussed in Section 2.6.6. All of these lines of evidence were collectively used to define and map the vessel scour footprint.

Additionally, in the navigation channel, smaller features oriented with the axis of the channel are evident. It is important to note that although these bed forms are evident in many areas of the LDW and their depths vary from a few cm to over 30 cm in some areas, the majority of scour marks appear to have depths of less than 10 cm (i.e., within the depth of the active mixing zone). These smaller features in the navigation channel may represent effects of tug maneuvering to position vessels into berthing areas. This analysis provides information on net scour, but not on absolute scour occurring during individual events. Areas that are scoured as vessels maneuver may immediately fill in as the sides of the trench are sloughed. Therefore, an observed net depth of 10 cm may not capture deeper immediate scour depths. Areas with more than 10 cm of relief (forming ridges and furrows in the sediment surface) are primarily associated with berthing areas, where tugs maneuver barges, bulk carriers, and container ships. As a point of comparison, the STM (QEA 2008) predicts a maximum 100-year high-flow net erosion depth of 22 cm.

These anthropogenic bedform features are dynamic; old features are filled in by sedimentation and/or reworked by the creation of new features. This analysis represents a "snapshot" in time (2003) that is coincident with collection of the bathymetric data and provides only a general pattern of vessel scour. Detailed evaluations of vessel scour are more appropriate on a location-specific basis. This analysis is considered to be representative of ambient conditions.

2.3.1.2 Net Sedimentation Rates

Net sedimentation rates were determined in the STM (QEA 2008) and validated using empirical evidence from the RI and historical cores. The STM quantified sedimentation rates on a grid-cell basis using bed sediment properties (e.g., grain size and scour potential) and incoming total suspended solids (TSS) and bed loads (Figure 2-11[8]).

Results of the predictive model and empirical geochronology analysis are summarized as follows (QEA 2008):

- Net sedimentation rates in the intertidal and subtidal bench areas were estimated to range from 0.2 cm/year to greater than 2.0 cm/year, with those in the intertidal areas being on the order of 0.5 cm/year. The cores having lower estimated net sedimentation rates were generally collected from areas with shallower water depths (i.e., intertidal elevations above -4 ft MLLW) than the other geochronology cores, suggesting that these areas may be subject to relatively low deposition.

- Net sedimentation rates in the navigation channel exceeded 2 cm/year, reaching up to >50 cm/year in the Upper Turning Basin, where the maximum estimated net sedimentation rate was 150 cm/year. The Upper Turning Basin behaves as a trap for sediment entering the LDW from upstream and is dredged on an approximate biennial schedule to remove accumulated sediment. If the Upper Turning Basin were not dredged

[8] Figures 5-4 and F-2 compare net sedimentation rates estimated from cores with those predicted by the STM.

periodically, net sedimentation rates would likely be lower because some of the sediment would move farther downstream before depositing. This would likely increase net sedimentation rates in areas downstream of the Upper Turning Basin.

- Evidence of potential disturbances (e.g., episodic erosion and deposition, dredging, slumping) was observed in some of the geochronology cores.

Empirical evidence of net sedimentation rates, as reported in Appendix F of the STAR (Windward and QEA 2008), including chemical and physical time markers identified in sediment cores collected in the LDW, was used to validate the net sedimentation rates in the STM (QEA 2008). In most of the cores, there is generally strong agreement between the empirical lines of evidence and the STM estimates. However, in some locations, the STM estimates greater sedimentation than the empirical evidence does, and in other locations, the reverse occurs. This is discussed in more detail in Appendix F of this FS. Areas with lower net sedimentation rates (less than 2 cm/year) are scattered throughout the LDW, as dictated by channel geography, intertidal areas, and near-field scour events. Some uncertainty may exist in the observed vertical profiles of cores, but generally the empirical evidence supports the findings from the STM (QEA 2008).

2.3.2 Chemical CSM (Nature and Extent of Contamination in Sediment)

An understanding of the distribution of COC concentrations in the LDW follows the development of the physical CSM (Section 2.3.1).

2.3.2.1 COC Concentrations

The baseline HHRA (Windward 2007b) identified four human health risk drivers: PCBs, arsenic, cPAHs, and dioxins/furans. These risk drivers are evaluated in this FS at three spatial scales appropriate to human exposure: site-wide (netfishing), in potential clamming areas, and in beach play areas. Further, 41 of the 47 contaminants (including total PCBs and arsenic), for which SMS criteria are available, are risk drivers for benthic invertebrates because detected concentrations of these contaminants in surface sediments exceeded SMS criteria at one or more sediment stations (these data are hereinafter referred to as SMS chemistry data). SMS contaminants are evaluated on a point basis, as relevant to benthic invertebrate exposure. Total PCBs are also a risk driver for river otters and are evaluated on a site-wide basis for this receptor. Section 3 provides a summary of the ERA, HHRA, including the COCs, risk drivers, and appropriate exposure scales.

Tables 2-3 and 2-6 summarize minimum and maximum detections, average concentrations, and detection frequencies of human health risk drivers and other COCs, respectively, in the LDW FS dataset. In both the RI and FS baseline datasets, total PCBs were detected at 94% of the locations where PCB Aroclors were analyzed. In the RI baseline dataset, detected total PCB concentrations ranged from 1.6 to

223,000[9] micrograms per kilogram dry weight (µg/kg dw). In the FS baseline dataset, concentrations ranged from 2.2 to 2,900,000 µg/kg dw. Two samples with total PCB concentrations of 2,900,000 and 230,000 µg/kg dw were excluded from the spatial interpolation as outliers. Arsenic was detected at 93% and 94% of the locations where arsenic was analyzed in the RI and FS baseline datasets, respectively. In both datasets, the range of detected arsenic concentrations was 1.2 to 1,100 milligrams per kilogram dry weight (mg/kg dw), and the mean was 17 mg/kg dw. cPAHs were detected at 94% and 96% of the locations where cPAHs were analyzed in the RI and FS baseline datasets, respectively. In both datasets, the maximum cPAH concentration was 11,000 micrograms toxic equivalent per kilogram dry weight (µg TEQ/kg dw). The minimum detected cPAH concentration was the same in both the FS and RI datasets (9.7 µg TEQ/kg dw) and the mean concentration was lower in the FS baseline dataset than in the RI baseline dataset (460 µg TEQ/kg dw versus 500 µg TEQ/kg dw). Contaminants with SMS exceedances (Table 2-6) are represented only as point concentrations in the FS, while total PCBs, cPAHs, dioxins/furans, and arsenic are represented both as point concentrations and as spatially-weighted average concentrations (SWACs).

The FS baseline SWAC for total PCBs is 346 µg/kg dw[10] compared to the RI baseline SWAC of 350 µg/kg dw.[11] The FS baseline SWAC for cPAHs is 388 µg toxic equivalent (TEQ)/kg dw, compared to the RI baseline SWAC of 380 µg TEQ/kg dw. The FS baseline SWAC for arsenic is 15.6 mg/kg dw based on inverse distance weighting (IDW) interpolation, discussed below. The RI baseline SWAC for arsenic was 15 mg/kg dw, see Section 4 of the RI for all risk drivers; Windward 2010.

Dioxins/furans were detected in all surface sediment samples in which they were analyzed. The LDW-wide baseline SWAC (based on Thiessen polygons) is 25.6 nanograms (ng) TEQ/kg dw. Dioxins/furans were not spatially interpolated in the RI. The average of the 54 dioxin/furan surface sediment samples in the RI baseline dataset was 82 ng TEQ/kg dw (Windward 2010). A total of 119 surface sediment samples with dioxin/furan data are in the FS baseline dataset. Following finalization of the RI baseline dataset in 2006, additional dioxin/furan surface sediment samples were

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[9] This value was rounded to 220,000 µg/kg dw for presentation in the RI.

[10] Two outliers in the Trotsky inlet (RM 2.2) were not used in the interpolation to generate this LDW-wide SWAC. When all FS baseline data are considered, the SWAC is 1,313 µg/kg dw. These two outlier samples were not in the RI baseline dataset because those data were not available until after that dataset was finalized.

[11] The FS and RI SWACs are not calculated over the same area. For the FS, baseline SWACs were calculated over the area extending from RM 0.0 to RM 5.0. For the RI, baseline SWACs were generally calculated over the area from RM 0.0 to RM 6.0.

collected in 2009 and 2010, which are described in Table 2-2 and in the memorandum *2009/2010 Surface Sediment Sampling Results for Dioxins and Furans* (Windward 2010a).

For the SMS chemistry data, a total of 633 locations (44% of the 1,438 FS baseline surface sediment locations from RM 0.0 to 5.0) had detected concentrations of at least one SMS contaminant that exceeded the sediment quality standard (SQS) of the SMS. For some of these locations, the exceedances are only for total PCBs, being the only contaminant analyzed in those samples. Approximately half (316) of the locations with exceedances of SMS criteria are in EAAs. Outside of the EAAs, 317 sampling locations had surface sediment chemistry data that exceeded the SQS, based on chemistry alone.[12]

Sediment toxicity tests were conducted on surface sediment samples collected by LDWG from 48 locations for the RI. Thirty additional surface sediment samples were collected during the Ecology SPI event and subjected to toxicity testing. Two of the RI toxicity samples were co-located with newer toxicity data in the FS baseline dataset. Therefore, these older toxicity data were removed from the FS baseline dataset, yielding a total of 76 toxicity samples,[13] 44 of which passed for all biological endpoints tested. Of these 44 locations passing the toxicity tests, 41 represented either SQS or cleanup screening level (CSL) exceedances based only on chemistry. When evaluating surface sediment data relative to SMS exceedances, toxicity testing results override chemistry results. However, the chemistry data are retained for other FS purposes, such as mapping of human health risk drivers and source control evaluations. These 41 locations with toxicity passes, but chemistry exceedances, were identified as being below the SQS for mapping purposes.

Figures 2-12a through 2-12e display the exceedances of the SQS or CSL for any SMS contaminant in each sample of each core. Tables in Appendix G (Tables G-1 to G-3) list the SMS contaminants, and the concentrations responsible for those exceedances. It

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[12] One SMS contaminant, 2,4-dimethylphenol, was not identified as a benthic risk driver in the RI (Windward 2010) and ERA (Windward 2007a) because it did not exceed the SQS in the RI baseline dataset. However, this contaminant exceeded the SQS and CSL (which are both 29 µg/kg dw) in the Ecology SPI event. This contaminant was detected above the SQS and CSL in 25 of 30 SPI event samples. However, 20 of these samples have toxicity data passing the SQS biological effects criteria, so they are not considered SQS exceedances, following the data rules.

[13] One 2005 Round 2 RI location where toxicity data are available is co-located with a 2003 Duwamish/Diagonal EAA perimeter monitoring location. The chemistry data for this Round 2 location are not in the FS baseline dataset (because in the RI baseline this location was described as being influenced by the EAA removal activities and thus did not represent baseline conditions). However, to expand the toxicity dataset, the toxicity test results for this location (LDW-SS22) were used in the FS baseline dataset. This is more protective, because the 2003 chemistry results are below the SQS, but the 2005 toxicity test result is a CSL exceedance; therefore, this location is coded as exceeding the CSL.

should be noted that there are no toxicity test overrides for subsurface sediment data. The following observations were made regarding these subsurface sediment data:

- Forty-eight percent (728 of 1,504) of the subsurface sediment samples analyzed for PCBs had detected total PCB concentrations above the SQS.
- Five percent (28 of 531) of the subsurface sediment samples analyzed for arsenic had detected concentrations above the SQS.
- Twenty-five percent (81 of 535) of the subsurface sediment samples analyzed for bis(2-ethylhexyl) phthalate (BEHP) had detected concentrations above the SQS. Although BEHP is not a human health risk driver, it is being mapped because, other than total PCBs (515), it has the most SQS exceedances (104) in the surface sediment dataset (Table 2-6).
- Forty-nine percent (785 of 1,585) of the subsurface sediment samples had detected concentrations above the SQS for at least one of the SMS contaminants.

In general, the average concentrations of total PCBs and arsenic are higher in subsurface sediments than in surface sediments, while the reverse is true for cPAHs and dioxins/furans (Table 2-3). However, it is noted that concentrations in surface sediment are more appropriately compared to concentrations in subsurface sediment on a core-by-core basis. Core-by-core comparisons are provided in Appendix F as part of the discussion of empirical evidence for natural recovery.

2.3.2.2 Interpolative Mapping of Risk-Driver Contaminants

Spatially interpolated data are used in this FS for several evaluations, including the estimation of contaminated sediment volumes, natural recovery modeling, and delineation of the AOPCs (as discussed in Section 6). This section provides additional detail on the methods of spatially interpolating surface sediment data for the risk drivers, using the FS baseline dataset. Spatial interpolation of data generates a value for every location within the study area, rather than only at the discrete locations sampled. This interpolation is especially important for chemistry data that are applied to site-wide exposure scenarios and used as model inputs. Uncertainty related to spatial interpolation is also discussed in Section 6.

Human Health Risk-Driver Contaminants

The FS baseline dataset includes the following surface sediment sample counts between RMs 0 and 5.0: total PCB data for 1,392 stations, arsenic data for 916 stations, and cPAH data for 891 stations. For these three human health risk drivers, the data were spatially interpolated to generate a network of continuous 10-ft^2 grid cells. The IDW method used for the interpolations applies adjustable parameters to create the grid-based output for the whole LDW area. The parameters chosen and the methods used to optimize these parameters are discussed in Appendix A. The resulting IDW interpolations for total PCBs, arsenic, and cPAHs are displayed in Figures 2-13 through 2-15.

There are 119 discrete surface sediment grab samples for dioxins/furans included in the FS baseline dataset for interpolation.[14] Thiessen polygons were selected as the method for spatially representing these surface sediment data across the study area because the dataset is relatively small compared to that for the other risk drivers. The use of Thiessen polygons is a method by which a polygon is drawn around every data point. The boundaries of each polygon are drawn at the mid-points between the data point of interest and each surrounding data point. All surface sediment within each polygon is then assigned the concentration of the empirical data point contained within it; thus, a spatial extent is assigned to sample data at a given location. This method has inherent uncertainty because, unlike IDW interpolation, a concentration gradient is not estimated between data points. However, IDW interpolation is not appropriate for dioxins/furans because of the sparse dataset, as discussed in Appendix A. The dioxin/furan data for surface samples in the FS baseline dataset are shown in Figure 2-16; the dioxin/furan data for subsurface samples, as well as the Thiessen polygons mapped for the surface sediment data, are shown on Figure 2-17.

Interpolated data for total PCBs, arsenic, cPAHs, and dioxins/furans are used in the BCM (discussed in Section 5) to predict surface sediment quality over time.

SMS Chemistry

Thiessen polygons were also selected to spatially represent exceedance status relative to SMS criteria for chemistry and toxicity data at each location. There are 1,438 surface sediment samples with SMS contaminant data. However, some of these samples were analyzed only for PCBs. Of these samples, 891 were analyzed for all SMS contaminants (or the majority of the SMS contaminants), and thus this smaller dataset was used to delineate the spatial extent of SMS exceedances.

A polygon with more than one data point contained within it (e.g., one station with SMS chemistry data and a second station with only PCB data) was assigned the highest exceedance status of the two stations (pass, SQS, or CSL). The maximum exceedance status for individual SMS contaminants at each station was used to assign a status to that station's Thiessen polygon. For example, the polygon around a station with a CSL exceedance for fluoranthene, SQS exceedances for four other PAHs, and no exceedances for any other SMS contaminants, was designated as exceeding the CSL.

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[14] The composite sediment samples collected from beach areas were not included in the spatial interpolation of the baseline; but were included in Section 3 risk estimates and Section 8 technology assignments.

For mapping the AOPCs (Section 6) and remedial alternative footprints (Section 8), data are mapped as points with the spatial extent assigned by Thiessen polygons. The IDW method is not used because it is too labor intensive to interpolate the surface sediment concentrations of all SMS contaminants, which involves multiple steps of adjusting interpolation parameters and calculating error metrics for each set of parameters.

Where toxicity and chemistry data are both available within a polygon, toxicity results override chemistry results. For example, a polygon with a toxicity pass, but a chemical SQS exceedance, was assigned a pass. The toxicity data were used to assign the SMS status to the entire polygon, even if two stations are located within the polygon.[15] This override is relevant only to assigning exceedance status to Thiessen polygons relative to the SMS; it does not exclude chemistry data from other evaluations, such as the IDW interpolation of total PCBs described above.

Figures 2-13 through 2-16 show the distributions of total PCBs, arsenic, cPAHs, and dioxins/furans in surface sediment, respectively. The distribution of BEHP surface sediment sample locations and concentrations is shown in Figure 2-18.[16] The distributions of SMS chemistry and toxicity data in the surface sediment are shown in Figure 2-19. Figures 2-20a through 2-20g display the SMS contaminant concentrations in both dry weight and organic-carbon normalized units, where appropriate, that exceeded the SQS. Figure 2-21 presents the interpolation of the SMS exceedance status (by Thiessen polygon) in surface sediment.

2.3.2.3 Contaminant Distribution Patterns

Based on the surface sediment data, the LDW can be characterized as having localized areas of relatively high contaminant concentrations ("hot spots") separated by relatively large areas with lower contaminant concentrations. The distribution of concentrations in these hot-spot areas were different among the risk drivers, as described below. The top one hundred samples with the highest total PCB concentrations (ranging from 2,970 to 2,900,000 µg/kg dw) were all collected from within and near the EAAs and other hot spots (Trotsky Inlet at RM 2.2W, RM 3.8E, and RM 1.0 in the navigation channel). The average total PCB concentration of the remaining samples outside of these areas is 307 µg/kg dw (1,292 samples excluding the top 100 concentrations and the samples above RM 5.0) compared to 1,136 µg/kg dw for 1,390 samples (excluding the two outlier samples). The average PCB concentration in the FS baseline dataset is 3,383 µg/kg dw with all 1,392 samples included.

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[15] Extrapolation of toxicity test results across stations for the purpose of defining AOPCs in the FS should not be construed to imply that this practice will be acceptable in defining cleanup areas in the remedial design phase.

[16] BEHP data are included in the evaluation of SQS exceedances (benthic invertebrate risk driver).

The highest arsenic concentrations are localized mostly within discrete areas at RM 0.1, RM 1.0 (Slip 1), RM 1.3 – 1.45 (in the vicinity of Glacier Northwest, Inc.), RM 2.2 (Slip 3) and RM 3.8E (Figure 2-14). Fourteen stations exceed the CSL for arsenic and are located in these areas. The average arsenic concentration, excluding these nine stations, is 12 mg/kg dw, compared to 16 mg/kg dw with all data (918 samples).

The samples with the highest cPAH concentrations are more widespread (Figure 2-15). There are 48 samples at or above 1,500 µg TEQ/kg dw. The average cPAH concentration, excluding these 48 stations, is 333 µg TEQ/kg dw, compared to 459 µg TEQ/kg dw with all data (891 samples).

The five highest dioxin/furan sample concentrations are located within an EAA and two hot-spot areas: one concentration of 180 ng TEQ/kg dw (Duwamish/Diagonal EAA); three concentrations of 460, 570, and 2,100 ng TEQ/kg dw at RM 1.5W; and 410 ng TEQ/kg dw in Trotsky Inlet (RM 2.2W) (Figures 2-16 and 2-17). All other dioxin/furan concentrations are at or below 120 ng TEQ/kg dw. The average dioxin/furan concentration, excluding the five highest concentrations, is 11 ng TEQ/kg dw, compared to 42 ng TEQ/kg dw with all data from RM 0 to 5 (119 samples).

The highest surface sediment concentrations of the human health risk drivers often co-occur, typically within the EAAs and other hot spots, as noted by area below:

- **Duwamish/Diagonal EAA**: Preremedy sediments in the Duwamish/Diagonal EAA contained some of the highest concentrations of three of the four human health risk drivers: total PCBs, cPAHs, and dioxins/furans. The fifth highest total PCB concentration in the FS baseline dataset (56,200 µg/kg dw) and the fifth highest dioxin/furan concentration (180 ng TEQ/kg dw) were collected in this area. Five of the cPAH samples collected in this EAA exceeded 1,500 µg TEQ/kg dw.

- **Terminal 117 and Boeing Plant 2/Jorgensen Forge EAAs**: Of the ten samples with the highest total PCB concentrations, five (26,000 to 110,000 µg/kg dw) were collected from the sediments in the Terminal 117 and Boeing Plant 2/Jorgensen Forge EAAs; four of the samples with the highest cPAH concentrations (3,400 to 11,000 µg TEQ/kg dw) were also from these areas. A sample with an elevated dioxin/furan concentration (101 ng TEQ/kg dw) was also collected in the Boeing Plant 2/Jorgensen Forge EAA.

- **Slip 4 EAA**: Thirteen total PCB samples exceeded 1,300 µg/kg dw, and 5 cPAH samples exceeded 1,500 µg TEQ/kg dw.

- **Norfolk EAA Area**: A sample downstream of the Norfolk Area at RM 4.85 had the third highest total PCB concentration (223,000 µg/kg dw).

- **Trotsky Inlet (RM 2.2W)**: The two highest concentrations of total PCBs (2,900,000 and 230,000 µg/kg dw) were collected in 2007 at RM 2.2W (Trotsky Inlet; SAIC 2009). However, they were removed from the total PCB dataset as outliers for the purposes of IDW interpolation. These samples remain in the FS baseline dataset, but were excluded from the interpolation and any reported SWACs. The sample with the fourth highest dioxin/furan concentration (410 ng TEQ/kg dw) was also collected in the Trotsky Inlet.

- **RM 3.8E**: The highest arsenic concentration (1,100 mg/kg dw) was collected at RM 3.8E. This area also had elevated cPAH concentrations (>1,500 µg TEQ/kg dw).

- **Glacier Northwest, Inc. (RM 1.5W)**: Samples with the three highest dioxin/ furan concentrations (463, 565, and 2,100 ng TEQ/kg dw) were collected from sediments in the embayment adjacent to Glacier Northwest, Inc. (RM 1.5W). This embayment (and the downstream area to RM 1.3) also contained elevated arsenic concentrations (>93 mg/kg dw).

Some other areas in the LDW with high concentrations of co-located human health risk drivers include:

- The Ash Grove Cement Area (RM 0.1E) for arsenic, cPAHs, and total PCBs
- The head of Slip 1 for arsenic and cPAHs
- The navigation channel just upstream of RM 1.0 for total PCBs and dioxins/ furans.

Some areas listed in the bullets above exhibited high COC concentrations in both subsurface and surface sediment, coincident with low net sedimentation rates calculated in the STAR (Windward and QEA 2008) and supported by the STM (QEA 2008). In a few areas where higher net sedimentation rates were estimated, the presence of high COC concentrations near the surface could be the result of localized disturbances or recent, ongoing sources of contamination.

2.3.3 Sources and Pathways

After the physical and chemical settings are described, the third component of a CSM evaluates the source of the contaminants and the likely pathways by which these contaminants are transported into and within the LDW. Although the source control program and this FS address a much broader list of contaminants, this section focuses on the sources and pathways for the four human health risk drivers identified in the RI (Windward 2010).

2.3.3.1 Historical and Ongoing Sources of Contaminants

Today, many sources of historical origin, including direct discharges of municipal and industrial wastewater and spills, have been identified and controlled to some extent, by enhanced regulatory requirements, improved housekeeping practices, and technological advances. The reduction of some contaminants, such as PCBs, is due in part to banned production and use in the U.S.; however, significant contamination of historical origin is still present in the environment, and releases are ongoing. Such PCB legacies include older paints, caulks, and building materials still on or in existing structures, as well as soils and groundwater that were contaminated while PCBs were still actively used and produced in the U.S. Historical sources likely contributed much of the sediment contamination in the LDW, and historically impacted media/materials remain in the drainage basin and continue to be transported to the LDW.

Potential sources of PCBs, arsenic, PAHs, and dioxins/furans are summarized below:

- Although PCB production was banned in 1979, historical PCB use continues to affect the LDW today in a number of ways, including flaking paints, caulking, and building materials that contain PCBs and contaminated soils and groundwater. Historical sources of PCBs to the LDW include dielectric fluids, waste oils, hydraulic oils, paints, and sealants. PCBs were also historically released with cement kiln emissions, along with dioxins/furans. PCBs also come from industrial, commercial, and residential properties (e.g., hydraulic fluid in historical equipment). PCBs are present in the LDW drainage basin in sources such as contaminated soils and building materials such as paint and caulk (e.g., the former Rainier Brewery building, now known as Rainier Commons, which has paint on its exterior walls with total PCB concentrations greater than 10,000 mg/kg dw).

- Arsenic was historically (and is currently) used in lumber treatment and is released with other metals during watercraft repair. Arsenic was also released historically in air emissions from smelters, wood-treating facilities, and distillate oil combustion. Atmospheric releases of arsenic have been significantly minimized by the closure of smelters. Releases of arsenic and other metals to the LDW have been reduced by housekeeping practices and controls on wastewater discharge at facilities that practice activities such as ship maintenance.

- PAHs are generated from the burning of organic matter, fossil fuels, and charcoal (pyrogenic) and are present in refined petroleum products (petrogenic). Therefore, PAHs are continually generated and released to the LDW drainage basin and airshed through petroleum use and combustion. In addition, PAHs were historically released from brick manufacturing operations, hydraulic equipment manufacturing, machine shops, and from repair and fueling of vehicles, airplanes, trains, and watercraft. They can continue to be released by most of these sources; but best management practices (BMPs) controlling spills and leaks have reduced input from these sources. Finally, timber piles and dolphins (groups of closely driven piles used as a fender for a dock, a mooring, or a guide for boats) in the LDW and utility poles and railroad ties in the watershed were treated with creosote, which can deposit PAHs directly into the LDW as these structures degrade or onto impervious surfaces in the watershed.

- Dioxins/furans are not used in manufacturing operations but are unintentionally formed as byproducts of incineration when chlorine and organic material are present. They were historically (and are currently) released from the burning of waste and from paper mills, cement kilns, and drum recycling. Historically, dioxins/furans were byproducts of pentachlorophenol (used in wood treating) and pesticide production; neither activity is present in the LDW drainage basin today.

2.3.3.2 Pathways to the LDW

To identify and manage sources, it is important to understand sources (discussed above) and pathways to the LDW sediments. Contaminated media from within the LDW drainage basin can affect sediments in several ways, which can be organized into seven general types based on the affected media, the origin of contamination, and pathways to sediments:

- Direct discharge (e.g., CSOs, storm drains)
- Surface water runoff or sheet flow
- Spills and/or leaks to the ground, surface water, or directly into the LDW
- Groundwater migration/discharge
- Bank erosion/leaching
- Atmospheric deposition
- Transport of resuspended contaminated sediments.

These pathways, as they relate to the four human health risk drivers, are discussed in more detail below. Not all pathways are complete or significant at all locations or at all times. Ongoing sources include those associated with industrial and general urban use within the watershed. Examples of contaminants and their sources include PAHs (fossil fuels), phthalates (plastics), zinc (tire wear), and copper (brake pads). Ongoing sources also include legacy contamination from historical upland operations, which continue to impact the LDW via ongoing pathways, such as groundwater migration/discharge and bank erosion. Contaminants released to media such as air, soil, groundwater, and surface water or to impervious surfaces may migrate to the LDW through various pathways.

Historically, controls on wastewater discharges and use of BMPs were not common. PCB discharges in particular are expected to have been of a greater magnitude historically before commercial PCB production was banned in 1979. However, trends for other contaminants such as BEHP and PAHs suggest rising levels due to increased urbanization. Appendix F presents historical risk-driver trends in Puget Sound sediments, and Appendix J evaluates recontamination potential to the LDW by direct discharge pathways (CSOs and storm drains).

Direct Discharge

Discharge from public or private storm drain systems, CSOs, and emergency overflows (EOFs) is a pathway for contaminants to enter the LDW. The locations of CSOs and EOFs are displayed on Figure 2-22 (along with other outfalls and the source control areas discussed in Section 2.4). CSOs and EOFs can discharge wastewater (residential, commercial, and industrial) and stormwater runoff. CSO discharges generally occur only during large storm events when the capacity of the combined sewer system is exceeded and not all flow can be successfully conveyed to a treatment plant. EOF discharges are not storm-related; those overflows occur as a result of mechanical failure, pipe obstruction, or power failure. The LDW drainage basin is served by a combination of separated storm drains and sanitary sewers and combined sewer systems. The total combined (storm and sanitary) sewer service area is 19,800 acres. The separated storm drainage basin covers 8,936 acres. However, these areas overlap in many places; the total area discharging to the LDW is approximately 20,400 acres (Schmoyer, personal communication, 2011a). Approximately 208 direct discharge points occur along the LDW shoreline, of which 203 are public or private outfalls, and 5 are ditches, creeks, or streams. In addition, 7 major seeps and 22 abandoned outfalls have been identified during shoreline surveys (Schmoyer, personal communication, 2011b).

Stormwater pollution is generated when rain contacts pollutants that have accumulated in or on exposed soils and surfaces, or comes from illegal discharges or illicit connections to storm drains, which convey stormwater only. Storm drains convey stormwater runoff collected from streets, parking lots, roof drains, and other impervious surfaces to the LDW. A wide range of contaminants may become dissolved or suspended in stormwater as it flows over surfaces. Contaminated solids that collect in storm drains/pipes, ditches, or creeks may be carried to the LDW by stormwater. Activities in urban areas generate particulates, dust, oil, asphalt, rust, rubber, metals, pesticides, detergents, or other materials that can be flushed into storm drains during wet weather events. Storm drains also convey materials generated by business activities such as outdoor manufacturing, outdoor storage of equipment and waste materials, vehicle washing, runoff from landscaped areas, erosion of contaminated soil, groundwater infiltration, and illegal discharge of materials into the sewer. Some businesses have National Pollutant Discharge Elimination System (NPDES) industrial stormwater permits. In the LDW drainage basin, approximately 90 general and individual NPDES permits have been granted for industrial stormwater discharges to storm drains or the LDW. However, not all businesses in the stormwater drainage area are required to obtain such permits. The City of Seattle, City of Tukwila, the Port of Seattle, and King County are NPDES permittees for stormwater discharged via municipal outfalls.

Some areas of the LDW are served by combined sewer systems, which carry both stormwater and municipal/industrial wastewater in a single pipe. Under normal rainfall conditions, wastewater and stormwater are conveyed through this combined sewer pipe to a wastewater treatment facility. During large storm events, however, the total volume of wastewater and stormwater sometimes exceeds the conveyance and treatment capacity of the combined sewer system. When this occurs, the combined sewer system is designed to overflow through relief points, called CSOs. The CSOs prevent the combined sewer system from backing up and creating flooding problems. Untreated municipal and industrial wastewater and stormwater can be discharged through CSOs to the LDW during storm events. CSO discharges can carry contaminants that affect sediments. The City's CSO network has its own NPDES permit; the County's CSOs are administered under the NPDES permit established for the West Point Wastewater Treatment Plant.

Stormwater is discharged to the LDW from approximately 200 public and private storm drains, CSOs, ditches, and streams, and contaminants discharged from any of these may affect LDW sediments. Most of the waterfront properties within the LDW are served by privately-owned drainage systems that discharge stormwater directly to the LDW. Upland areas not adjacent to the waterway are served by a combination of privately- and publicly-owned drainage systems. However, the private storm drains in the upland areas typically connect to a publicly-owned system before discharging to the LDW. The City of Seattle and King County stormwater and CSO systems overlap throughout the LDW drainage basin in complex ways. The following paragraphs summarize characteristics of these various systems and outfalls in the LDW source area. Section 2.4 describes the source control strategy for addressing direct discharges from both public and private drainage systems in the LDW drainage basin.

The City of Seattle's storm drain system services approximately 61% of the LDW drainage basin (8,936 acres), which is a separated or partially separated storm drain system. Other public storm drains service about 24% of the drainage basin, and the remaining 15% of the drainage basin is serviced by small, private waterfront storm drain systems.

The City of Seattle owns and operates the local sanitary sewer collectors and trunk lines, while King County owns and operates the larger interceptor lines that transport flow from the local systems to the West Point Wastewater Treatment Plant.

The City of Seattle operates two CSOs in the LDW: #116 at South Brighton Street and #111 at Diagonal Way South. CSO #111 consists of eight separate overflow points discharging to a single outfall. The City also operates three EOFs in the LDW. The City of Seattle began monitoring the frequency and volume of discharges from its CSOs in 1999. CSO #116 has not overflowed since 1999. Over a 6-year period of record (1999 to 2005), the total annual discharge volume from CSO #111 has ranged from 0.6 to 74 million gallons. In 2005, Seattle Public Utilities modified the overflow structure on CSO #111's largest overflow point (#111 D) to allow more water to enter the King County treatment system and release less water to the LDW (Seattle Public Utilities 2008). In 2008, no overflow events were recorded from CSO #111; in 2009, five events, releasing a total of 2.1 million gallons, were recorded (Tetra Tech 2010b).

King County also operates nine CSOs and two EOFs that discharge to the LDW. For the period from 1999 to 2005, one of these CSOs had no recorded overflows. For the remaining eight CSOs discharging to the LDW, the average total monthly overflow volumes ranged from 0.12 million gallons (July) to 14 million gallons (November). King County has no record of an overflow event ever occurring at the pump station EOF located at the E. Marginal Way S outfall. The Duwamish East CSO/EOF also functions as an emergency bypass for a pump station; this CSO/EOF has not experienced an emergency overflow since 1989 (Nairn 2007). This location also contains an EOF for the siphon that traverses the LDW. This EOF had one overflow in 2005 and one in 2007 (King County 2010c).

Historically, direct sanitary sewer discharges were reduced as King County eliminated raw sewage outfalls and redirected wastewater to the West Point Wastewater Treatment Plant. Many industrial discharges were also rerouted from the LDW to the West Point Wastewater Treatment Plant. King County also developed industrial waste pretreatment and CSO reduction programs in accordance with the Clean Water Act. Since 1969, those programs have reduced contaminant discharges to the sewers and reduced CSO discharges of contaminants to the LDW.

Infrastructure improvements have greatly improved system storage capacity and reduced the number of discharges from the combined sewer systems (those that may include contributions of stormwater, sewage, and industrial waste streams). These combined systems are still in operation in some areas adjacent to the LDW, but their existence is very limited (Windward 2010). Continuing efforts to increase infiltration and treatment of stormwater and to educate businesses and residents are all designed to reduce pollutants entering the LDW. However, regional development and population growth may increase source loads of PAHs and other COCs (Ecology 2005).

Surface Water Runoff or Sheet Flow

Surface runoff is a potentially complete pathway for transport of COCs to the LDW. In areas adjacent to the LDW and lacking collection systems, contaminated soils or contaminants improperly stored either as raw or as waste materials could be carried directly over impervious surfaces (surface runoff) or through creeks and ditches to the LDW. For properties not adjacent to the shoreline, sheet flow generally enters a publicly-owned conveyance before discharging to the LDW.

Spills and/or Leaks to the Ground, Surface Water, or Directly into the LDW

Infrastructure and activities over or near the LDW have the potential to release COCs to adjacent sediments. Overwater activities occur on shoreline structures such as piers, wharves, and dolphins (discussed in Section 2.6.3). Historical industrial practices included dumping and sweeping waste from piers and through floor hatches in overwater buildings into the LDW. These practices have resulted in accumulation of contaminants in sediments near these structures. These practices are no longer common because BMPs are now required under environmental regulations. Contaminants in soils, surface water, or groundwater that resulted from spills or leaks at the properties adjacent to the LDW may reach the LDW.

Groundwater Migration/Discharge

Groundwater migration/discharge is a potentially complete pathway for transport of COCs to the LDW. Contaminated groundwater has been documented at several properties in the LDW drainage basin where groundwater flows toward the LDW. Seep and porewater sampling conducted in 2004 for the RI identified 82 seep locations throughout the LDW; 18 of these locations were selected for chemical analyses. The results of this study were discussed in the RI (Windward 2010). EPA and Ecology may further evaluate seeps as part of their continuing upland site cleanup and source control efforts.

Determining whether a contaminant identified in groundwater will reach sediment and surface water in the LDW is a complex process. The potential for groundwater transport to be a significant pathway at some locations will be assessed as part of facility-specific remedial investigations implemented under the 2004 source control strategy (Ecology 2004). For example, at the Boeing Isaacson/Thompson properties (RM 3.8), where high concentrations of arsenic were detected both in groundwater and in the sediments immediately offshore, the groundwater-to-sediment pathway will be investigated as part of the remedial investigation for that facility.

As part of the Phase 1 RI completed in 2003, a preliminary pathway assessment, based on the information available at the time for 12 upland facilities, was conducted to evaluate the potential for groundwater contamination to reach the LDW and contaminate sediment. Groundwater information through 2002 was summarized for the 12 upland facilities[17] identified by EPA and Ecology as preliminary facilities of interest for the RI. The Final RI,[18] completed in 2010, expanded the list to 45 facilities, adding shoreline properties associated with one of the 11 source control areas (SCAs) discussed in Appendix I of the RI[19] and those identified by Ecology as being facilities of interest for groundwater. The RI provided updated information on contaminants found in groundwater at 28 of the facilities for which groundwater data were available as of 2008. Groundwater data collected at these facilities were compared to contaminant concentrations in receiving sediments, but the potential for groundwater contaminants to affect LDW sediments was not assessed further in the RI. The following results were noted:

- At 7 of the 12 facilities evaluated in the 2003 preliminary assessment, evidence was found for metals accumulation in sediment to concentrations greater than SMS criteria or DMMP guidelines in potential groundwater discharge zones. The RI lists 20 facilities with detected metals in groundwater, and at 9 of these facilities one or more of these metals were also detected in nearby sediments at concentrations above the SQS.
- PCBs were not identified as a COC in groundwater in the 2003 preliminary assessment based on groundwater data available at the time and the known high retardation factors for PCB transport in groundwater. However, more recent data summarized in the RI have revealed detectable concentrations of PCBs in groundwater under eight facilities (Terminal 106, Duwamish Marine Center, Boeing Plant 2, PACCAR, Georgetown Steam Plant, North Boeing Field, Terminal 117, and Industrial Container Services). PCBs were detected in nearby sediments at all of these facilities.

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[17] The 12 facilities are Advance Electroplating (RM 4.1), Boeing Developmental Center (RM 4.8), Boeing Isaacson (RM 3.8), Boeing Plant 2 (RM 3.6), Great Western International (RM 2.4), Long Painting (RM 3.1), Terminal 117 (RM 3.7), PACCAR (former Kenworth Truck, RM 4.0), Philip Services/Burlington Environmental (RM 1.4), former Rhône-Poulenc (RM 4.2), South Park Landfill (RM 2.6), and Terminal 108 (RM 0.7). EPA is also evaluating groundwater from the Boeing Electronics Manufacturing Facility (EMF; upland site near RM 3.4). It was evaluated in the RI in the context of the Boeing Plant 2/Jorgensen Forge EAA SCA because groundwater from the EMF flows under these properties.

[18] In this FS, the Final RI (Windward 2010) is simply referred to as the RI.

[19] In 2002, only 11 source control areas were identified. By 2010, Ecology and the Source Control Work Group had identified 24 separate source control areas based on the extent of municipal storm and sanitary drain infrastructure.

- Elevated concentrations of PAHs were not detected in the groundwater or adjacent sediments at any of the 12 facilities in the 2003 assessment. Additional data included in the RI indicate detected concentrations of PAHs in groundwater at 9 facilities along the LDW, and at 6 of these facilities, PAHs were found above the SQS in nearby sediments.
- Volatile organic compounds (VOCs) were detected in groundwater at 18 facilities, including Great Western International (RM 2.3 to 2.4E), where chlorinated ethenes were detected in porewater and seeps. This facility is documented as having elevated VOCs in groundwater, but fate and transport analyses for VOCs indicated extensive degradation prior to discharge to the LDW (Windward 2010). Boeing Plant 2/Jorgensen Forge (and Boeing Electronics Manufacturing Facility [EMF]) also had elevated concentrations of VOCs in groundwater, and VOCs were also detected in seeps and sediments.

All of these assessments are preliminary. The source control program will prepare more detailed, facility-specific assessments of the potential for groundwater contaminants to contribute to sediment contamination.

Bank Erosion/Leaching

Unprotected shoreline banks are susceptible to erosion by wind, surface water, and surface runoff, creating a pathway for contaminated soils to reach LDW surface sediments. Shoreline armoring and vegetation may reduce the potential for bank erosion. Currently, the majority of the LDW shoreline is armored with constructed steel and concrete bulkheads, sheet-pile walls, and riprap banks, limiting bank erosion in many areas. Bank erosion is more likely to occur in unarmored areas such as the banks of Kellogg Island, the shoreline east of the island, and areas to the south near the Upper Turning Basin.

Much of the material behind the riprap, seawalls, and other armoring is fill, placed during industrial/commercial development of the LDW. Historically, the source and quality of fill materials was not tracked, which leads to potential source control issues in these areas based on the lack of knowledge about their nature (i.e., historical contamination). Unknown contaminant concentrations in historical fill materials may be related to potential pathways such as erosion, groundwater/tidal communication to the LDW, and infiltration to storm drains or other discharge infrastructure.

Shoreline structures and conditions are discussed in more detail in Section 2.6.4. However, because of the limited amount of data available for the banks, this pathway was not evaluated in the FS from a contamination perspective. It is discussed only in reference to the physical conditions of the banks (i.e., whether they may be erodible, but not whether the bank soils are contaminated). Both the physical conditions and potential contamination of the banks will be important on a case-by-case basis at the remedial design level and will be addressed as part of location-specific cleanups and through ongoing source control efforts.

Atmospheric Deposition

Atmospheric deposition allows air pollutants to enter the LDW directly, and to reach the LDW via stormwater from the watershed. Air pollutants may be transported over long distances by wind, and can be deposited on land and water surfaces by precipitation or particle deposition. Global atmospheric transport of PCBs from parts of the world where they are still used represents an ongoing pathway. Additional information on recent and ongoing atmospheric deposition studies in the LDW area is summarized in the LDW Source Control Status Reports (Ecology 2007c, 2008a, 2008b, 2009, and subsequent updates). Ecology will continue to monitor these efforts.

Air pollutants may be generated from direct or indirect sources. Direct sources include industrial smokestacks and activities such as painting, sandblasting, loading/unloading of raw materials, and other activities. Indirect sources include dispersed sources such as vehicle emissions, aircraft exhaust, resuspension of particulates, and off-gassing and degradation of common materials such as plastics and building materials.

Section 9 of the RI (Windward 2010) reported (based on Puget Sound Clean Air Agency records) that over 200 businesses in the Duwamish Valley (the airshed of the LDW[20]) are registered as active sources of air pollution. Motor vehicle traffic on Interstate 5, State Routes 99 and 509, and local roads also produces nitrous oxide, black carbon (i.e., soot), and other emissions through the burning of fossil fuels.

Atmospheric releases of PCBs have been significantly minimized by the United States ban on production of PCBs in 1979. However, PCBs contained within old paints, caulks, and other building materials remain in the watershed, and thus represent ongoing sources, with releases from these media via off-gassing (to the atmospheric deposition pathway) and physical degradation (transported via stormwater discharge and runoff pathways).

Transport of Resuspended Contaminated Sediments

Sediments in one part of the LDW that are scoured and transported can contaminate sediments in other parts of the LDW, including remediated areas. The STM (QEA 2008) delineates areas where sedimentation is predicted to bury historically impacted

[20] The Duwamish Valley (bounded to the west by West Seattle and to the east by Beacon Hill) is smaller than the LDW drainage basin. The combined sewer and storm drainage systems, discharging to the LDW, extend beyond the Duwamish Valley. For example, the Duwamish/Diagonal CSO/SD basin extends north and east into the International/Central District and Beacon Hill neighborhoods of Seattle.

sediment. However, in scour areas or areas disturbed by mechanical actions, contaminated subsurface sediments may become exposed, and either surface or previously subsurface sediments may be transported. Section 2.3.1.1 discussed both high-flow and ship-induced scour.

Additionally, migration from upstream sources to the LDW continues via inflow of suspended sediments and surface water that contain contaminants.

2.4 Source Control Strategy

The LDW source control strategy (Ecology 2004) describes the process for identifying source control issues and implementing effective source controls for the LDW. The strategy is used to identify and manage sources of potential contamination and recontamination in coordination with sediment cleanups. The goal is to limit sediment recontamination that exceeds LDW sediment cleanup goals. Existing administrative and legal authorities will be used to perform inspections and required source control actions.

The LDW source control strategy (Ecology 2004) focuses on controlling contamination that affects LDW sediments. It is based on the principles of source control for sediment sites described in *Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites* (EPA 2002b) and Washington's SMS. The first principle is to control sources early, starting with identifying all ongoing sources of contaminants to the site. It is anticipated that the Record of Decision (ROD) will require that sources of sediment contamination to the LDW be evaluated, investigated, and controlled as necessary. Dividing source control work into specific Source Control Action Plans (SCAPs) and prioritizing actions within those plans to coordinate with sediment cleanups will address the guidance and regulations and will be consistent with the remedial alternatives in this FS.

Ecology is the lead agency for implementing source controls in the LDW and works in cooperation with local jurisdictions and EPA to create and implement source control strategy and action plans and to prioritize upland cleanup efforts in the LDW. In 2002, these entities formed the LDW Source Control Work Group (SCWG), which conducts several different source control activities within the LDW area. Primary members of the group include EPA, Seattle Public Utilities, King County, and the Port of Seattle. The LDW source control strategy (Ecology 2004) also identifies various regulatory programs at EPA and Ecology that are called upon as needed for source control as well as several ad hoc members of the SCWG, including the City of Tukwila, Puget Sound Clean Air Agency, and Washington State Departments of Transportation and Health. All LDW SCWG members are public entities with various source control responsibilities and the collective purpose is to share information, identify issues, develop action plans for source control tasks, coordinate implementation of various source control measures, and share progress reports on these activities.

The LDW source control strategy describes how recontamination of LDW sediments will be controlled to the maximum extent practicable. The goal is to limit sediment recontamination that exceeds site-specific standards, where feasible. The LDW source control efforts are designed to identify and manage sources of contaminants to waterway sediments in coordination with sediment cleanups. This strategy provides the framework and process for identifying source control issues and implementing practical control of contaminant sources. The strategy also serves three other primary functions. First, it sets up the reporting process for tracking and documenting all of the source control work performed throughout the LDW source area. This information is necessary for EPA's administrative records and remedial decisions. Second, the strategy broadly prioritizes source control work according to the schedules proposed for sediment cleanups (e.g., EAAs, other areas to be identified in the ROD). These priorities or "tiers" for source control efforts are listed below. Finally, the strategy identifies the basic steps for performing source control: 1) identify, 2) characterize, and 3) control sources and pathways of contamination to the LDW.

The success of the strategy depends on the coordination and cooperation of all public agencies with responsibility for source control in the LDW area, as well as prompt compliance by businesses that must make the necessary changes to control releases from their properties. The strategy is being implemented through the development of a series of detailed SCAPs that will be coordinated with sediment cleanups, beginning with the EAAs. The SCAP for each source control area describes potential sources of sediment contaminants and the actions needed to control them. Each SCAP evaluates whether ongoing sources are present that could recontaminate sediments after cleanup. In addition, the SCAPs describe source control actions that are planned or currently underway, including sampling and monitoring activities to identify additional sources. The tiers are defined as follows:

- **Tier One –** Source control work associated with EAAs[21]
- **Tier Two –** Source control work associated with sediment cleanup areas identified for final or long-term cleanup through the RI process or in the LDW decision document
- **Tier Three –** Source control work associated with drainage basins discharging to LDW sediments that have not been identified for Tier One or Tier Two source control activities through the RI/FS process

[21] The Tier 1 areas published in the Phase 1 RI (Windward 2003a) included two areas that were not carried forward as EAAs because remedial actions for sediments are not scheduled to begin before the issuance of the LDW ROD. Five of the Tier 1 areas are currently EAAs. The other two areas are included in the Tier 2 areas.

- **Tier Four –** Source control work associated with sediment areas that are remediated and become subsequently recontaminated above SMS criteria or LDW cleanup goals based on post-cleanup monitoring.

Since 2002, the SCWG has identified 24 SCAs, which are generally based on stormwater and CSO infrastructure and drainage to the LDW study area (Figure 2-22). These 24 SCAs are based on drainage to ensure that source control will be conducted for the whole LDW, not just the areas identified for sediment cleanup. Ecology develops SCAPs for each SCA that describe potential sources of contamination that may affect sediments. They also describe source control actions that are planned or underway, and sampling and monitoring that must be done. The source control actions are subdivided into high, medium, and low priority tasks. Ecology and the other agencies identify those responsible for contamination and work with them and relevant SCWG partners to control contamination.

Ecology continues to develop SCAPs for the LDW. The first step in developing a SCAP is to summarize existing information and find out what is missing (data gaps). As of July 2011, Ecology had published SCAPs for 18 of the 24 SCAs. Ecology is currently working with its consultants to develop data gap reports and SCAPs for the remaining SCAs. Many source control documents are available on Ecology's LDW Source Control webpages, which launch from the Toxics Control Program tab on Ecology's home page at http://www.ecy.wa.gov. King County, the City of Seattle, and the Port of Seattle also have web content about their respective roles and work in LDW source control.

The status of the source control efforts within the LDW drainage basin as of September 2010 is described below (Ecology 2011b). Facilities named below are displayed on Figure 2-22:

- One hundred ninety-six confirmed or suspected contaminated upland facilities within the LDW drainage basin have been identified.
- Thirteen facilities along or near the LDW are under agreed orders in Ecology's cleanup process (MTCA). The facilities are:
    - Jorgensen Forge (uplands)
    - North Boeing Field/Georgetown Steam Plant
    - 8801 East Marginal Way (former Kenworth Truck)
    - South Park Landfill
    - Fox Avenue Cleanup
    - Glacier Northwest, Inc./Reichhold
    - Crowley Marine Services

- Duwamish Shipyard
- Industrial Containers/Trotsky/NW Cooperage
- Douglas Management Properties
- Boeing Isaacson-Thompson
- Port of Seattle Terminal 115 North
- Duwamish Marine Center.

♦ Ecology conducted site investigations at:

- South Park Marina (formerly A&B Barrel)
- Basin Oil
- Industrial Container Services (formerly Northwest Cooperage)
- Douglas Management Company/Alaska Marine Lines
- Washington Liquor Control Board Warehouse.

♦ Four voluntary cleanups under MTCA are occurring at the following facilities along or near the LDW:

- Port of Seattle Terminals 106/108
- Boeing Developmental Center uplands and sediments (Section 2.7.2)
- General Services Administration – Federal Center South
- City of Seattle 7th Avenue Pump Station.

♦ Five additional facilities in the LDW SCAs are under agreed orders administered by Ecology's Hazardous Waste Treatment and Reduction (HWTR) program:

- Art Brass Plating
- Blaser Die Casting
- Capital Industries
- General Electric-Dawson Street Plant
- Philip Services Georgetown.

♦ Nine facilities along or near the LDW are under an EPA cleanup process. These facilities are:

- Boeing Plant 2 (Resource Conservation and Recovery Act [RCRA] corrective action)

- Jorgensen Forge shoreline (Comprehensive Environmental Response, Compensation, and Liability Act [CERCLA] removal action)
- Stormwater outfall along Boeing Plant 2/Jorgensen property line (CERCLA removal action)
- Rhône-Poulenc/Monsanto (RCRA corrective action)
- Port of Seattle Terminal 117 (CERCLA removal action)
- Slip 4 (CERCLA removal action)
- Boeing EMF (CERCLA removal action)
- North Boeing Field/King County International Airport Storm Drain Treatment System (CERCLA removal action)
- Tully's/Rainier Commons (Toxic Substance Control Act).

- From 2003 to 2005, the City of Seattle and King County conducted a joint business inspection program in the Diagonal Ave S CSO/SD area to evaluate stormwater, industrial wastewater, spill containment, and hazardous waste management practices at each property and to bring businesses in compliance with local code requirements. During that time, 1,100 inspections were completed at approximately 625 businesses. The City took over the business inspection program in 2006, and King County continued to inspect the businesses in the LDW that are permitted under its Industrial Waste Program. King County also provides technical assistance to Seattle Public Utilities (SPU) as needed on issues related to industrial waste and hazardous waste. In 2010, the City completed the first round of inspections at the approximately 1,275 high-risk pollutant generating sites in the LDW drainage basin. Between 2003 and September 2011, approximately 2,900 inspections were completed at businesses throughout the LDW drainage basin. The LDWG partners have also collected sediment samples from storm drains and combined sewer systems to help identify and characterize sources discharging to the storm and combined sewer[22] collection systems in the LDW. As of June 2011, over 1,000 samples had been collected, mostly by SPU.

- Approximately 500 combined hazardous waste and water quality inspections have been completed under the Ecology LDW Urban Waters Initiative (March 2007 through July 2010). From October 2009 through September 2010, water quality inspections numbered 66. Of these, 33

[22] King County has also collected CSO water samples in the Duwamish River Basin.

notices of violation have been issued, 4 administrative orders have been issued, and 4 penalties have been assessed.

- Approximately 105 facilities in the LDW drainage basin have Ecology water quality discharge permits (NPDES); approximately 90 facilities are regulated under a general industrial stormwater permit; 2 active facilities have individual industrial stormwater permits; 2 facilities operate under general discharge permits for boatyards; and 4 facilities operate under general discharge permits for sand and gravel facilities.

- Four local governments have municipal stormwater general discharge permits (Phase I for the City of Seattle and King County, secondary permittee under Phase I for the Port of Seattle, and Phase II for the City of Tukwila).

- Two local governments (the City of Seattle and King County) have individual discharge permits for their CSO/SD systems.

- Several MTCA agreed orders have been issued by Ecology to evaluate upland properties in the LDW watershed (Figure 2-22).

Source control is an iterative process. Early steps are often revisited and conclusions refined by information gathered later. Source identification in one basin may influence source control investigations in another basin. Addressing each potential source may involve one or more of the following elements: source control investigations, upland site assessment and cleanup, inspections, source tracing, sampling, and monitoring.

In conjunction with source control activities led by Ecology, the City of Seattle is conducting a source-tracing study and has collected storm drain sediment samples (from catch basins and within storm drain systems) within areas of the LDW drainage basin.[23] The City of Seattle compiled data from storm drain sediment samples collected by Seattle Public Utilities, King County, and The Boeing Company for use in this FS as part of the modeling efforts described in Section 5 and in Appendix C, Part 3. PCBs were detected in 84% of 953 samples. Through this source tracing exercise, PCBs have also been found in various building materials (e.g., paint, caulk, and other sealants). Unlike other contaminants, PCBs exhibited a distinct geographic distribution, with hotspots identified at Terminal 117, Rainier Commons, North Boeing Field/ Georgetown Steam Plant, and Boeing Plant 2/Jorgensen Forge. The latter two have been sampled extensively and make up a significant portion of the overall source-tracing dataset. Other activities conducted by municipalities and property operators

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[23] Other parties, such as The Boeing Company and the Port of Seattle, have also been collecting source-tracing samples at their sites.

include inspections, NPDES-required stormwater discharge sampling, development of stormwater pollution prevention plans and source control strategy plans, use of BMPs, and other activities.

Arsenic was detected in 52% of 576 sediment samples collected from within storm drain systems that discharge to the LDW. Arsenic concentrations were fairly uniform and relatively low, with only 5 percent of the samples exceeding the SQS (57 mg/kg dw) and only 3 percent exceeding the CSL (93 mg/kg dw). Samples containing elevated arsenic concentrations were not clustered in any particular geographic area.

cPAHs were detected in 93% of 543 storm drain sediment samples. Concentrations did not display a distinct geographic distribution. cPAHs were present at concentrations exceeding 25,000 µg TEQ/kg dw (used as a screening level) at various locations throughout the drainage basin, typically in on-site drainage structures (catch basins and oil/water separators) at facilities engaged in transportation-related activities (e.g., bus and airport operations), maintenance facilities, service stations, foundries, and fast food facilities.

In 2004 and 2005, dioxins/furans were analyzed in nine storm drain sediment samples in catch basins and maintenance holes, one storm drain sediment sample upstream of an oil-water separator, and one street dirt sample. Concentrations ranged from 6.2 to 26 ng TEQ/kg dw in the storm drain sediment samples and 91 ng TEQ/kg dw in the street dirt sample (Integral 2008). The median value for all samples was 18 ng TEQ/kg dw. Appendix C and Section 5 present summary statistics for storm drain and CSO data collected within the LDW basin and used in the chemical modeling.

2.5 Key Observations and Findings from the RI

Key findings from the RI (Windward 2010) are summarized below.

- Over the past 100 years, the LDW has been highly modified from its natural configuration to support urban and industrial development. Changes have included reductions in and control of water flow, significant shoreline modifications, loss of intertidal habitat, and installation of riprap, pier aprons, and sheet pile walls. Some limited areas of natural shoreline still exist within the LDW.

- Industrial and commercial facilities occupy most of the shoreline; one residential community (South Park) is also located along the shoreline, and another community (Georgetown) is nearby.

- The LDW is currently used as an industrial navigational corridor. It also supports recreational uses such as boating, kayaking, fishing, and beach play. The LDW is also part of Tribal Usual and Accustomed fishing areas.

It is also one of the locations of the Muckleshoot Tribe's commercial, ceremonial, and subsistence fishery for salmon, and the Suquamish Tribe actively manages aquatic resources north of the Spokane Street Bridge, located just north of the LDW study area. The Duwamish Tribe uses Herring's House Park and other parks along the Duwamish for cultural gatherings.

- Despite significant alterations in habitat and areas with elevated COC concentrations, the LDW contains a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators.
- The majority of high arsenic and total PCB concentrations in surface sediment are located within fairly well-defined areas. The locations of the highest arsenic and total PCB concentrations are generally not in the same areas, indicating that sources likely differ for these two contaminants. Areas with the highest cPAH concentrations are located in many of the same areas identified for arsenic and total PCBs, but are also more dispersed. Several areas have high dioxin/furan concentrations in surface sediments.
- Sediment is continually depositing within the LDW, with almost all new sediment (99%) originating from the Green/Duwamish River system. The STM (QEA 2008) estimates that over 200,000 metric tons of sediment per year enter the LDW. Approximately 50% of this total deposits in the LDW. STM modeling runs indicate that approximately 90% of the total bed area in the LDW receives 10 cm of new sediment (from the combined Green/Duwamish River and lateral sources) within 10 years or less. This sediment is mixed with the existing surface sediment through various processes, including bioturbation and propeller wash.
- A few areas in the LDW will be scoured during high-flow events. Based on the STM, the maximum scour depth is relatively shallow, and is generally limited to sediment in the top 20 cm; thus, deeper sediment would not be exposed as a result of high-flow events. Scour to these relatively shallow depths is estimated to occur in relatively small areas of the LDW. The STM did not account for scour from localized activities, such as discharges from outfalls, tugboat maneuvering, or anchor dragging, which could have caused localized erosional environments. Routine boat traffic is expected to mix the top few cm of sediment, which is part of the biologically active zone also mixed by benthic invertebrates, whereas tugboat maneuvering is a potential source of localized erosion that could disturb sediment at greater depths in small areas. In addition, in some areas, ships may have caused localized erosion from physical forces (e.g., anchor dragging)

unrelated to propeller-driven scour. Location-specific information, in addition to the STM results, will be evaluated in any future remedial design.

- The physical CSM of net depositional environments is supported by both physical and chemical lines of evidence, including lithology and chemistry profiles in sediment cores. The depths of most (70%) peak PCB concentrations were consistent with the estimated sediment deposition rates, with a few exceptions.
- Based on the STM and with ongoing source control in the LDW basin, LDW surface sediment is generally expected to become more similar in character over time to the sediment being transported by the Green/Duwamish River system; localized areas may continue to be influenced by inputs from sources in the LDW basin.

2.6 Additional Considerations for the FS

Data presented in the RI (Windward 2010) are expanded upon in this section for the purposes of this FS. This section also discusses information not presented in the RI that may be relevant to selecting remedial technologies and developing remedial alternatives.

2.6.1 Sediment Physical Properties

The geotechnical and physical properties of sediment (such as sediment grain size and the presence of debris) are important for developing appropriate remedial technologies. Some of the important technology considerations affected by sediment physical properties include:

- Dredgeability or "digability"
- Production rates
- Sediment handling
- Sediment dewatering
- Slope stability
- Bearing capacity for cap placement.

Grain size composition, total organic carbon (TOC), other geotechnical properties such as porosity and bulk density, and the presence of debris were evaluated to provide evidence of the manner in which sediment will behave when handled during remediation. In addition, TOC is determined so that dry weight concentrations of non-polar organic compounds can be organic carbon-normalized for direct comparison to the SMS criteria. TOC also affects the bioavailability of contaminants.

2.6.1.1 Grain Size Composition and Total Organic Carbon

Sediment composition varies throughout the LDW, ranging from sand to mud (fine-grained silt and clays) with varying amounts of organic material, depending on the source of the sediments and the local current velocity. Silt and organic silt are the dominant sediment types, based on Atterberg limits tests, in much of the LDW main channel and in the slips. A mixture of silt and sand dominates the subsurface sediment upstream of the Upper Turning Basin and downstream of Kellogg Island. Sand is predominant from RM 1.1 to 1.8 (mostly west of the navigation channel, but also within it from RM 1.1 to 1.5), on the western side of the navigation channel from RM 2.2 to 2.5, and across the LDW from RM 3.2 to 3.4. The sediment type in the upper 4 ft presented in Figure 2-23 is based on an interpretation from 59 cores collected for the RI in 2006. There is some uncertainty associated with spatially interpolating the extent of physical characteristics between these cores.

Surface sediment toward the mouth of the LDW and on mudflats consists predominantly of fine-grained silts. Overall, the fines (silt [3 to 6.25 micrometers (µm)]+clay [<3 µm]) content of surface sediment in the LDW has been reported to be highly variable, with an average content of 53%. Surface sediment in the navigation channel has a higher fines content than other sediment. The average fines content in the navigation channel was 62%; the 10^{th} and 90^{th} percentile fines contents were 29 and 82%, respectively. Fines content was more variable outside of the navigation channel (excluding the slips), with 10^{th} and 90^{th} percentile contents generally ranging from about 13 to 87%, respectively, and an average content of 53%. Average fines contents have been calculated using point-based averages. Figure 2-24 displays an interpolation of the surface sediment fines content.

Three of the five slips along the LDW had high fines contents relative to the overall LDW average. Slips 1, 3, and 6 had average fines contents of 79, 71, and 87%, respectively. The fines contents of Slips 2 and 4 were lower, with average values of 41% and 57%, respectively. The area upstream of RM 5.0 had a much lower average fines content (approximately 11.5%).

Fines content in the upper 4 ft of the subsurface sediment ranges from 2% to 97%, with a mean of 54% in the 56 RI cores.

TOC content in surface sediment does not vary widely throughout the LDW, and has an average value of 1.9% (Figure 2-25). Outside the navigation channel, the 10^{th} and 90^{th} percentiles were 0.80 and 2.9%, respectively. The TOC content in the navigation channel was less variable than the TOC content outside the navigation channel, with 10^{th} and 90^{th} percentiles of 1.2 and 2.6%, respectively. The average TOC content (1.9%) was the same within and outside the navigation channel. The TOC content in Slips 1, 3, 4, and 6 was slightly higher than the LDW-wide average, with average TOC contents of 2.3, 2.2, 2.6, and 2.7%, respectively. In Slip 2, the average TOC content (1.5%) was lower than the LDW-wide average. Average TOC content was calculated using point-based averages. The area upstream of RM 5.0 had a lower average TOC content (0.84%).

2.6.1.2 Other Geotechnical Characteristics

To understand the engineering properties of sediment that could be the subject of remediation, geotechnical parameters were determined for the upper 4 ft of a subset of sediment cores collected in 2006. These parameters included grain size distribution, moisture content, specific gravity, Atterberg limits (i.e., liquid limit, plastic limit, plastic index), bulk density (dry and wet), and porosity.

Analysis of the grain size distributions of the sediment cores indicated that the median grain size (D_{50}) in the upper 4 ft ranged from 6 µm (or 0.006 millimeter [mm]) to 520 µm (0.52 mm).[24] This grain size range is classified as fine silt to medium sand. Sediment grain size was generally finer in the navigation channel, and coarser in the higher, intertidal zones. In the channel, the D_{50} in the upper 4 ft ranged from 6 µm (0.006 mm) to 320 µm (0.32 mm), which is fine silt to fine sand. In the subtidal bench areas, the D_{50} in the upper 4 ft ranged from 9 µm (0.009 mm) to 410 µm (0.41 mm). In the intertidal areas, the D_{50} in the upper 4 ft ranged from 10 µm (0.01 mm) to 520 µm (0.52 mm). The D_{50} did not vary substantially with depth in the channel and subtidal bench areas. In the intertidal area cores, however, the average of the D_{50} values in the upper 2 ft was 150 µm (0.15 mm), while the average D_{50} values in the lower (2 to 4 ft) sample intervals was closer to 260 µm (0.26 mm).

Sample results for specific gravity, porosity, and wet density did not vary notably with depth, indicating that sediment texture in the upper 4 ft is relatively uniform. The mean particle density of all subsurface sediment samples across similar core intervals ranged from 2.64 grams per cubic centimeter (g/cm^3) to 2.66 g/cm^3. The mean sediment porosity ranged from 59% to 64%, and the mean wet bulk density ranged from 102 pounds per cubic foot (lb/ft^3) to 104.4 lb/ft^3.

Other geotechnical properties varied with depth:

- The mean moisture content of all samples was 75% dw at the surface, decreasing to 63% dw below the 2-ft interval, consistent with the decrease in water content with depth as noted on the core logs.

[24] The D_{50} in the top 4 ft of the sediment is an important consideration when evaluating remedial technologies, such as soil washing.

- The mean dry bulk density across similar core intervals increased with depth from 60.4 lb/ft^3 to 67.2 lb/ft^3, again, consistent with the decrease in water content (Windward and RETEC 2007).
- Atterberg limits tests were performed on fine-grained sediments and revealed that the mean liquid limit of all subsurface sediment samples ranged from 61.2% dw to 70.7% dw, and the mean plastic limit ranged from 35.0% dw to 39.3% dw. Subsurface sediment samples exhibited medium to high plasticity, with the mean plasticity index varying from 26.2% dw to 32.5% dw, consistent with most of the core logs with noted organic compressible texture (Windward and RETEC 2007).

Other geotechnical information is available from past studies that evaluated the engineering feasibility of construction projects in and around the LDW. Table 2-7 lists studies conducted around the LDW for which in-water cores (or upland cores used in cross sections discussed in Section 6) were collected.

2.6.1.3 Debris

Submerged and emergent debris and obstructions can have a substantial impact on the selection and application of appropriate remedial technologies and overall performance of the LDW remediation, particularly as it relates to dredge production rate and the generation of residuals. Encountering debris and submerged objects can damage dredge buckets and clog cutterheads, slow production, cause substantial material release of sediments out of partially opened buckets or flushed hydraulic pipelines, and, in general, impact the ability of a dredging operation to achieve cleanup standards in an effective manner. Industrial waterways such as the LDW typically contain significant amounts (thousands of tons) of debris, deposited over decades of waterway use.

It is not feasible to characterize and quantify the type and extent of all the debris that will be encountered during dredging until dredging is under way; however, design-level assessment may include side-scan sonar, magnetometer, and diver surveys to assist in qualitatively assessing buried debris. Debris sweeps are assumed to be a part of the dredging activities for all remedial alternatives (see Section 8).

Scattered wood and anthropogenic debris (e.g., glass shards, sand blast grit) were identified in 34 of the 56 cores collected for the RI. Six cores (SC17, SC28, SC40, SC47, SC50, and SC54) were sampled with the vibracorer because the MudMole™ sampler (which was the sampling device used for the other cores) was not able to penetrate layers of sand or gravel to depths of 10 ft below the mudline.

The cores with more than 50% visually identified anthropogenic material or debris by volume included SC2 (rock flour), SC26 (gravel), SC28 (sand blast grit), and SC38 (wood and sheen). Trace to moderate hydrocarbon-like sheens were also observed in several cores at depth. Table 2-8 and Figure 2-23 summarize these findings.

2.6.2 Dredging and Capping Events

Historical dredging and capping events were evaluated in the FS for a number of reasons:

- Material accumulated after dredging events can provide evidence of sedimentation rates, sediment transport, and characteristics of sediment contributed from upstream sources (when an area at the upstream end of the LDW, such as Delta Marine, is repeatedly dredged).
- Project dredging depths in both the navigation channel and berthing areas provide information regarding the operational depths necessary for safe vessel navigation. These required depths are important to understand when considering capping remedies.
- Historical dredging records often describe equipment that has been used successfully within the LDW.
- Historical dredging activities often describe material types and quantities that have been removed from the LDW.
- Monitoring conducted at capping sites provides useful data to evaluate the long-term viability of capping in the LDW and recontamination potential.

The dredging projects conducted to maintain navigable depths and the contaminated sediment projects discussed below are valuable case studies that provide information regarding successful dredging and capping methodologies employed in the LDW. Relevant projects are reviewed in greater detail in Section 7 to assist in evaluating remedial technologies.

2.6.2.1 Navigation Channel

An understanding of the dredging that has occurred in the navigation channel is important for the FS because it describes the quantity and nature of sediment originating from the upstream Green/Duwamish River system. Contaminant data associated with the dredging events characterize the quality of these sediments. Because the LDW is a navigational waterway, numerous dredging events have occurred to maintain appropriate depths. These events generally began in the early 1900s when the Lower Duwamish River was straightened into a navigation channel. Most navigation channel dredging since the 1950s has occurred in the upstream portions of the LDW above RM 3.3.

Today, the USACE is responsible for maintaining the navigation channel to the following authorized depths and widths (see Figure 2-26):

- -30 ft MLLW and 200 ft wide from Harbor Island (RM 0.0) to the First Avenue South Bridge (RM 2.0), also known as the Harbor Island and Georgetown Reaches
- -20 ft MLLW and 150 ft wide from the First Avenue South Bridge (RM 2.0) to Slip 4 (RM 2.8), also known as the First Avenue South Reach
- -15 ft MLLW and 150 ft wide from Slip 4 (RM 2.8) to the Upper Turning Basin (RM 4.7), also known as the South Park and 14th Avenue Bridge Reaches. The authorized dimensions of the navigation channel portion of the Upper Turning Basin are 250 ft wide by 500 ft long (USACE 2006).

To maintain navigation depths, the USACE conducts dredging every one to three years in the upstream areas. The area typically dredged under this program is the Upper Turning Basin and downstream to approximately RM 4.0.

Without routine maintenance dredging of the LDW, shoaling would create a shallower channel and inhibit the safe passage of vessels. The Upper Turning Basin acts as a settling basin for sediments that would normally migrate downstream. Routine maintenance dredging keeps sediments from accumulating beyond the holding capacity of the basin. Without the current maintenance dredging, the sediment would continue to migrate downstream via bed load transport and settle in downstream areas. This shoaled material, generally consisting of fine- to medium-grained sand with some silt,[25] is currently dredged in the Upper Turning Basin before it migrates downstream, thereby minimizing the need for maintenance dredging in the lower portion of the LDW.

Table 2-9 summarizes recent maintenance dredging events in the LDW navigation channel between 1986 and 2010. Figure 2-27 shows the locations of the dredging events. The yearly volumes of sediment dredged from the LDW have varied widely, from a minimum of 34,000 cubic yards (cy) dredged in 1986 to a maximum of 200,000 cy in 1992. For the most recent event (February to March 2010), 60,371 cy was dredged from RM 4.18 to the Upper Turning Basin (USACE 2010a).

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[25] Figure 2-24 illustrates fine-grained material in the surface sediment of the navigation channel. Subsurface sediment in the navigation channel, particularly the Upper Turning Basin, is coarser and is primarily fine- to medium-grained sand, with some silt.

2.6.2.2 Dredging Events at Berthing Areas

Berthing areas are typically adjacent to piers, wharves, and dolphins where vessels are moored for temporary parking or unloading/loading. Berthing areas are important to consider in the FS because they represent areas where:

- Specific navigable depths must be maintained.
- Maneuvering vessels may cause scour.
- Remediation and data collection may be difficult because of the presence of moored vessels, overwater structures, or other physical obstructions.

Most berthing areas are within Reach 1 (RM 0 to 2.2). The 2002 Port Series No. 36 publication (USACE 2002), a periodic inventory of shipping facilities within all waters operated by the Port of Seattle, lists berthing areas in the LDW. Table 2-10 and Figure 2-28 summarize these berthing areas, which were generated based on this publication, communications with the Port of Seattle, historical dredging records, established tug routes, and field surveys.

Dredging occurs in these berthing areas to maintain depths for shipping and marina uses. The depths at which these areas are maintained also must be considered when developing remedial alternatives. Evidence of this dredging was obtained from Dredged Material Management Office memos, sampling and analysis plans, and, to a lesser extent, post-dredging confirmation reports. Table 2-11 summarizes the locations, dates, depths, volumes, and other details of private maintenance dredging events in the LDW since 1980. Most dredging in private berthing areas occurs in the downstream portions of the LDW below RM 3.0 because of the large vessels that transit that area. Private dredging has removed about 160,000 cy of material since 1980. Almost 72% of this material, based on reported volumes, was deemed acceptable for open water disposal, based on sediment quality testing. The LDW also has several berthing areas where dredging either has not occurred or has not been documented.

2.6.2.3 Contaminated Sediment Dredging and Capping with Clean Material

Several dredging and capping projects have been conducted in the LDW or made use of clean dredged materials from the LDW for the purpose of capping contaminated sediment. It is important to review these projects for the FS because they strongly relate to the evaluation of remedial technologies and alternatives for the LDW cleanup projects. Prior dredging conducted in the LDW for the purpose of sediment remediation can provide:

- Information regarding the chemical and physical characteristics of the removed sediments.

- Descriptions of equipment and remedial approaches that have been used within the LDW. These records provide information on a number of technical performance areas related to the removal of contaminated sediments, including dredge production rates, impacts of debris, sediment transportation and off-loading methods, sediment treatability and disposal methods, and environmental impacts.
- An understanding of the ability of a remedial operation to achieve cleanup goals and of the factors (e.g., debris, residuals) that may have an effect on that ability.

Sediment remediation projects completed in the LDW in the past 30 years are briefly described below and in Table 2-12.

- In September 1974, 260 gallons of Aroclor® 1242 where spilled into Slip 1. In October 1974, an emergency removal operation was undertaken by EPA, in which divers recovered approximately 70 to 90 gallons of the PCBs using hand-held pumps. This Phase 1 removal operation reduced the pre-dredging surficial Aroclor® 1242 concentration from greater than 30,000 mg/kg wet weight (ww) to about 1,500 mg/kg ww.[26] A subsequent Phase 2 remediation was undertaken by the USACE in March 1976 as the first major dredging operation in the United States to remove PCB-contaminated sediments. Prior to the Phase 2 dredging, the average surficial Aroclor® 1242 concentration was 4 mg/kg ww in the target area. A Pneuma dredge pump, deployed from the USACE vessel *Puget*, was used to remove sediment, resulting in a 10-ft-deep hole. The post-dredging surficial Aroclor® 1242 concentrations at the stations monitored ranged from 0.01 to 8 mg/kg ww (Blazevich et al. 1977).
- The first contained aquatic disposal (CAD) project in Puget Sound was conducted in 1984. In this project, 1,100 cy of PCB-contaminated sediments were dredged from a portion of the LDW navigation channel at RM 0.5, bottom-dumped into a CAD site in the West Waterway, and covered by 4,200 cy of clean sand dredged from the Upper Turning Basin (Battelle 2001, USACE 1994).
- Four sediment remediation projects were conducted in the LDW either as EAAs or before the AOC was signed (i.e., Norfolk CSO/SD, Boeing Developmental Center south storm drain area, Duwamish/Diagonal EAA), and Slip 4/EAA. Sediments were dredged and capped in these areas. These projects are described in more detail in Section 2.7.

[26] Note that data from these reports are reported in wet weight.

Sediment remediation projects that utilized LDW sediment as capping material are summarized below:

- Beginning in 1984, sediments dredged from the upstream portions of the LDW for navigation maintenance have been used as capping material for several nearshore remediation projects in Elliott Bay and in the West Waterway (Battelle 2001). These projects used "clean" sands, generally from upstream portions of the LDW, for capping to cover and isolate *in situ* contaminated sediment or for CAD projects (Battelle 2001, USACE 1994).
- Between 1989 and 1994, four contaminated sediment capping projects were conducted along the Seattle waterfront, each with varying COCs and COC concentrations. These included the Pier 51 Ferry Terminal Expansion, Denny Way CSO, Pier 53-55 Sewer Outfall, and Pier 64/65 capping projects. The capping material for each project, ranging from about 10,000 cy (Pier 51) to about 22,000 cy (Pier 53-55), was obtained from LDW maintenance dredging (Battelle 2001).
- In 2004, approximately 67,000 cy of dredged material from the Upper Turning Basin was beneficially used as capping material to remediate the 58-acre Pacific Sound Resources (PSR) marine operable unit (located in Elliott Bay just outside of the West Waterway). PSR is the site of a former wood-treating facility. The sandier portion of the Upper Turning Basin material was used in nearshore areas where it met design specifications for grain size; finer material was used for deeper parts of the PSR cap.

2.6.3 Overwater and In-water Structures

The majority of upland areas adjacent to the LDW have been industrialized for many decades. Overwater and in-water structures, primarily in the form of wharves, piers, docks, utility crossings, dolphins, and piles are prevalent along the LDW to support industrial and commercial activities. Overwater structures occupy about 19,700 linear ft or 3.7 miles, representing about 24% of the total LDW shoreline (see Figures 2-28 and 2-29).

Existing overwater structures have been catalogued using the 2002 Port Series No. 36 publication (USACE 2002), the Duwamish Waterway Shoreline Inventory (Terralogic and Landau 2004), high-resolution ortho-rectified aerial photographs, oblique aerial photographs available at public internet sites (MSN live search), and field observations. Table 2-10 summarizes available details of these overwater structures.[27]

The distribution and types of overwater and in-water structures within the LDW are important to consider in this FS because they represent areas where:

- Remediation and data collection may be difficult because of restricted access, vessel interference, and armored conditions of the sediment/shoreline. Few FS baseline samples are available from beneath overwater structures, and additional data collection in these areas will be needed during the remedial design phase.
- Sediment contamination from various sources (e.g., bank erosion, stormwater discharges, groundwater/seep transport, spills, poor BMPs, or sediment deposition) could accumulate over time. This represents a data gap that will be filled, where necessary, during the remedial design phase.
- Marine structures such as piles, sheet-pile walls, pipelines, cables, and foundations may be damaged or undermined by sediment removal.
- Remedial alternatives may have to be engineered to allow navigation depths to be maintained.
- Vessel maneuvering, including vessels used for remediation, can cause scour.
- Piles, moored vessels, floating docks, and other structures may need to be removed or modified to implement the remediation.
- Vertical and horizontal clearances may impact traffic related to remedial operations (e.g., delivery of dredged material to an off-loading facility or of capping material to the project site).

Necessary remediation in areas with overwater and in-water structures will be coordinated with source control efforts and other remediation work.

The majority of overwater structures in the LDW are within Reach 1 (RM 0.0 to RM 2.2). The primary overwater structures in this reach are wharves used for the shipment and receipt of bulk materials such as cement, coal, gypsum, sand and gravel, rock lime, lumber products, and scrap metal. In total, 8 such land-based companies operate along the LDW, and 12 associated wharves or piers on both sides of the LDW currently serve

[27] Approaches for cleanup near and beneath overwater structures are discussed in Section 7 of this FS as they relate to the evaluation of remedial technologies and development of applicable remedial alternatives for the LDW.

these operations within Reach 1. Other overwater structures in operation within Reach 1 support the shipment and/or receipt of seafood, containerized and other cargo, and construction equipment, as well as the moorage of private and commercial vessels. The Duwamish Shipyard, located on the west side of the LDW at about RM 1.4, formerly operated a wharf, marine railway, graving dock (dry dock), and two floating dry docks. The graving dock was subsequently filled in after the shipyard ceased operations. In-water structures include a pile field and pile and dolphin groups at RM 0.2 and around Kellogg Island. Overhead utility crossings occur at two locations in this reach (RM 0.4 and RM 1.95). Submerged sewer lines are located near the downstream end of this reach at RM 0.4, while submerged cable and pipeline crossings are located further upstream at RM 1.9. The First Avenue Bridge (State Route 99) crosses the LDW in two spans at RM 2.1 to RM 2.2. Its supporting structures are located in-water, with barrier walls restricting vessel traffic from navigating too close to the bridge supports.

Within Reach 2 (RM 2.2 to RM 4.0), the primary overwater structures are wharves used for the shipment and/or receipt of scrap metal, lumber, and containerized cargo, as well as the moorage of floating equipment. In total, five land-based companies and seven associated wharves on both sides of the LDW serve these operations within Reach 2. Overwater structures in this reach also include buildings constructed on in-water supports (e.g., Boeing Plant 2). A new South Park Bridge is under construction between RM 3.3 and RM 3.4 just downstream of the former bridge location, and is scheduled to be finished in the fall of 2013. An overhead utility crossing is located at RM 3.6, and submerged cable and pipeline crossings occur in two areas (RM 2.85 to RM 3.0 and RM 3.15 to RM 3.4).

Within Reach 3 (RM 4.0 to RM 4.8), only three major overwater structures exist: the Duwamish Yacht Club floating docks, the Delta Marine Industries wharf, and the Boeing Slip 6 wharf. These facilities currently support moorage for recreational vessels, recreational and commercial vessel construction and repair, and barge moorage, respectively. There is also a timber pier along the west bank of the Upper Turning Basin at RM 4.6 on property owned by the Muckleshoot Tribe. An overhead utility crossing is located at RM 4.4.

2.6.4 Shoreline Conditions

The LDW study area contains a number of different types of shoreline features that will need to be considered in developing remedial alternatives for the site (e.g., riprap fronted by dock face). Known shoreline conditions of the LDW are displayed in Figure 2-29.

The extensive shoreline development affects the remedial alternatives that may be used. Open shoreline areas are also important to consider when evaluating remedial alternatives. They represent areas where habitat restoration can more easily be combined with remedial actions. However, currently armored shorelines, which may be removed for remedial activities, also present opportunities for habitat improvements. These features are also important to consider in the FS because they represent locations where:

- Pile-supported structures, outfalls, engineered or unengineered steep slopes, and vertical bulkhead walls may be damaged or undermined by sediment remediation or removal.
- Associated shoreline armoring and debris may impact the selection and implementation of remedial alternatives.
- Outfalls may require armoring of adjacent sediment caps or backfill material.
- Intertidal and riparian bank soils may contain contaminants and require remediation.
- Remediation and data collection may be encumbered because of restricted access or hardened surfaces.
- Associated shoreline armoring materials and debris may impact the implementation of remedial alternatives.
- Piles, debris, and derelict structures may have to be removed to achieve remediation goals.
- Shoreline armoring and debris may impact the selection and implementation of remedial alternatives.
- Staging of remediation equipment may be feasible.

Shoreline armoring (e.g., engineered and unengineered riprap, cobbles, broken concrete, asphalt), bulkheads (e.g., steel sheet pile, timber pile, concrete) and exposed bank fill are the general types of shoreline that exist along the LDW. Of the total 79,580 ft (15.1 miles) of LDW shoreline, represented by the east and west banks, Kellogg Island, and the southern end of Harbor Island, approximately 53,400 ft (10.1 miles) are armored shoreline, 5,280 ft (1.0 mile) are vertical bulkhead, 1,400 ft (0.3 mile) are dock face, and 19,300 ft (3.7 miles) are exposed shoreline. Dock face also overlaps the shoreline over 24,200 ft (4.6 miles). Figure 2-29 displays these features and notes the total dock face frontage (25,900 ft or 4.9 miles).

2.6.5 Shoreline and Nearshore Habitat Features

Remedial alternatives in this FS consider impacts to nearshore habitat that may occur as a result of sediment remediation activities. The substantive requirements of a number of state and federal laws and regulations impose basic constraints on nearshore in-water work including (but not limited to):

- No net loss of aquatic habitat
- Preference for intertidal (-4 to +11.3 ft MLLW), shallow subtidal (-4 to -10 ft MLLW) habitat creation
- Preference for shallow slopes
- Preference for finer substrate
- Importance of riparian vegetation.

General approaches for nearshore remediation are considered in this FS, sufficient for feasibility-level definition and evaluation of alternatives. Detailed approaches for nearshore areas would be developed in the remedial design phase.

In addition, federal, state, and tribal Natural Resource Trustees will be working to restore damaged habitat in the LDW under the Natural Resource Damages (NRD) provisions of CERCLA. To the extent possible, implementation of remedial actions will be coordinated with NRD habitat restoration activities.

2.6.6 Vessel Traffic Patterns

Various vessel traffic operates within the LDW, including tugboats moving alone or with barges/derricks, fishing vessels, bulk cargo vessels, recreational vessels such as sailboats and motor yachts, and miscellaneous vessels such as fireboats, passenger boats, and research vessels. The LDW is also frequently used by recreational boaters in kayaks.

Five bridges span the LDW and the West Waterway. Three are located in the West Waterway: the high-level West Seattle Bridge, a railroad bridge, which remains open unless a train is traversing the waterway, and the Spokane Street Bridge. Bridge opening logs for the other two bridges that cross the LDW (First Avenue Bridge and the former South Park Bridge[28]) and the Spokane Street Bridge are discussed in this section. These are opened periodically to allow the passage of vessels that exceed clearance heights. The Spokane Street Bridge (downstream of the LDW near its mouth) is operated by the Seattle Department of Transportation (SDOT). The First Avenue Bridge (at RM 2.0) is operated by the Washington State Department of Transportation

[28] The former South Park Bridge was closed and demolished in 2010. A new South Park Bridge is under construction between RM 3.3 and RM 3.4 just downstream of the former bridge location and is scheduled to be finished in the fall of 2013.

(WSDOT). The former South Park Bridge (at RM 3.3) was operated by the King County Department of Transportation (KCDOT). Logs of bridge openings quantify the number, duration, and frequency at which large vessels move under the bridges while open. These records were reviewed to assess the degree to which vessel traffic varies throughout portions of the LDW (SDOT 2006, KCDOT 2006, WSDOT 2006).

Bridge opening logs for the Spokane Street Bridge, which has a 55-ft clearance above mean high water, record the number of vessels entering and exiting the LDW through the West Waterway and every occasion the bridge is opened. For the analysis of potential vessel impacts on the LDW, only openings for motorized vessels other than sailboats were tabulated for the period 2003 to 2005 (Table 2-13). Motorized vessels include tugboats, which have a maximum displacement of 500 tons and an average displacement of 200 tons, and container ships, which can reach 29,000 tons and have an average displacement of 3,500 tons.

Logs for the Spokane Street Bridge for the period 2003 to 2005, portions of which are summarized in Table 2-13, recorded monthly bridge openings for large motorized vessels, ranging from 93 openings in February 2005 to 261 openings in March 2003. The average number of monthly openings during the period is 146, or approximately 5 per day. Most of these openings were for tugboat-escorted vessels and barges, representing 75 to 140 per month, with an average of 104, or approximately 3 per day (SDOT 2006). These counts represent bridge openings for large vessels entering the LDW; vessels with a low clearance do not require the bridge to be opened.

Vessels entering and leaving the LDW could disturb bottom sediments while transiting the navigation channel. Multiple vessels passing in close time proximity might create a net scour effect by preventing suspended sediment from resettling to the bed. To evaluate this possibility, an analysis was conducted to determine the frequency with which vessels enter or leave the LDW within 1 hour of each other. For motorized vessels exceeding 100 tons in displacement during the period from 2003 to 2005, the average number of times per month when 2 bridge openings occurred within 1 hour was 28, representing approximately once per day, or 40% of the openings. The conclusion from this analysis is that cumulative scour potential is expected to be minimal because vessels often do not enter the LDW within 1 hour of the prior vessel entrance and because most sediment is expected to resettle in the same place given the low frequency. The logs show that regular vessel traffic is spaced from one to several hours apart, providing minimal potential for cumulative propeller scour from several subsequent passing ships.

Records for the two drawbridges located within the LDW provide evidence of vessel traffic at least as far upstream as each bridge's location:

- The First Avenue Bridge crosses the LDW at RM 2.0. It has a 41-ft clearance at the center span and 24-ft clearance at the side spans. It opened over

1,300 times annually in both 2005 and 2006, averaging less than 4 openings daily.

- The former South Park Bridge (also referred to as the 14^{th} Avenue Bridge, which was demolished in 2010) was located at RM 3.3. It had a 34-ft clearance at the center span and 21-ft clearance at the side span; the draw spans were removed in the summer of 2010 as part of the bridge's demolition. It was opened between 700 and 800 times annually in 2005 and 2006, approximately twice daily.

Comparison of the annual openings of the Spokane Street Bridge (approximately 2,000; KCDOT 2006) and the First Avenue Bridge (approximately 1,500; WSDOT 2006) indicates that about 75% of the vessel traffic that enters the LDW berths downstream of RM 2.0 (i.e., in Reach 1). Comparison of the number of Spokane Street Bridge openings to the annual openings of the former South Park Bridge shows that 35% to 40% of the vessels entering the LDW continue upstream at least as far as RM 3.3 (former South Park Bridge) (700 to 800 annual openings compared to 2,000 at the Spokane Street Bridge, with the assumption that each opening represents one vessel).

2.6.7 Bathymetric Coverage

Bathymetric data are used to determine mudline elevations, which in turn are used to calculate sediment volumes and compare current conditions against permitted maintenance dredging depths.

Bathymetric soundings were collected for the RI in 2003 (Windward and DEA 2004). However, the spatial extent of data collection was restricted in areas where vessels and overwater structures blocked access. As a result, the GIS grid generated to display mudline elevations was incomplete because of missing data.

Thus, in this FS, data from other sources were used to complete the bathymetry coverage. These data sources included:

- A U.S. Fish and Wildlife Service GIS shapefile of the extent of the intertidal zone, based on an aerial photograph in which sediments exposed at low tide could be observed
- Mudline elevations recorded in the field during RI sample collection (by calculation of water depth and tide level)
- Soundings recorded on National Oceanic and Atmospheric Administration (NOAA) electronic nautical charts (NOAA 2008)
- Elevations recorded during a 2003 USACE bathymetry survey (USACE 2003a).

2.7 Status of Early Action Areas

In 2003, LDWG proposed seven areas as candidates for early cleanup actions (Windward 2003b). Of the seven initially proposed, five areas (or portions of them) have been designated as EAAs by EPA and Ecology and are referred to as the EAAs in this FS. The parties responsible for the five EAAs have conducted a study of each one, and cleanups have occurred at three of the five EAAs: the Duwamish/Diagonal EAA (King County 2010a), the Norfolk EAA (King County 1999b, Calibre 2009), and Slip 4 (Integral 2012). Remedy decisions have been issued by EPA for Terminal 117 and Boeing Plant 2/Jorgensen Forge. These cleanups are being implemented under EPA Consent Orders. The purpose of this section is to provide an update on the five EAAs. All five EAAs have published SCAPs and have identified investigations and work with MTCA, RCRA, or CERCLA orders, or voluntary actions for major contaminant sources and pathways. The two candidate EAAs that were not carried forward as EAAs are included in the areas being considered for remediation in this FS.

2.7.1 Duwamish/Diagonal

In 2003 and 2004, the Duwamish/Diagonal EAA at RM 0.4E was dredged (68,000 cy). In 2004, the dredged area (7 acres) was capped. These actions were conducted by King County for the Elliott Bay/Duwamish Restoration Program (EB/DRP), which was established in 1991 to implement an NRD Consent Decree. The COCs that triggered these actions were total PCBs, mercury, BEHP, and butyl benzyl phthalate. The cleanup action did not address all the contamination present in this area.

Analysis of post-action sampling data from the perimeter stations in March 2004 revealed that the 2003/2004 project dredging activities had increased surface sediment PCB concentrations around the margin of the southwestern portion of the dredge/cap area (for technology performance discussion see Section 7; for time trends, see Appendix J). The occurrence of dredging residuals in this area was consistent with observations made regarding initial dredging operations. The BMPs that were required to minimize the spread of dredging residuals were not consistently employed, which resulted in elevated PCB concentrations around the dredge footprint. After consultation with Ecology and EPA, King County selected the thin-layer placement option, also known as ENR, as the best way to reduce the elevated PCB concentrations most expediently within the 4-acre dredging residual area adjacent to the dredge/cap area. This option was implemented in 2005, when a thin layer of clean sand was placed to a minimum thickness of 6 inches over this area. Annual monitoring was performed for five years to document the effectiveness of this option and to compare it to natural recovery rates in the area surrounding the dredge/cap area, which had significantly lower dredging residuals.[29] The most recent monitoring event (2009) showed BEHP

[29] Five years (2005 to 2009) of post-remedy monitoring data for the cap and ENR area are presented in Appendix J. Appendix F presents perimeter monitoring data for this time span.

exceedances of the SQS in 1 of 8 cap samples and in 1 of 7 ENR area samples. No other contaminants exceeded the SQS in the 2009 cap or ENR samples. The need for further cleanup for this 4-acre area is considered part of the development of the remedial alternatives. Appendix J discusses time trend data in this area and on the sediment cap. Section 7 discusses diver probing observations made of the ENR thickness during post-remedy surveys. No further action is anticipated in the FS for the 7-acre cleanup area.

The SCAP, published in 2004, identified 446 facilities in the Duwamish/Diagonal EAA drainage basin that needed to be evaluated for their potential to recontaminate sediments. Ongoing source control efforts include source tracing and business inspections as well as an Ecology study (including sampling) of exterior building paints in the Diagonal Avenue S drainage basin. Terminals 108 and 106, adjacent to the EAA, are being evaluated for potential source control actions under Ecology's Voluntary Cleanup Program (VCP). The RI provides further details on source control activities occurring in the Duwamish/Diagonal drainage basin (Windward 2010).

2.7.2 Norfolk EAA: Norfolk CSO/SD and Boeing Developmental Center South Storm Drain

A partial cleanup at the Norfolk EAA was conducted by King County in 1999. The action was conducted for EB/DRP in the vicinity of the Norfolk CSO/SD. However, this action predates the AOC for the LDW RI/FS. During this action, 5,190 cy of contaminated sediment were excavated with dredging as deep as 9 feet in one portion of the area in an attempt to remove all contamination. The area was then backfilled with 6,700 cy of clean material. Bank stability concerns precluded further excavation, leaving some sediment in place that exceeded the CSL for total PCBs. This area was backfilled up to the original grade, resulting in backfill material to depths of 9 ft or more below the mudline (King County 1999b).

In 2001, total PCB concentrations on the Norfolk CSO/SD cleanup area ranged from 31 µg/kg dw to 1,330 µg/kg dw in the upper 10 cm of sediment and reached up to 1,900 µg/kg dw in a 0- to 2-cm sample. The highest concentrations were detected in samples near the Boeing Developmental Center's south storm drain, and a source investigation was initiated.

Under Ecology's VCP, a small area immediately offshore of the Boeing Developmental Center at RM 4.9E was excavated and capped in 2003 to address the recontamination of the Norfolk CSO/SD cleanup area. During this event, Boeing removed 60 cy of sediment from a 0.04-acre area inshore of the Norfolk CSO/SD cleanup area, and in the vicinity of the Boeing Developmental Center's south storm drain just downstream of the Norfolk CSO/SD. The excavation was then backfilled with clean sand overlying a geotextile liner containing activated carbon. The cleanup did not address all the contamination present in the broader Norfolk EAA. Subsequent monitoring of surface sediment on both caps shows that PCB concentrations have since decreased. Temporal trends in contaminant concentrations in these cleanup areas are discussed in later sections of the FS. No further action is anticipated in the FS for these cleanup areas; the nearshore and downstream areas of the Norfolk EAA are being evaluated in this FS. The RI includes a discussion of the source control activities occurring in the drainage basin (Windward 2010).

2.7.3 Slip 4

The head, or eastern 3-acre sediment and riverbank portion, of the 6-acre Slip 4 (the Slip 4 EAA) was actively cleaned up by the City of Seattle under an EPA Consent Order from October 2011 through January 2012. The cleanup included:

- Dredging/excavating approximately 10,260 cy of sediments and bank material with off-site disposal
- Overexcavating bank areas to expand intertidal and riparian habitat
- Conducting pier demolition
- Removing piling and debris
- Capping the entire 3.6-acre area with 30,700 cy of clean sand and gravel to obtain a 12-in minimum cap thickness, including armor rock and 3,500 cy of granular activated carbon amended filter material
- Implementing institutional controls and long-term monitoring.

As part of implementing the selected remedy, the City of Seattle purchased much of the affected portion of Slip 4. In the summer of 2009, the City of Seattle cleaned out and replaced the Georgetown Steam Plant Flume with a pipe, which still discharges stormwater to Slip 4.

Other source control actions included cleaning catch basins and storm drain lines at King County International Airport (KCIA) and inspecting businesses and facilities at KCIA to verify that they comply with applicable regulations and BMPs. In addition to this work, from 2004 to 2007, the Boeing Company removed approximately 89,000 linear feet of PCB-contaminated concrete joint material from North Boeing Field, and in 2010 they removed an additional 3,900 linear feet of this material from the northern area of the property (Ecology 2011a). Construction of the Slip 4 EAA cleanup was undertaken after completion of these and other source control actions within the Slip 4 drainage basin.

Four surface sediment samples were collected in 2006 as part of the 2007 100% design submittal, and 13 subsurface samples were collected in 2008 for the 2010 design update (in addition to those in the RI baseline dataset). These samples are included in the FS baseline dataset.

The southwestern portion of Slip 4 is being addressed as part of the Boeing Plant 2 RCRA corrective action, which will include a habitat restoration project pursuant to an NRD settlement between the natural resource trustees and Boeing.

2.7.4 Boeing Plant 2/Jorgensen Forge

Since the early 1990s, various soil, groundwater, and sediment investigations have been conducted within the Boeing Plant 2/Jorgensen Forge EAA under RCRA for Boeing Plant 2 and under MTCA and CERCLA for Jorgensen Forge. Boeing Plant 2 is a RCRA hazardous waste treatment, storage, disposal (TSD) facility subject to RCRA permitting and regulation. A component part of all RCRA permitting is the performance of all necessary corrective action or cleanup of hazardous waste or constituents released at or from the TSD. EPA issued a RCRA AOC to Boeing in January 1994, requiring the performance of a RCRA Facility Investigation/Corrective Measures Study (RFI/CMS), to determine the nature and extent of hazardous constituent releases at or from Plant 2 requiring corrective action (also called corrective measures) and an analysis of alternative corrective measures to address those releases, as well as the implementation of Interim Measures to mitigate or correct ongoing or continuing releases in a manner consistent with future corrective action. A RCRA RFI/CMS is the functional equivalent of a CERCLA or MTCA RI/FS.

Surface sediment exceedances of the SMS criteria in this EAA included total PCBs, PAHs, phthalates, cadmium, chromium, copper, lead, mercury, phenol, silver, and zinc. Boeing Plant 2 sediments have some of the highest concentrations (thousands of µg/kg dw) of total PCBs in the LDW. Investigations of upland portions of Boeing Plant 2 have identified over 40 hazardous constituents in upland soil, groundwater, seeps, and source tracing samples.

To date, several potential sources identified during upland investigations of Boeing Plant 2 have been controlled or removed as RCRA Interim Measures under the AOC (e.g., stormwater lines have been removed and/or cleaned, and catch basins connected to the storm drain conveyance system have been routinely sampled and cleaned as needed). Soils and groundwater in some areas with elevated hazardous constituent concentrations have been removed, remedied, or contained. There have also been very limited hot-spot removals of contaminated sediments in the intertidal area offshore of Boeing Plant 2. Eleven surface sediment and 355 subsurface sediment samples were collected from 2007 to 2009 and have been included in the FS baseline dataset. These samples are in addition to those collected earlier and included in the RI baseline dataset. EPA recently approved Boeing's CMS for remediation of contaminated sediments adjacent to Plant 2 (2010). A RCRA Statement of Basis (the RCRA equivalent of a Proposed Plan for Remedial Action) containing EPA's proposed corrective action for Boeing Plant 2 sediments was released in spring 2011; this document describes alternatives for sediment remediation, with a range of 114,000 to 142,000 cy to be dredged from the northern portion of the EAA and a range of 43,000 to 86,000 cy from the southern area of the Boeing Plant 2 portion of the EAA (EPA 2011b).

The 22-acre Jorgensen Forge facility is located south (upstream) of Boeing Plant 2. In 2007, Ecology and the Jorgensen Forge Corporation (the current owner of Jorgensen Forge) negotiated an agreed order to conduct a source control investigation at the facility. Underground storage tank removals and some upland soil investigations have occurred (Ecology 2007a). Also, in 2003, EPA issued an AOC to the Earle M. Jorgensen Company (a former owner of Jorgensen Forge) for investigation and preparation of an Engineering Evaluation/Cost Analysis (EE/CA) for a non-time-critical removal action for sediments and associated shoreline bank soils. The final EE/CA was completed and approved by EPA in 2011 (Anchor QEA 2011). EPA anticipates issuing an Action Memorandum following public comment on the EE/CA, and selecting a remedy compatible with its proposed remedy for Boeing Plant 2. Amendments to Boeing's and Jorgensen's AOCs with EPA require that the Boeing Plant 2 and Jorgensen cleanups be fully coordinated to address sediments in this EAA.

2.7.5 Terminal 117

The Terminal 117 (T-117) upland area at RM 3.5W was historically used for the manufacture and storage of asphalt products. The Duwamish Manufacturing Company began manufacturing asphalt roofing materials at T-117 in the late 1930s at a location that generally corresponds with the present-day western half of the upland portion of T-117. The business and property were sold in 1978 to the Malarkey Asphalt Company, which continued operating until 1993 when industrial operations ceased. During the Duwamish Manufacturing Company's operation of the facility from the late 1960s through the mid 1970s, used oils, some of which contained PCBs, were used as fuel for the boilers in the asphalt manufacturing process. Some of the used oils came from Seattle City Light (Windward et al. 2010).

Soils on the upland portion of T-117 with elevated concentrations of PCBs were removed by the Port of Seattle with EPA oversight pursuant to separate AOCs issued by EPA for time-critical removal actions in 1999 and 2006. In addition, PCB-contaminated areas in the rights-of-way were paved, and a temporary stormwater collection system was voluntarily installed by the City of Seattle, without EPA oversight, which conveys most runoff from the roadways adjacent to T-117 to the combined sewer system.

The EE/CA for T-117 was approved by EPA on June 3, 2010. It included an analysis of alternative non-time critical removal actions for three study areas: adjacent in-water sediments, the former industrial facility upland soil and groundwater, and adjacent city streets and residential yards; along with an assessment of potential recontamination of the nearby Basin Oil facility and the South Park Marina. Data gap findings and groundwater occurrence and quality are presented in separate project documents. EPA issued its Action Memorandum for the T-117 EAA, containing removal actions for each of the three study areas, on September 30, 2010 (EPA 2010). An Administrative Settlement Agreement and Order on Consent for implementing the selected removal action was issued to the Port of Seattle and City of Seattle on June 9, 2011 (EPA 2011c).

Table 2-1 Chronology of Historical Events in the Lower Duwamish Waterway and River

Event	Event or Report Date	Notes
Duwamish River channelization	1901	Waterway construction began with filling of wetlands using regrade material from surrounding hills.
Dredging of East and West Waterways to create Harbor Island	1903-1905	
Channelization of Duwamish River into LDW	1909-1916	Present configuration by 1920.
Construction of Lake Washington Ship Canal	1916	Restricted flow of Lake Washington to the Duwamish River, redirected Cedar River from Black and Duwamish rivers to Lake Washington.
Commercial Waterway District established	pre-1920	District is responsible for maintenance of LDW.
USACE became responsible for maintenance dredging of navigation channel	1920	
Construction of Howard Hanson Dam	1961	Last flood event in 1959. Dam approximately 65 miles upstream of LDW.
Port of Seattle ownership of LDW begins	1963	
Shoreline filling	1966-1972	Slough at RM 0.5E filled, last evidence of Slips 5 and 7, first evidence of Slip 3 with geometric configuration.
Significant sewage treatment upgrades	1967-1969	Duwamish Siphon built under river to transport water from West Seattle to pump station. Duwamish Pump Station operations begin (pump water to West Point) and Diagonal Avenue Sewage Treatment Plant operation and direct discharge to the LDW ceases.
PCB transformer spill in Slip 1	1974	Sediment in and outside of Slip 1 dredged by EPA.
Last evidence of Diagonal Avenue Sewage Treatment plant structures on USACE conditions surveys	1981	
Renton Wastewater Treatment Plant no longer discharges treated effluent to the Green River.	1984	Notable releases to LDW from these sources cease.
Harbor Island secondary lead smelter closes.		
Norfolk CSO/SD EAA sediment removal and capping	1999	Sediment dredging and capping offshore of Norfolk CSO/SD at RM 4.9.
Listing of LDW as Superfund Site	2001	
Boeing Developmental Center South Storm Drain sediment removal and capping	2003	Inshore area adjacent to Norfolk CSO/SD remediated at RM 4.9.
Duwamish / Diagonal EAA sediment removal and capping	2003-2005	Dredging and capping of two areas in 2003-2004. Thin-layer of sand placement cap on adjacent area at RM 0.5 in 2005.

Sources: King County, Anchor, and EcoChem 2005b, *Duwamish/Diagonal Cleanup Study Report*; HistoryLink.org; USACE 1947 to 1981, Historical Conditions Surveys; Windward 2010, *Lower Duwamish Waterway Remedial Investigation*.

Notes:

CSO/SD = combined sewer overflow / storm drain; EAA = early action area; EPA = U.S. Environmental Protection Agency; LDW = Lower Duwamish Waterway; PCB = polychlorinated biphenyl; RM = river mile; USACE = U.S. Army Corps of Engineers

Table 2-2 FS Data Added to the RI Baseline Dataset

Sampling Event	Sample Date(s)	Number of Samples or Locations[a]	Party; Notes
Surface Sediment (FS baseline data averaged to location; count is number of locations)			
Duwamish/Diagonal EAA (perimeter stations)	2005-2009	13 (5 in 2005; 8 in 2009)	King County; most recent perimeter data in FS baseline dataset; pre-remedy cap and ENR area data in FS baseline dataset [same as in RI baseline dataset]; data from all monitoring events used in time trends analysis; post-remedy dioxin/furan composite sample from ENR area also in FS baseline dataset to increase breadth of dioxin/furan dataset.
Boeing Plant 2 EAA Western Boundary	2007	11	The Boeing Company; analyzed only for PCBs.
Terminal 117 EAA Boundary	2008	17	Port of Seattle; analyzed only for PCBs and dioxins/furans.
Slip 4 EAA Design	2006	4	City of Seattle.
LDWG dioxin/furan site-wide sampling	2009, 2010	41	LDWG; 7 discrete samples in beaches were also analyzed for PCBs, arsenic, and cPAHs; 1 additional sample (LDW-SS527, not in a beach) was analyzed for PCBs, arsenic, cPAHs, and the full suite of SMS contaminants. This event also included a collection of 6 beach sediment composite samples, but they are not part of the FS baseline dataset because they do not represent discrete samples. These data were used to update beach risk estimates and thus will be used for technology assignments (Section 8).
PACCAR / Kenworth, 8801 East Marginal Way	2006, 2008	41	Anchor QEA for PACCAR.
Industrial Container Services	2007	4	Industrial Container Services.
Terminal 115 Intertidal	2009	5	Port of Seattle.
Ecology Upstream Surface Sediment	2008	86 (8 in LDW at RM 4.9 and 5.0)	Ecology; 8 locations at RM 4.9 and 5.0 are a part of baseline dataset; locations upstream of RM 5.0 used in development of BCM input parameters (Section 5 and Appendix C).
Ecology SPI camera survey/chemistry and bioassay data (RM 0.0 to Slip 4)	2006	30	Ecology; locations also include toxicity data.
Total surface sediment chemistry location count		***174 in Study Area (does not include locations upstream of RM 5.0)***	

Table 2-2 FS Data Added to the RI Baseline Dataset (continued)

Sampling Event	Sample Date(s)	Number of Samples or Locations[a]	Party; Notes
Subsurface Sediment (number of samples; multiple samples in each core)			
Boeing Plant 2 Boundary and under building	2008, 2009	355	The Boeing Company.
PACCAR / Kenworth, 8801 East Marginal Way	2008	25	Anchor for PACCAR.
Slip 4 Early Action Area	2006, 2008	38	Landau for City of Seattle.
Terminal 115 Dredged Material Characterization	2008	11	Port of Seattle.
USACE Navigation Channel Dredged Material Characterization (data newer than RI baseline dataset)	2008, 2009	44	USACE; data used in development of BCM input parameters (Section 5).
USACE Navigation Channel Dredged Material Characterization (older data that were not in RI Baseline dataset)	1990, 1991, 1996	32	USACE; data used in development of BCM input parameters (Section 5).
Delta Marine Dredged Material Characterization	2007	4	Delta Marine.
Total subsurface sediment sample count		***509 in Study Area***	

Notes: See Figures 2-5 and 2-6a through 2-6i for sample locations.

a. Surface sediment data are averaged to location if both parent and duplicate samples exist at one location. Subsurface sediment counts are by sample because multiple samples are typically within each core. However, for core samples, parent and duplicate samples are also averaged.

BCM = bed composition model; cPAH = carcinogenic polycyclic aromatic hydrocarbons; EAA=Early Action Area; ENR = enhanced natural recovery; FS = feasibility study; LDWG = Lower Duwamish Waterway Group; PCBs = polychlorinated biphenyls; RI = remedial investigation; RM = river mile; SMS = Sediment Management Standards; SPI = sediment profile imaging; USACE = U.S. Army Corps of Engineers

Table 2-3 Statistical Summaries for Human Health Risk Drivers in Sediment

Data Type/Contaminant	Summary Statistics for Sediment in the LDW (RM 0.0 to 5.0)				Total Number of Sediment Samples in FS Baseline Dataset	
	Minimum Detect	Calculated Mean[a]	Maximum Detect	Spatially-Weighted Average Concentration[a]	Total	With Detected Values
Surface Sediment						
Total PCBs (µg/kg dw)	2.2	1,136[b]	2,900,000 230,000[b]	346[b,c]	1,392 (1,390)[b]	1,309
Arsenic (mg/kg dw)	1.2	17	1,100	15.6	918	857
cPAHs (µg TEQ/kg dw)[d]	9.7	459	11,000	388	893	852
Dioxins/Furans (ng TEQ / kg dw)[e]	0.25	42	2,100	25.6	123	119
Subsurface Sediment						
Total PCBs (µg/kg dw)	0.52	1,953	890,000	n/a	1,504	1131
Arsenic (mg/kg dw)	1.2	29	2,000	n/a	531	453
cPAHs (µg TEQ/kg dw)[d]	1.2	373	7,000	n/a	542	449
Dioxins/Furans (ng TEQ / kg dw)[e]	0.15	17	194	n/a	64	64

Source: FS baseline surface and subsurface sediment dataset dated April 28, 2010 (surface) and May 14, 2010 (subsurface).

Notes:

a. The calculated mean and the SWAC use one-half the reporting limit for undetected data.

b. Mean and SWAC for total PCBs calculated with two outliers (2,900,000 and 230,000 µg/kg dw in Trotsky inlet) excluded (n = 1,390). The highest remaining concentration in the FS baseline surface sediment dataset (223,000 µg/kg dw) is located in the Norfolk area. If the two outliers were not removed, the mean would be 3,383 µg/kg dw and the SWAC would be 1,313 µg/kg dw.

c. 95% upper confidence limits on the total PCB SWAC (ranging from 544 to 702 µg/kg dw) were calculated by Kern (2010) for the interpolated RI baseline dataset using various methods. No attempt has been made to calculate 95% upper confidence limits on the SWACs for the other risk drivers.

d. cPAH TEQ calculated using compound-specific potency equivalency factors (California EPA 1994).

e. Dioxin/furan TEQ calculated using World Health Organization (Van den Berg et al. 2005) mammalian toxic equivalent factors.

cPAH = carcinogenic polycyclic aromatic hydrocarbons; dw = dry weight; FS = feasibility study; kg = kilograms; µg = micrograms; mg = milligrams; n/a = not applicable; ng = nanogram; PCB = polychlorinated biphenyls; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent

Table 2-4 Human Health Risk-Driver Summary Statistics from Ecology Upstream Bedded Sediment Event

Risk Driver	Number of Samples (Number of Detections)	Range of Concentrations	Mean	Median	90th Percentile	UCL95
Total PCBs[a] (µg/kg dw)	73[a] (38)	2.7 U – 22	3	3	6	3
Arsenic (mg/kg dw)	74 (74)	3.7 - 16	7	6	10	7
cPAHs (µg TEQ/kg dw)	74 (60)	0.7U - 230	18	9	57	43
Dioxin/Furans (ng TEQ/kg dw)	74 (54)	0.07U - 8.4	1	0.3	3	2

Notes:

1. See Appendix C, Part 3 for all data; these data are all contained within the FS project database. Appendix C also provides a discussion of this event, its data, and how these data were used as a line of evidence for the Bed Composition Model upstream input parameters.

a. Outlier of 770 µg/kg dw for total PCBs was excluded from the dataset statistics, because it appeared to be related to an outfall.

cPAH = carcinogenic polycyclic aromatic hydrocarbons; dw = dry weight; Ecology = Washington State Department of Ecology; µg = micrograms; mg = milligrams; ng = nanograms; PCBs = polychlorinated biphenyls; TEQ = toxic equivalent; U = undetected at the reporting limit shown; UCL95 = 95% upper confidence level on the mean.

Table 2-5 Datasets Used in the FS that are not Part of FS Baseline Sediment Dataset

FS Dataset	Where Used	Where Found[a]
Composite surface sediment samples from 6 beaches collected by LDWG	Risk calculations described in Section 3 and Appendix B	Project database
CSO whole-water samples collected by King County and storm drain solids samples collected largely by Seattle Public Utilities	BCM input parameters discussed in Section 5 and Appendix C, Part 3	Additional Excel data files
Surface sediment samples and solids from centrifuged water samples collected upstream of the LDW by Ecology	BCM input parameters discussed in Section 5 and Appendix C, Part 3	Additional Excel data files
Whole water samples upstream of the LDW collected by King County	BCM input parameters discussed in Section 5 and Appendix C, Part 3	Additional Excel data files
2008 Puget Sound sediment PCB and dioxin survey (*OSV Bold* Survey) (conducted by DMMP)	Background calculations described in Section 4	Additional Excel data files
Resampled surface sediment stations in Norfolk Area EAA, Duwamish/Diagonal EAA and adjacent area (conducted by King County)	Surface sediment time trends; Section 7, Appendix F, and Appendix J	Project database
Puget Sound urban water body data (from EIM)	Surface sediment time trends described in Appendix J	Additional Excel data files
DMMP characterization of dioxins/furans outside of the LDW (provided by DMMP)	Surface sediment time trends described in Appendix J	Additional Excel data files
DMMP characterization in the LDW Upper Turning Basin (1990 – 2009; conducted by USACE)	BCM input parameters discussed in Section 5 and Appendix C, Part 3	Additional Excel data files
Data trumped by more recent co-located data in Duwamish/Diagonal or Norfolk (collected by King County)	Used in the FS baseline dataset for mapping surface sediment exceedances in Section 2, unless the data are located in Duwamish/Diagonal cap or ENR areas or in the Norfolk removal and backfill area; in these cases, the preremedy data are used in the FS baseline dataset[a]	Project database
Tissue data (compiled by LDWG)	Used in risk estimates; discussed in Appendix B	Project database
Seep and porewater data (collected by LDWG and others)	Section 4 and Appendix N	Project database

Notes:

a. The FS project database and additional Excel data files are available on http://www.ldwg.org in one zip file. The zip file also contains an index describing each dataset and its file location. Each table and dataset included in the FS project database has undergone rigorous quality control checks, as documented in technical memoranda (the most recent being Addendum 3; Windward 2012, review in progress). Other datasets are also included in the project files, but have not been formatted into the standardized set of fields included in the project database. These files are provided in Microsoft Excel format (often maintained in the same format in which they were received) and may not have undergone the same level of quality control checks as the database files.

BCM = bed composition model; CSO = combined sewer overflow; DMMP = Dredged Material Management Program; EAA = early action area; Ecology = Washington State Department of Ecology; EIM = environmental information management system managed by Washington State Department of Ecology; ENR = enhanced natural recovery; FS = feasibility study; LDW = Lower Duwamish Waterway; LDWG = Lower Duwamish Waterway Group; PCB = polychlorinated biphenyl; USACE = U.S. Army Corps of Engineers

Table 2-6 Statistical Summaries for Contaminants of Concern for Ecological Health

Contaminant	Summary Statistics for Surface Sediments			Total Number of Surface Sediment Samples in FS Baseline Dataset						Benthic Invertebrate Risk Driver[d]
	Minimum Detect	Maximum Detect	Mean[a]	Total	With Detected Values	Detection Frequency	>SQS, ≤CSL, detected[b]	>CSL, detected[b]	>SQS or CSL, detected[b,c]	
Metals and TBT (mg/kg dw)										
Arsenic	1.2	1,100	17	916	857	94%	5	9	14	yes
Cadmium	0.03	120	1.0	894	632	71%	2	12	14	yes
Chromium	4.80	1,680	42	906	906	100%	1	10	11	yes
Copper	5.0	12,000	106	908	908	100%	0	13	13	yes
Lead	2.0	23,000	139	908	908	100%	2	23	25	yes
Mercury	0.015	247	0.53	927	813	88%	20	30	50	yes
Nickel	5.0	910	28	836	836	100%	n/a	n/a	n/a	no
Silver	0.018	270	1.0	875	537	61%	0	10	10	yes
Vanadium	15	150	59	589	589	100%	n/a	n/a	n/a	no
Zinc	16	9,700	194	905	905	100%	26	19	45	yes
Tributyltin as ion	0.28	3,000	90	189	178	94%	n/a	n/a	n/a	no
PAHs (μg/kg dw)										
2-Methylnaphthalene	0.38	3,300	42	882	169	19%	1	4	5	yes
Acenaphthene	1.0	5,200	65	891	352	40%	16	4	20	yes
Anthracene	1.3	10,000	134	891	647	73%	2	0	2	yes
Benzo(a)anthracene	7.3	8,400	322	891	821	92%	10	6	16	yes
Benzo(a)pyrene	6.5	7,900	309	886	819	92%	7	5	12	yes
Benzo(g,h,i)perylene	6.1	3,800	165	891	763	86%	10	12	22	yes
Total benzofluoranthenes	6.6	17,000	732	885	829	94%	6	6	12	yes
Chrysene	12	7,700	474	891	846	95%	29	3	32	yes
Dibenzo(a,h)anthrecene	1.6	1,500	63	891	498	56%	18	6	24	yes
Dibenzofuran	1.0	4,200	54	889	276	31%	7	3	10	yes
Fluoranthene	18	24,000	889	891	868	97%	35	12	47	yes
Fluorene	0.68	6,800	78	891	431	48%	11	3	14	yes
Indeno(1,2,3-cd)pyrene	6.4	4,300	180	891	801	90%	16	13	29	yes

Table 2-6 Statistical Summaries for Contaminants of Concern for Ecological Health (continued)

Contaminant	Summary Statistics for Surface Sediments: Minimum Detect	Maximum Detect	Mean[a]	Total Number of Surface Sediment Samples in FS Baseline Dataset: Total	With Detected Values	Detection Frequency	>SQS, ≤CSL, detected[b]	>CSL, detected[b]	>SQS or CSL, detected[b,c]	Benthic Invertebrate Risk Driver[d]
Naphthalene	3.0	5,300	49	882	183	21%	0	2	2	yes
Phenanthrene	7.1	28,000	429	891	832	93%	27	3	30	yes
Pyrene	19	16,000	723	891	860	97%	2	6	8	yes
Total HPAH	23	85,000	3,809	891	873	98%	25	6	31	yes
Total LPAH	9.1	44,000	696	891	835	94%	4	3	7	yes
Phthalates (µg/kg dw)										
Bis(2-ethylhexyl) phthalate	5.4	17,000	590	886	704	79%	46	58	104	yes
Butyl benzyl phthalate	2.0	7,100	87	878	478	54%	80	10	90	yes
Dimethyl phthalate	2.0	440	25	878	186	21%	0	2	2	yes
Chlorobenzenes (µg/kg dw)										
1,2,4-Trichlorobenzene	1.6	940	19	871	6	1%	0	2	2	yes
1,2-Dichlorobenzene	1.3	670	19	871	19	2%	0	4	4	yes
1,4-Dichlorobenzene	1.5	1,600	23	871	50	6%	0	4	4	yes
Hexachlorobenzene	0.4	95	17	874	46	5%	4	2	6	yes
Other SVOCs and COCs (µg/kg dw)										
2,4-Dimethylphenol	6.1	290	44	869	29	3%	0	25	25	yes
4-Methylphenol	4.8	4,600	44	883	116	13%	0	4	4	yes
Benzoic acid	54	4,500	238	876	111	13%	0	9	9	yes
Benzyl alcohol	8.2	670	49	867	30	3%	9	7	16	yes
Carbazole	3.2	4,200	82	775	425	55%	n/a	n/a	n/a	no
n-Nitrosodiphenylamine	6.5	230	27	871	24	3%	0	2	2	yes
Pentachlorophenol	14	14,000	122	840	30	4%	1	1	2	yes
Phenol	10	2,800	91	886	282	32%	19	6	25	yes

Table 2-6 Statistical Summaries for Contaminants of Concern for Ecological Health (continued)

Contaminant	Summary Statistics for Surface Sediments			Total Number of Surface Sediment Samples in FS Baseline Dataset						Benthic Invertebrate Risk Driver[d]
	Minimum Detect	Maximum Detect	Mean[a]	Total	With Detected Values	Detection Frequency	>SQS, ≤CSL, detected[b]	>CSL, detected[b]	>SQS or CSL, detected[b,c]	
Pesticides (µg/kg dw)										
Total DDTs	0.72	77,000	462	216	87	40%	n/a	n/a	n/a	no
Total chlordanes	0.20	230	268	216	28	13%	n/a	n/a	n/a	no
Aldrin	0.01	1.6	27	216	4	2%	n/a	n/a	n/a	no
Dieldrin	0.10	280	29	218	8	4%	n/a	n/a	n/a	no
alpha-BHC	0.14	1.8	1.1	207	3	1%	n/a	n/a	n/a	no
beta-BHC	0.09	13	1.2	207	4	2%	n/a	n/a	n/a	no
gamma-BHC	0.05	8.6	27	216	12	6%	n/a	n/a	n/a	no
Heptachlor	0.12	5.2	27	216	6	3%	n/a	n/a	n/a	no
Heptachlor epoxide	0.47	4.9	2.8	207	4	2%	n/a	n/a	n/a	no
Toxaphene	340	6,300	111	205	2	1%	n/a	n/a	n/a	no
Total PCBs (µg/kg dw)										
Total PCBs[e]	2.2	223,000	1,136	1,390	1,309	94%	336	179	515	yes

Source: Feasibility study baseline surface sediment database queries, RM 0 to 5.0.

Notes:

a. Calculated mean concentration is the average of detected concentrations and one-half the reporting limit for non-detected results.
b. For non-polar organic compounds, comparisons to SQS and CSL were made using organic carbon-normalized concentrations. If total organic carbon in the sample was <0.5% or >4%, dry weight concentrations were compared to the LAET and 2LAET.
c. Sum of samples with SQS (but less than CSL) exceedances and samples with CSL exceedances.
d. Contaminants identified as risk drivers for the benthic invertebrate community (RAO 3) are those with one or more surface sediment samples with exceedances of the SQS. Three additional contaminants (total DDTs, total chlordanes, and nickel) that do not have SMS criteria were also identified as COCs for the benthic community.
e. Total PCB statistics and counts were generated with two outliers (2,900,000 and 230,000 µg/kg dw in Trotsky inlet) excluded. Sample count with outliers included is 1,395.

2LAET = second lowest apparent effects threshold; BHC = benzene hexachloride; COCs = contaminants of concern ; CSL= cleanup screening level; dw = dry weight; DDT = dichlorodiphenyl-trichloroethane; FS = feasibility study; HPAH = high molecular weight polycyclic aromatic hydrocarbon; kg = kilograms; LAET= lowest apparent effects threshold; LPAH = low molecular weight polycyclic aromatic hydrocarbon; µg = micrograms; mg = milligrams; n/a = not applicable; nc = not calculated; RAO = remedial action objective; SQS = sediment quality standard; SVOC = semivolatile organic compound; TBT = tributyltin; TEQ = toxic equivalent; VOC = volatile organic compound

Table 2-7 Upland Engineering Studies with In-Water Geotechnical Data and Borings

River Mile	Study/ Report Year	Study Name	Author	Area	Data Type(s)	No. of In-water Borings	Purpose of Study
0.0	1973	West Seattle Freeway Seismic Studies	SPU, Shannon and Wilson	RM 0.0-0.2	Boring logs, SPT	2	pile load test and seismic studies
0.0	1974	West Seattle Freeway Pile Load Test Program				1	
0.0	1968	Soils and Foundation Report Duwamish East Waterway Fill Industrial Terminal No. 2	Shannon and Wilson	RM 0.0E	Boring logs, SPT, grain size analysis, triaxial compression test, mohr strength envelope, consolidation test, liquid limit, subbottom profiling, bottom contour map, isopach of mud thickness	5	Fill area for terminal
0.1	1968	Lone Star Cement Site Plan	Shannon and Wilson, Soil Mechanics and Foundation Engineers	RM 0.0-0.2	Boring logs, cross sections, water content, grain size analysis	2	Proposed clinker storage silo and mill bldg construction
0.2	1993	Measured Sections and Drillhole Descriptions, Geologic Map of Surficial Deposits in the Seattle 30'x60' Quadrangle	Yount et al.	RM 0.0-0.2	Boring logs	2	Major unit mapping
0.4	1970	South Substation to Delridge Substation	Seattle Eng. Dept.	Kellogg Island	Several upland borings, no report text	1	no report text, could not determine purpose
0.4-0.5	1988	Report of Geotechnical Investigation, Port of Seattle, Terminal 108 Site, for LaFarge Canada	Dames and Moore	Kellogg Island	Boring logs, blow counts, shear test, grain size analysis	2	Proposed cement silos
0.4	1972	Diagonal	Seattle Eng. Dept.	Kellogg Island	Several upland borings, no report text	1	no report text, unknown purpose
0.4-0.5	1966-1971	Diagonal Yard	SPU	Kellogg Island	Boring logs, test pit logs, sludge pond probes	5	could not determine from materials

Table 2-7 Upland Engineering Studies with In-Water Geotechnical Data and Borings (continued)

River Mile	Study/ Report Year	Study Name	Author	Area	Data Type(s)	No. of In-water Borings	Purpose of Study
0.4-2.1	1965	SW Marginal Way between SW Spokane & S Kenyon St (GeoNW name, logs are by bridge, no report name given)	Seattle Eng. Dept.	First Ave Bridge	Only boring logs, no report text	3	no report text, unknown purpose
0.5-0.8	1968	Report of Preliminary Soils Investigation, Proposed Kellogg Island Development	Dames and Moore	Kellogg Island	Boring logs, cross sections, triaxial test (moisture content, dry density, cell pressure, deviator stress), direct shear test, consolidation test, moment coefficient	4	Development on Kellogg Island
0.6-1.0	1970	Soils and Foundation Investigation for Proposed Terminal 107 (Kellogg Island)	Twelker & Assoc.	Kellogg Island	Cross sections (poor scan quality)	6	Development of Terminal 107
1.4	1967	Foundation Investigation for Waterfront Development at 5900 West Marginal, Kaiser Cement and Gypsum	Twelker & Assoc.	Glacier Northwest, Inc.	One in-water boring to >90 ft below mudline, cross sections, SPT	1	Pier construction investigation
1.4-1.5	1979	Subsurface Exploration and Geotechnical Engineering Study for Proposed Additions to the Seattle Finish Grinding Facility	Hart Crowser	Glacier Northwest, Inc.	General description of subsurface conditions, cone penetration resistance, friction ratio, boring logs, SPT, grain size analysis, plasticity index vs. liquid limit, stress vs. strain	3	Proposed clinker storage silo, finish mill, feed bins, truck-rail unloading hopper, ship unloading facility, clinker conveyor system
2.0	1993	Geotechnical Report First Ave S Bridge Utilidor Relocate	Seattle Eng. Dept.; Shannon and Wilson	First Ave Bridge	Boring logs, cross section, SPT	2	Utilidor relocation in conjunction with seismic retrofitting of existing bascule bridge and construction of parallel bridge to the west
2.0-2.1	1992	Geotechnical Report, Preliminary Explorations and Engineering Studies, First Avenue South Bridge Over Duwamish	Shannon and Wilson	First Ave Bridge	Boring logs, SPT, cross sections, piezocone probe data, grain size analysis, plasticity index	2	Bridge construction

Table 2-7 Upland Engineering Studies with In-Water Geotechnical Data and Borings (continued)

River Mile	Study/ Report Year	Study Name	Author	Area	Data Type(s)	No. of In-water Borings	Purpose of Study
2.0-2.1	1972	Kenyon to First Avenue, Proposed 72" Utilities Tunnel	WSDOT	First Ave Bridge	Boring logs, deep cross section	5	Utilities tunnel construction
2.2	1961	Northwest Cooperage Foundation Exploration	Twelker & Assoc.	RM 2.2W	Boring logs	4	Foundation exploration
4.7	1988	Geotechnical Design Report, North Oxbow Bridge, Boeing Developmental Center	Rittenhouse-Zeman & Assoc.	RM 4.7	Written text (general riverbank condition), liquefaction test, SPT, logs, cross section, bathymetry	3	Bridge construction

Notes:

1. Logs and portions of reports from GeoNW website. http://geomapnw.ess.washington.edu/index.php

RM = river mile; SPT= standard penetrometer test; SPU= Seattle Public Utilities; WSDOT= Washington State Department of Transportation

Table 2-8 Trace to Abundant Debris and/or Sheen in 2006 RI Sediment Cores

2006 Core	Debris and/or Sheen Description	Debris	Sheen
SC-2	Trace hydrocarbon-like sheen from 1.2 to 4.1 ft; rock flour (100%) from 4.3 to 10.5 ft.	X	X
SC-4	Trace hydrocarbon-like sheen from 1.4 to 2.5 ft.		X
SC-11	Red chips, 1 piece of plastic & leather, 2 glass shards, and cedar chips from 0 to 0.9 ft; dark grey gravel from 4.1 to 4.9 ft.	X	
SC-13	Layer of shredded wood with fibrous peat-like material from 5.5 to 6.0 ft.	X	
SC-14	1/16" sheen florets from 0.3 to 3.7 ft and from 4.1 to 8.7 ft.		X
SC-15	Trace hydrocarbon florets and blebs up to 1/2" long from 1.2 to 2.0 ft.		X
SC-16	Trace 1/16" sheen florets from 1.3 to 2.0 ft; garbage bag at 0.5 ft; trace odor and sheen from 4.0 to 7.4 ft.	X	X
SC-17	Layers of wood and abundant debris from 0.9 to 12.3 ft; rainbow sheen on core side walls from 2.0 to 6.0 ft.	X	X
SC-18	Glass shard 0.2 ft long at 0.8 ft; subangular rock 0.3 ft long at 1.5 ft; 1" layer of wood fragments up to 3/4" long at 8.6 ft.	X	
SC-19	Rainbow sheen florets up to 1/4" long and wood fragments up to 1" long at 1.9 ft; rainbow sheen on side walls of core from 0.8 to 7.0 ft.	X	X
SC-20	Trace hydrocarbon-like sheen from 0.1 to 4.7 ft.		X
SC-21	Scattered wood layers up to 0.1 ft thick with orange-brown shredded wood from 10.1 to 12.7 ft.	X	
SC-22	Trace debris from 0 to 1.3 ft; moderate creosote-like sheen and hydrocarbon staining from 1.3 to 2.0 ft; abundant wood fragments at 2 ft; scattered debris from 2.0 to 9.3 ft w/ 3" brick fragment at 3.6 ft.	X	X
SC-23	Trace debris from 0 to 0.5 ft.	X	
SC-25	Layers of 4" long shredded wood fragments from 0.4 to 5.5 ft; glass shard at 1.8 ft.	X	
SC-26	Large gravels with hydrocarbon-like sheen and scattered debris from 7.9 to 9.1 ft; scattered debris and florets from 9.1 to 13.1 ft.	X	X
SC-28	Black, loose sand blast grit and scattered debris from 5.8 to 12.8 ft; grit left metallic sheen on core side walls from 4.0 to 11.3 ft.	X	X
SC-32	Trace wood fragments up to 4" long with slight hydrocarbon-like sheen florets from 3.5 to 5.1 ft.	X	X
SC-33	Trace black sheen from 0.3 to 1.8 ft; trace debris from 2.8 to 10.0 ft.	X	X
SC-34	Trace debris from 0.7 to 11.3 ft.	X	
SC-37	Trace debris and wood fragments from 0.3 to 2.6 ft; metallic and hydrocarbon sheens up to 1" long from 3.2 to 6.3 ft.	X	X
SC-38	Moderate to heavy hydrocarbon-like sheen in sand seams from 2.5 to 3.8 ft; scattered wood debris from 0.0 to 2.5 ft and from 3.8 to 5.6 ft.	X	X
SC-39	Trace debris from 0.4 to 2.5 ft; wood fragments up to 1/2" long from 3.9 to 7.5 ft; trace wood fragments up to 7" long from 7.5 to 10.3 ft.	X	
SC-40	Trace debris from 1.7 to 13 ft.	X	
SC-41	Wood fragments up to 4" long and scattered 1/2" sheen florets from 2.2 to 6.9 ft.	X	X
SC-42	Shredded wood from 0.2 to 4.0 ft; 3" layer of silt with black sheen at 3.4 ft; black sheen from 8.0 to 11.8 ft; piece of plastic at 11.0 ft.	X	X
SC-44	1" glass shards and little debris from 2.4 to 4.8 ft; 6" subangular conglomerate at 3.3 ft.	X	
SC-45	Scattered rainbow sheen florets from 2.2 to 4.1; 2" long concrete piece at 4.8 ft; trace debris from 5.2 to 7.5 ft; drive 2 close to shore had heavy sheen in gravel layer at 4 ft and free phase blebs.	X	X
SC-46	Trace debris from 2.3 to 7.9 ft; metallic sheen at 2.8 ft; rainbow sheen at 3.6 ft.	X	X
SC-47	Up to 1" long trace debris from 0.7 to 2.9 ft.	X	
SC-50	2" layer of black gravel at 1.3 ft; subangular gravel at wood fragments and gravel from 1.3 to 4.2 ft.	X	
SC-51	Scattered hydrocarbon-like sheen florets and streaks up to 1" long from 0.0 to 0.4 ft; scattered debris including brick fragment from 1.7 to 5.0 ft.	X	X
SC-53	Trace possible anthropogenic fibers at 5.2 ft.	X	
SC-56	Trace hydrocarbon-like sheen above 1" silt seam at 1 ft; abundant wood fragments 3.8 to 7.0 ft.	X	X

Notes:

Significant (>50% by volume) anthropogenic material / debris or abundant large gravels.

ft = feet; RI = remedial investigation

Table 2-9 LDW Navigation Channel Maintenance Dredging (1986 to 2010)

River Mile	Dredge Date Start	Dredge Date End	Volume Dredged (cy)	Paydepth / Overdepth (ft MLLW)	Survey Dates Pre-Dredge	Survey Dates Post-Dredge	Side Slope
4.19 to 4.38	03/11/86	03/29/86	33,637	-16 / -18	—	—	—
4.38 to 4.65	06/19/86	7/15/1986	126,470	-16 / -18	—	—	—
4.38 to 4.65	02/24/87	03/24/87	80,160	-18 / -20	—	—	—
3.97 to 4.65	02/28/90	03/30/90	127,619	-17	—	—	2:1
3.34 to 4.65	02/06/92	03/21/92	199,361	-15 / -17	—	—	3:1
4.33 to 4.65	03/07/94	03/28/94	57,243	-15 / -17	1/21/1994	4/6/1994	2:1
4.02 to 4.48	02/22/96	03/30/96	90,057	-15 / -16	2/14/1996	4/2/1996	2:1
4.26 to 4.65	02/05/97	03/31/97	89,011	-15 / -16	1/23/1997	—	2:1
3.43 to 4.65	03/11/99	06/29/99	165,116	-15 / -16	3/5/1999	7/8/1999	2:1
4.27 to 4.65	01/14/02	02/09/02	96,523	-15 / -16	1/3/2002	2/20/2002	2:1
4.33 to 4.65	01/15/04	02/16/04	75,770	-15 / -17	12/17/2003	2/14/2004	3:1
4.27 to 4.65	12/11/07	01/10/08	140,608	-15 / -16	—	—	—
4.18 to 4.65	02/19/10	3/30/10	60,371	-15 / -17	Oct. 2008 and Aug. 2009	5/24/2010	—

Sources:

USACE Dredge Summary and Analysis Reports (USACE 2005), *2009 Suitability Determination* (DMMP 2009a), and *2010 Payment Summary* (USACE 2010a).

Notes:

1. See Figure 2-27 for locations of dredging events.

cy = cubic yards; ft = feet; MLLW = mean lower low water; RM = river mile; USACE = U.S. Army Corps of Engineers; — = unknown or no survey conducted

Table 2-10 Overwater Structures, Moorages, and Other Physical Structures

Structure	River Mile	River Side	General Type[a]	Use	Recorded Water Depth (ft MLLW)	Authorized Navigation Channel Depth Adjacent to Berthing Area (ft MLLW)	Breasting Distance (ft)[b]
Harbor Island Marina	0	W	Marina	Recreational and commercial vessel moorage	—	-30	—
Glacier Northwest South Wharf (Terminal 103)	0	W	Timber bulkhead with solid fill fronted by timber pile wharf, steel transfer bridge	Receipt of sand, gravel, and stone	-10	-30	240 (face)
Ash Grove Cement North Wharf	0.1	E	Timber pile, concrete decked wharf	Shipment of bulk cement	-25	-30	600 with dolphins
Ash Grove Cement South Pier	0.2	E	Steel pile, timber decked pier	Receipt of coal, gypsum, gravel, and rock lime	-25	-30	225
Berth No. 1 Wharf (Terminal 105)	0.3	W	Steel sheet pile bulkhead, asphalt-surfaced solid fill	Receipt of scrap metal	-40	-30	660 (face)
Berth No. 2 Wharf (Terminal 105)	0.4	W	Timber bulkhead with solid fill fronted by timber pile timber-decked wharf	Mooring vessels	-15	-30	450
Tilbury Cement East Marginal Terminal Wharf	1.0, adjacent to Manson wharf	E	Concrete pile, concrete-decked wharf	Receipt of bulk cement and gravel	-17	-30	300
U.S. Government Wharf	1.0, north side Slip 1	E	Timber bulkhead, solid fill, concrete-decked extensions	Mooring vessels / previously used for containerized shipments	-26 (face); 0 to -26 (west side); -15 to -26 (head of slip)	-30	642 (face); 165 (head of slip)
Manson Construction Wharf	South side Slip 1 and 1.0, just south of Slip 1	E	Concrete bulkhead, solid fill, concrete-decked extensions	Mooring floating equipment and dredge, moving supplies to and from barges	-12 to -20 (face); -20 (west side dolphins)	-30	550 (face); 300 (west side dolphins)
Lafarge Corporation Raw Materials Wharf	1.0 to 1.25	W	Steel sheet pile, cellular bulkhead	Receipt of limestone, shale, coal, and slag	-30	-30	1,100
Lafarge Corporation Cement Wharf	1.0, south of Kellogg Island	W	Three timber piles, timber decked offshore wharves, connected by timber catwalks	Receipt and shipment of bulk cement	-32	-30	645 with dolphins (center wharf)
J.A. Jack and Sons Wharf	1.2	E	Offshore row of 6 timber dolphins, catwalk	Receipt of limestone	-20	-30	250
Alaska Marine Lines Dock No. 1	1.25	W	Concrete, timber, steel piles, concrete-decked wharf	Containerized general cargo	-20 to -25	-30	325
Duwamish Shipyard Graving Dock Wharf	1.3	W	Wharf: concrete and timber pile bulkhead; historical graving dock (subsequently filled in): steel sheet pile retaining walls, concrete floor, steel gate	Mooring vessels for repair / previous shipment of concrete fabrications and mooring vessels	-20 (pier)	-30	400 by 138 (graving dock); 60 (pier)
General Construction Mooring	1.4	E	Offshore row of 11 timber dolphins	Mooring floating equipment and barges	-17	-30	800
Duwamish Shipyard Wharf	1.4	W	Irregularly shaped timber pile, timber-decked offshore wharf, timber floats connect dolphins, dredged basin at rear of dolphins on south side	Mooring vessels for repair, mooring dry docks	-25 (face); -20 to -25 (basin)	-30	500 with dolphins
Glacier Northwest West Terminal Wharf	1.5	W	Concrete pile, concrete-decked offshore wharf with concrete-decked approach	Receipt of bulk cement	-34 to -40	-30	467
James Hardie Gypsum Wharf	1.6	E	Steel and timber pile, timber-decked wharf extending from steel sheet pile bulkhead with solid fill	Receipt of bulk cement and gypsum rock	-30 to -31 (face); -6 to -32 (south face); -11 to -32 (north side)	-30	400 with dolphins
Northland Services (Terminal 115)	1.5 to 1.9	W	Berth 1: Piers A and C center timber pier, Pier B ramp support structure and A-Frame and upgrade fendering systems.	Barge loading and unloading	-15	-30	Proposed modification, not constructed yet
International Terminal North Wharf (Terminal 115)	1.6 to 1.8	W	Concrete piles support 103-ft wide concrete apron over water. Riprap slope and sheet pile bulkhead on inner land side.	Containerized general cargo and heavy lift items; receipt of steel products; receipt and shipment of forest products	-40	-30	1,200
Glacier Northwest Slip 2 Wharf	1.7, north side Slip 2	E	Timber pile, timber-decked offshore wharf, adjustable transfer bridge	Receipt of sand and gravel	-16 to -17	-30	325 with dolphins
South Wharf (Terminal 115)	1.8	W	Three timber pile, timber-decked loading platforms fronting concrete bulkhead	Containerized general cargo and heavy lift items	-14	-30	490
Filter Engineering Wharf	1.8, south side Slip 2	E	Steel/timber pile, timber-covered, concrete-decked offshore wharf	Moving construction equipment to and from barges	-12	-30	130 with dolphins
Seafreeze Limited Partnership Wharf (Terminal 115)	1.9	W	Concrete pile, concrete-decked offshore wharf with concrete approach and steel catwalks	Receipt of fish and seafood	-20	-30	100
Alaska Marine Lines Dock No. 2	2.1	W	Concrete pile, concrete-decked wharf	Containerized general cargo; mooring vessels	-15	-20	400 with dolphins
Northland Services Fox Avenue Terminal Wharf	2.1 to 2.2, south of and on south side of Slip 3	E	Concrete pile, concrete-decked wharf extending from sheet pile bulkhead	Conventional and containerized general cargo	-18	-20	475 (slip side); 500 (river side)
Silver Bay Logging South River Street Wharf	2.1, north side Slip 3	E	Timber pile, timber-decked wharf extending from timber bulkhead	Mooring barges	-15	-20	215
Boyer Alaska Barge Line Mooring	2.3	W	Two offshore breasting dolphins fronting natural bank	Mooring floating equipment	-10	-20	175
MC Halverson Marina	2.3	W	Marina	Residential vessel moorage	—	-20	—
Seattle Iron & Metals North Wharf	2.4	E	Timber pile, asphalt-surfaced, timber-decked wharf extending from steel sheet pile bulkhead	Receipt of scrap metal by barge	-12 to -13	-20	125

Table 2-10 Overwater Structures, Moorages, and Other Physical Structures (continued)

Structure	River Mile	River Side	General Type[a]	Use	Recorded Water Depth (ft MLLW)	Authorized Navigation Channel Depth Adjacent to Berthing Area (ft MLLW)	Breasting Distance (ft)[b]
Boyer Alaska Barge Line Seattle Wharf	2.4	W	Timber bulkhead, asphalt surfaced solid fill with timber pile, timber-decked extension	Containerized general cargo, lumber, mooring tugs and barges	-10	-20	300 with dolphins
Seattle Iron & Metals South Wharf	2.5	E	Timber pile, asphalt-surfaced, timber-decked wharf extending from steel sheet pile bulkhead	Receipt of scrap metal by barge	-16	-20	300
Alaska Washington Building Materials Co. Wharf	2.5	W	Irregularly shaped concrete bulkhead with solid fill, fronted by three timber dolphins	Not used / previously receipt of sand and gravel	-2 to -12	-20	100+25
Hurlen Construction Mooring	2.65	W	Natural bank with shore moorings	Mooring floating equipment, moving supplies to and from barges	-8 to -20	-20	200
Hurlen Construction Wharf	2.7	W	Timber pile, timber-decked wharf	Mooring floating equipment, moving supplies to and from barges	-20	-20	280 with dolphins
Northland Services 8th Avenue Terminal Wharf	2.8, north side Slip 4	E	Concrete pile, concrete-decked wharf	Conventional and containerized general cargo	-13 to -15	-15	165, 390, 480 along face
Silver Bay Logging 8th Avenue Wharf	2.9	W	Steel pile, steel beam, timber and steel grating decked wharf	Receipt of lumber by barge	-18	-15	400 with dolphins
Boeing Plant 2	3.1 - 3.5	E	Two buildings	Historical overwater buildings	n/a	-15	n/a; not used for moorage
South Park Marina	3.4	W	Marina	Moorage of commercial and recreational vessels	-8	-15	~900
McElroy George and Assoc.Inc.	4.0	W	Marina	Vessel moorage	—	-15	—
Northwest Container Services	4.1	E	Dolphins for mooring	Moorage of barges	—	-15	—
Duwamish Yacht Club	4.1	W	Marina	Moorage of recreational vessels	-8	-15	620 x 320
Delta Marine Industries Wharf	4.2	W	Offshore row of permanently moored floats, approach from concrete-paneled bulkhead	Mooring vessels for outfitting and repair; fiberglass vessels manufactured on site	-10	-15	284 (face); 160 (rear); 230 (bulkhead)
The Boeing Company Seattle Wharf	4.3, Slip 6	E	Six concrete pile, concrete-decked, asphalt-surfaced loading platforms	Mooring barges; previously not used	-18	-15	650 total
Various structures	0.15 to 0.2	W	Abandoned pile fields associated with historical vessel launch facilities. At least 500 abandoned single piles appear to be in this area.		n/a		
	0.43 to 0.48	Both	Submerged sewer line crossings				
	0.38 to 0.47	Both	Overhead power cable crossings. Authorized vertical clearances are in excess of 90 ft at each installation.				
	1.95						
	3.6						
	4.4						
	0.6 to 0.9	W	Pile group along Kellogg Island's west side				
			Pile and dolphin groups along Kellogg Island's east side				
	1.8 to 2.1	Both	Submerged cable and pipeline area				
	2.1 to 2.2	Both	First Avenue bascule bridges. The west and east bridges have 145-ft horizontal clearance closed and 120-ft horizontal clearance open. Vertical clearance is 22 ft (39 ft at center) when closed.				
	2.85 to 3.0	Both	Submerged cable area				
	3.15 to 3.4	Both					
	3.3 to 3.4	Both	South Park bascule bridge. Also known as the 14th/16th Ave South Bridge, this bridge had a 92-ft horizontal clearance, and 21-ft vertical clearance (34 ft at center) (NOAA 2008). The former bridge was demolished and a new bridge is under construction just downstream of the former bridge location, with completion scheduled for fall of 2013.				
	Throughout		Abandoned and working piles and dolphins throughout the LDW				

Source: *Port Series No. 36* (Revised 2002) – Port of Seattle, Washington; U.S. Army Corps of Engineers Institute for Water Resources; NOAA 2008. NOAA Chart 18450, Edition 18, 2/1/2004, Updated 2/2/2008; additional sources used include: field surveys, 2002 aerial photograph, DMMO memos, and Remedial Investigation (Windward 2010).

Notes:

1. See Figure 2-28 for locations of berthing areas.

a. Structure type is general. See Port Series for additional details.

b. Breasting distance is the length in ft of the portion of the structure to which a vessel berths.

DMMO = Dredged Material Management Office; E = east; ft = feet; LDW = Lower Duwamish Waterway; MLLW = mean lower low water; NOAA = National Oceanic and Atmospheric Association; RM = river mile; W = west

Table 2-11 History of Private Maintenance Dredging Events in the LDW (1980 to 2008)

Project/Site Name	River Mile	River Side	Dredge Year	Volume Dredged (cy)	Pay Depth / Overdepth (ft MLLW)	Purpose	Suitable for Open-water Disposal?	Source Type		Permit
								Pre-dredge Documents[a]	Post-dredge Confirmation[b]	
Terminal 103	0.46 to 0.56	W	2005	1,350	-14/-15	Navigation	—	x		—
Lone Star/Current Ash Grove Location	0.2	E	began in March 1980 (with add'l in 1983)	5,000 allowed; 4,000 by 1981	-35	Maintenance dredging event for clinker ship unloading	Dredged material used as raw material in cement kiln	x	x	071-0YB-1-005983: issued in 1980
Lafarge	0.98	W	2009	1,000	—	Maintenance dredging event	—			—
Lehigh Northwest	1.0 to 1.1	E	2004	9,000	-20 / -21	Maintenance dredging event	DMMUs 1 and 3 (6,000 cy) suitable DMMU 2 (3,000 cy) not suitable	x		—
Duwamish Shipyard	1.39 to 1.42	W	Last event in 1982	—	-25 to -15	Maintain depth of basin behind dolphins	—		x	—
Glacier Northwest, Inc.	1.42 to 1.54	W	2005	9,920	-34 (pay depth authorized to -35)	Maintenance dredging and thin-layer cap	DMMU 1 (3,250 cy) suitable DMMUs 2 and 3 (6,670 cy) not suitable (capped)	x	x	92-2-00452: 3,900 cy in 1993; 4,000 cy in 1997 (in original permit, but removed from revision, so assume did not occur, also not mentioned in 2005 document); can go up to 10,000 additional cy to 2003 (with permit revision); permit allows maintenance to -35' in whole area, but shows only small area dredged in 1993.
Lone Star Northwest-West Terminal	1.43 to 1.52	W	1993	3,900	-35 / -36	Maintenance dredging event	Yes	x	x	
Lone Star Northwest-West Terminal	1.43 to 1.52	W	1986	—	—	Maintenance dredging event	No, taken to upland site		x, mentioned in reports for later events	
James Hardie Gypsum	1.56 to 1.75	E	1999	10,000 permitted	-31	Maintenance dredging event	4,540 of 7,042 cy suitable	x		Same permit as 95-2-00837 below, issued in 1996, authorized 10 years of dredging, 1999 is first dredging event since 1996
Lone Star-Hardie / Kaiser	1.55 to 1.75	E	1996	18,000	-30 / -31	Maintenance dredging event & dock upgrade	DMMUs 1-3 (9,375 cy) not suitable DMMUs 4 and 5 (8,625 cy) suitable	x	x	95-2-00837: 95-4-00837 revision (August 1996) for 3 dolphins, 28 piles, and walkway extension; annually dredge additional 9,000 cy for upland disposal in upstream portion of footprint (DMMUs 1-3); shows previous dredge at downstream end (DMMUs 4-5 for in-water disposal).
Lone Star-Hardie / Kaiser	1.6 to Slip 2	E	1986 (unconfirmed)	26,000	-30 (dock), -16 (Slip 2)	Ramp, conveyer, dolphin construction	—		x	071-OYB-2-009121: area in front of dock and Slip 2 to construct ramps, conveyers, dolphins; no confirmation this occurred. PCB concentrations too high for open-water disposal. Dredging footprint modified. No map found of dredging footprint.
Glacier Ready Mix	Slip 2	E	2001	4,900	-15 /-16	Maintenance dredging event	Yes			—
Lone Star Northwest Slip 2	Slip 2	E	1990	1,600	-14	Maintenance dredging event	Yes	x	x	071-OYB-2-013065: 1,600 cy first year (1990) then 1,000 cy each year for 9 years for a max of 10,600 cy, 1994 modification to 3,000 cy; HPA #B2-13065-03: issued in 1990 and revised in 1994 to retrieve spilled aggregate; 1994 Dept. of Ecology water quality modification #DE 94ER-008.
		E	1991	1,100	Not specified	Maintenance dredging event	No, taken to upland site	x	x	
		E	1994	3,000	-14	Maintenance dredging event	No, taken to upland site	x	x	
	Adjacent to Slip 2	E	1994	2,000	Not specified	Retrieve spilled aggregate	Dredged material used as raw aggregate	x		
Terminal 115	1.78 - 1.95 (2 areas)	W	1993	3,000	-15	Maintenance dredging event, dolphin construction	Yes	x		92-2-01363
	1.5 - 1.9	W	2009	3,000	-15 / -17	Reconstruction of Berth 1 for Northland Services lease	No	x		SEPA DNS; creosote timber piles will be removed; Pier B will be demolished
Boyer	2.45 to 2.47	W	2004	—	—	Dock replacement	Yes, not confirmed by DMMO memo	x		Nationwide permit #3 200200607

Table 2-11 History of Private Maintenance Dredging Events in the LDW (1980 to 2008) (continued)

Project/Site Name	River Mile	River Side	Dredge Year	Volume Dredged (cy)	Pay Depth / Overdepth (ft MLLW)	Purpose	Suitable for Open-water Disposal?	Source Type: Pre-dredge Documents[a]	Source Type: Post-dredge Confirmation[b]	Permit
Boyer	2.39 to 2.49	W	1998	8,000	-10	Maintenance dredging event	DMMUs 5, 6 suitable DMMUs 1-4 at Hurlen site	x	x	98-2-00477: permit allows dredging to -8 ft MLLW; but 1998 dredging extended to -10 ft.
Hurlen	2.64 to 2.77	W	1998	15,000	-10	Maintenance dredging event	DMMUs 1, 4 suitable DMMUs 2, 3 not suitable	x		98-2-00476
Crowley	Slip 4		1996	13,000	-15	Maintenance dredging event	DMMU 2 (3,250 cy) suitable DMMUs 1, 3, 4 (9,750 cy) not suitable	x		95-2-00537
Morton	2.86 to 2.97	W	1992	7,980	-18	Maintenance dredging event	Yes	x		OYB-2-013054, City of Seattle shoreline permit #8903261-1991
South Park Marina	3.36 to 3.44	W	1993	15,500 permitted	-8 / -9	Maintenance dredging event	1991 DMMO memo states all 8,000 cy suitable (permit allows 15,500 to be dredged)	x		OYB-2-012574
Duwamish Yacht Club	4.03 to 4.15	W	1999	24,000	-8	Maintenance dredging event	Yes	x		071-0YB-2-008104 and 071-OYB-2-012184 authorized to -7 to -11 ft MLLW at 1V:6H slope.
Delta Marine	4.17 to 4.24	W	2004	7,000	-10 / -11 in 0.89-acre area	Maintenance dredging event	Yes	x	x	NWS-200200175: periodic to -10 ft MLLW; march 2008 requested deepening of portion of area dredged in 2004 to -15 ft. NWS-2008320-NO: expansion to adjacent 0.29-acre area (boat basin), also to -5 ft; revision to allow four dredge cycles beginning in 2008 over 10 years (3,550 cy per year). Material from deepening and expansion found suitable for open water disposal under interim dioxin/furan guidelines.
			2008	11,905	-10 / -11 (dredged in January 2008) in portion of area previously dredged in 2004, -5 / -17 to deepen other area previously dredged in 2004 (2,629 cy not yet dredged); -15 / -17 to new 0.29-acre area in permit revision (expansion of boat basin; 3,905 cy not yet dredged)	Maintenance dredging, deepening, and expansion of basin	DMMO memo indicated all 11,905 cy suitable for open water disposal; permit calls for 3,550 cy per event to be dredged; recency extension memo indicates all suitable for open water disposal	x	x	
						Total for all projects	118,384 cy suitable	74%		percentage of cy suitable for open water disposal
						Total for all projects	41,797 cy not suitable	26%		percentage of cy not suitable for open water disposal

Notes:

1. See Figure 2-27 for locations of dredging events.

a. Pre-dredge documents have been reviewed. These documents include: Sampling and Analysis Plans, Suitability Determination Reports, Dredged Materials Characterization Reports, Request for Comments on Proposed Work in CERCLA Area, and Sediment Characterization Reports, and SEPA DNS of Proposed Action.

b. Post-dredge documents have been reviewed. These documents include: Remediation Reports and Dredging Summary and Analysis Reports; USACE inspection reports; recency extensions; the Port Series 2003, piers, wharves, and docks tables; and later DMMO memos or later sampling plans that document previous dredging.

— = unknown / not documented; cy= cubic yards; CERCLA = Comprehensive Environmental Response, Compensation, and Liability Act; DMMO = Dredged Material Management Office; DMMU = dredged material management unit; DNS = Determination of Non-Significance; E = east; ft = feet; LDW = Lower Duwamish Waterway; MLLW = mean lower low water; SEPA = State Environmental Policy Act; USACE = United States Army Corps of Engineers; W = west

Table 2-12 Dredging Events for Contaminated Sediment Removal

Project/Site Name	River Mile	River Side	Dredging Year	Volume Dredged (cy)	Notes
Contaminated Sediment Removal from LDW					
Slip 4 EAA	2.8	E	2011	10,260	Removed sediment and bank soil. Disposed of off-site. Constructed 3.6-acre cap using clean sand mixed with granular activated carbon; completed cap construction January 30, 2012.
Duwamish/Diagonal EAA	0.4 – 0.7	E	2003	68,250	Two areas were dredged and capped in 2003-2004. An adjacent area was covered with a thin-layer cap of sand in 2005.
Norfolk EAA: Norfolk CSO/SD	4.9	E	1999	5,190	Backfill material consisted of 6,700 cy of clean sand derived from the navigational dredging of the Upper Turning Basin.
Norfolk EAA: Boeing Developmental Center South Storm Drain	4.9	E	2003	60	Sediment was removed from the 0.04-acre area adjacent to and inshore of the Norfolk CSO cap by land-based excavation. A portion of the excavation was then backfilled with clean fill.
USACE Navigation Channel Dredging	0.6 – 0.7	navigation channel	1984	1,100	Material deposited in CAD site in West Waterway, covered with capping material from Upper Turning Basin.
Slip 1	1.0	E	1974	50,000	260 gallons Aroclor® 1242 spilled in 1974 when an electric transformer was dropped and broke on the north pier of Slip 1.
Use of LDW Sediment as Capping Material in Elliott Bay					
Elliott Bay and West Waterway	n/a		1984	unknown	Sediment dredged from Upper Turning Basin used as capping material.
Pier 51, Denny Way CSO, Pier 53-55, Pier 64-65			1989 – 1994	10,000 – 22,000 per event	
Puget Sound Resources, Elliott Bay			2004	67,000	

Notes:

See Figure 2-27 for locations of dredging events listed in the table with the exception of the Slip 4 EAA dredging, which occurred in late 2011.

CAD = contained aquatic disposal; CSO/SD = combined sewer overflow / storm drain; cy = cubic yards; EAA = early action area; E = east; LDW = Lower Duwamish Waterway; n/a = not applicable; USACE = U.S. Army Corps of Engineers.

Table 2-13 Number of Monthly LDW Bridge Openings (2003 – 2006)

Year	Openings	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Monthly Average	Daily Average
Spokane Street Bridge															
2003	All motorized vessels	228	208	261	207	193	165	133	139	95	143	122	103	166	5.5
	Tugboat-escorted vessels and barges	93	83	124	106	140	112	105	113	76	109	84	79	102	3.4
	Openings within 1 hour	68	41	81	58	50	42	20	31	16	17	21	17	39	1.3
2004	All motorized vessels	121	105	133	139	138	145	164	115	112	149	152	152	135	4.5
	Tugboat-escorted vessels and barges	95	85	97	113	111	101	133	105	98	109	94	110	104	3.4
	Openings within 1 hour	16	9	18	23	35	26	40	8	16	23	37	23	23	0.8
2005	All motorized vessels	117	93	142	133	152	166	131	160	142	143	136	105	135	4.4
	Tugboat-escorted vessels and barges	80	77	115	113	112	131	104	132	115	103	107	75	105	3.5
	Openings within 1 hour	19	10	26	29	34	33	15	38	19	22	27	10	24	0.8
First Avenue Bridge															
2005	All openings	108	119	175	158	168	147	116	135	115	92	93	124	129	4.3
2006	All openings	112	83	129	145	155	142	182	146	139	125	—	—	136	4.5
Former South Park Bridge[a]															
2005	All openings	39	63	76	47	42	59	95	76	80	53	35	46	59	2.0
2006	All openings	39	42	42	82	101	88	125	98	81	59	—	—	76	2.5

Sources:

Seattle Department of Transportation (SDOT) 2006. *Spokane Street Bridge Opening Logs, 2003-2005*. Provided by Bridge/Structures Maintenance and Operations Manager. November 15, 2006.

Washington State Department of Transportation. *First Avenue Bridge Opening Logs, 2005-2006*. Provided by Bridge/Structures Maintenance & Operations Manager. December 18, 2006.

King County Department of Transportation. *South Park Bridge Opening Logs, 2005-2006*. Provided by Bridges/Structures Operations & Maintenance Manager. December 18, 2006.

Notes:

1. During most openings, vessels moving through the opened bridge include 1 large vessel and 1 to 3 tugs.

a. This bridge was closed and demolished in 2010. A new bridge is under construction, just downstream of the former bridge location, with completion scheduled for fall of 2013.

— = data not available at time it was requested.

LDW = Lower Duwamish Waterway

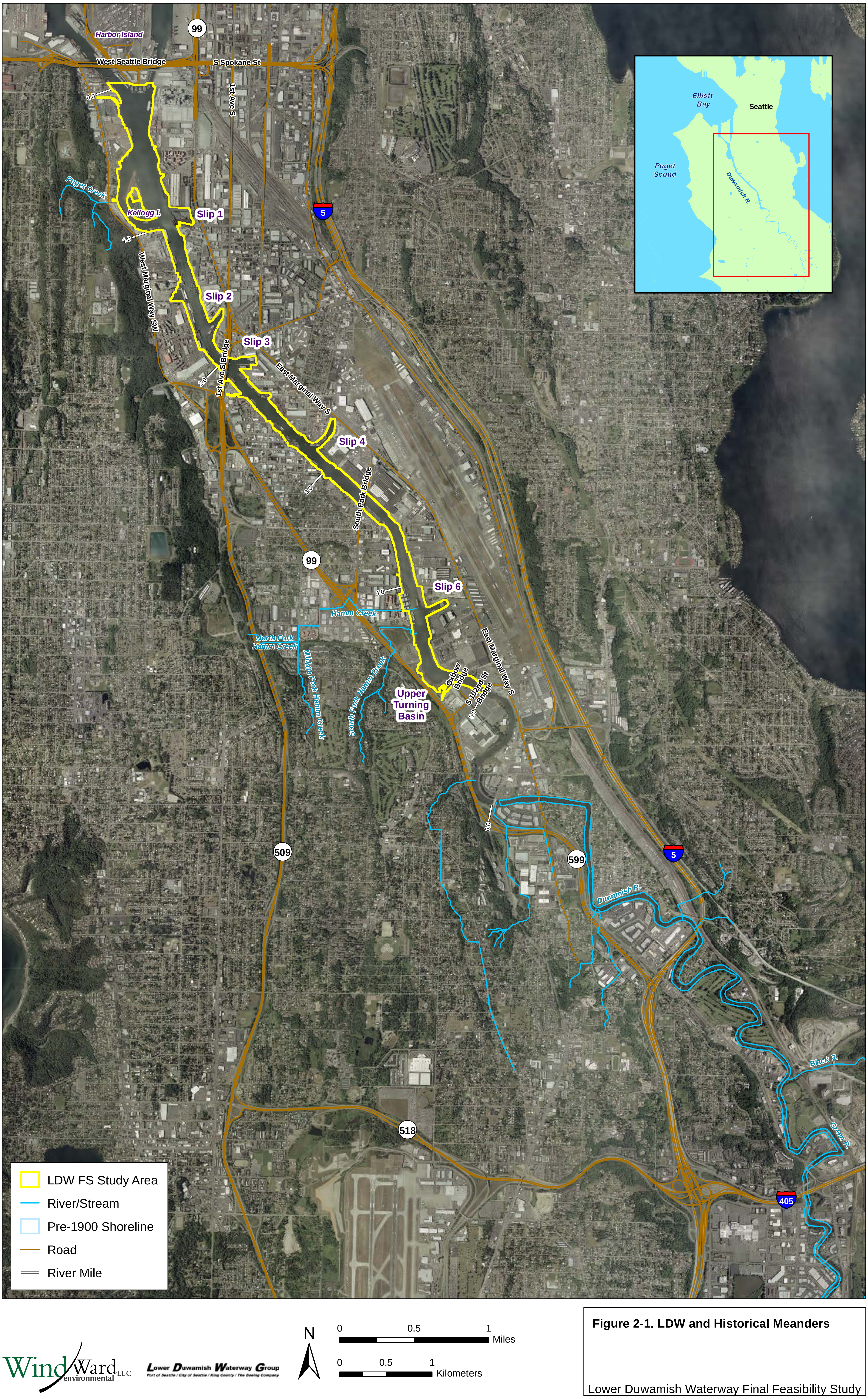


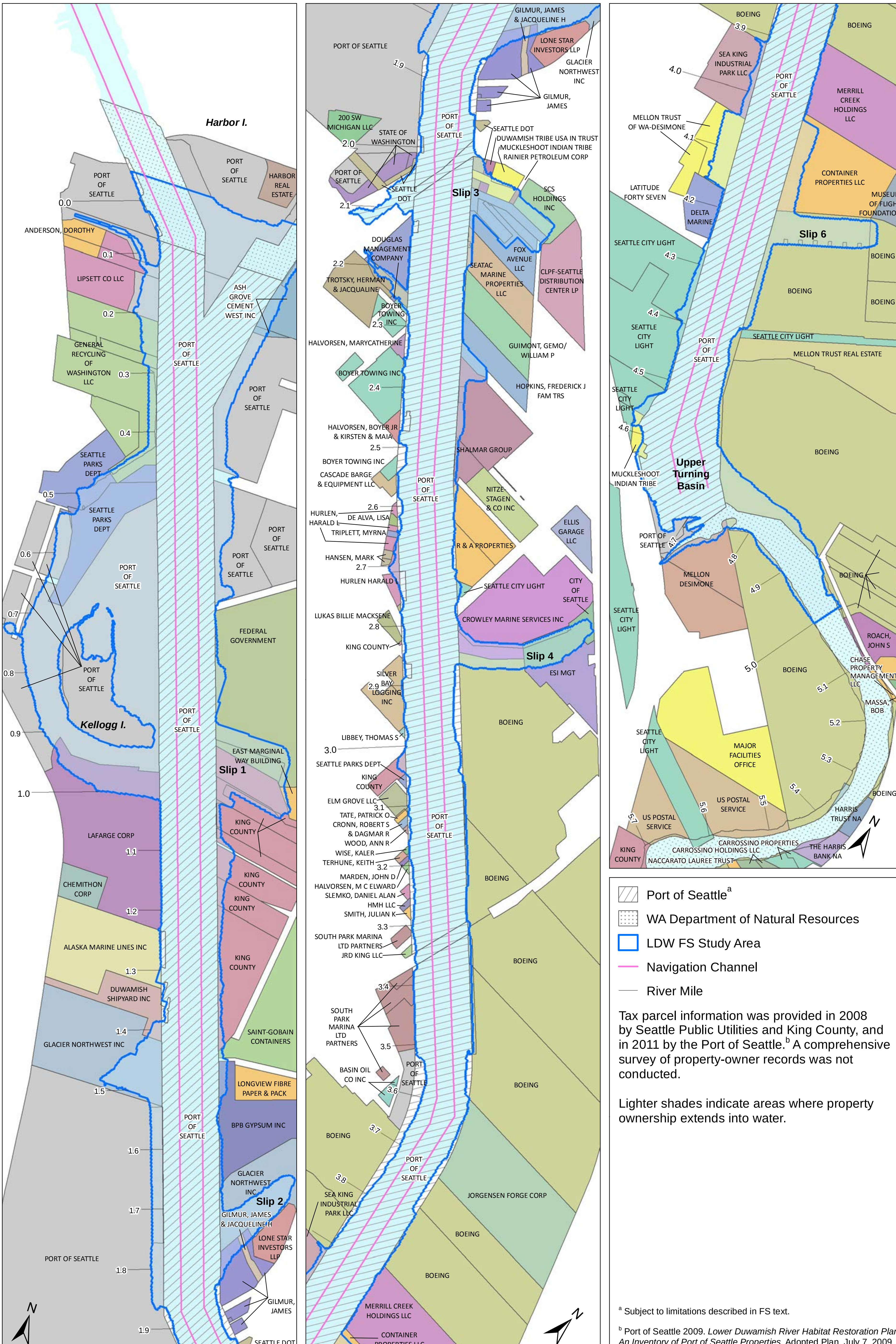
Figure 2-1. LDW and Historical Meanders

Lower Duwamish Waterway Final Feasibility Study

Windward environmental LLC

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Photo source: USGS High Resolution Orthoimage, Seattle/Tacoma WA, 1 m resolution. Photo date 06/11/2002.



Tax parcel information was provided in 2008 by Seattle Public Utilities and King County, and in 2011 by the Port of Seattle.[b] A comprehensive survey of property-owner records was not conducted.

Lighter shades indicate areas where property ownership extends into water.

[a] Subject to limitations described in FS text.

[b] Port of Seattle 2009. *Lower Duwamish River Habitat Restoration Plan. An Inventory of Port of Seattle Properties*. Adopted Plan. July 7, 2009.

0 0.1 0.2 Miles

0 0.1 0.2 Kilometers

Scale is the same for each inset map

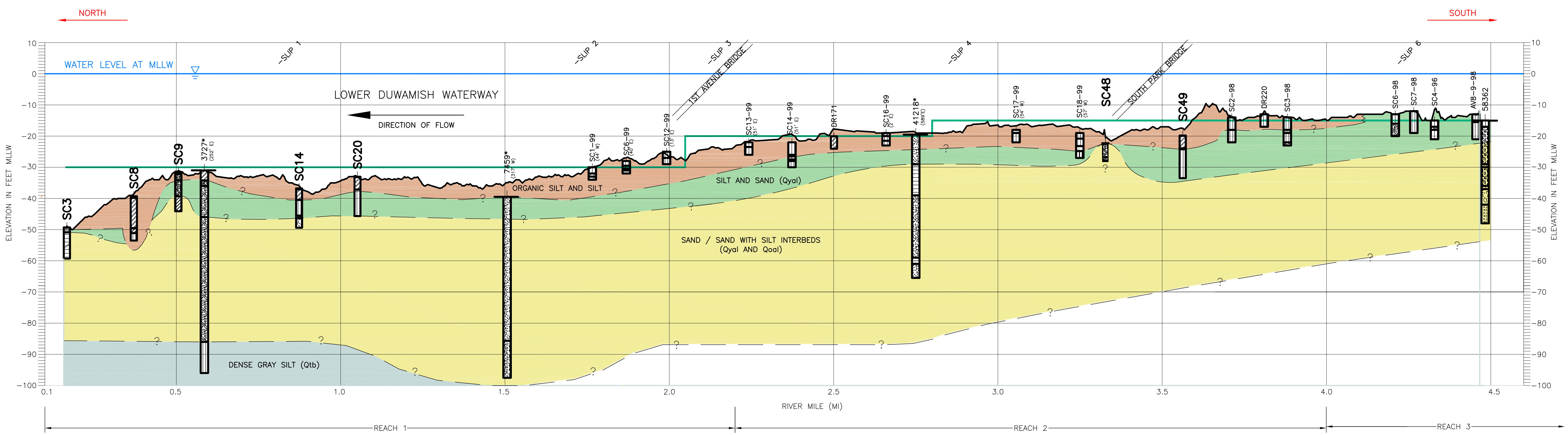
Windward environmental LLC

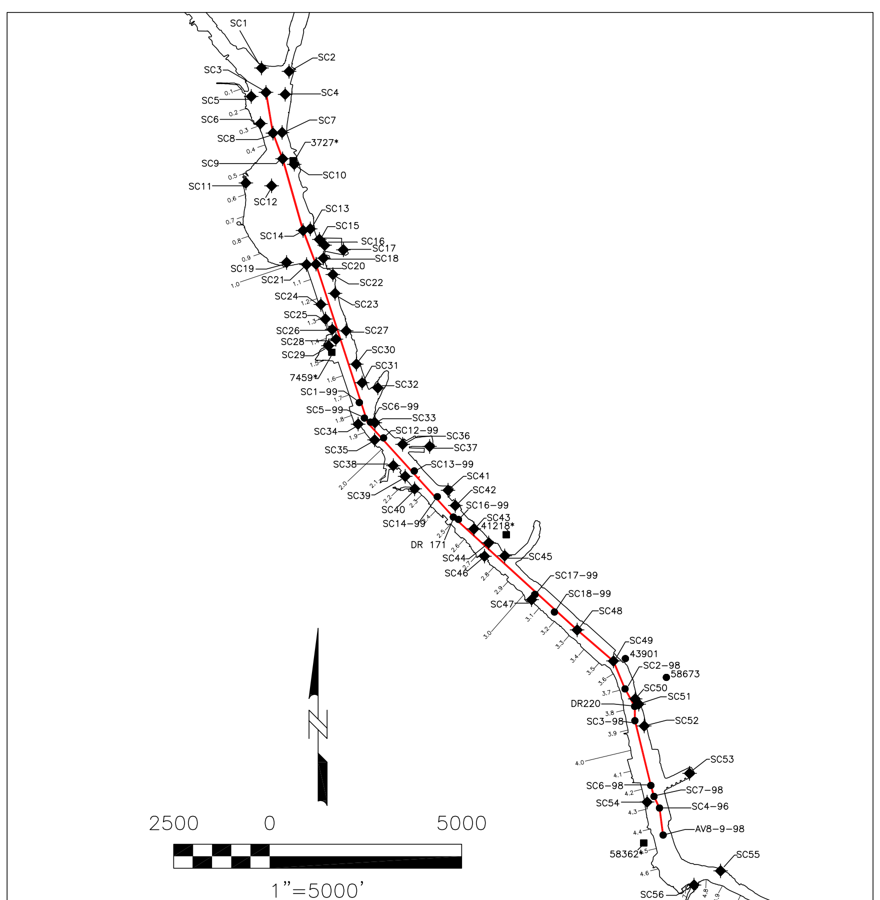
Lower Duwamish Waterway Group — Port of Seattle / City of Seattle / King County / The Boeing Company

Prepared by inmann, 10/31/2012; L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\2012-10-15 LDW FS WW GIS Maps and Data\Fig 2-02 2851 Land and water ownership1.mxd

Figure 2-2. Upland, Intertidal, and Subtidal Land Ownership

Lower Duwamish Waterway Final Feasibility Study





800 0 1600

HORIZONTAL SCALE IN FEET

VERTICAL EXAGGERATION = 50:1

LEGEND

DR-171 ● Historical Subsurface Core
- SC4-96 = PSDDA96 study
- SC2-98 = PSDDA98 study
- SC6-99 = PSDDA99 study
- DR171 = EPASI study

3727* ■ Upland Core
- 7459 = Hart Crowser 1979 study
- 41218 = Yonemitsu Geological Services 1979 study
- 3727 = Dames and Moore 1988 study
- 58362 = Seattle Public Utilities (CPT-D77_01) 1985 study

SC48 ◆ 2006 Subsurface Core

Authorized navigation depth (ft MLLW)

Mid-channel mudline elevation (based on 2003 bathymetric survey, ft MLLW; Windward and DEA 2004)

Water level at MLLW (ft)

Top of projected core not shown

KEY

SC14-99 (51' E)
- Core Name (Distance from section to Core)
- Top of Core
- Observed Contacts
- Bottom of Core

STRATIGRAPHY**

- RECENT
- UPPER ALLUVIUM/TRANSITION
- LOWER ALLUVIUM
- DENSE POST-GLACIAL AND GLACIAL UNIT

Qtb = TRANSITIONAL SILTS
Qyal = YOUNGER ALLUVIUM
Qoal = OLDER ALLUVIUM

LITHOLOGY

- Clay/Silt
- Organic silt
- Silt
- Sand with silt
- Sand with silt interbeds
- Sand
- Gravel, sandy gravel

NOTES

1) MLLW – mean lower low water

2) 2006 Phase 2 cores (SC3, SC8, SC9, SC14, SC20, SC48, SC49) are labeled with bold font in cross section.

3) AV8-9-98 is an average of Cores 8 and 9 from PSDDA (1998).

4) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

5) Cross sections represent regional geology. See core logs in the Subsurface Sediment Data Report (Windward 2007) for detailed stratigraphy.

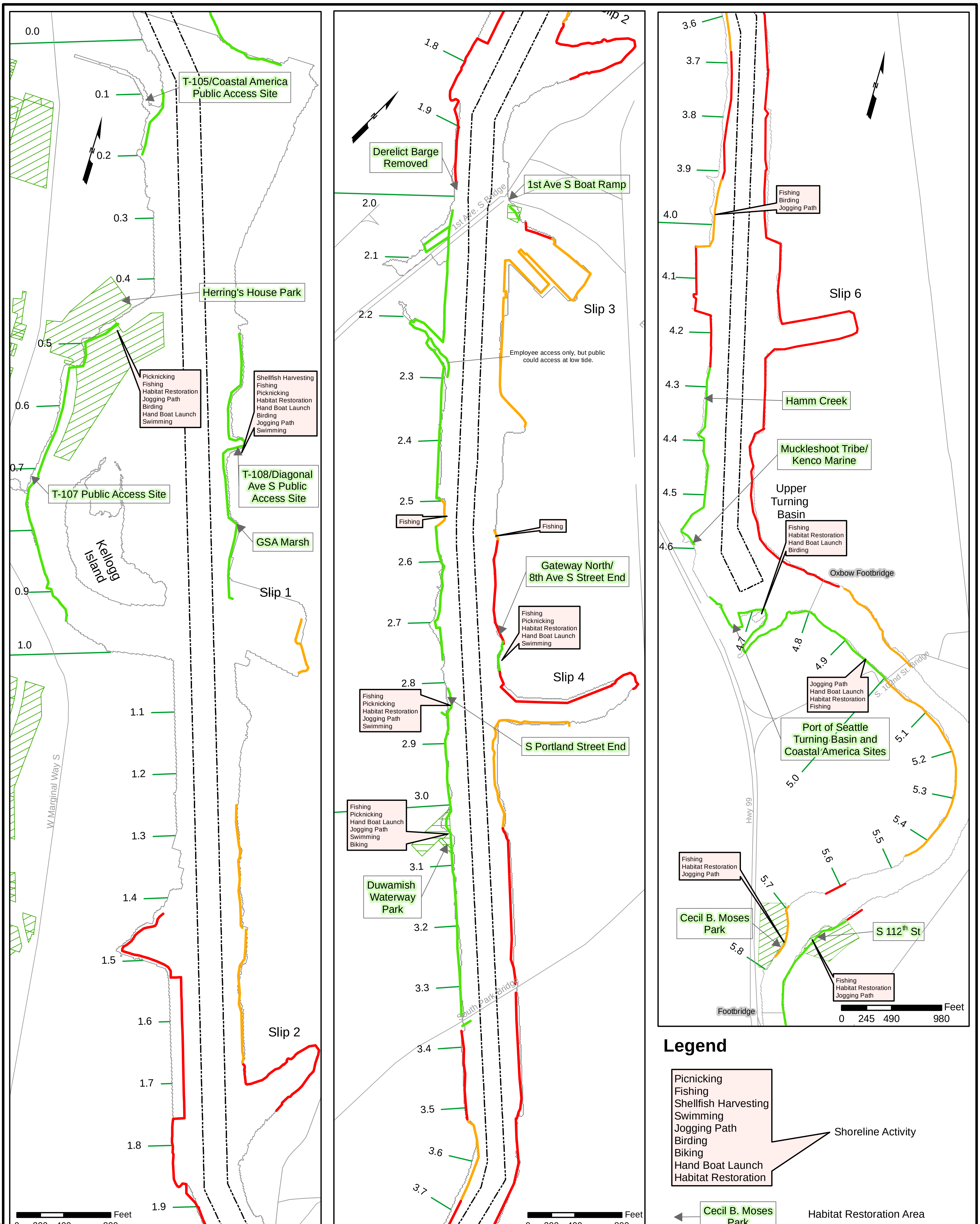
* Upland cores projected into navigation channel. Lower portions of upland cores used to bound deeper sediment units.

** Stratigraphy terminology is used in Subsurface Data Report (Windward and RETEC 2007) and Sediment Transport Analysis Report (Windward and QEA 2008).

AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

LOWER DUWAMISH WATERWAY FINAL FEASIBILITY STUDY 60150279-14.34	NAVIGATION CHANNEL LONGITUDINAL CROSS SECTION
DATE: 10/31/12 DRWN: E.M./MO/SEA	FIGURE 2-3



Legend

Picnicking
Fishing
Shellfish Harvesting
Swimming
Jogging Path
Birding
Biking
Hand Boat Launch
Habitat Restoration — Shoreline Activity

Cecil B. Moses Park — Habitat Restoration Area

Park

Road or Bridge

Navigation Channel

River Mile Marker

Shoreline Use

Easy Public Access

Difficult Public Access

Restricted Access

Notes:
1. Shoreline use and activities from Windward (2005) Human Use Survey. Activities based on questionnaires by residents. Activities and locations where activities are engaged in are not all inclusive.
2. Restoration areas and parks from RI Map 2-9 and LDWG member interviews.
3. Easy public access is where there are waterfront homes, public parks, street ends, or other areas that can be readily accessed by the public on foot. Shoreline areas with difficult access designation are either not accessible by land or the access is unknown. Restricted public access designates areas accessible by employees or members of businesses and marinas, respectively.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-4ParksRestorationShoreAccess.mxd

AECOM

Lower Duwamish Waterway Group
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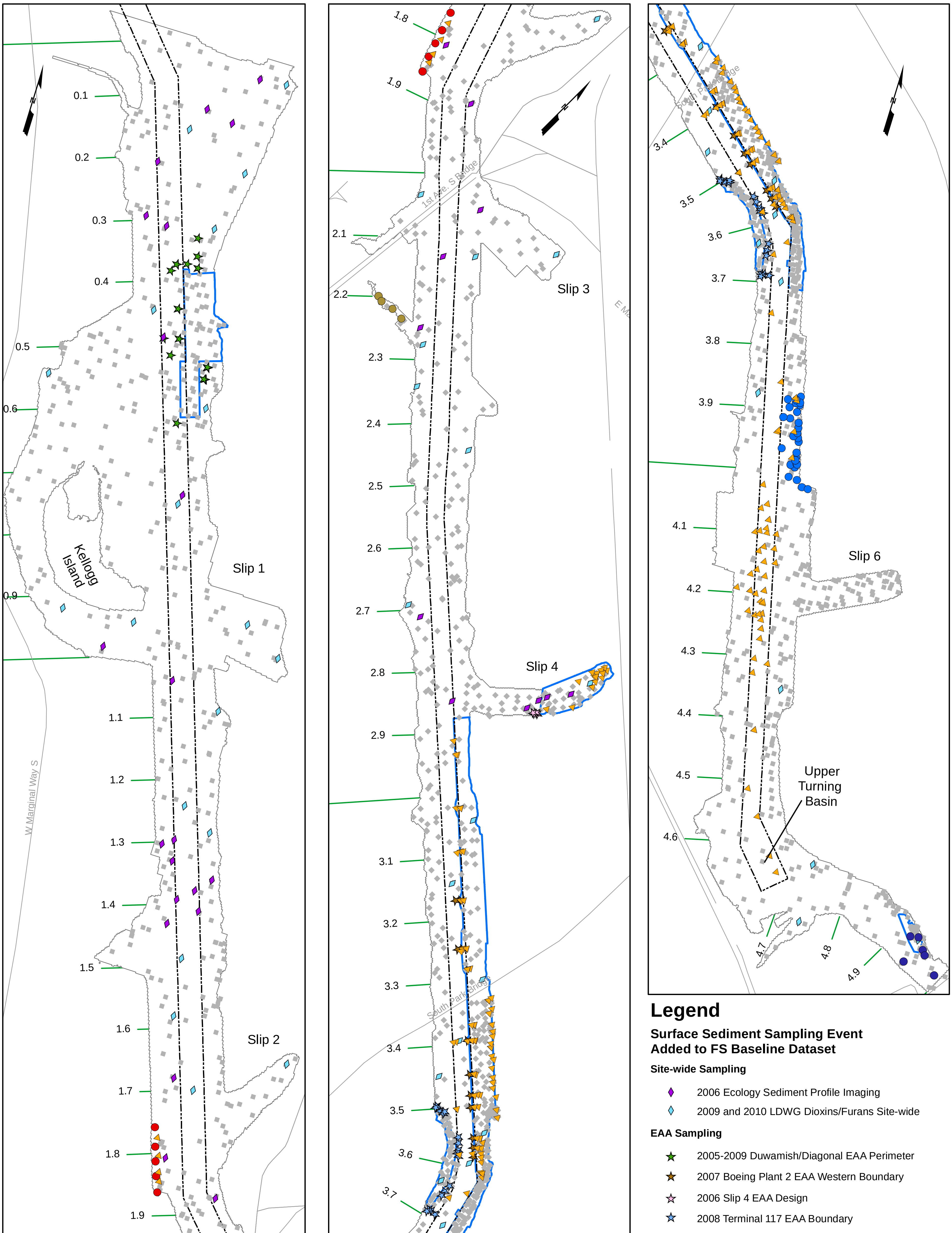
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Habitat Restoration Areas, Parks, and Shoreline Access
DATE: 10/31/12	DWRN: MVI/sea	Revision: 1	FIGURE 2-4



Legend

Surface Sediment Sampling Event Added to FS Baseline Dataset

Site-wide Sampling

- 2006 Ecology Sediment Profile Imaging
- 2009 and 2010 LDWG Dioxins/Furans Site-wide

EAA Sampling

- 2005-2009 Duwamish/Diagonal EAA Perimeter
- 2007 Boeing Plant 2 EAA Western Boundary
- 2006 Slip 4 EAA Design
- 2008 Terminal 117 EAA Boundary

Other Area Sampling

- 2006 and 2008 8801 E Marginal Way (formerly Kenworth PACCAR)
- 2007 Industrial Container Services
- 2008 Ecology Upstream Surface Sediment
- 2009 Terminal 115 Intertidal

- RI Baseline Surface Sediment Location
- Core Added to FS Dataset

- Early Action Area (EAA)
- Navigation Channel
- River Mile Marker
- Road

Feet
0 200 400 800

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60150279-14.34

DATE: 10/31/12 | DWRN: mvi/SEA | Revision: 0

Surface Sediment Sampling Locations Added Since October 2006 to Generate FS Baseline Dataset

FIGURE 2-5

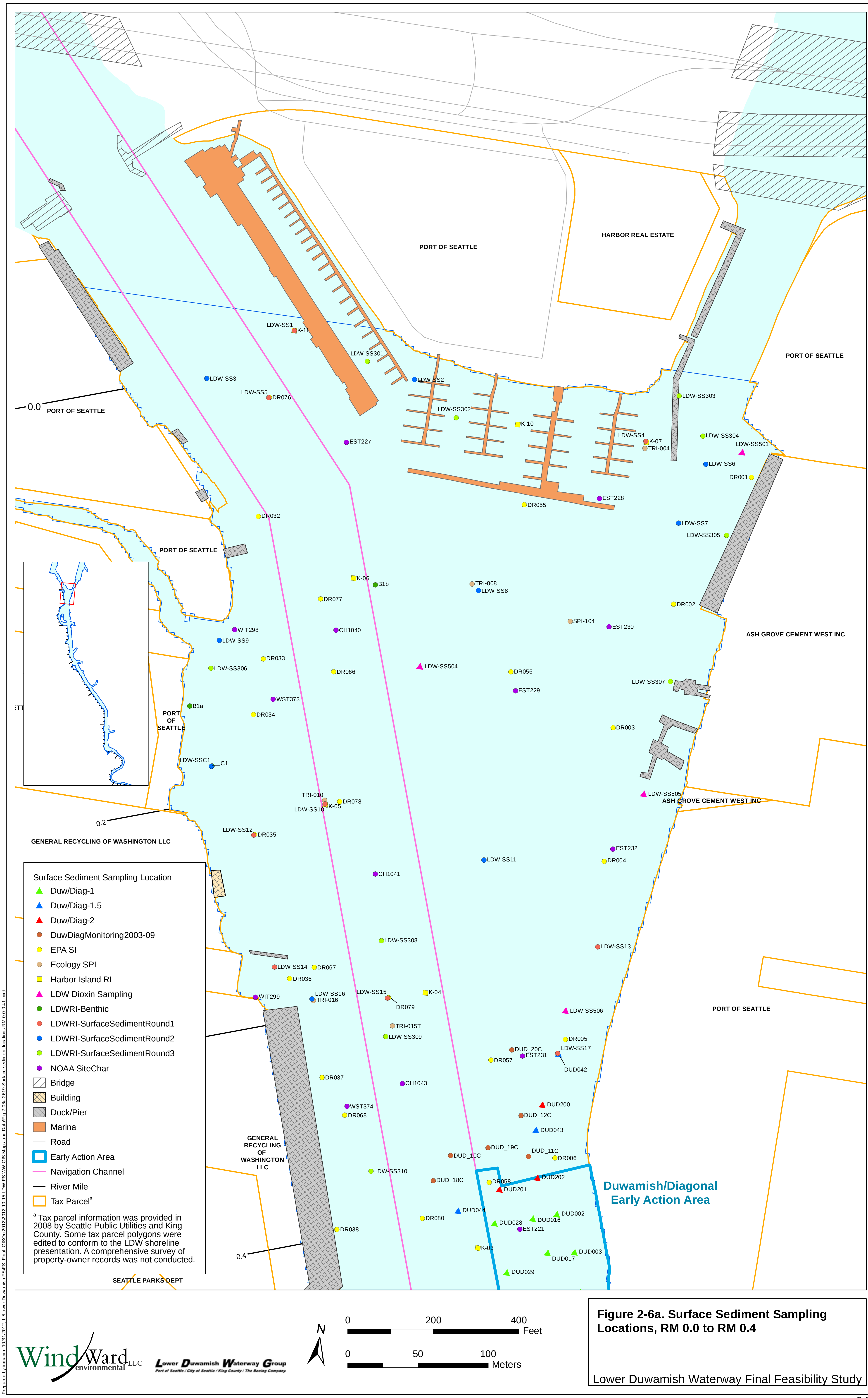
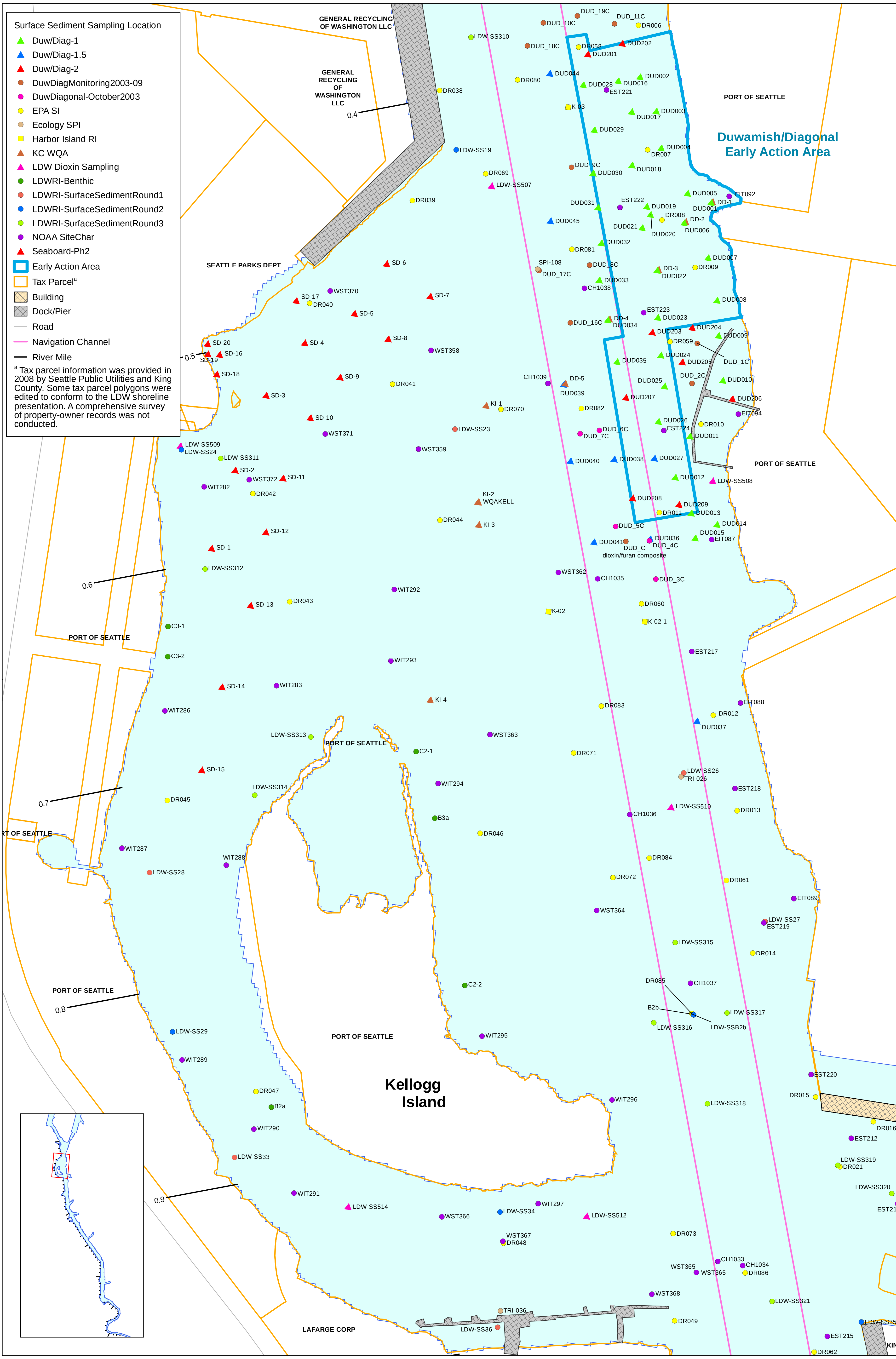


Figure 2-6a. Surface Sediment Sampling Locations, RM 0.0 to RM 0.4

Lower Duwamish Waterway Final Feasibility Study



0 200 400 Feet

0 50 100 Meters

N

Figure 2-6b. Surface Sediment Sampling Locations, RM 0.4 to RM 0.9

Lower Duwamish Waterway Final Feasibility Study

Prepared by mmamm, 10/31/2012; L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\2012-10-15 LDW FS WW GIS Maps and Data\Fig 2-6b 2620 Surface Sediment locations labeled RM 0.4-0.9.mxd

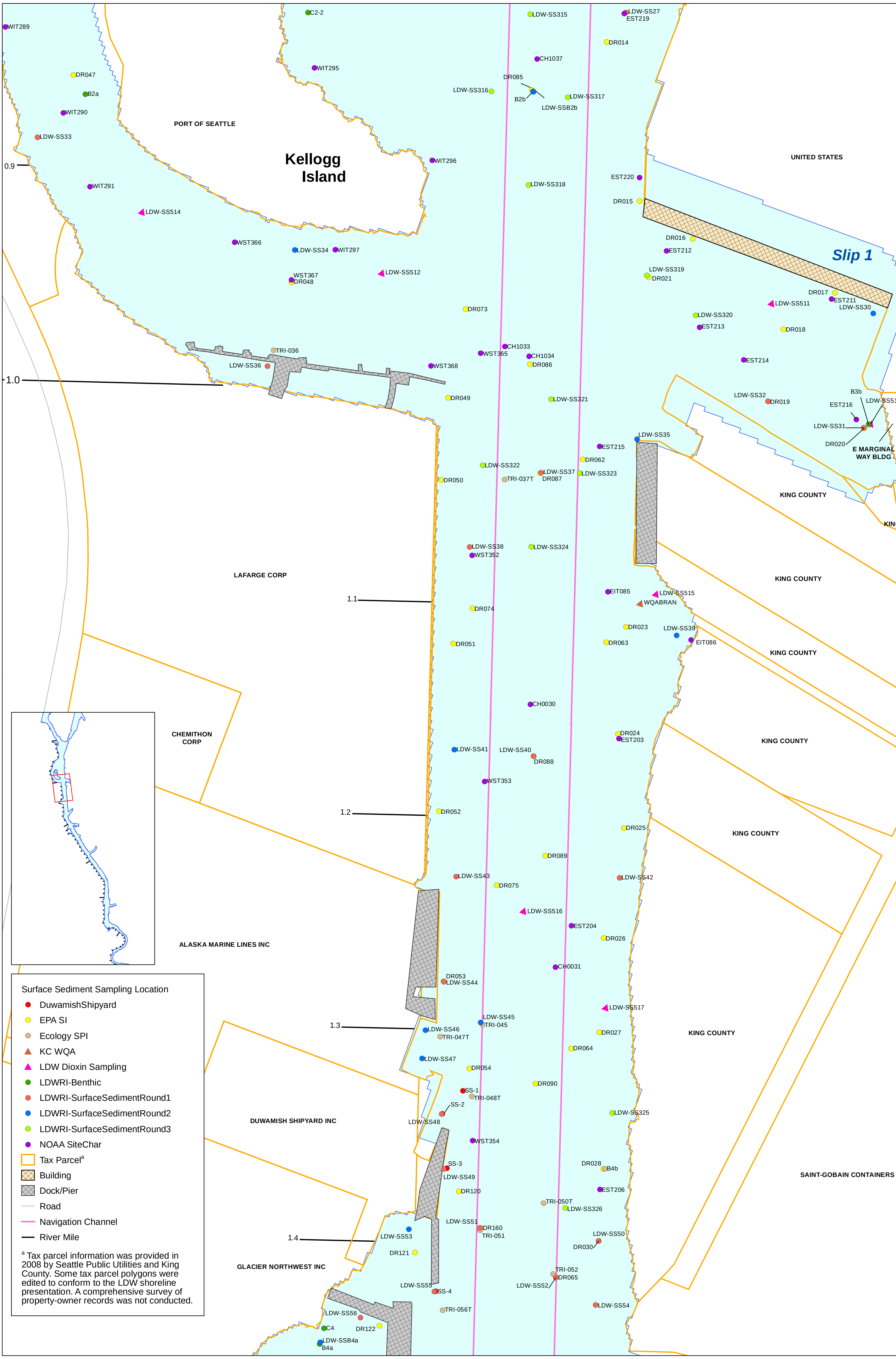


Figure 2-6c. Surface Sediment Sampling Locations, RM 0.9 to RM 1.4

Lower Duwamish Waterway Final Feasibility Study

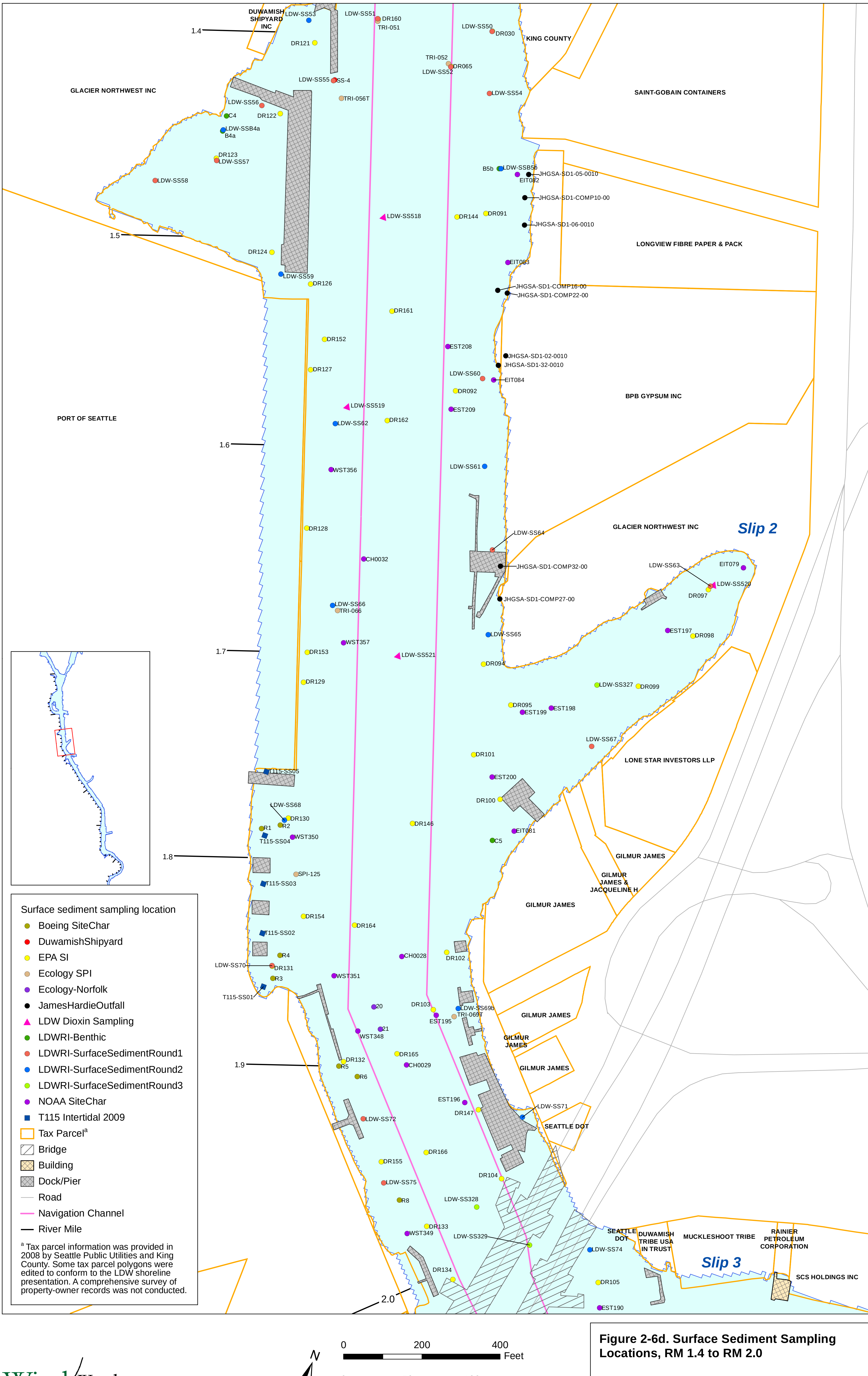


Figure 2-6d. Surface Sediment Sampling Locations, RM 1.4 to RM 2.0

Lower Duwamish Waterway Final Feasibility Study

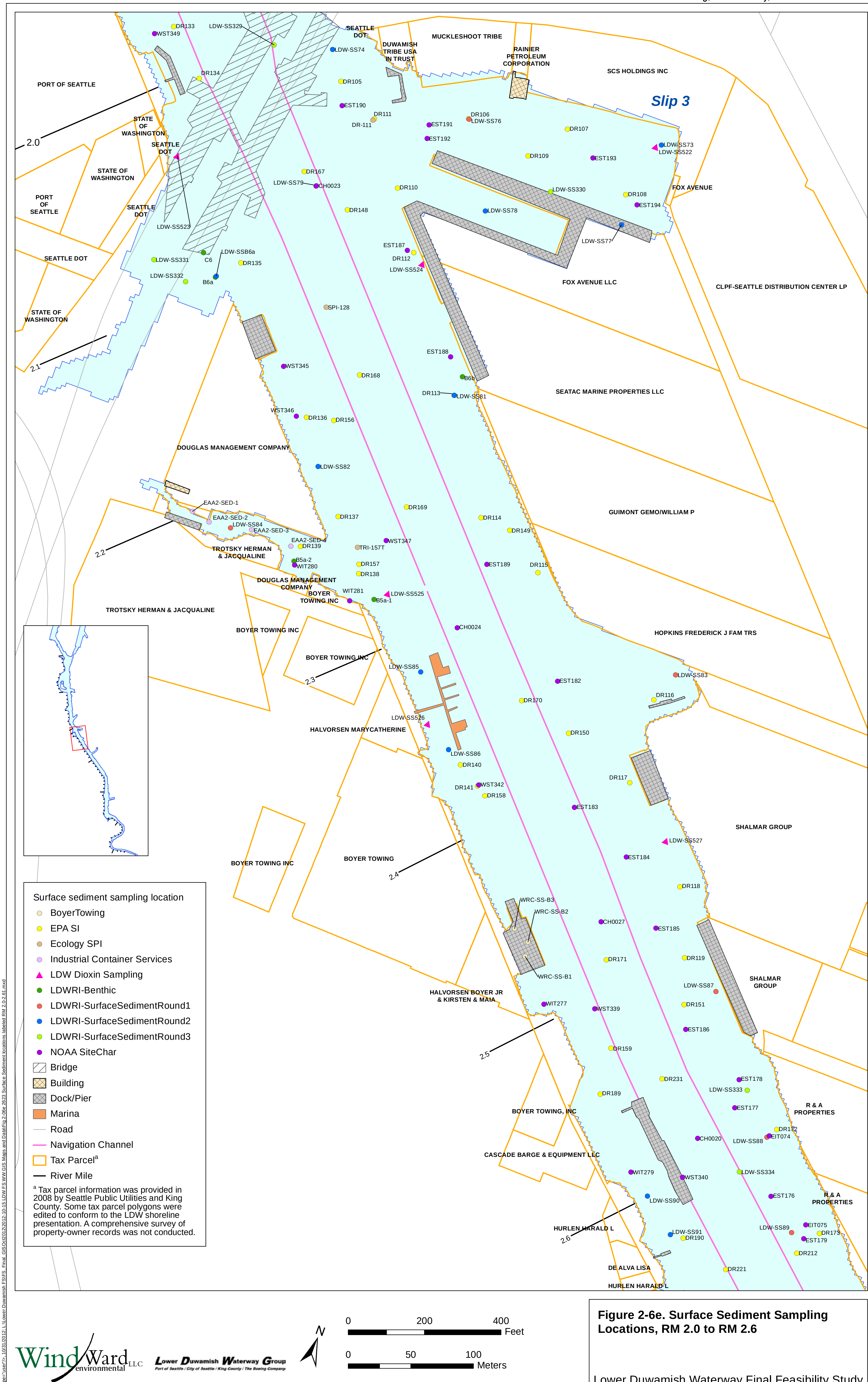


Figure 2-6e. Surface Sediment Sampling Locations, RM 2.0 to RM 2.6

Lower Duwamish Waterway Final Feasibility Study

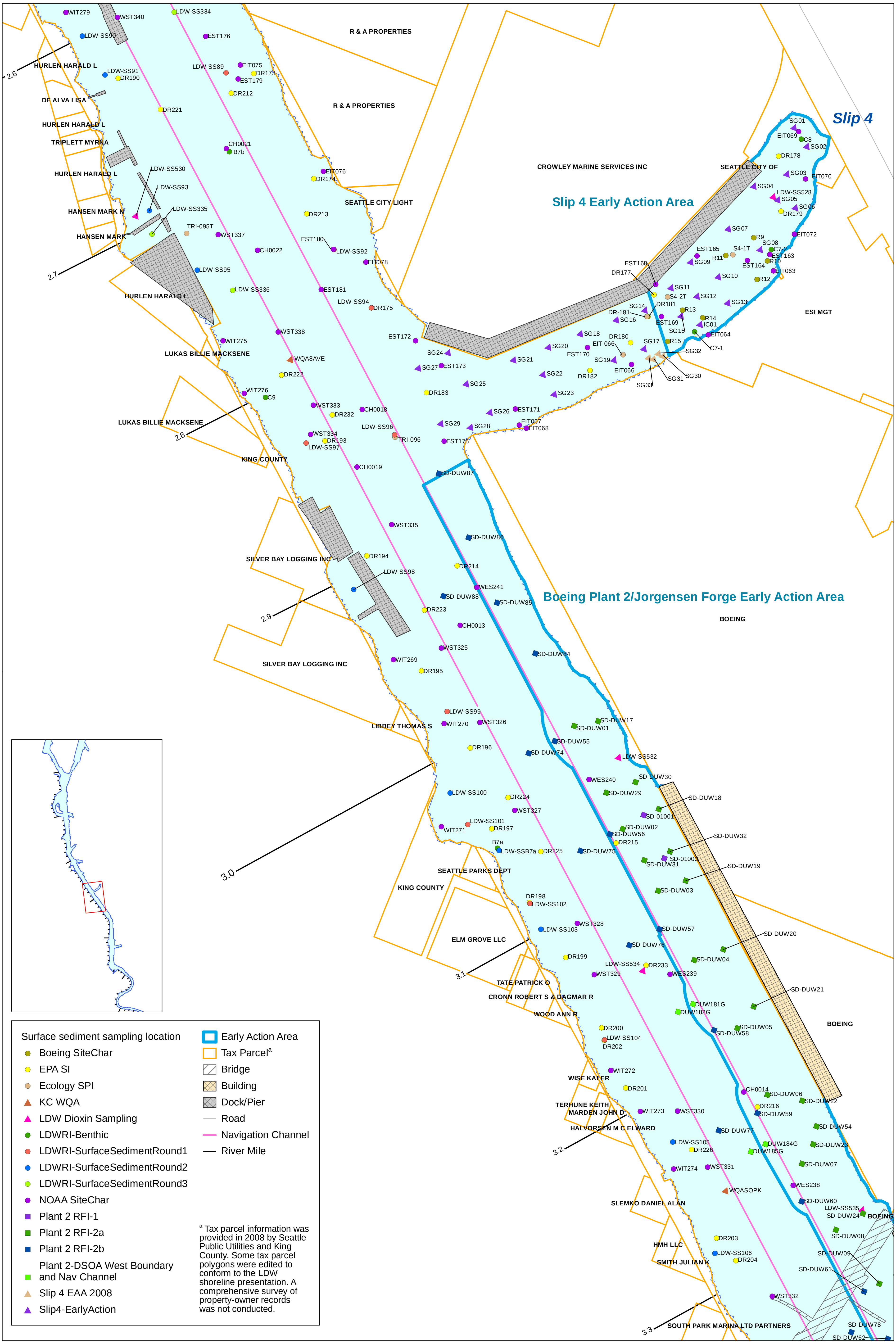


Figure 2-6f. Surface Sediment Sampling Locations, RM 2.6 to RM 3.3

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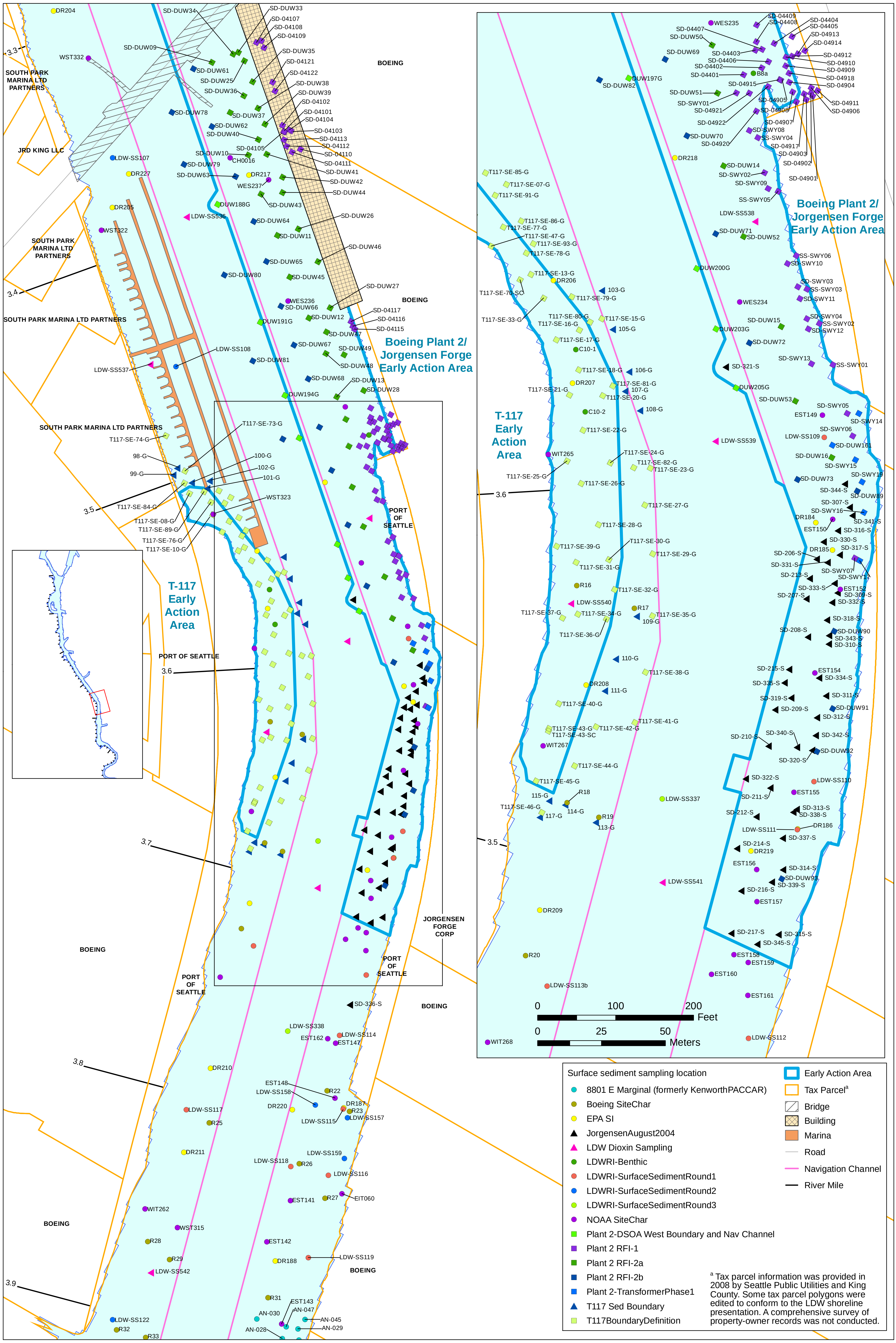

Figure 2-6g. Surface Sediment Sampling Locations, RM 3.3 to RM 3.9

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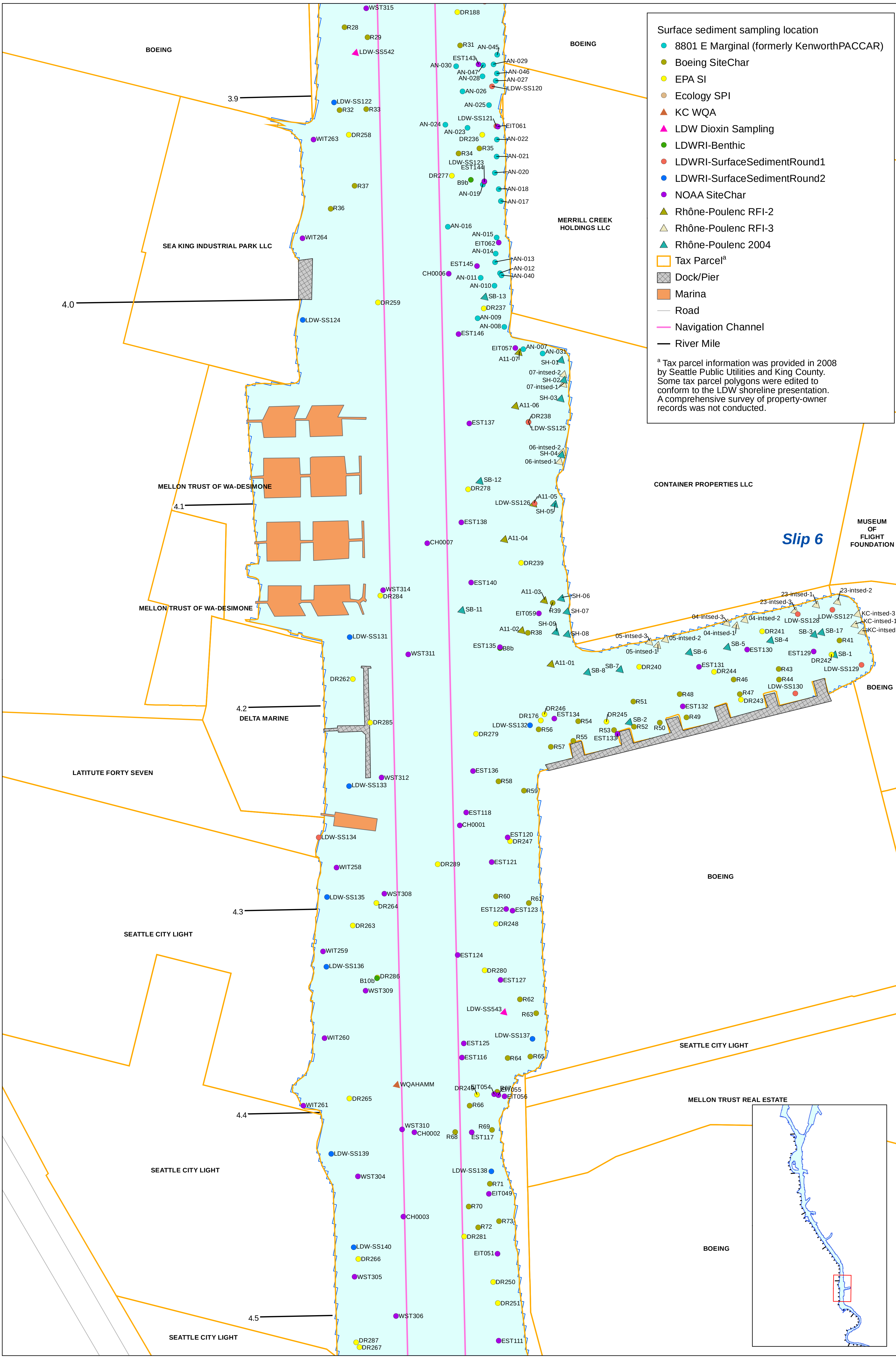


Figure 2-6h. Surface Sediment Sampling Locations, RM 3.9 to RM 4.5

Lower Duwamish Waterway Final Feasibility Study

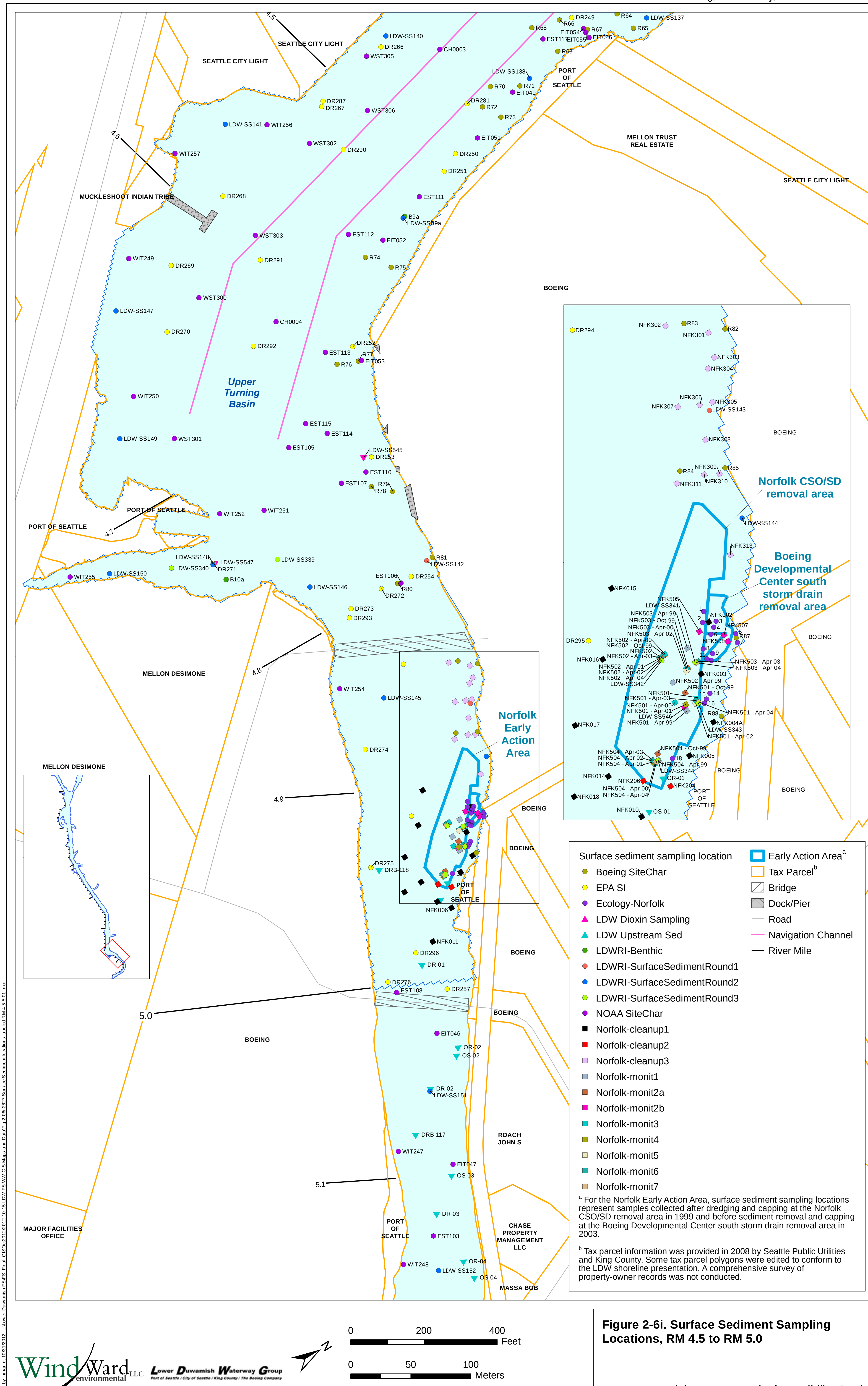
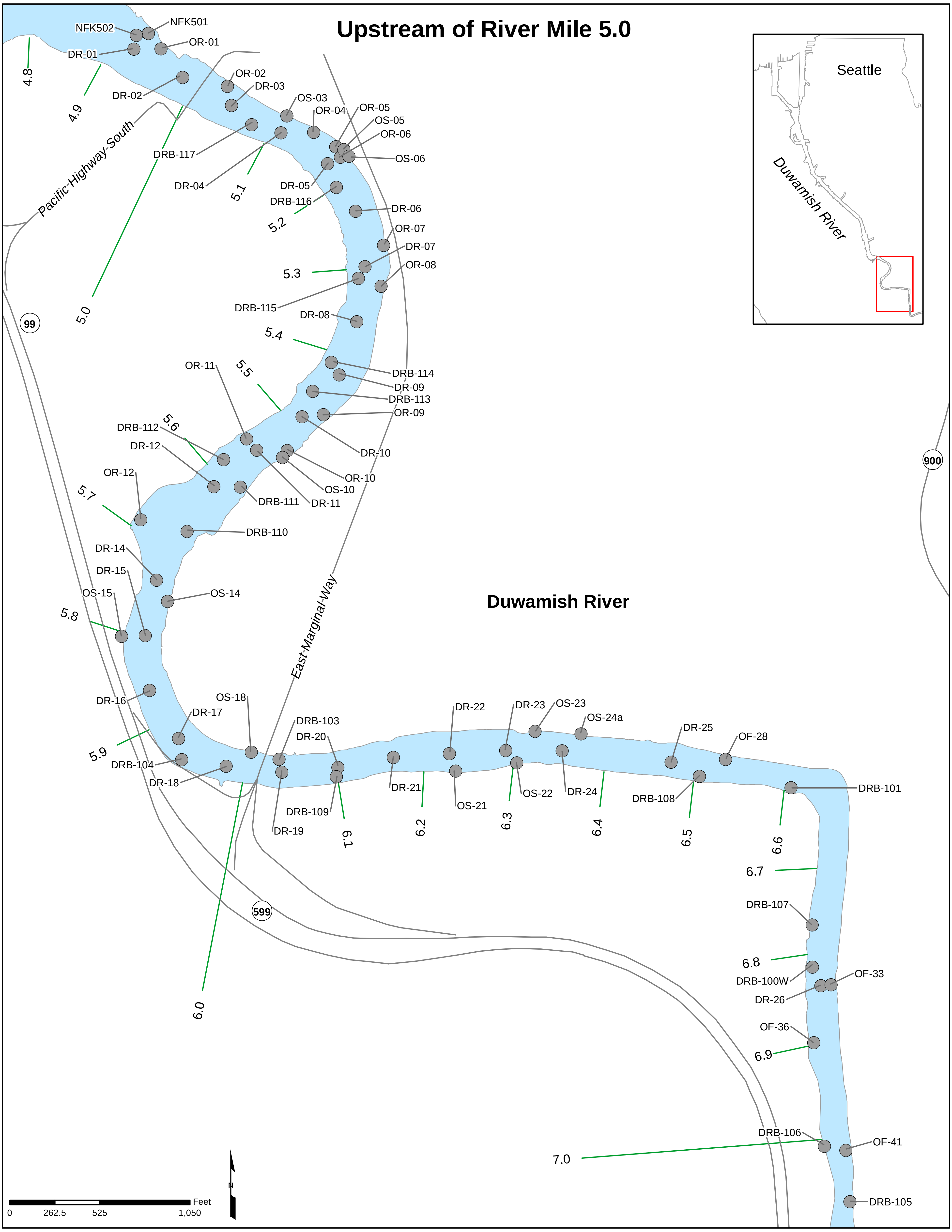


Figure 2-6i. Surface Sediment Sampling Locations, RM 4.5 to RM 5.0

Lower Duwamish Waterway Final Feasibility Study



Notes:
1. Surface sediment samples from RM 4.9 to 6.5 are from 2008 Ecology study. DR = center channel samples, OS = samples near the discharge points of outfalls, OR = samples within the Duwamish River approximately 15 meters downstream of outfall discharge points, DRB = bank samples that appear to be depositional environments, OF = bank samples at discharge points of selected newly identified outfalls upstream of RM 6.5, NFK = samples near the Norfolk combined sewer overflow.
2. Surface sediment samples collected at depths between 0 and 10 cm.

Legend

Upstream Surface Sediment Sample Location

- Ecology 2008 Study
- River Mile Marker
- State Highway

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			**Ecology 2008 Upstream Surface Sediment Sampling Locations**
DATE: 10/31/12	DWRN: MVI/sea	Revision: 0	**FIGURE 2-7**

Figure 2-8a LDW Conceptual Site Model for Reach 1

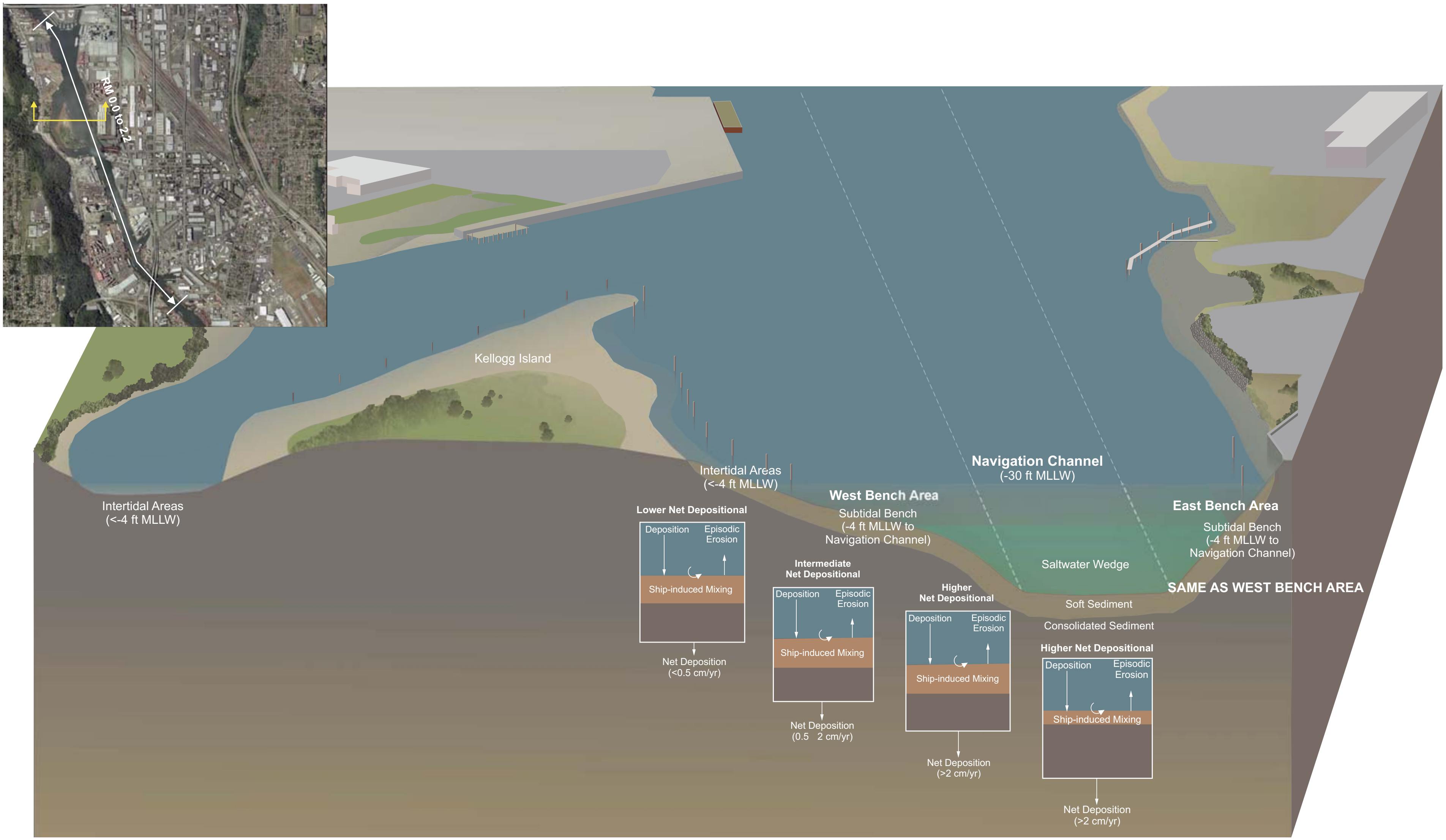


Figure 2-8b LDW Conceptual Site Model for Reach 2

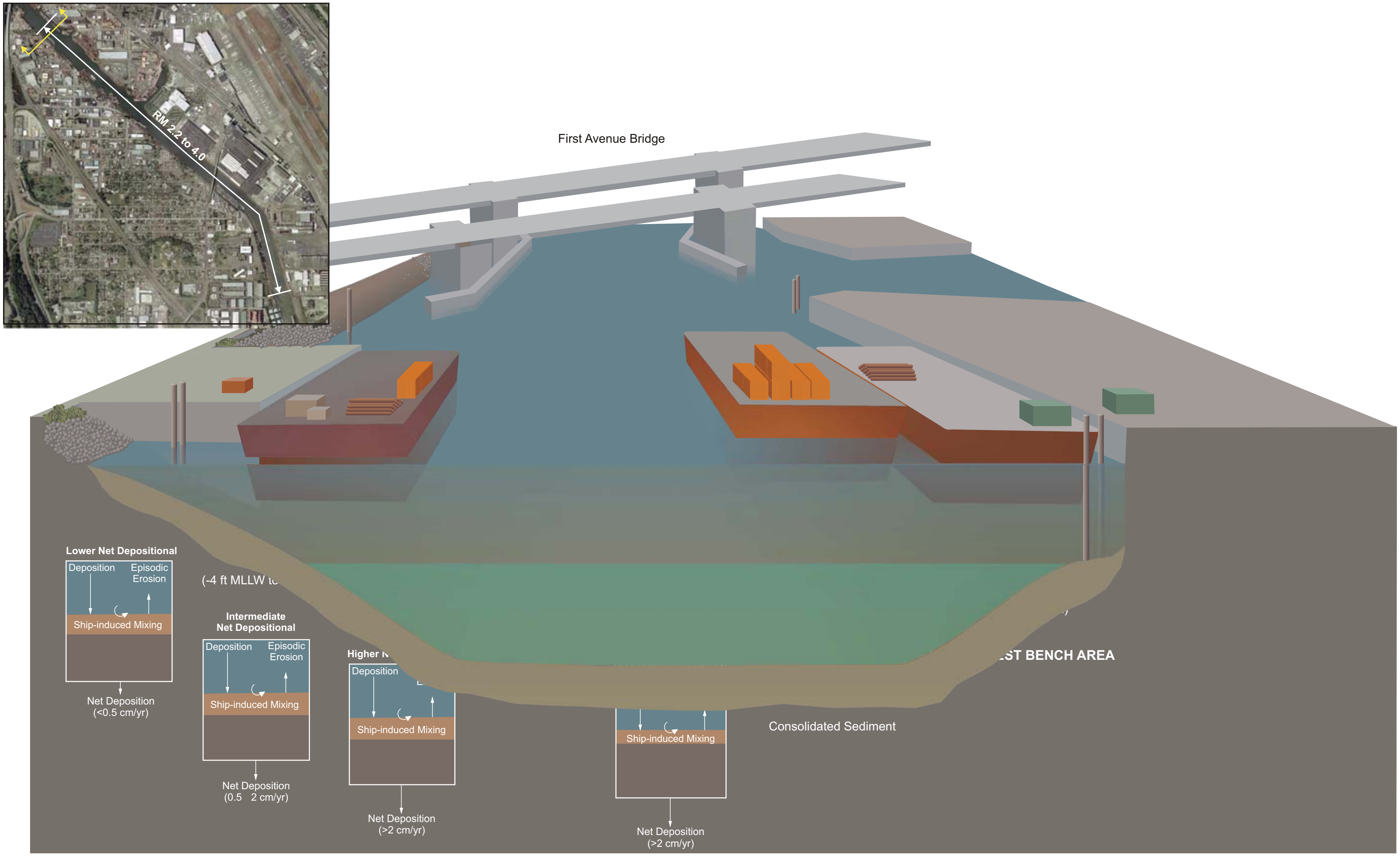
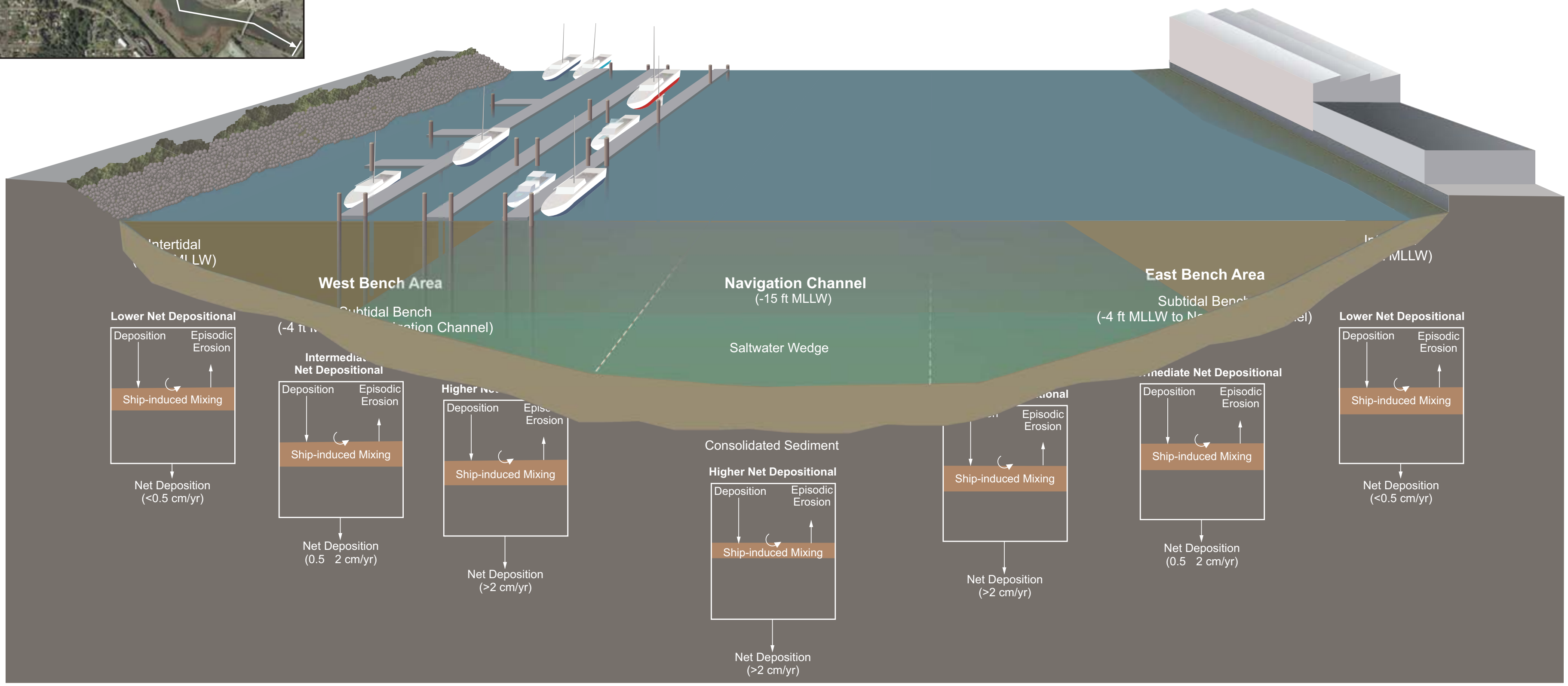
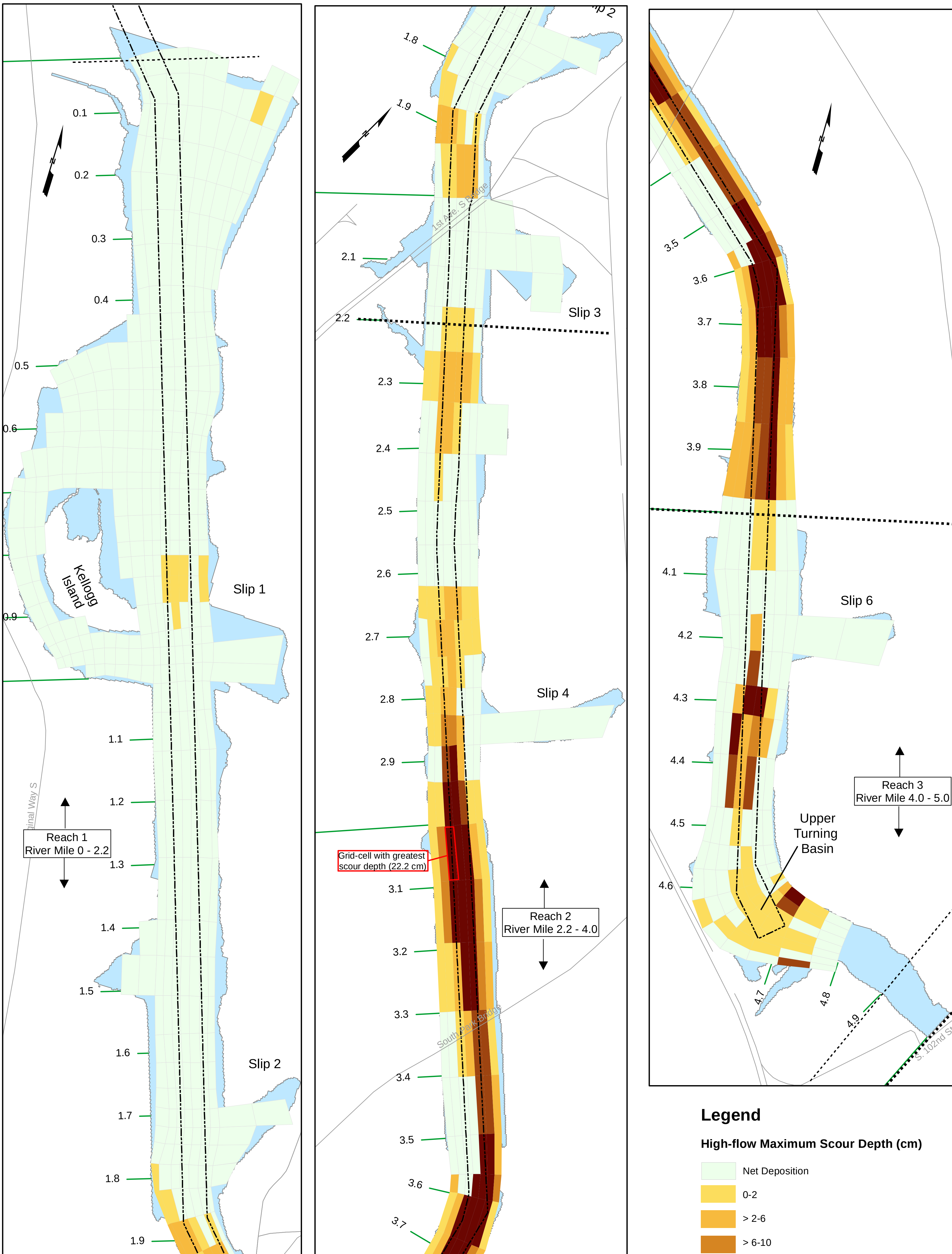


Figure 2-8c LDW Conceptual Site Model for Reach 3





Legend

High-flow Maximum Scour Depth (cm)

- Net Deposition
- 0-2
- > 2-6
- > 6-10
- > 10 - 15
- > 15 - 22
- Outside of Model Domain

- Extent of Conceptual Site Model Reach
- Road
- Navigation Channel
- River Mile Marker

Feet
0 200 400 800

Notes:
1. Maximum scour depths from 100-year high-flow event shapefile dated June 2008 (QEA 2008).
2. Maximum scour depth is the depth to which sediment is scoured from the bed any time during a 100-year high-flow event.

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Predicted Bed Scour Depth During 100-Year High-Flow Event

FIGURE 2-9

Notes:
1. Sun-illumination layer generated in ArcGIS 9.2 using October 2003 David Evans and Associates bathymetric survey using the following settings: azimuth 30°, altitude 45°.
2. Overwater structures data created in 2004 by Terralogic GIS, Inc. and Landau Associates, Inc. and modified using 2007 high resolution oblique aerial photography and field investigations.
3. Because the sun-illumination transparency was applied to the mudline elevation, the legend appears darker than the map.

Legend

- Evidence of Propeller Wash Scour
- Overwater Structure
- No 2003 Bathymetric Data Coverage

Mudline Elevation (ft MLLW)

High : 9.5

Low : -53.6

- Road
- Navigation Channel
- River Mile Marker

Feet
0 200 400 800

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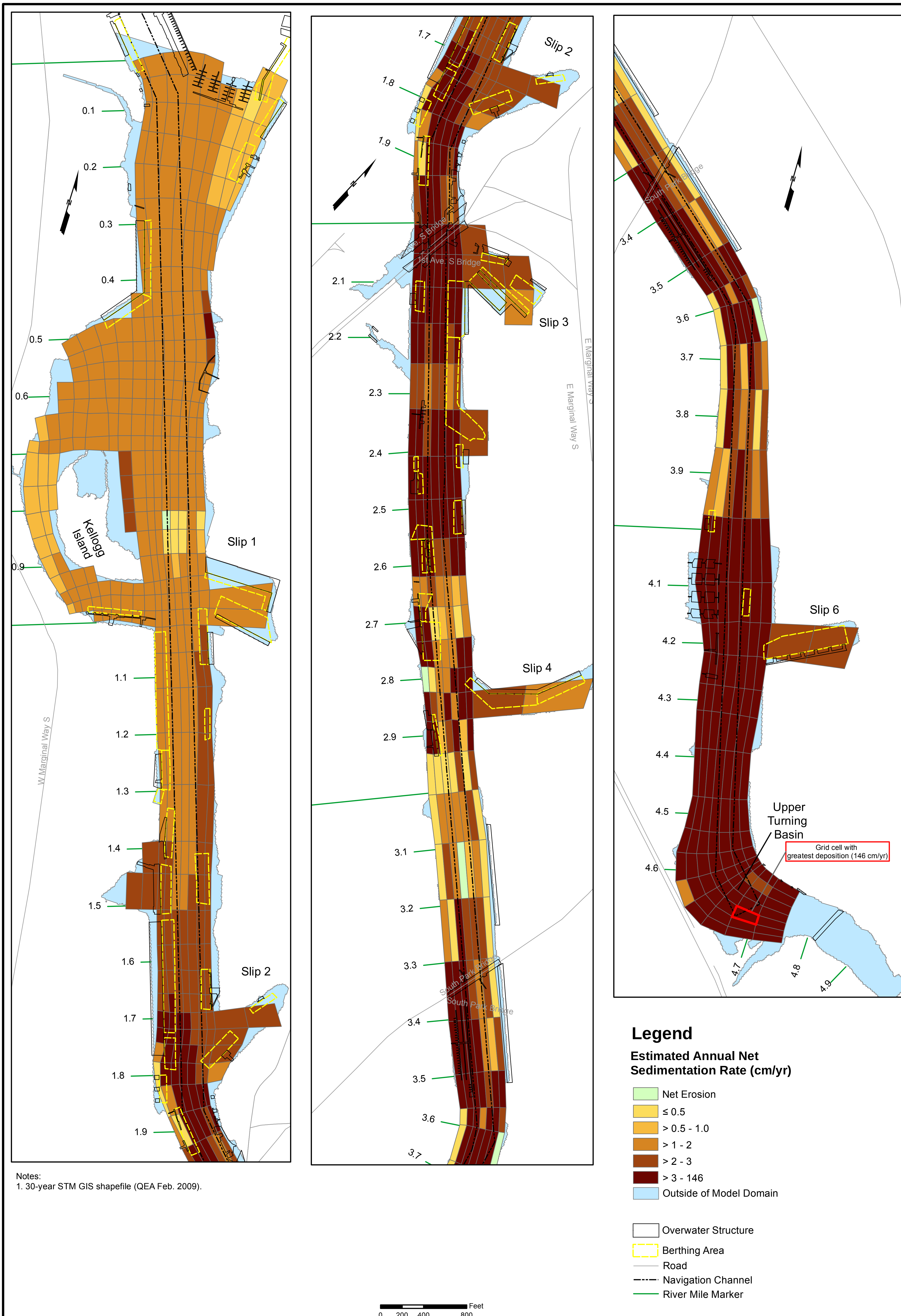
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			2003 Bathymetry, Overwater Structures, and Areas with Evidence of Propeller Wash Scour
DATE: 10/31/12	DWRN:MVI/sea	Revision: 1	FIGURE 2-10

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-10PropScour.mxd



Notes:
1. 30-year STM GIS shapefile (QEA Feb. 2009).

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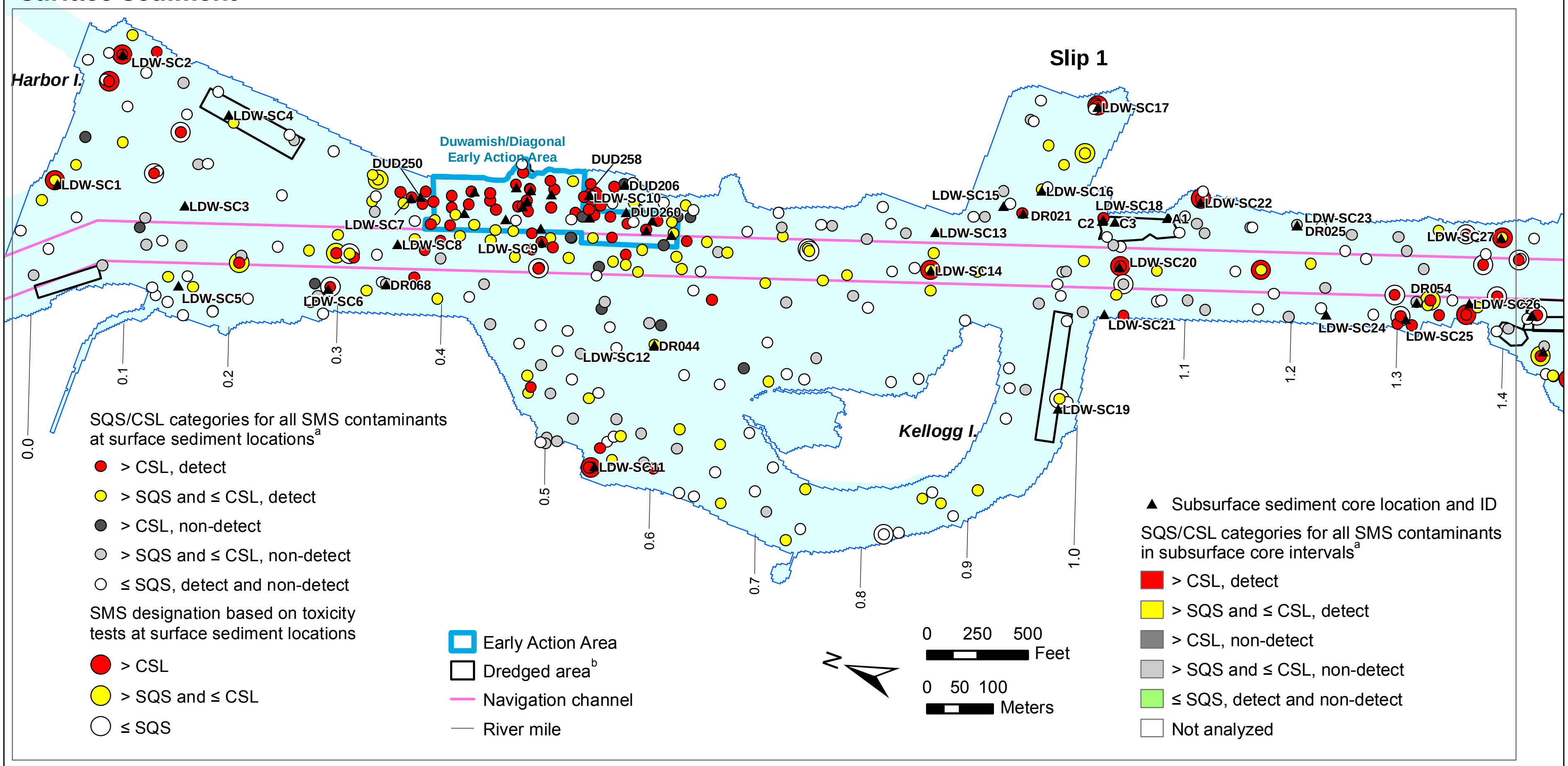
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Estimated Annual Net Sedimentation Rates
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 2-11

Subsurface sediment core locations and exceedances of SQS and CSL (chemical criteria and toxicity combined) in surface sediment



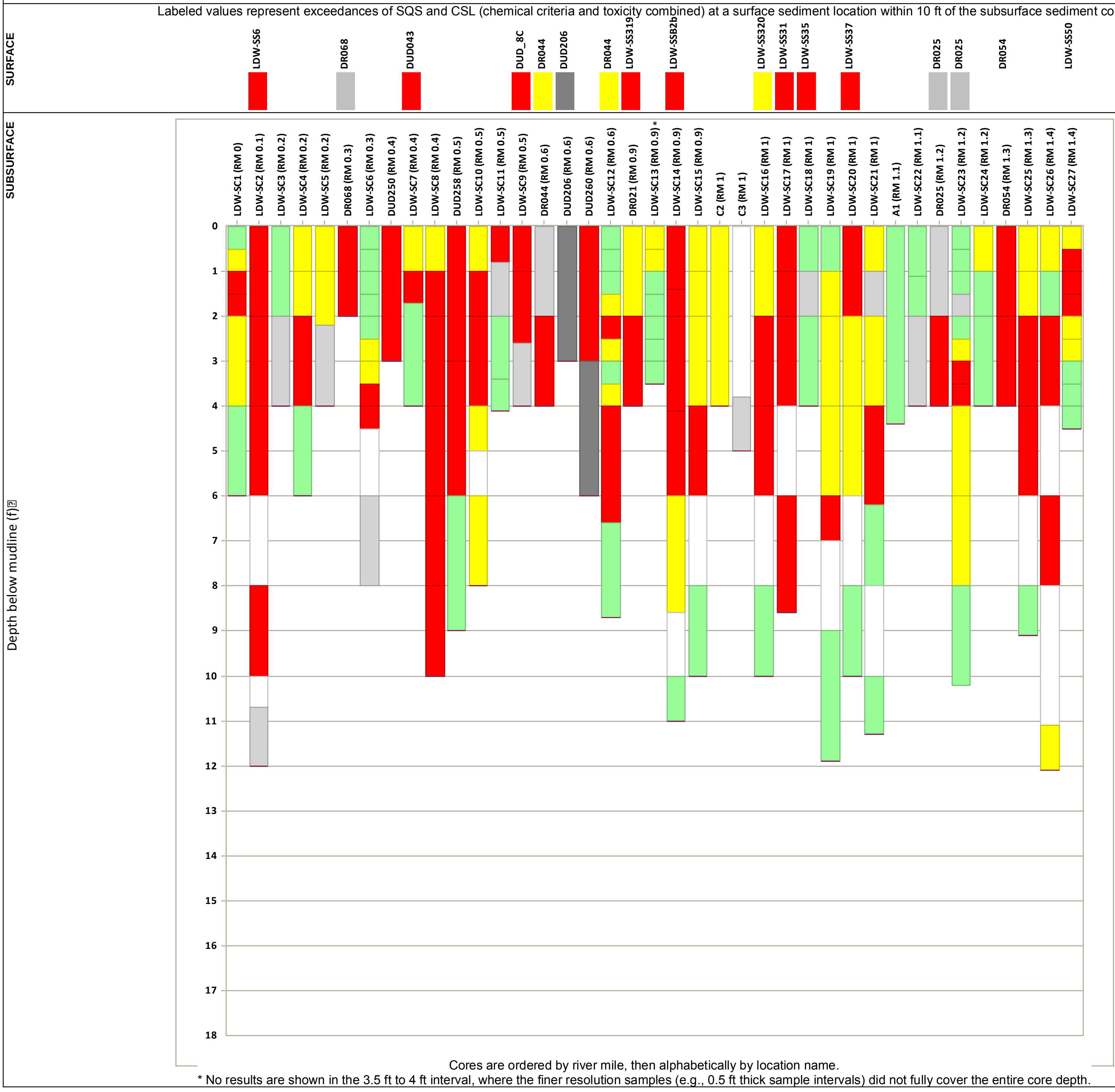
[a] When oc-normalization was not appropriate because TOC content was < 0.5% or > 4.0%, dry-weight concentrations for these locations were compared instead to the LAET and 2LAET.

[b] Subsurface sediment data in the Duwamish/Diagonal Early Action Area were collected prior to dredging and capping or thin-layer placement. In other dredged areas, subsurface data were collected prior to dredging.

Note: This map does not include samples in the Duwamish/Diagonal dredged and capped areas.

Exceedances of SQS and CSL in subsurface sediment cores and co-located (within 10 ft) surface sediment samples

Labeled values represent exceedances of SQS and CSL (chemical criteria and toxicity combined) at a surface sediment location within 10 ft of the subsurface sediment core.



Cores are ordered by river mile, then alphabetically by location name.

* No results are shown in the 3.5 ft to 4 ft interval, where the finer resolution samples (e.g., 0.5 ft thick sample intervals) did not fully cover the entire core depth.

Figure 2-12a. Comparisons of Concentrations of all SMS Contaminants to SMS Criteria (SQS or CSL) in Subsurface Sediment Cores, RM 0.0 to RM 1.4

Lower Duwamish Waterway Final Feasibility Study

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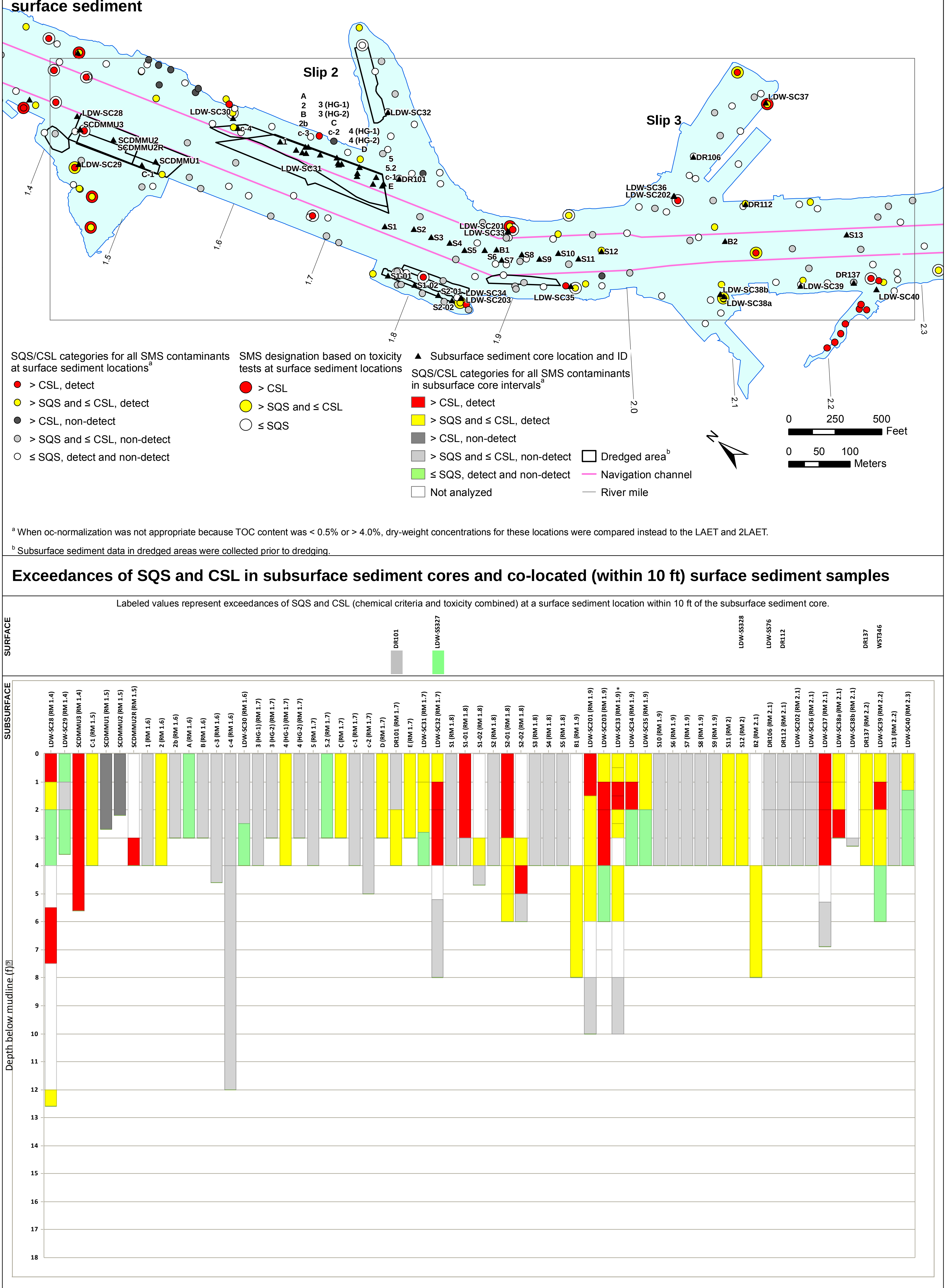
Subsurface sediment core locations and exceedances of SQS and CSL (chemical criteria and toxicity combined) in surface sediment



[a] When oc-normalization was not appropriate because TOC content was < 0.5% or > 4.0%, dry-weight concentrations for these locations were compared instead to the LAET and 2LAET.

[b] Subsurface sediment data in dredged areas were collected prior to dredging.

Exceedances of SQS and CSL in subsurface sediment cores and co-located (within 10 ft) surface sediment samples

Labeled values represent exceedances of SQS and CSL (chemical criteria and toxicity combined) at a surface sediment location within 10 ft of the subsurface sediment core.

Cores are ordered by river mile, then alphabetically by location name.

* No results are shown in the 3 ft to 4 ft interval, where the finer resolution samples (e.g., 0.5 ft thick sample intervals) did not fully cover the entire core depth.

Figure 2-12b. Comparisons of Concentrations of all SMS Contaminants to SMS Criteria (SQS or CSL) in Subsurface Sediment Cores, RM 1.4 to RM 2.3

Lower Duwamish Waterway Final Feasibility Study

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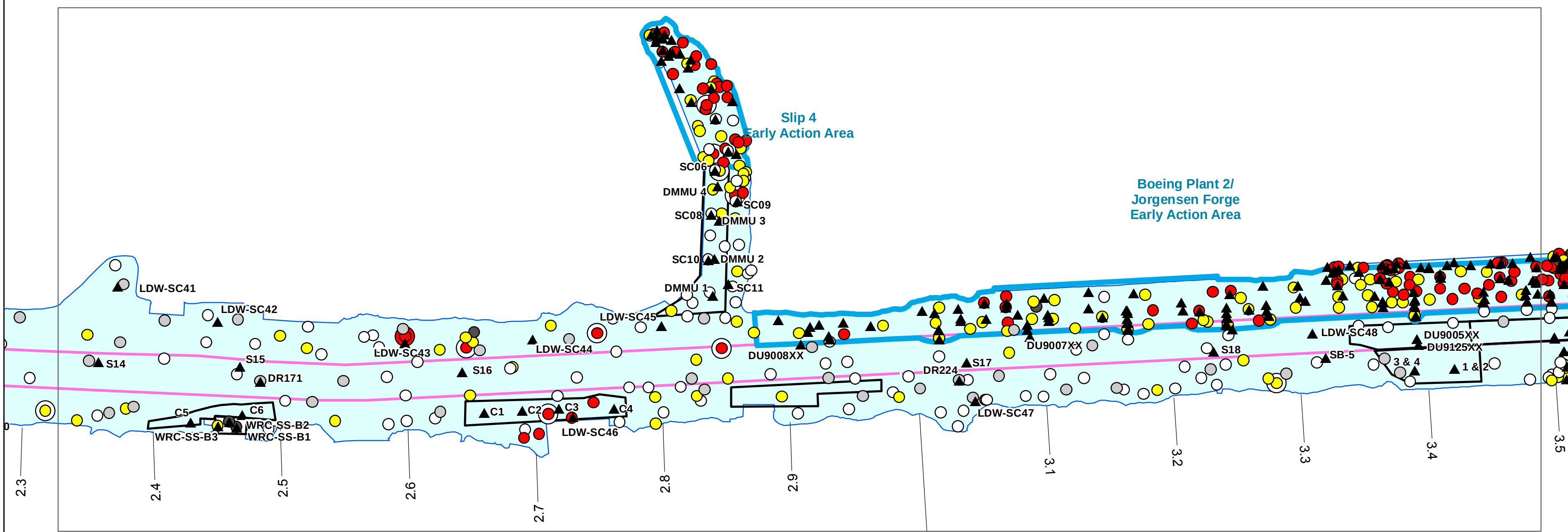
Subsurface sediment core locations and exceedances of SQS and CSL (chemical criteria and toxicity combined) in surface sediment



SQS/CSL categories for all SMS contaminants at surface sediment locations[a]
- > CSL, detect
- > SQS and ≤ CSL, detect
- > CSL, non-detect
- > SQS and ≤ CSL, non-detect
- ≤ SQS, detect and non-detect

SMS designation based on toxicity tests at surface sediment locations
- > CSL
- > SQS and ≤ CSL
- ≤ SQS

▲ Subsurface sediment core location and ID

SQS/CSL categories for all SMS contaminants in subsurface core intervals[a]
- > CSL, detect
- > SQS and ≤ CSL, detect
- > CSL, non-detect
- > SQS and ≤ CSL, non-detect
- ≤ SQS, detect and non-detect
- Not analyzed

- Early Action Area
- Dredged area[b]
- Navigation channel
- River mile

0 250 500 Feet

0 50 100 Meters

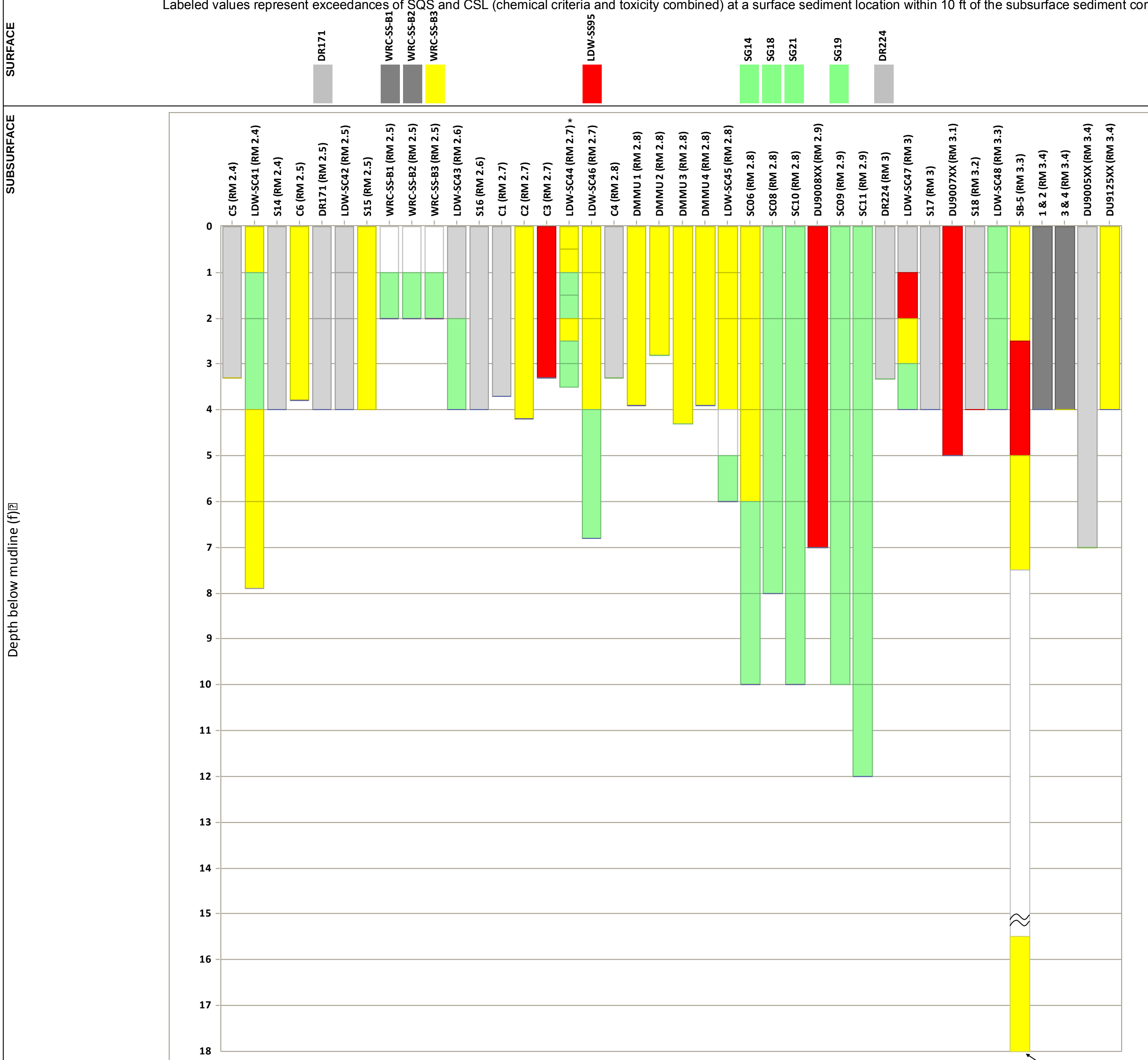
[a] When oc-normalization was not appropriate because TOC content was < 0.5% or > 4.0%, dry-weight concentrations for these locations were compared instead to the LAET and 2LAET.

[b] Subsurface sediment data in dredged areas were collected prior to dredging.

Note: This map does not include samples in the Slip 4 or Boeing Plant 2/Jorgensen Forge Early Action Areas.

Exceedances of SQS and CSL in subsurface sediment cores and co-located (within 10 ft) surface sediment samples

Labeled values represent exceedances of SQS and CSL (chemical criteria and toxicity combined) at a surface sediment location within 10 ft of the subsurface sediment core.



Cores are ordered by river mile, then alphabetically by location name.

Core recovered to a depth of 75 ft.

* No results are shown in the 3.5 ft to 4 ft interval, where the finer resolution samples (e.g., 0.5 ft thick sample intervals) did not fully cover the entire core depth.

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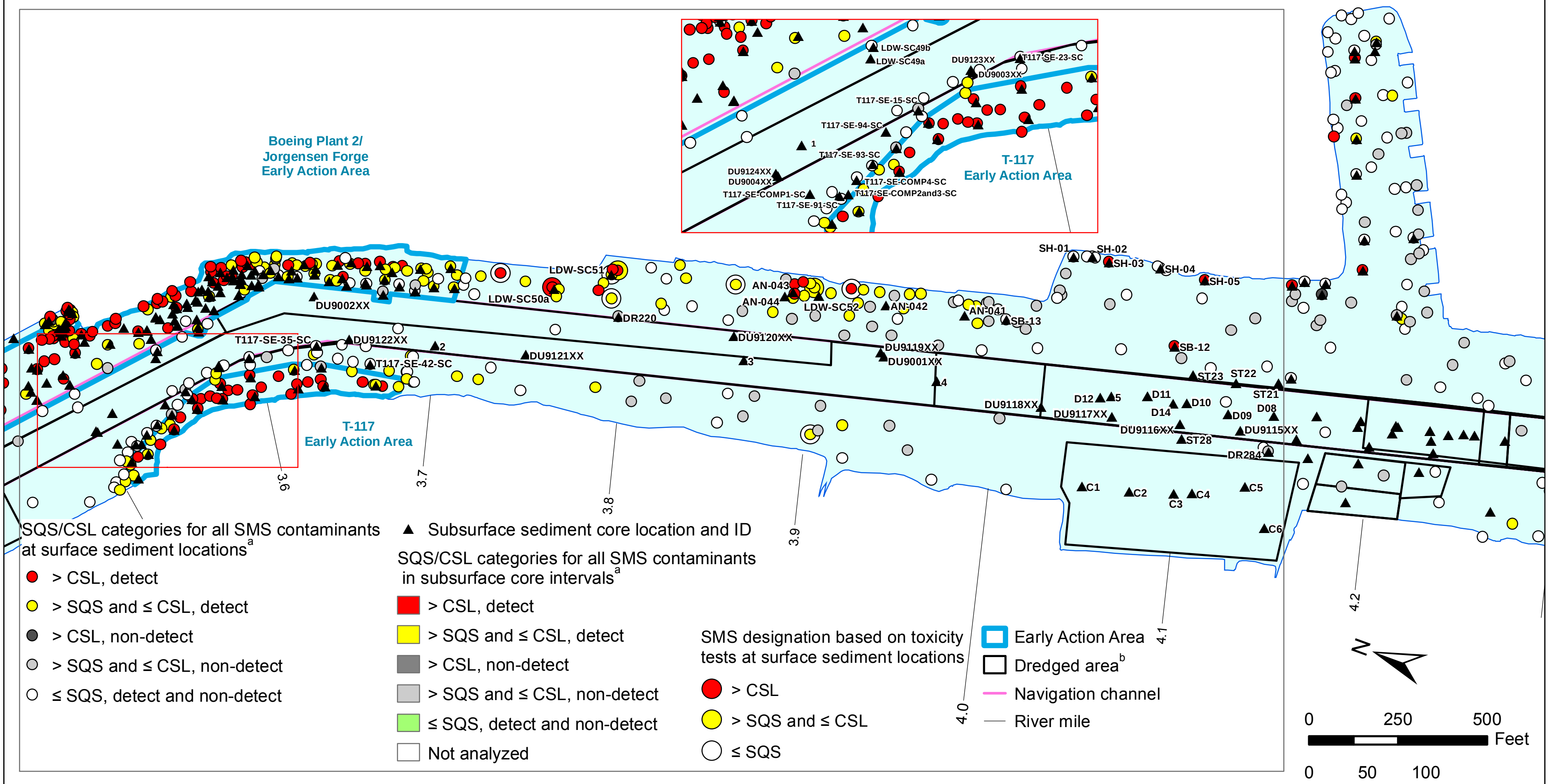
Figure 2-12c. Comparisons of Concentrations of all SMS Contaminants to SMS Criteria (SQS or CSL) in Subsurface Sediment Cores, RM 2.3 to RM 3.5

Lower Duwamish Waterway Final Feasibility Study

Subsurface sediment core locations and exceedances of SQS and CSL (chemical criteria and toxicity combined) in surface sediment



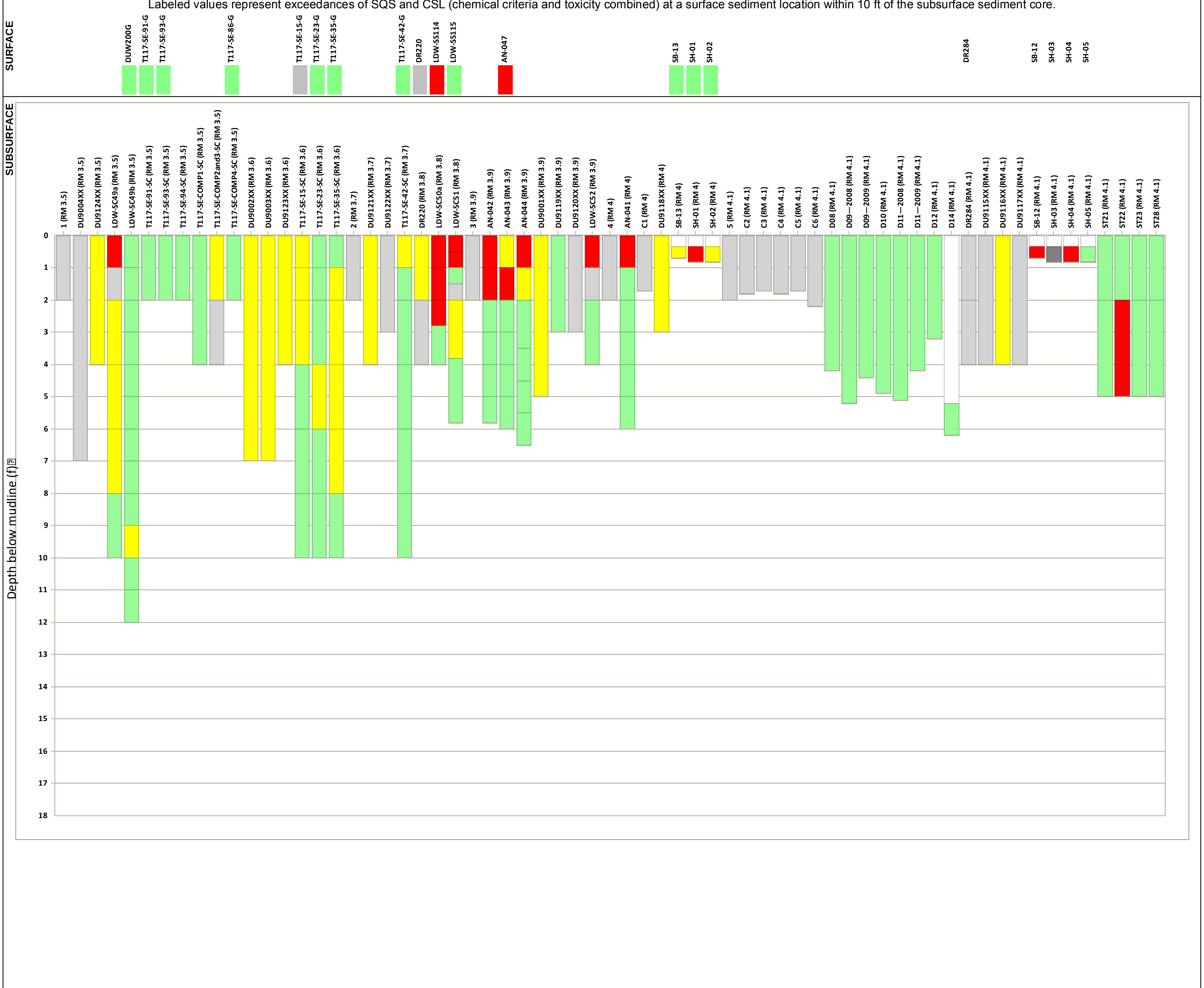
[a] When oc-normalization was not appropriate because TOC content was < 0.5% or > 4.0%, dry-weight concentrations for these locations were compared instead to the LAET and 2LAET.

[b] Subsurface sediment data in dredged areas were collected prior to dredging.

Note: This map does not include samples in the Boeing Plant 2/Jorgensen Forge or T-117 Early Action Areas.

Exceedances of SQS and CSL in subsurface sediment cores and co-located (within 10 ft) surface sediment samples

Labeled values represent exceedances of SQS and CSL (chemical criteria and toxicity combined) at a surface sediment location within 10 ft of the subsurface sediment core.



Cores are ordered by river mile, then alphabetically by location name.

Figure 2-12d. Comparisons of Concentrations of all SMS Contaminants to SMS Criteria (SQS or CSL) in Subsurface Sediment Cores, RM 3.5 to RM 4.3

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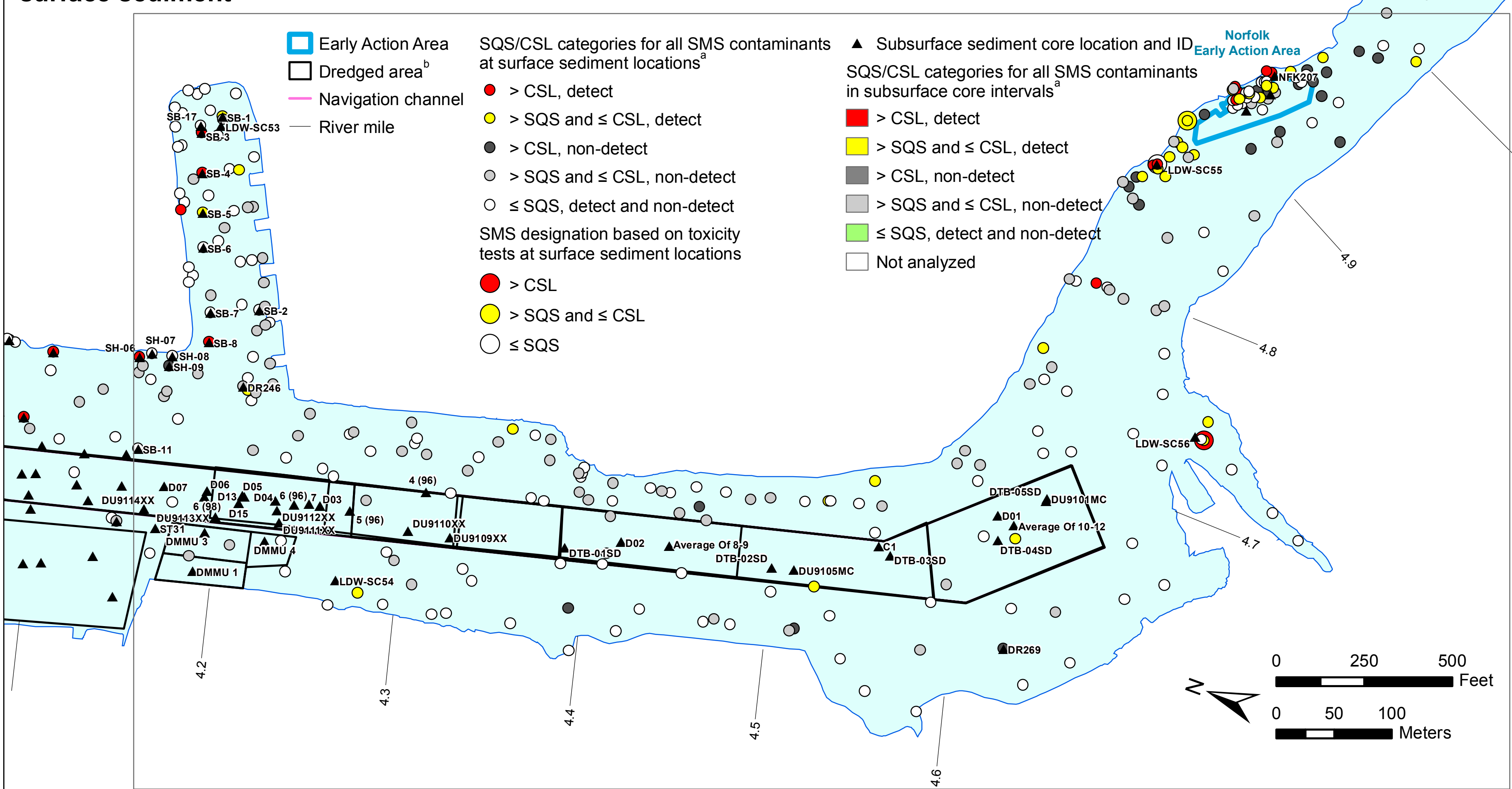
Subsurface sediment core locations and exceedances of SQS and CSL (chemical criteria and toxicity combined) in surface sediment

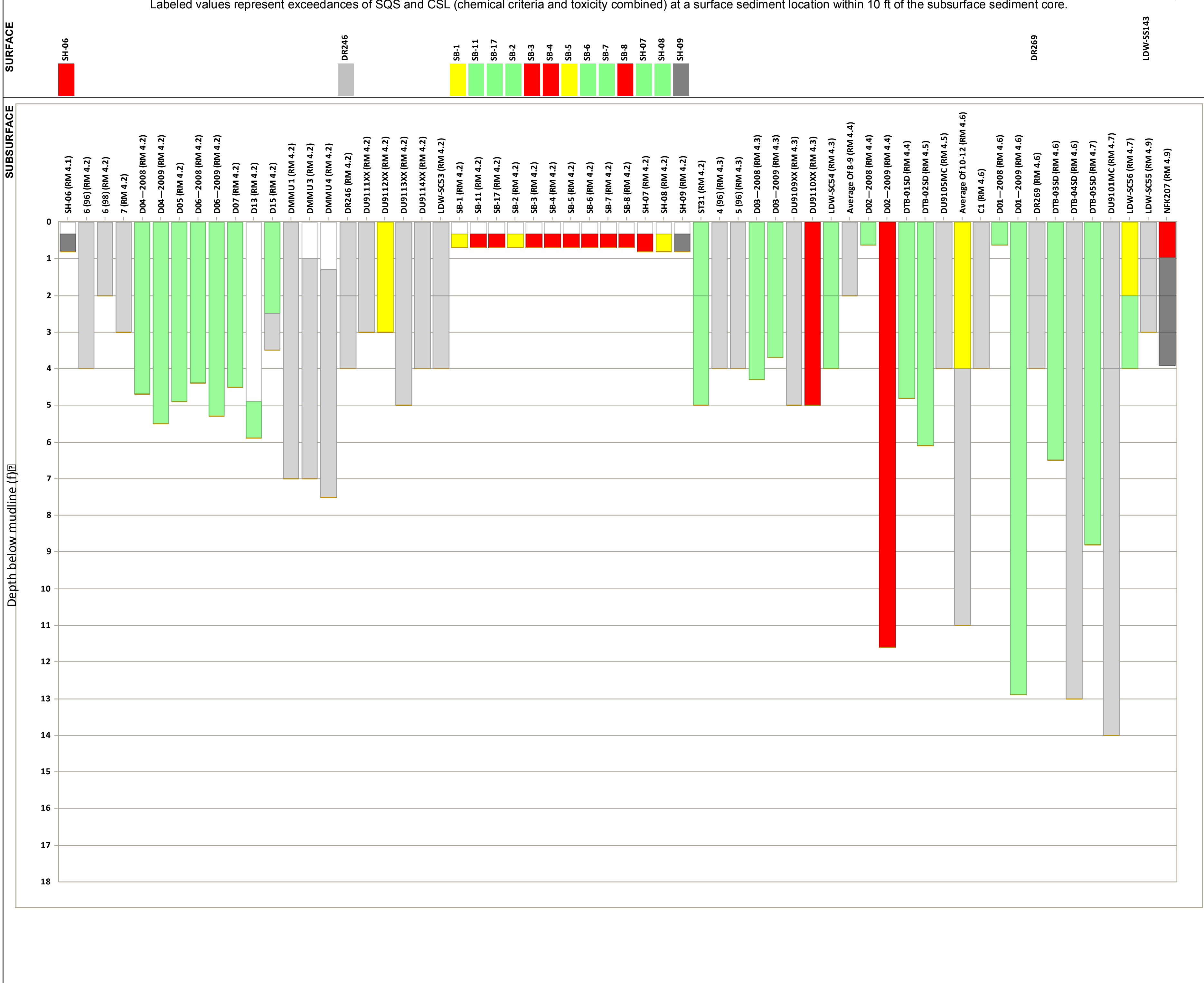


[a] When oc-normalization was not appropriate because TOC content was < 0.5% or > 4.0%, dry-weight concentrations for these locations were compared instead to the LAET and 2LAET.

[b] Subsurface data in the Norfolk Early Action Area were collected prior to dredging and capping. In other dredged areas, subsurface data were collected prior to dredging.

Exceedances of SQS and CSL in subsurface sediment cores and co-located (within 10 ft) surface sediment samples

Labeled values represent exceedances of SQS and CSL (chemical criteria and toxicity combined) at a surface sediment location within 10 ft of the subsurface sediment core.



Cores are ordered by river mile, then alphabetically by location name.

Figure 2-12e. Comparisons of Concentrations of all SMS Contaminants to SMS Criteria (SQS or CSL) in Subsurface Sediment Cores, RM 4.3 to RM 5.0

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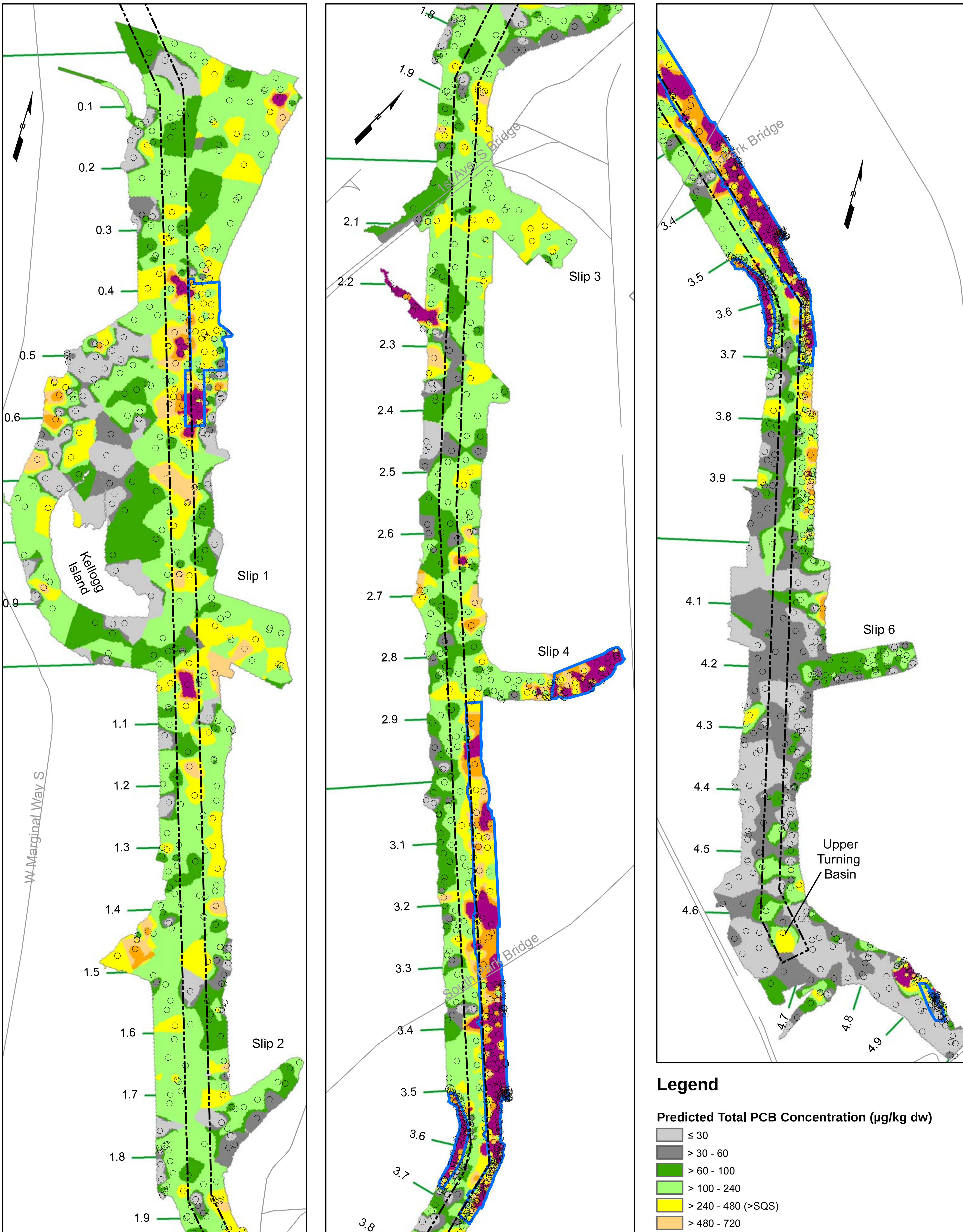

Notes:
1. Grid interpolated using the following parameters: Power 5, nearest neighbors 10/1, search radius 150x150 ft.
2. Sampling dates of the data range from 1991 to 2010.
3. PCB data from FS baseline dataset dated April 28, 2010.
4. SQS value of 240 µg/kg dw based on conversion of 12 mg/kg oc to a dry weight value using 2% TOC.
5. Two outliers (2,900,000 and 230,000 µg/kg dw) at the head of the Trotsky inlet (RM 2.2) were excluded from use in the interpolation.

Legend

Predicted Total PCB Concentration (µg/kg dw)

- ≤ 30
- > 30 - 60
- > 60 - 100
- > 100 - 240
- > 240 - 480 (>SQS)
- > 480 - 720
- > 720 - 1,300
- > 1,300 (>CSL)

PCB Sample Location
Road
Navigation Channel
River Mile Marker
Early Action Area

Feet
0 200 400 800

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Interpolated Total PCB Distribution in Surface Sediment

DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 2-13

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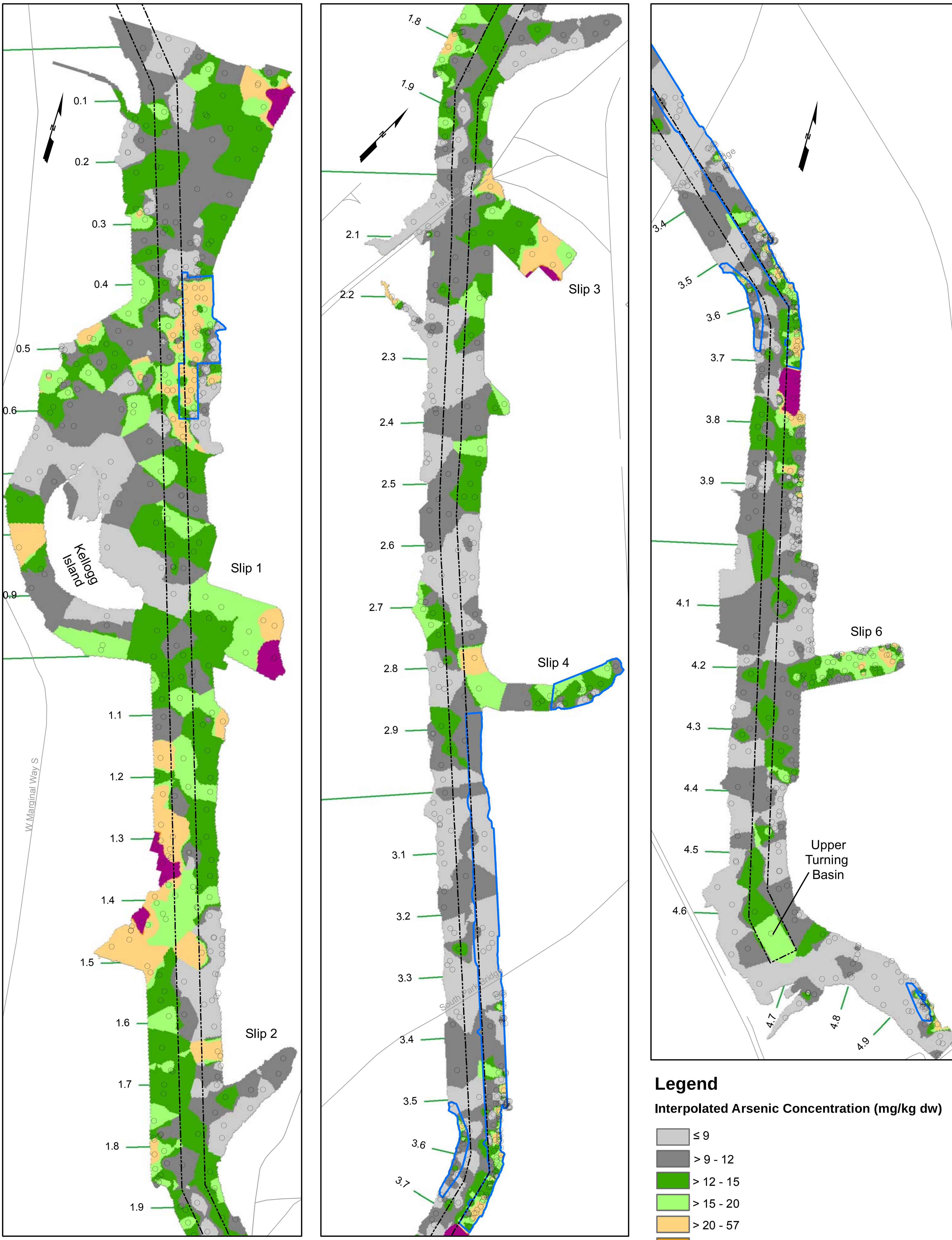

Legend

Interpolated Arsenic Concentration (mg/kg dw)

- ≤ 9
- > 9 - 12
- > 12 - 15
- > 15 - 20
- > 20 - 57
- > 57 - 93 (>SQS)
- > 93 (>CSL)

Arsenic Sample Location

Road

Navigation Channel

River Mile Marker

Early Action Area

0 200 400 800 Feet

Notes:
1. Grid interpolated using the following parameters: Power 5, nearest neighbors 10/1, search radius 150x150 ft.
2. Arsenic data from FS baseline dataset dated April 28, 2010.
3. Sampling dates of the data range from 1991 to 2010.

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			**Interpolated Arsenic Distribution in Surface Sediment**
DATE: 10/31/12	DWRN: MVI/sea	Revision: 0	**FIGURE 2-14**

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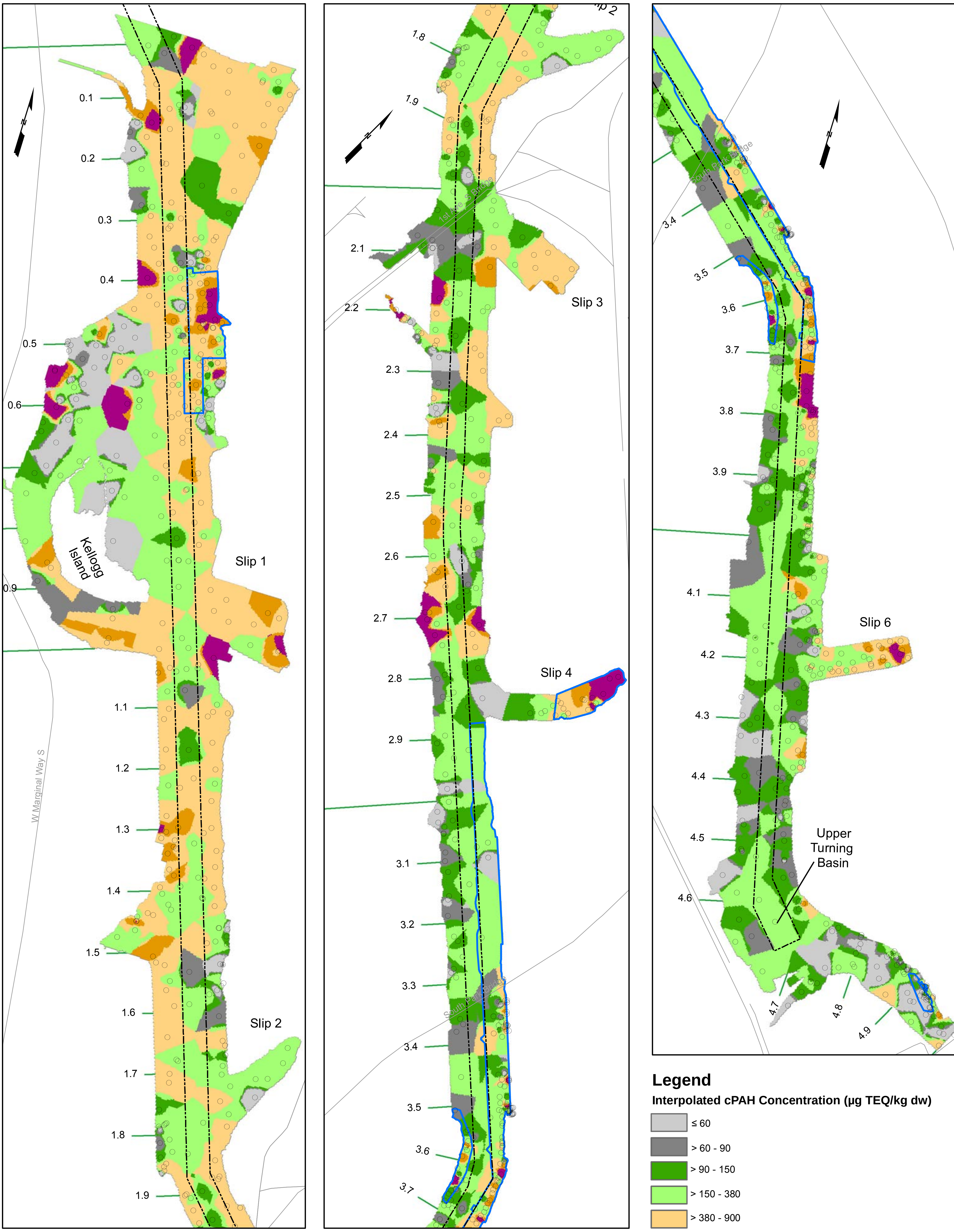

Legend

Interpolated cPAH Concentration (µg TEQ/kg dw)

- ≤ 60
- > 60 - 90
- > 90 - 150
- > 150 - 380
- > 380 - 900
- > 900 - 1,500
- > 1,500
- cPAH Sample Location
- Road
- Navigation Channel
- River Mile Marker
- Early Action Area

Feet
0 200 400 800

Notes:
1. Grid interpolated using the following parameters: Power 6, nearest neighbors 10/1, search radius 150x150 ft.
2. cPAHs calculated with PEFs from Calif. EPA (1994).
3. cPAHs binned based on 10^{-6} beach play, tribal clamming, and netfishing RBTC values, which are 90, 150, and 380 µg TEQ/kg dw, respectively.
4. Sampling dates of the data range from 1991 to 2010.
5. cPAH data from FS baseline dataset dated April 28, 2010.

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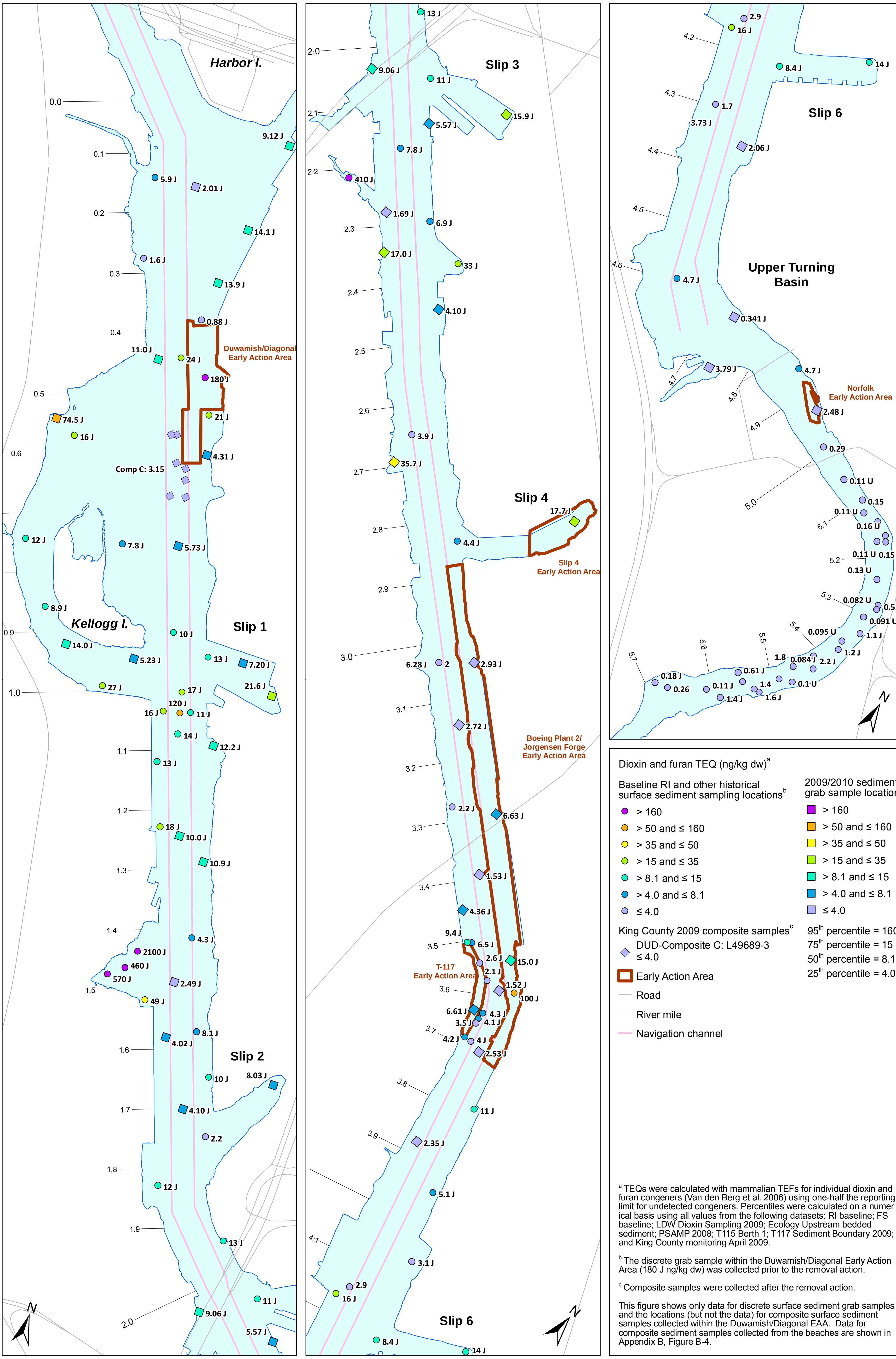
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Interpolated cPAH Distribution in Surface Sediment
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 2-15

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-15cPAHInterp.mxd



a TEQs were calculated with mammalian TEFs for individual dioxin and furan congeners (Van den Berg et al. 2006) using one-half the reporting limit for undetected congeners. Percentiles were calculated on a numerical basis using all values from the following datasets: RI baseline; FS baseline; LDW Dioxin Sampling 2009; Ecology Upstream bedded sediment; PSAMP 2008; T115 Berth 1; T117 Sediment Boundary 2009; and King County monitoring April 2009.

b The discrete grab sample within the Duwamish/Diagonal Early Action Area (180 J ng/kg dw) was collected prior to the removal action.

c Composite samples were collected after the removal action.

This figure shows only data for discrete surface sediment grab samples and the locations (but not the data) for composite surface sediment samples collected within the Duwamish/Diagonal EAA. Data for composite sediment samples collected from the beaches are shown in Appendix B, Figure B-4.

0 0.1 0.2 Miles

0 0.1 0.2 Kilometers

Scale is the same for each inset map

Figure 2-16. Dioxin and Furan TEQ Results for the 2009/2010 LDW Surface Sediment Sampling Event, Including Results from Historical Surface Sediment Sampling Events

Lower Duwamish Waterway Final Feasibility Study

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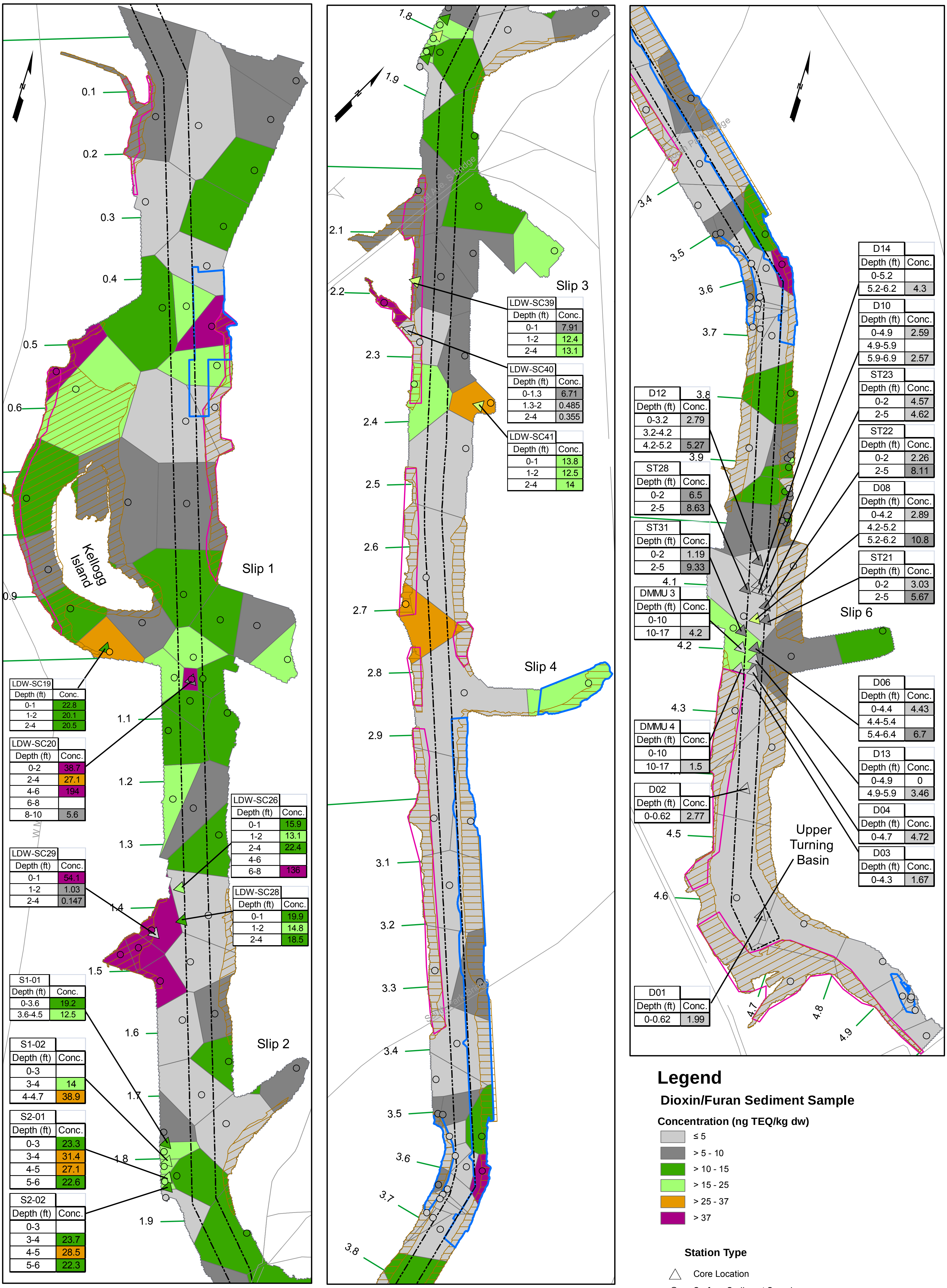
Prepared by inmamn, 10/31/2012; L:\Lower Duwamish FS\FS_Final_GIS\Od\2012\2012-10-15 LDW FS WW GIS Maps and Data\Fig 2-16 4110 All dioxin and furan data1.mxd



Legend

Dioxin/Furan Sediment Sample

Concentration (ng TEQ/kg dw)

- ≤ 5
- > 5 - 10
- > 10 - 15
- > 15 - 25
- > 25 - 37
- > 37

Station Type

- Core Location
- Surface Sediment Sample

- Potential Clamming Area
- Beach Play Area
- Early Action Area
- Road
- Navigation Channel
- River Mile Marker

Core ID

DMMU 3	
Depth (ft)	Conc.
0-10	
10-17	4.2

Skipped Interval — Sample Interval — No Data — Concentration (ng TEQ/kg dw)

0 200 400 800 Feet

Notes:
1. Thiessen polygons derived from 123 surface sediment locations; dataset includes the following surface sediment data: 25 RI samples, 41 2009/2010 LDWG samples, 26 EPA Site Investigation samples, 8 T117 perimeter samples, 5 T115 intertidal samples, 5 Ecology bedded sediment samples, 12 Kenworth PACCAR samples, and 1 location from Duwamish/Diagonal 2009 composite C.
2. Sampling dates range from 1998 to 2010.
3. Core locations symbolized by concentration in shallowest interval analyzed. The cores were not used to draw the Thiessen polygons.
4. The Thiessen polygons shown in this figure are based only on data for discrete surface sediment grab samples. Data for composite sediment samples collected from the beaches are shown in Appendix B, Figure B-4.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-17Diox.mxd

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60150279-14.34

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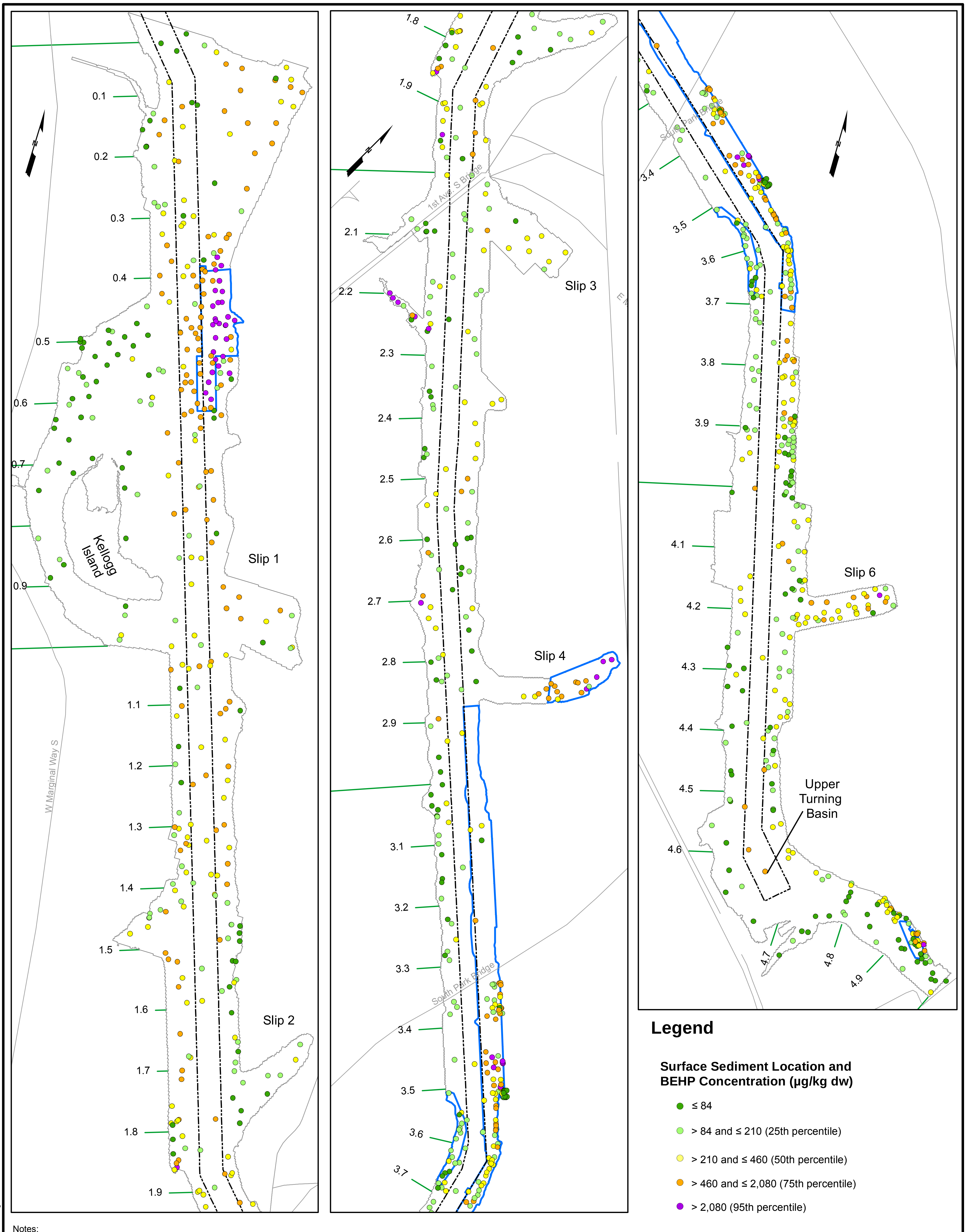
Distribution of Dioxins/Furans in Surface and Subsurface Sediment

FIGURE 2-17



Legend

Surface Sediment Location and BEHP Concentration (µg/kg dw)

- ≤ 84
- > 84 and ≤ 210 (25th percentile)
- > 210 and ≤ 460 (50th percentile)
- > 460 and ≤ 2,080 (75th percentile)
- > 2,080 (95th percentile)

- Road
- Navigation Channel
- River Mile Marker
- Early Action Area

Feet
0 200 400 800

Notes:
1. BEHP = bis(2-ethylhexyl)phthalate.
2. BEHP concentrations binned by percentiles.
3. Sample dates for the data range from 1991 to 2009.
4. BEHP data from FS baseline dataset dated April 28, 2010.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-18BEHP.mxd

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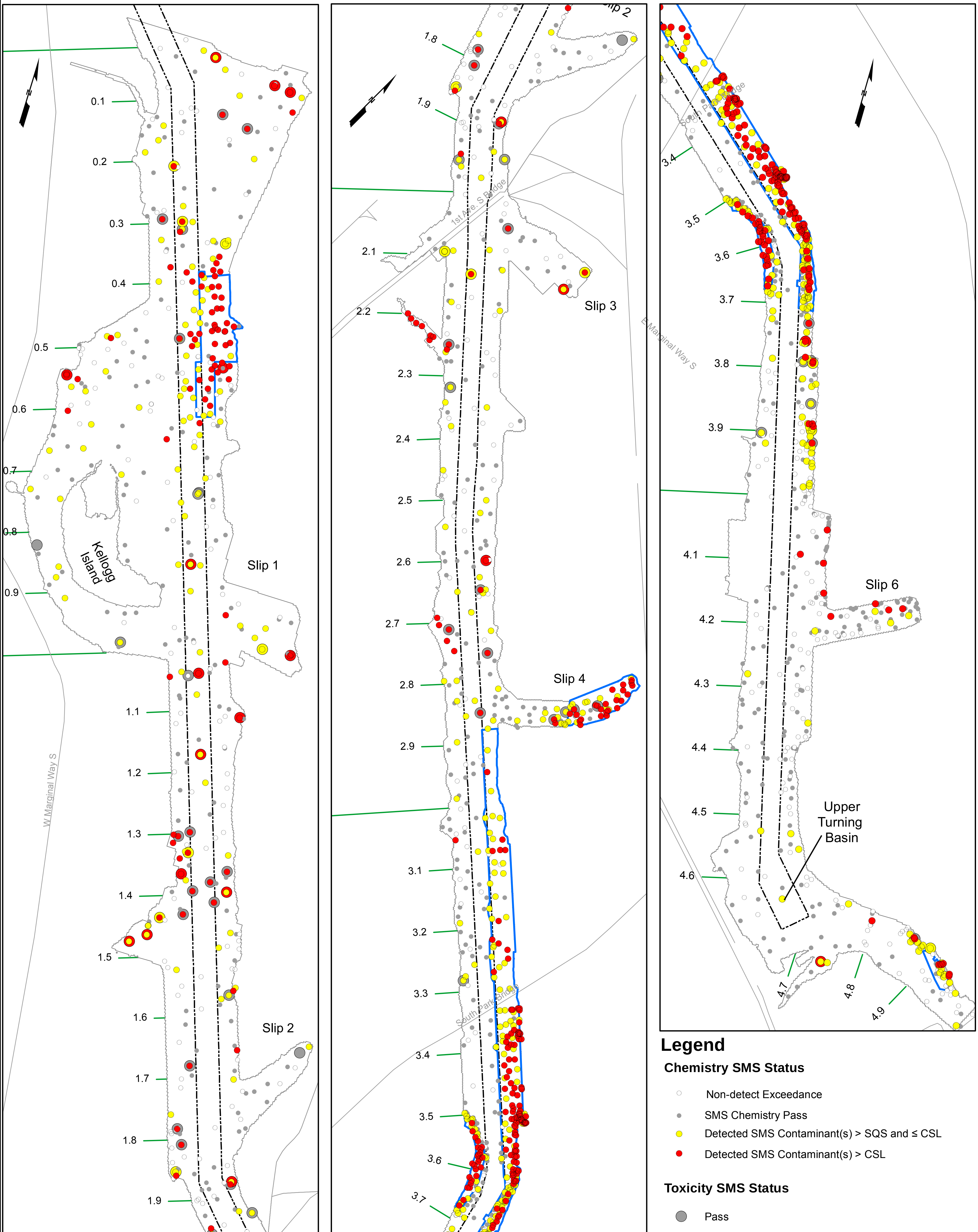
Distribution of BEHP in Surface Sediment

FIGURE 2-18



Legend

Chemistry SMS Status

- Non-detect Exceedance
- SMS Chemistry Pass
- Detected SMS Contaminant(s) > SQS and ≤ CSL
- Detected SMS Contaminant(s) > CSL

Toxicity SMS Status

- Pass
- > SQS and ≤CSL
- >CSL

- Road
- Navigation Channel
- River Mile Marker
- Early Action Area

Notes:
1. Chemistry exceedances address all detected SMS contaminant(s), including total PCBs and arsenic.
2. The sampling dates of the surface sediment data range from 1991 to 2010.
3. Contaminant exceedance data from FS baseline dataset dated April 28, 2010.

Feet
0 200 400 800

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-19ToxExceed.mxd

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Distribution of SMS Toxicity and Chemistry Status in Surface Sediment
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 2-19

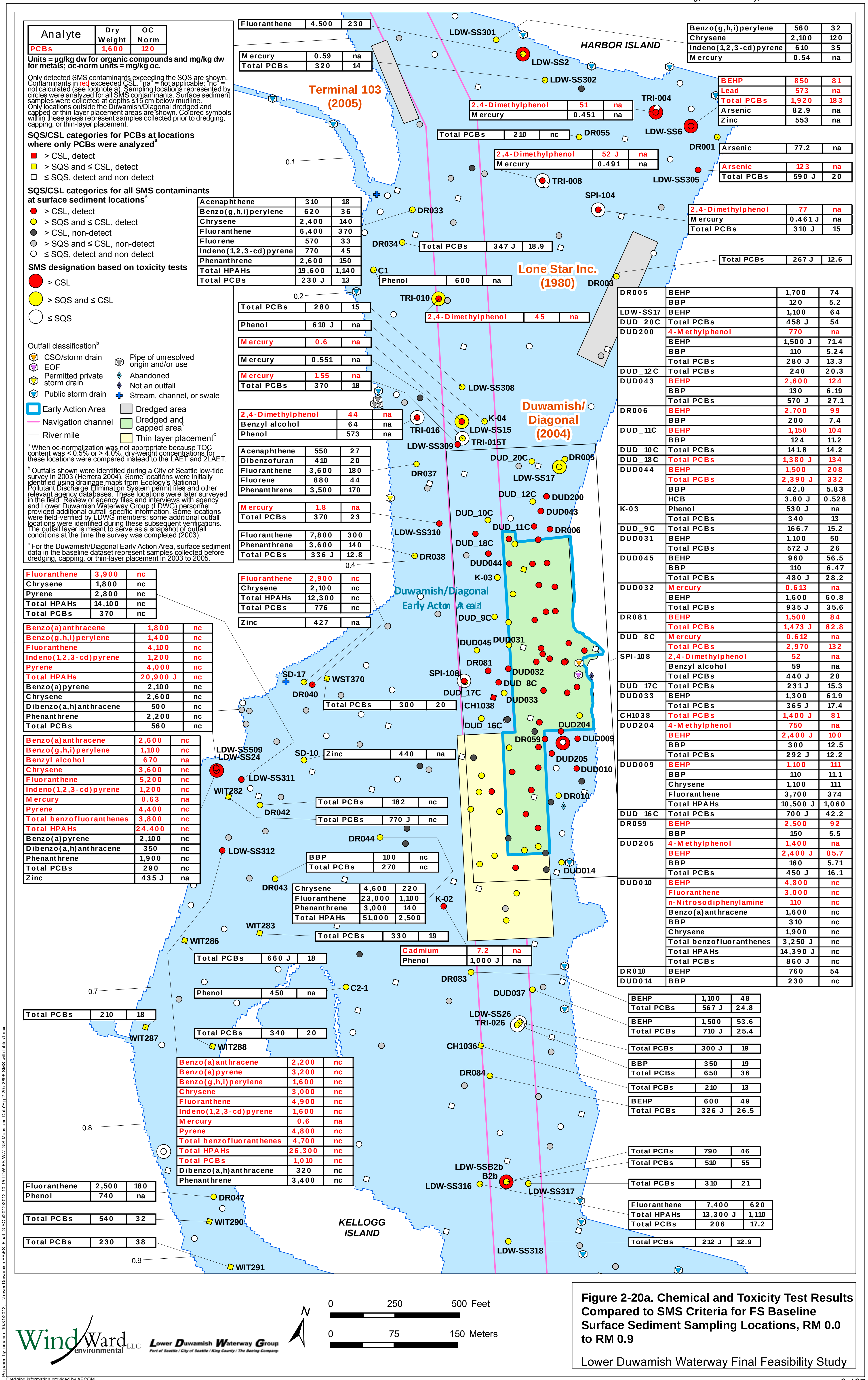

Figure 2-20a. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 0.0 to RM 0.9

Lower Duwamish Waterway Final Feasibility Study

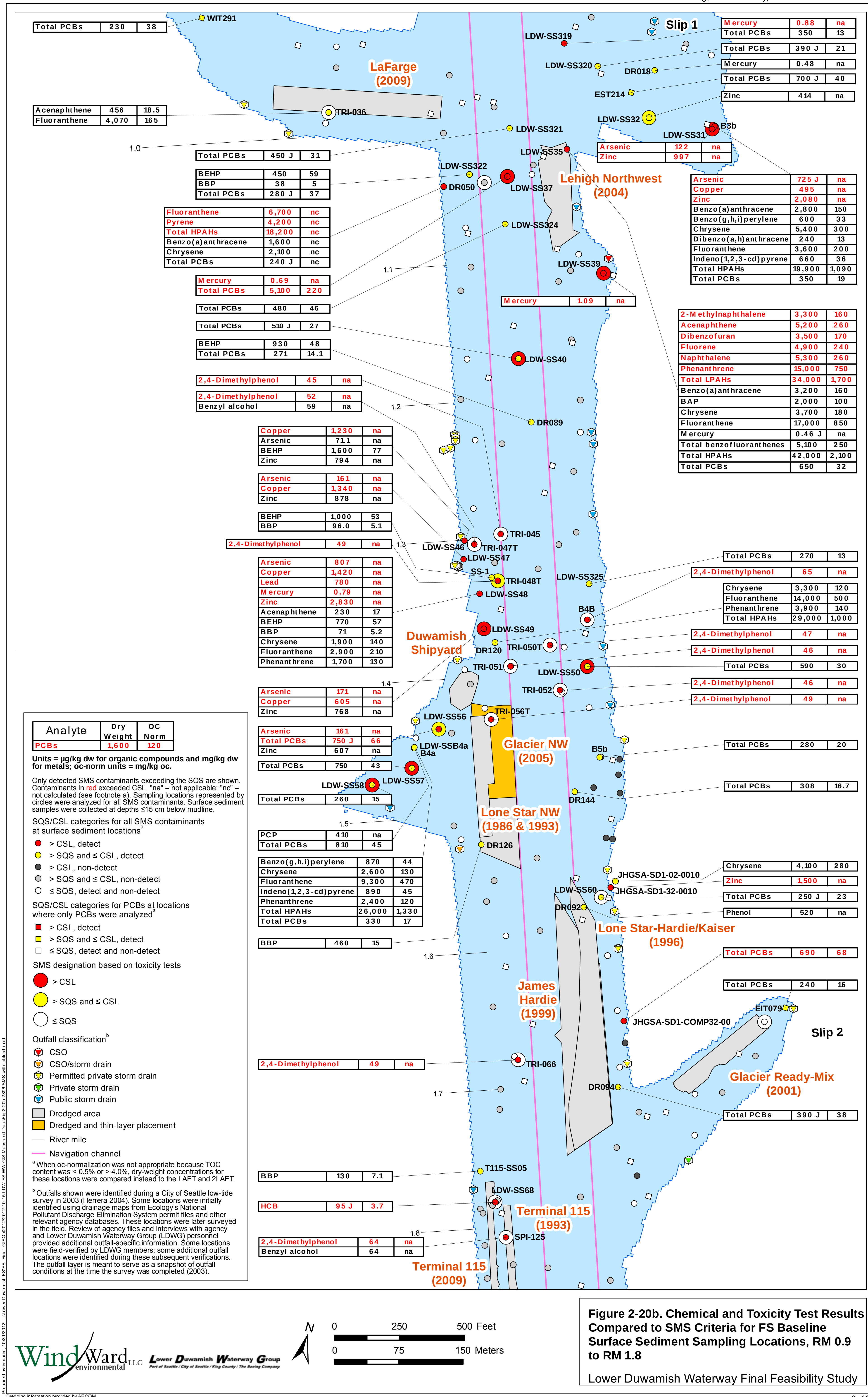

Figure 2-20b. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 0.9 to RM 1.8

Lower Duwamish Waterway Final Feasibility Study

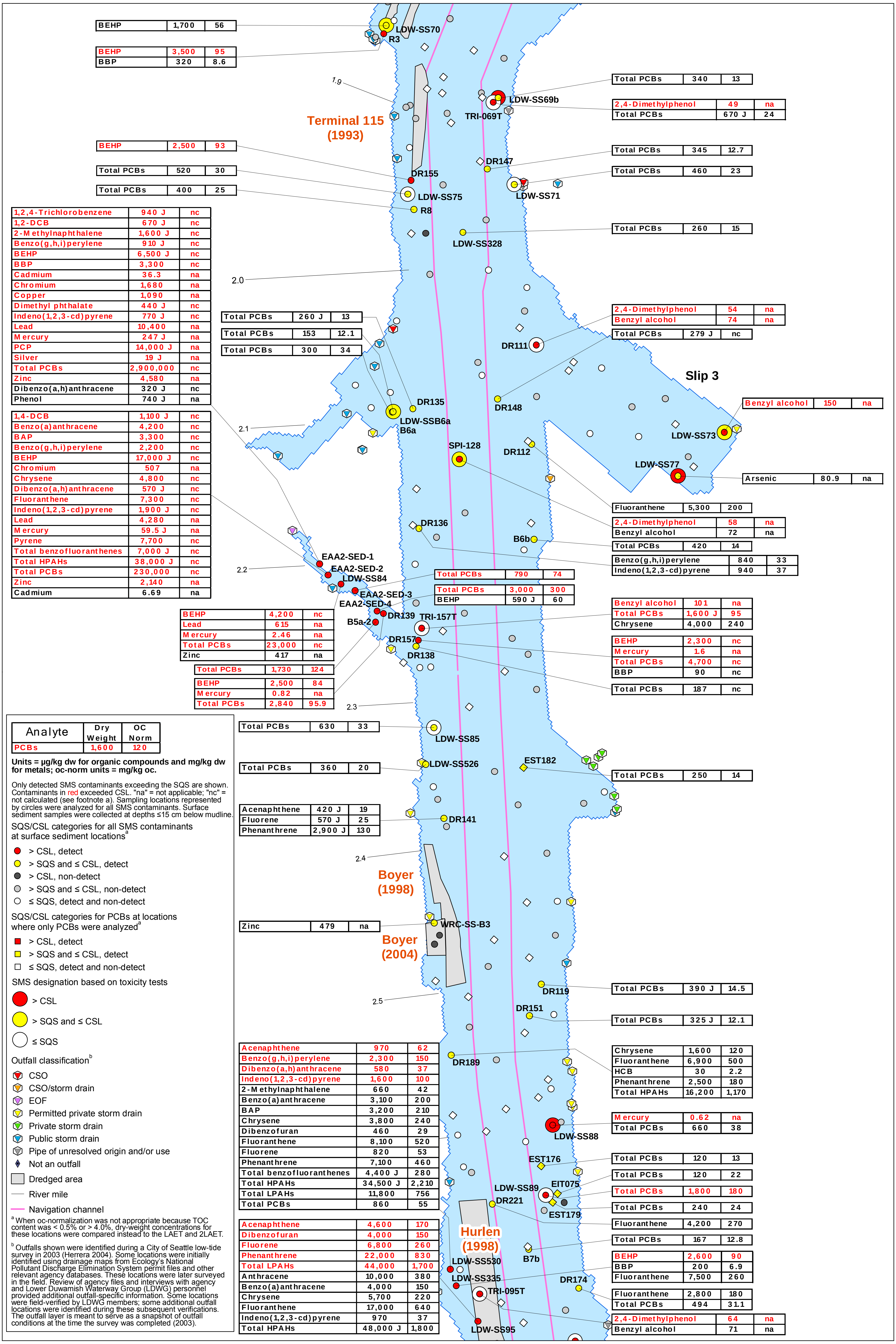

Figure 2-20c. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 1.8 to RM 2.7

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Dredging information provided by AECOM.

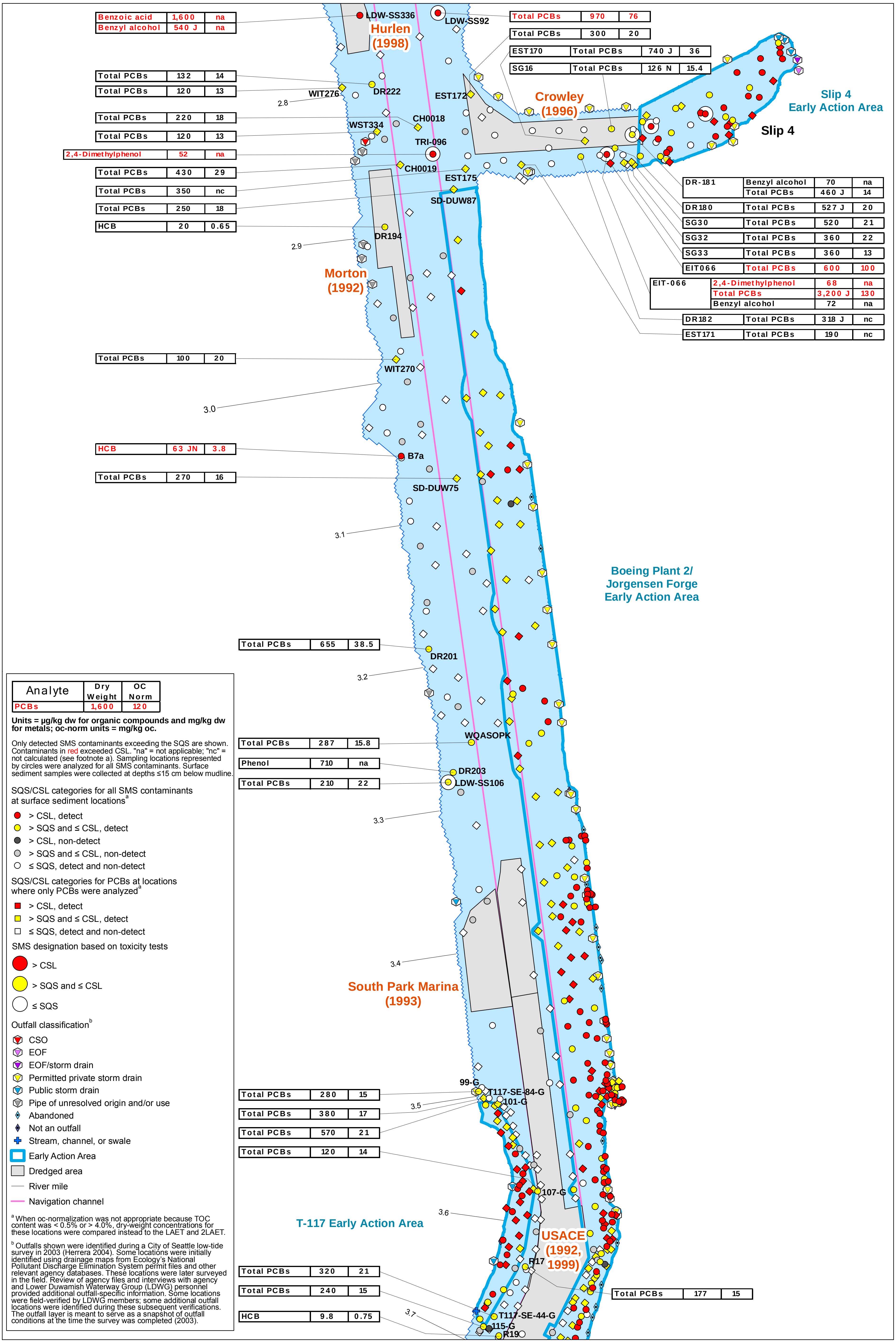

Figure 2-20d. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 2.8 to RM 3.7

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Dredging information provided by AECOM.

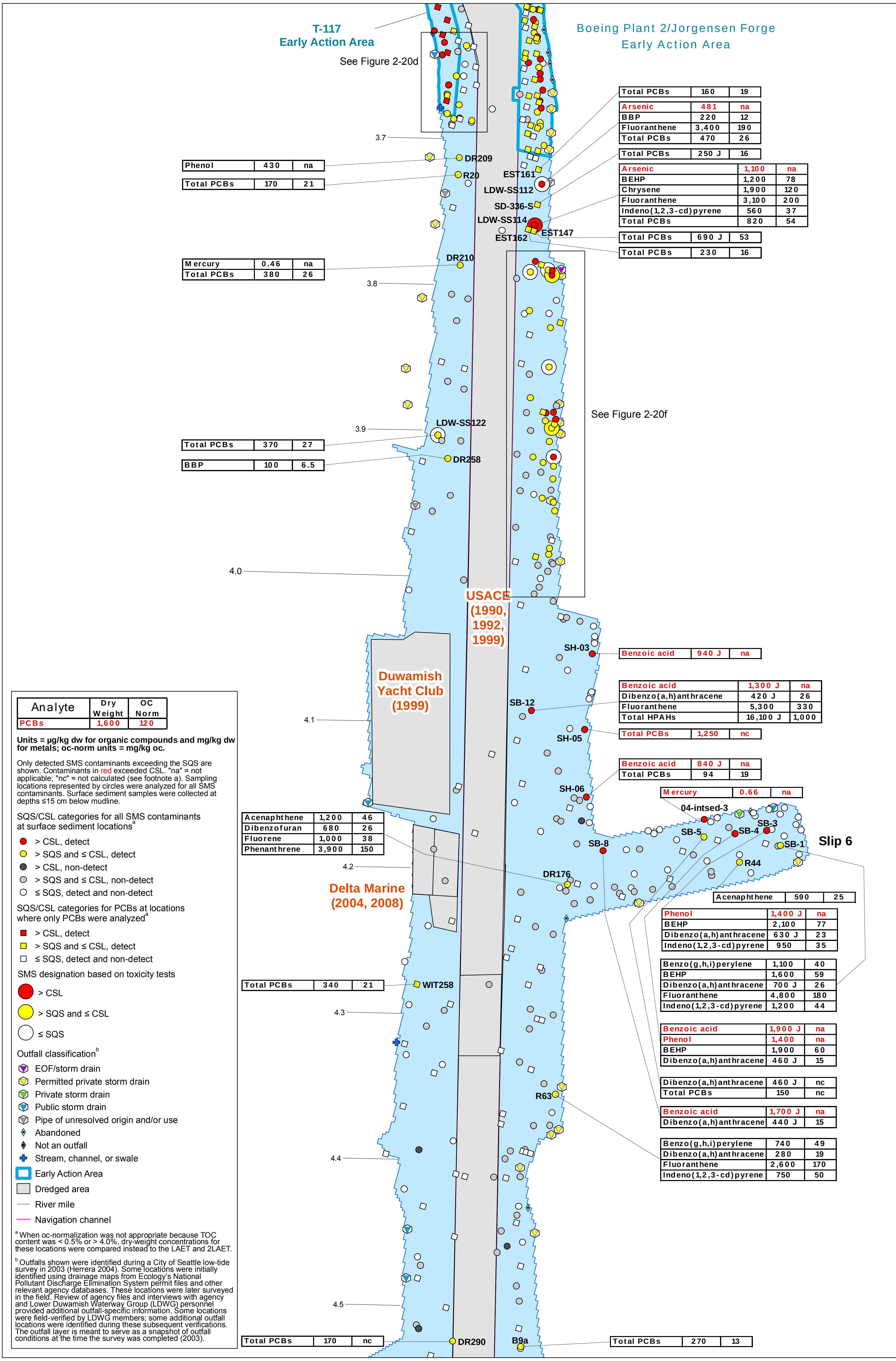

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Figure 2-20e. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 3.7 to RM 4.5

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Dredging information provided by AECOM.

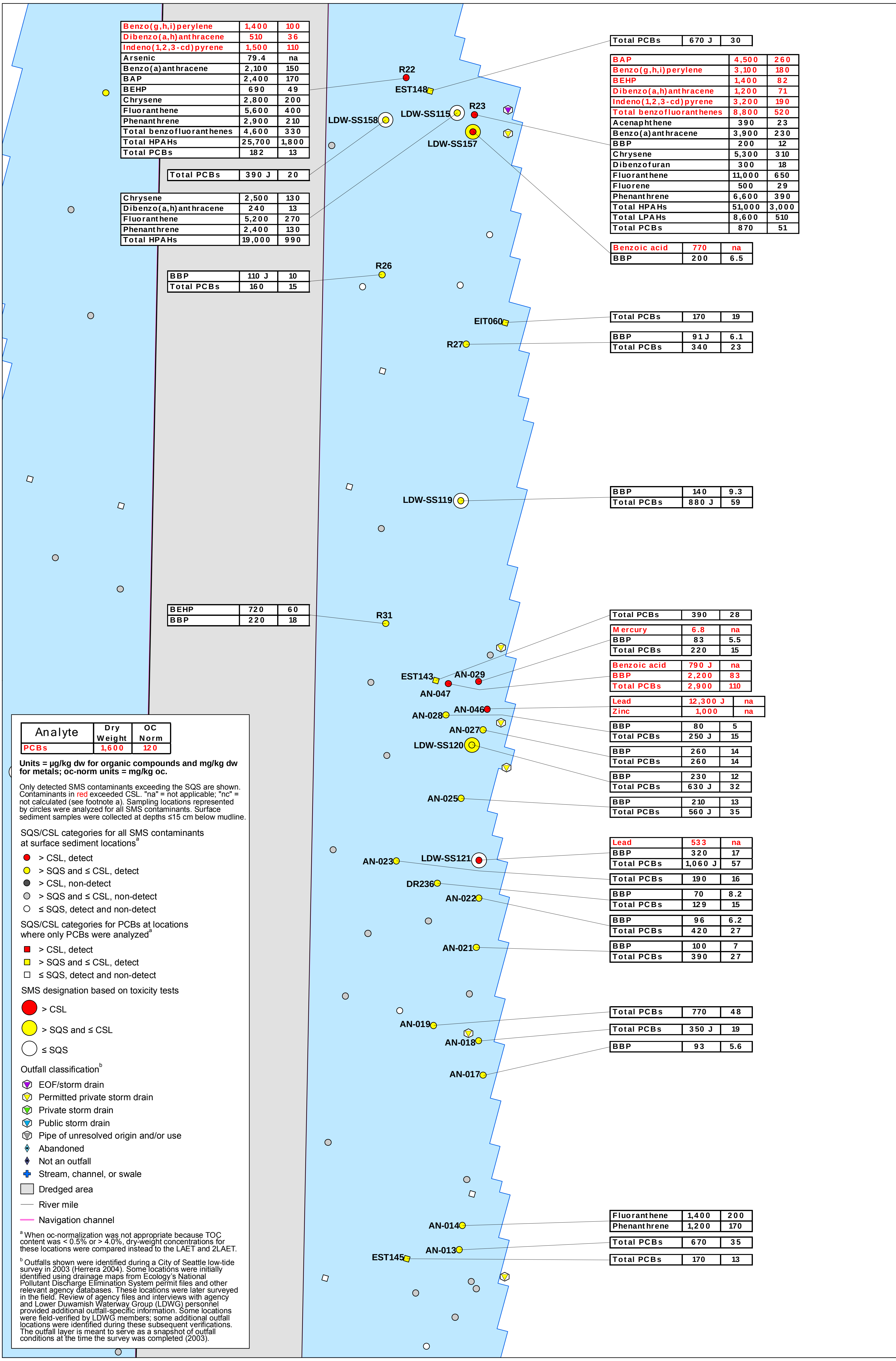

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Figure 2-20f. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 3.8 to RM 4.0

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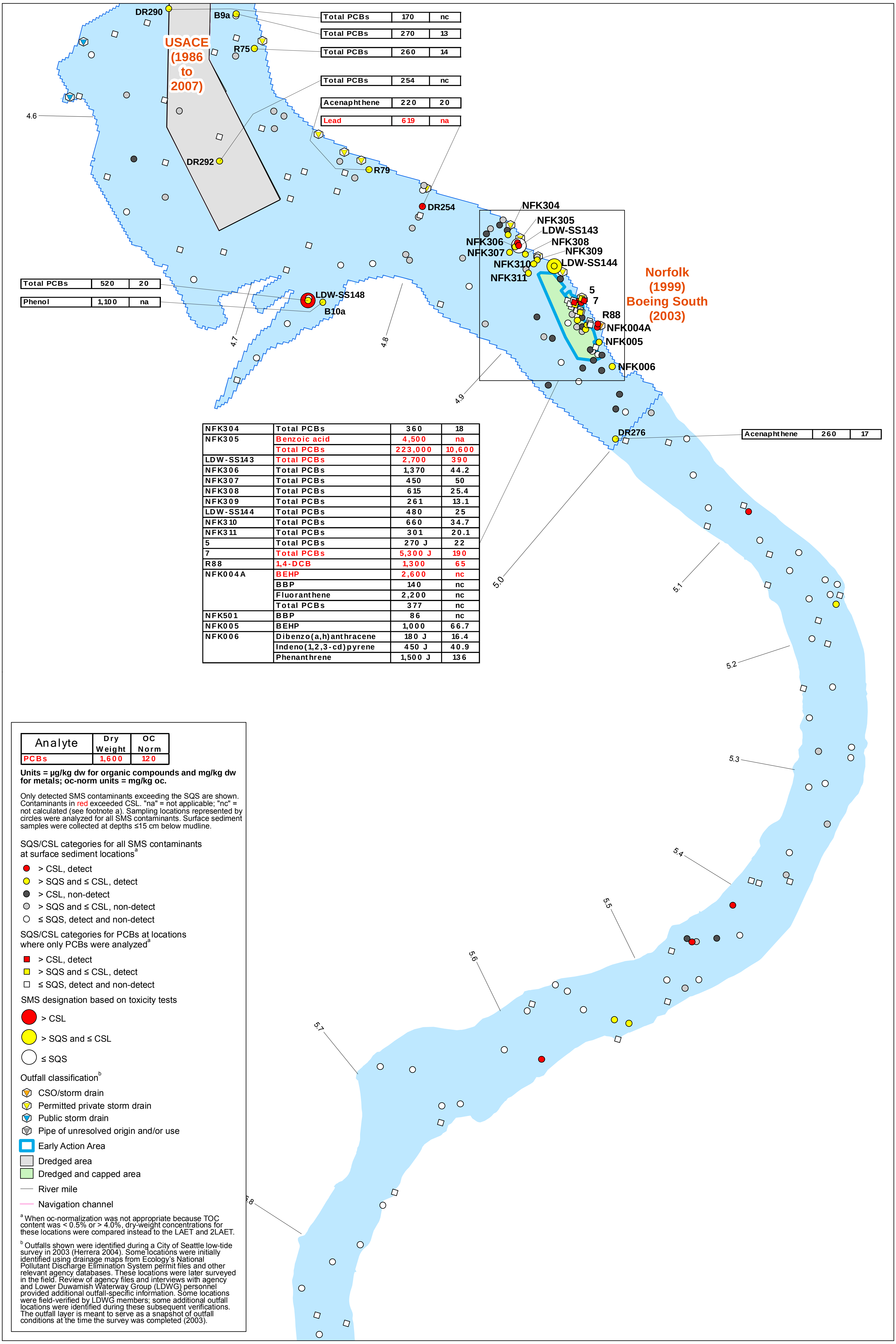
Dredging information provided by AECOM.



NFK304	Total PCBs	360	18
NFK305	Benzoic acid	4,500	na
	Total PCBs	223,000	10,600
LDW-SS143	Total PCBs	2,700	390
NFK306	Total PCBs	1,370	44.2
NFK307	Total PCBs	450	50
NFK308	Total PCBs	615	25.4
NFK309	Total PCBs	261	13.1
LDW-SS144	Total PCBs	480	25
NFK310	Total PCBs	660	34.7
NFK311	Total PCBs	301	20.1
5	Total PCBs	270 J	22
7	Total PCBs	5,300 J	190
R88	1,4-DCB	1,300	65
NFK004A	BEHP	2,600	nc
	BBP	140	nc
	Fluoranthene	2,200	nc
	Total PCBs	377	nc
NFK501	BBP	86	nc
NFK005	BEHP	1,000	66.7
NFK006	Dibenzo(a,h)anthracene	180 J	16.4
	Indeno(1,2,3-cd)pyrene	450 J	40.9
	Phenanthrene	1,500 J	136

N
0 250 500 Feet
0 75 150 Meters

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Figure 2-20g. Chemical and Toxicity Test Results Compared to SMS Criteria for FS Baseline Surface Sediment Sampling Locations, RM 4.5 to RM 5.0

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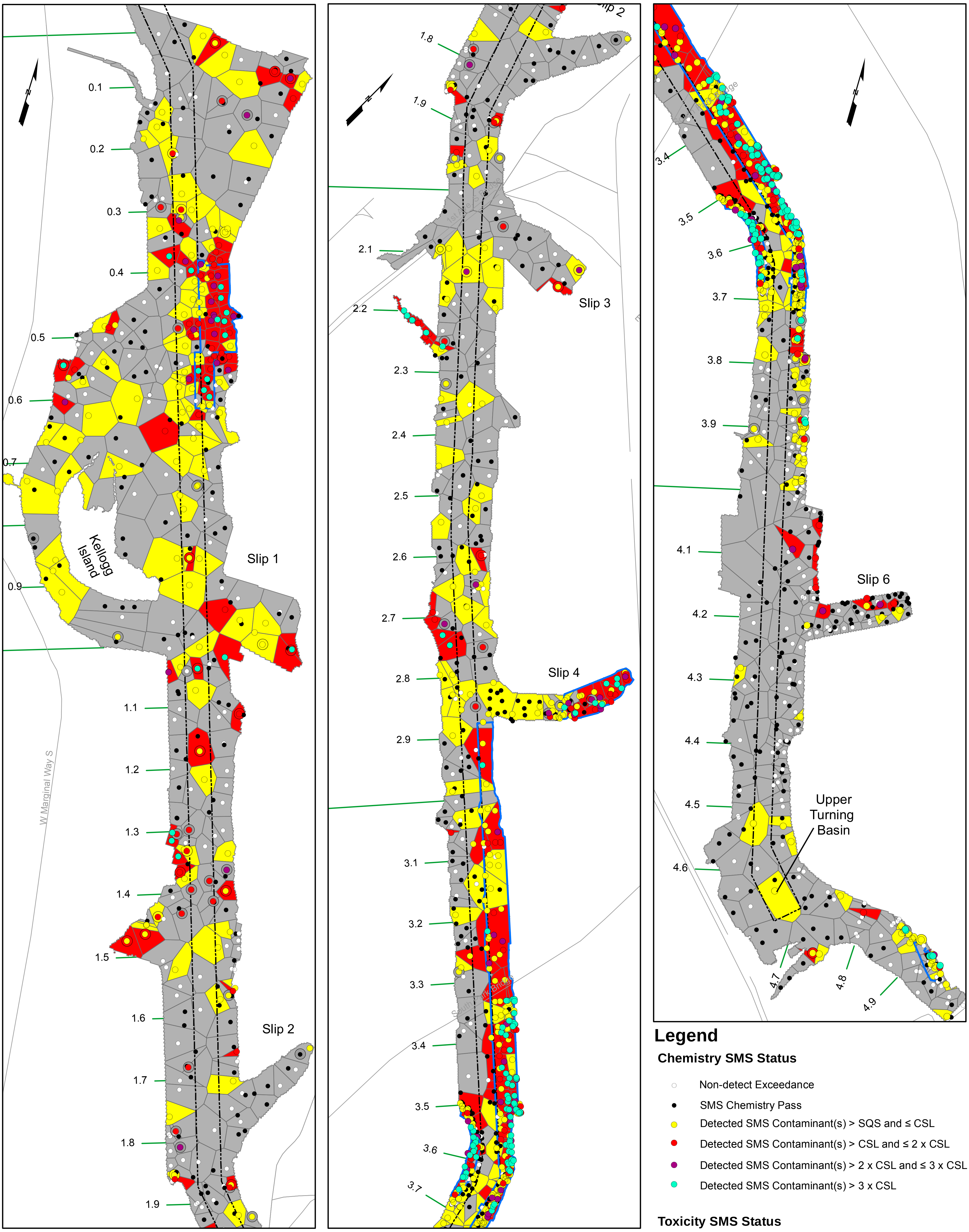
Dredging information provided by AECOM.



Legend

Chemistry SMS Status

- Non-detect Exceedance
- SMS Chemistry Pass
- Detected SMS Contaminant(s) > SQS and ≤ CSL
- Detected SMS Contaminant(s) > CSL and ≤ 2 x CSL
- Detected SMS Contaminant(s) > 2 x CSL and ≤ 3 x CSL
- Detected SMS Contaminant(s) > 3 x CSL

Toxicity SMS Status

- Pass
- > SQS and ≤ CSL
- > CSL

Thiessen Polygon SMS Status

- Pass
- > SQS and ≤ CSL
- > CSL

- Road
- Navigation Channel
- River Mile Marker
- Early Action Area

0 200 400 800 Feet

Notes:
1. Chemistry exceedances address all detected SMS contaminant(s), including total PCBs and arsenic.
2. The sampling dates of the surface sediment data range from 1991 to 2010.
3. Contaminant exceedance data from FS baseline dataset dated April 28, 2010.
4. Thiessen polygon SMS status is assigned to match that for the highest exceedance status for any contaminant for any point within that polygon. If a toxicity sample is co-located with a chemistry sample, the toxicity data override the chemistry results for that polygon.

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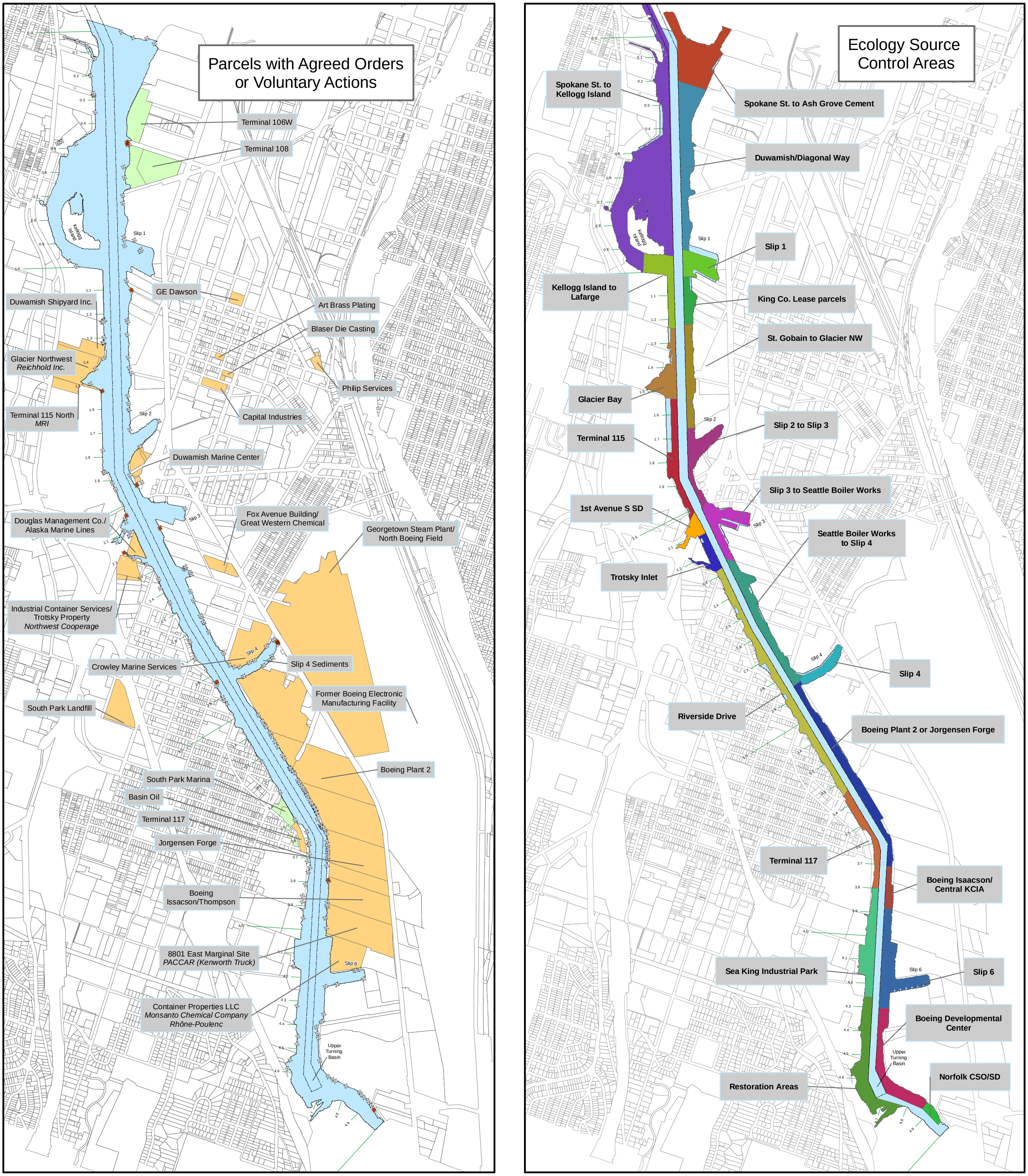
Interpolation by Thiessen Polygon of SMS Status in Surface Sediment

DATE: 10/31/12 | DWRN:MVI/sea | Revision: 0

FIGURE 2-21

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-21ToxExceedT_Poly.mxd



Notes:
1. King County Assessor parcel data received from King County on Sept. 22, 2004.
2. Ecology's source control areas were last updated in Sept. 2010.
3. *Italics*= historical operator.
4. Parcels in first panel are under, or are in negotiation for, either MTCA orders or EPA CERCLA or RCRA orders. They include Agreed Orders, Removal Orders, Enforcement Orders, and Consent Decrees.
5. CERCLA= Comprehensive Environmental Response, Compensation, and Liability Act; CSO= combined sewer overflow; EOF= Emergency Overflow; MTCA= Model Toxics Control Act; RCRA= Resource Conservation and Recovery Act; SD= storm drain.

Legend

Owner or Site Name — Glacier Northwest
Historical Operator — *Reichhold Inc.*

- Site Investigation or Voluntary Action
- MTCA, RCRA, or CERCLA Agreed Order Parcel
- Ecology Source Control Area (color varies by area)
- Lower Duwamish Waterway
- Parcel
- Navigation Channel
- River Mile Marker
- CSO/EOF Location
- Other Outfall Location

Feet: 0 500 1,000 2,000

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-22SourceContFocusAreas.mxd

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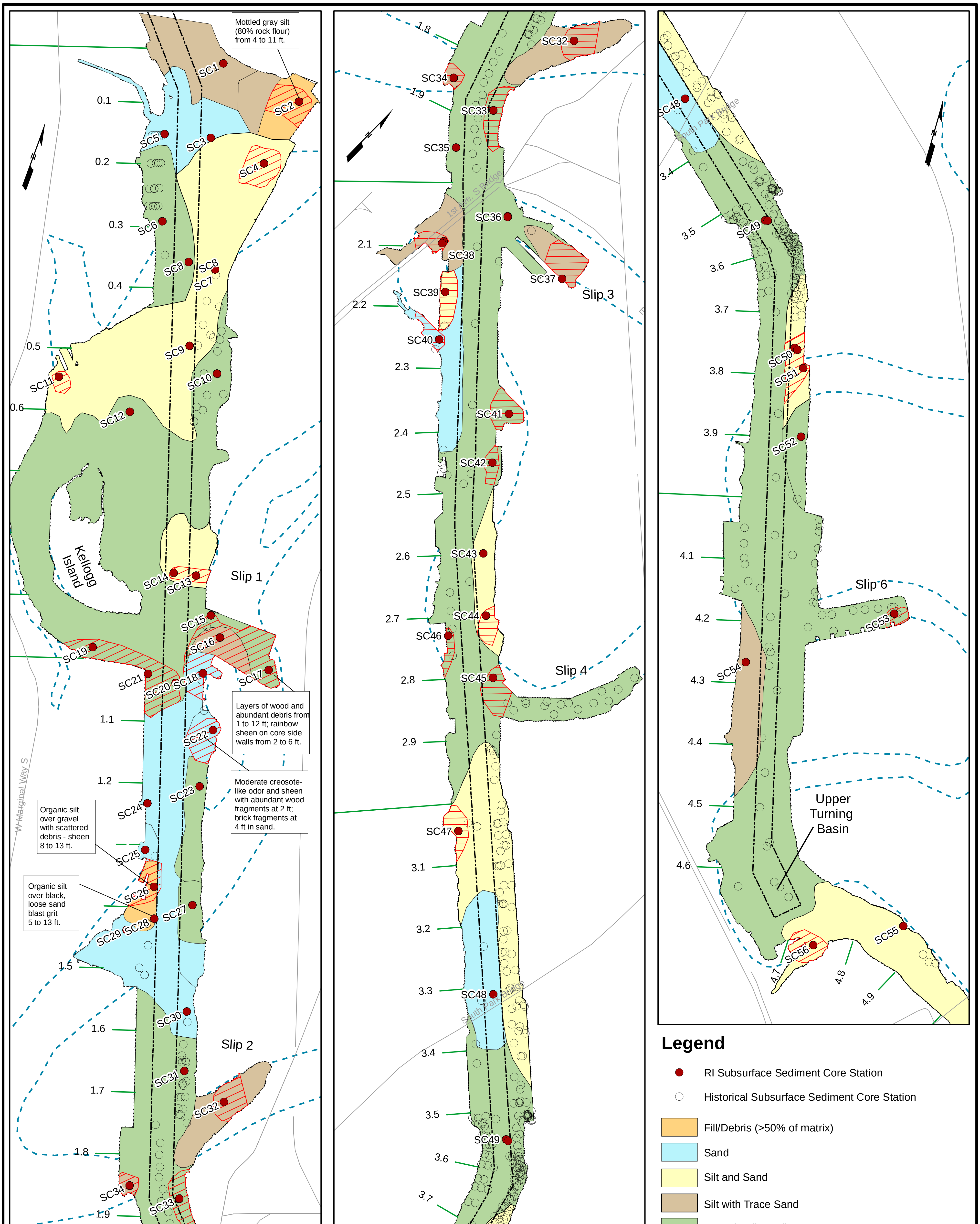
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			**Agreed Orders/Voluntary Actions and Ecology Source Control Areas**
DATE: 10/31/12	DWRN: MVI/sea	Revision: 0	**FIGURE 2-22**



Notes:
1. Distribution of debris and subsurface facies based on 2006 core logs, percent fines, physical data, and bathymetry. Facies represent common lithology across an interval of 4-ft depth. Groupings based on visual USC soil classifications (> 50% of matrix), ASTM grain size results, and Atterberg Limit results.
2. All contacts are inferred, and spatial extents are very approximate.
3. Depths in recovered feet below mudline.
4. Debris layer based on visual observations in cores at any depth and visual interpolation between cores.
5. See Table 2-8 for description of debris/sheen identified in RI cores. Most cores identified with hatching contain both debris and odor/sheen.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Fig2-23AnthroSubSurf.mxd

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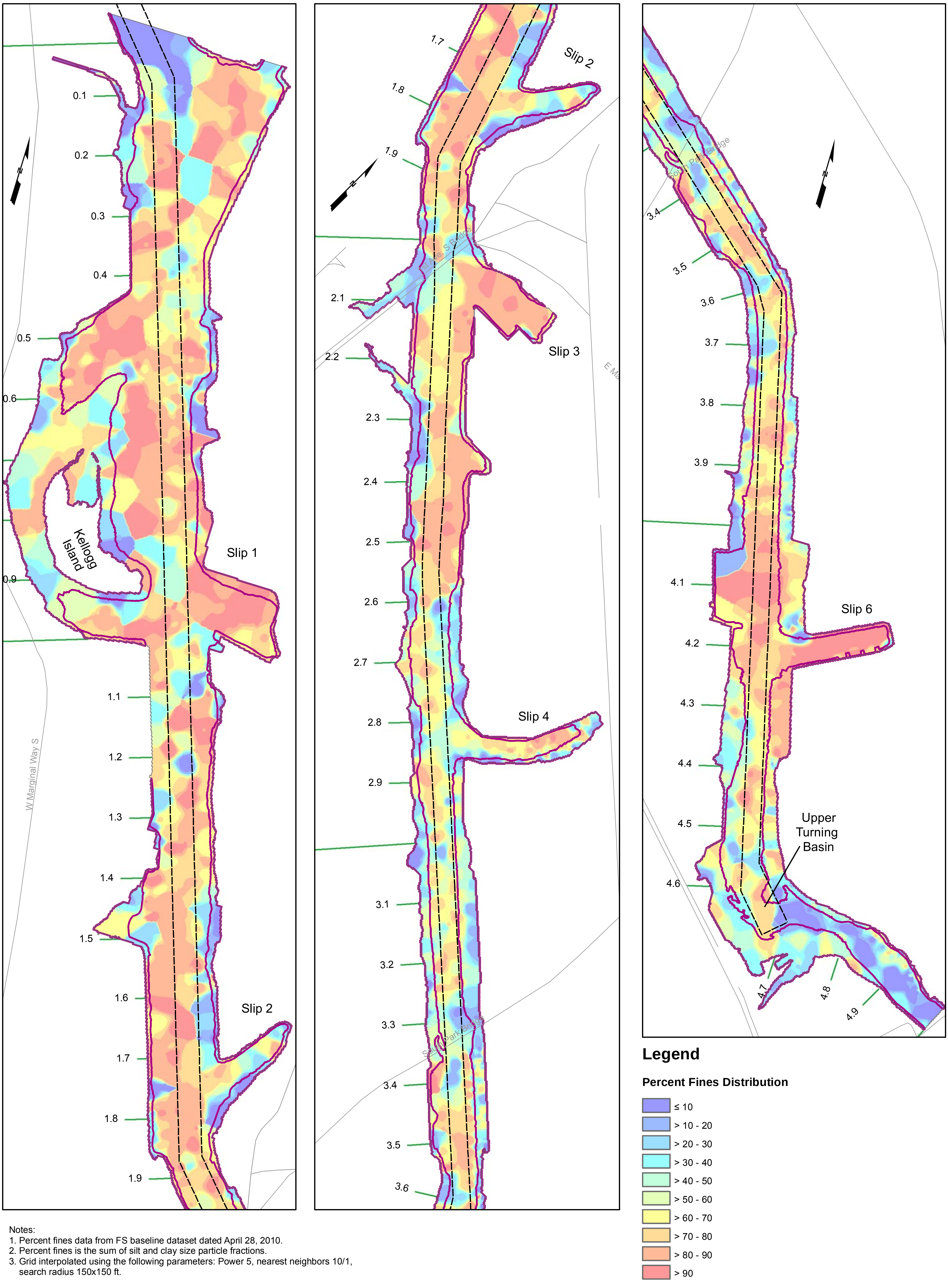
Generalized Subsurface Sediment Properties and Debris

FIGURE 2-23



Notes:
1. Percent fines data from FS baseline dataset dated April 28, 2010.
2. Percent fines is the sum of silt and clay size particle fractions.
3. Grid interpolated using the following parameters: Power 5, nearest neighbors 10/1, search radius 150x150 ft.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-24PercentFines.mxd

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Final Feasibility Study
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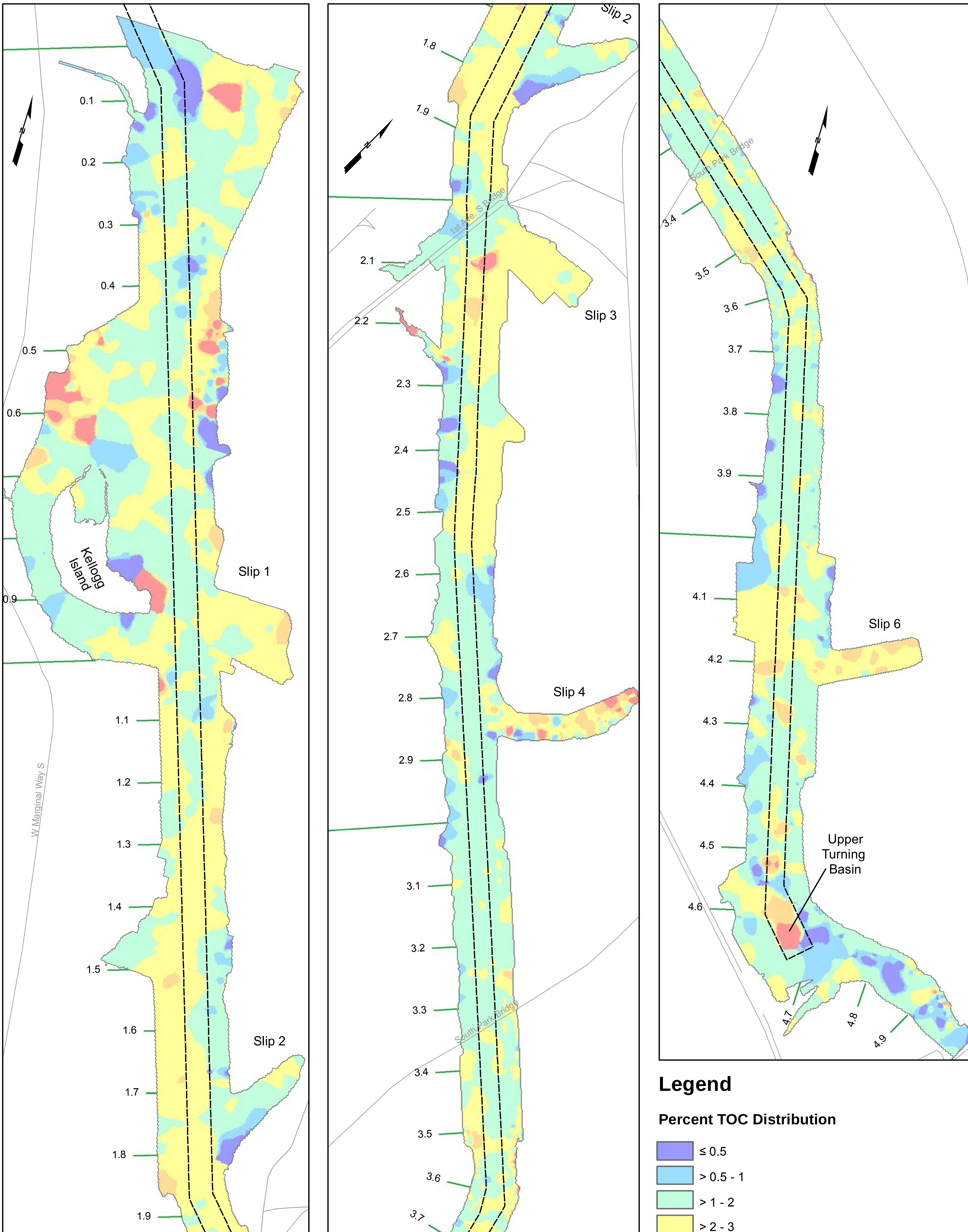
Distribution of Fine-Grained Surface Sediments

FIGURE 2-24



Legend

Percent TOC Distribution

- ≤ 0.5
- > 0.5 - 1
- > 1 - 2
- > 2 - 3
- > 3 - 4
- > 4

- Road
- Navigation Channel
- River Mile Marker

Notes:
1. TOC data from FS dataset dated April 28, 2010.
2. Grid interpolated using the following parameters: Power 5, nearest neighbors 10/1, search radius 150x150 ft.

Feet
0 200 400 800

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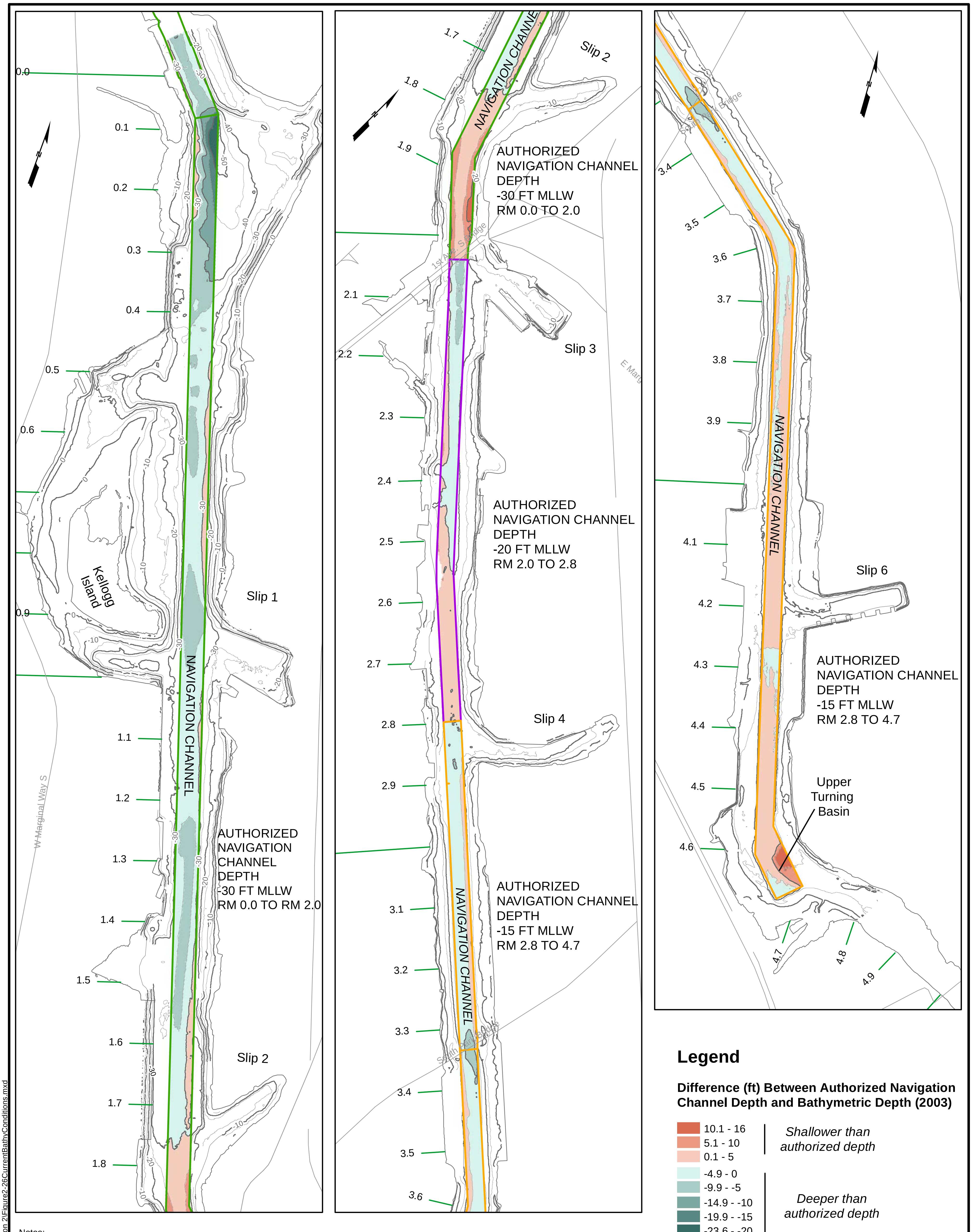
Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Distribution of Total Organic Carbon in Surface Sediments
DATE: 10/31/12	DWRN: MVI/sea	Revision: 0	FIGURE 2-25

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-25TOC.mxd



Legend

Difference (ft) Between Authorized Navigation Channel Depth and Bathymetric Depth (2003)

- 10.1 - 16
- 5.1 - 10
- 0.1 - 5

Shallower than authorized depth

- -4.9 - 0
- -9.9 - -5
- -14.9 - -10
- -19.9 - -15
- -23.6 - -20

Deeper than authorized depth

Outside of Authorized Navigation Channel

Authorized Navigation Channel Depth (ft MLLW)

- -30
- -20
- -15

Road
River Mile Marker
Bathymetric Contour (ft MLLW)

Notes:
1. Bathymetric data from David Evans and Associates October 2003 survey.
2. Navigation channel authorized depths and locations from NOAA chart 18450, Edition 18, Edition date 2/1/2004, Updated date 2/2/2008.

Feet
0 200 400 800

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-26CurrentBathyConditions.mxd

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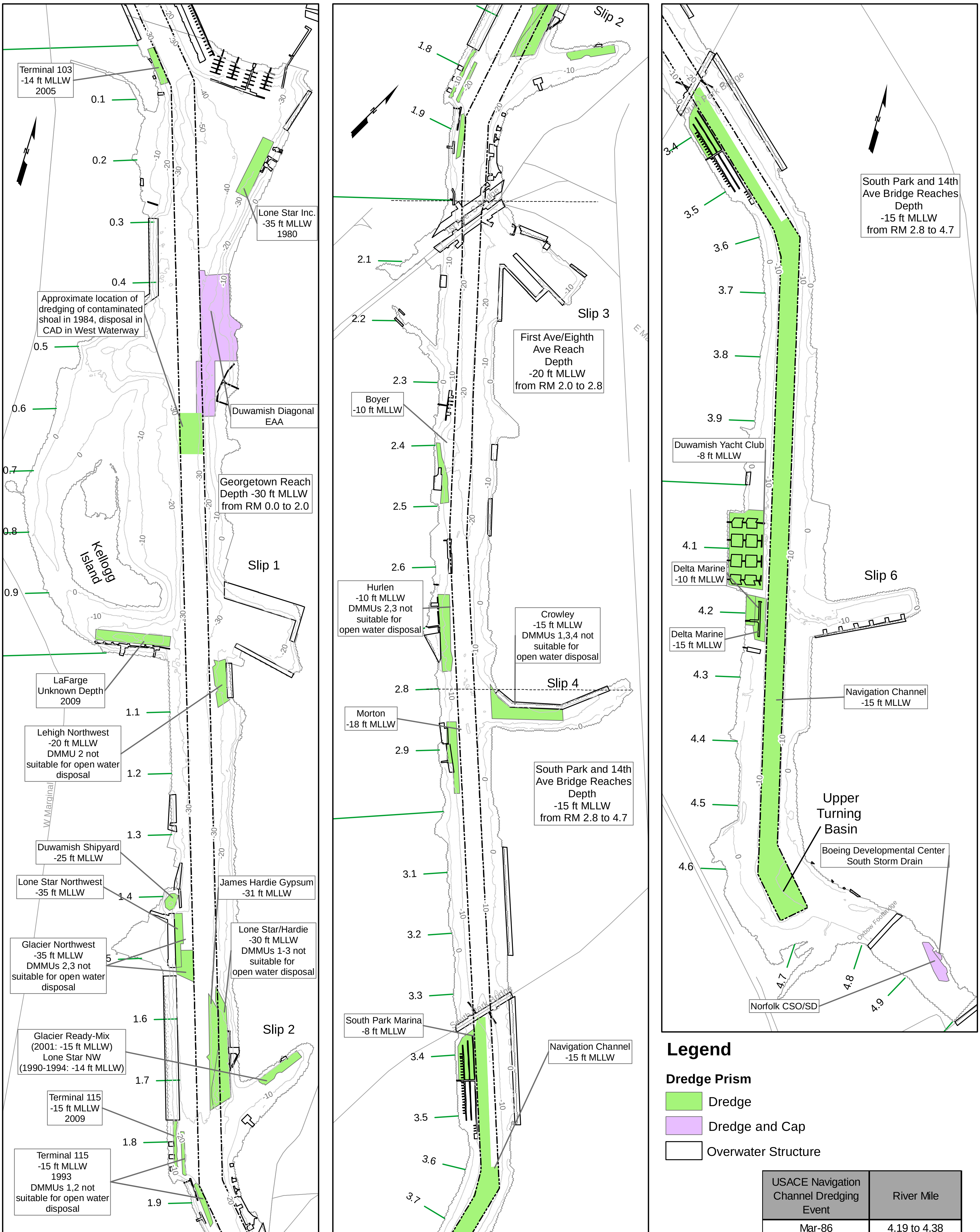
2003 Bathymetric Conditions Relative to Authorized Navigation Channel Depths

FIGURE 2-26



Notes:
1. See Tables 2-9, 2-11, and 2-12 for additional information on dredging events and sources of information.
2. No authorized maintenance dredging depth for removal projects: Duwamish/Diagonal EAA, Norfolk CSO/SD, and Boeing Developmental Center South Storm Drain.
3. Overwater structures data created in 2004 by Terralogic GIS., Inc. and Landau Associates, Inc. and modified using 2007 high resolution oblique aerial photography and field investigations.
4. Approximate location of 1984 dredging from USACE Waterway Experiment Station, 1994. Material deposited in contained aquatic disposal (CAD) site in West Waterway.
5. DMMU= dredged material management unit.

Road
Navigation Channel
River Mile Marker
Bathymetric Contour (ft MLLW)
Extent of Navigation Channel Authorized Depth

Navigation Channel
-15 ft MLLW
Project Name
Authorized elevation (ft MLLW)

USACE Navigation Channel Dredging Event	River Mile
Mar-86	4.19 to 4.38
Jul-86	4.38 to 4.65
1987	4.38 to 4.65
1990	3.97 to 4.65
1992	3.34 to 4.65
1994	4.33 to 4.65
1996	4.02 to 4.48
1997	4.26 to 4.65
1999	3.43 to 4.65
2002	4.27 to 4.65
2004	4.33 to 4.65
2007	4.27 to 4.65
2010	4.05 to 4.65

Feet
0 200 400 800

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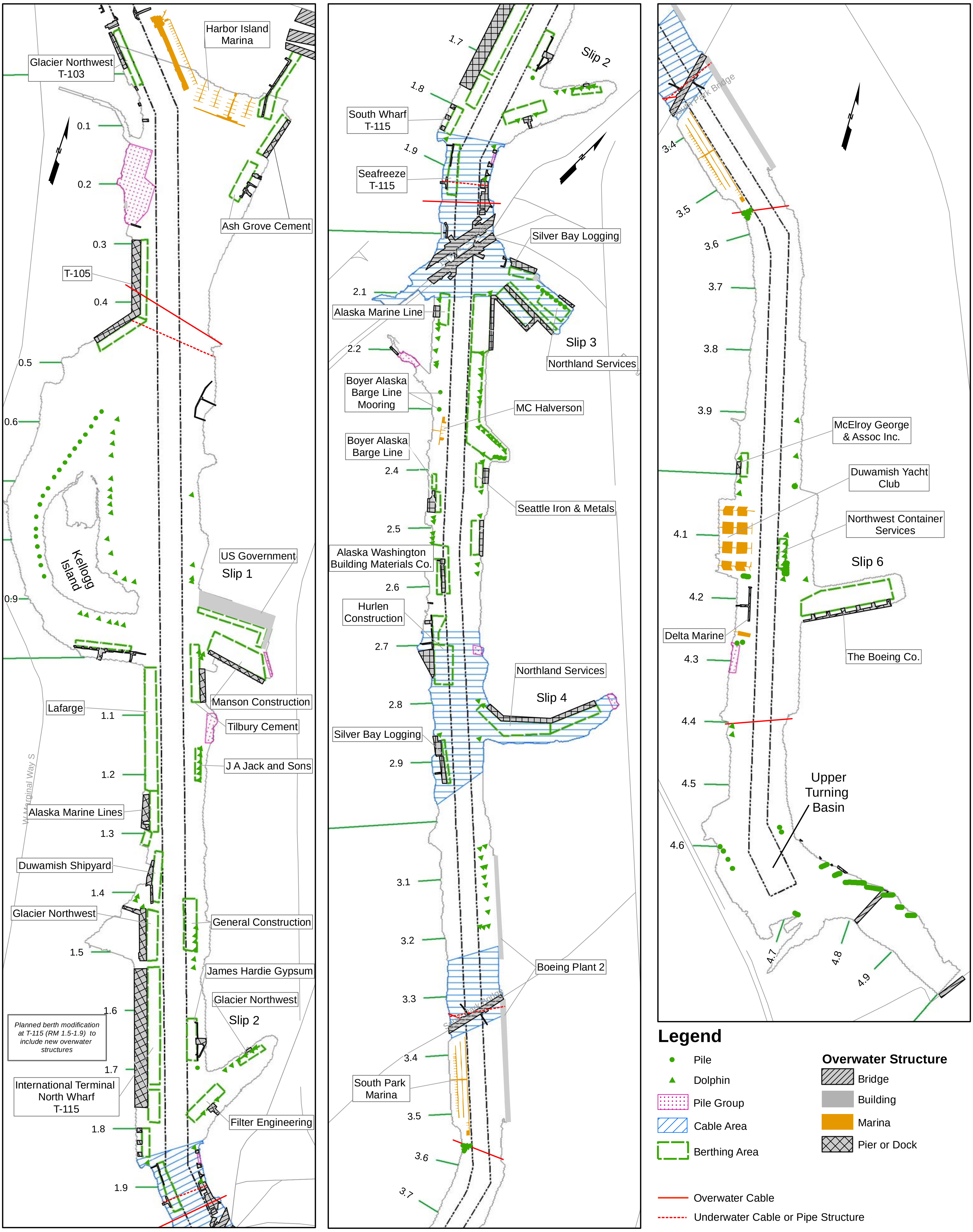
Extent and Depth of Authorized Dredging Events (1980 to 2010)

FIGURE 2-27

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2_27HistoricDredge.mxd



Legend

- Pile
- Dolphin
- Pile Group
- Cable Area
- Berthing Area

Overwater Structure

- Bridge
- Building
- Marina
- Pier or Dock

- Overwater Cable
- Underwater Cable or Pipe Structure
- Navigation Channel
- River Mile Marker
- Road

Berthing area or marina ownership

International Terminal North Wharf T-115

Notes:
1. Berthing area data from USACE, Port Series No. 36 Revised 2002; Port of Seattle, Washington.
2. Overwater structures data created in 2004 by TerraLogic GIS, Inc. and Landau Associates, Inc., and modified using 2007 high resolution oblique aerial photography and field investigations.
3. Pile and dolphin data created using 2005/2006 high resolution oblique aerial photography.
4. Utilities data (underwater and overwater cable and pipelines and areas) created using NOAA chart 18450, Edition 18, Edition date 2/1/2004, Updated date 2/2/2008.
5. See Table 2-10 for additional physical structures and berthing area information.

Feet
0 200 400 800

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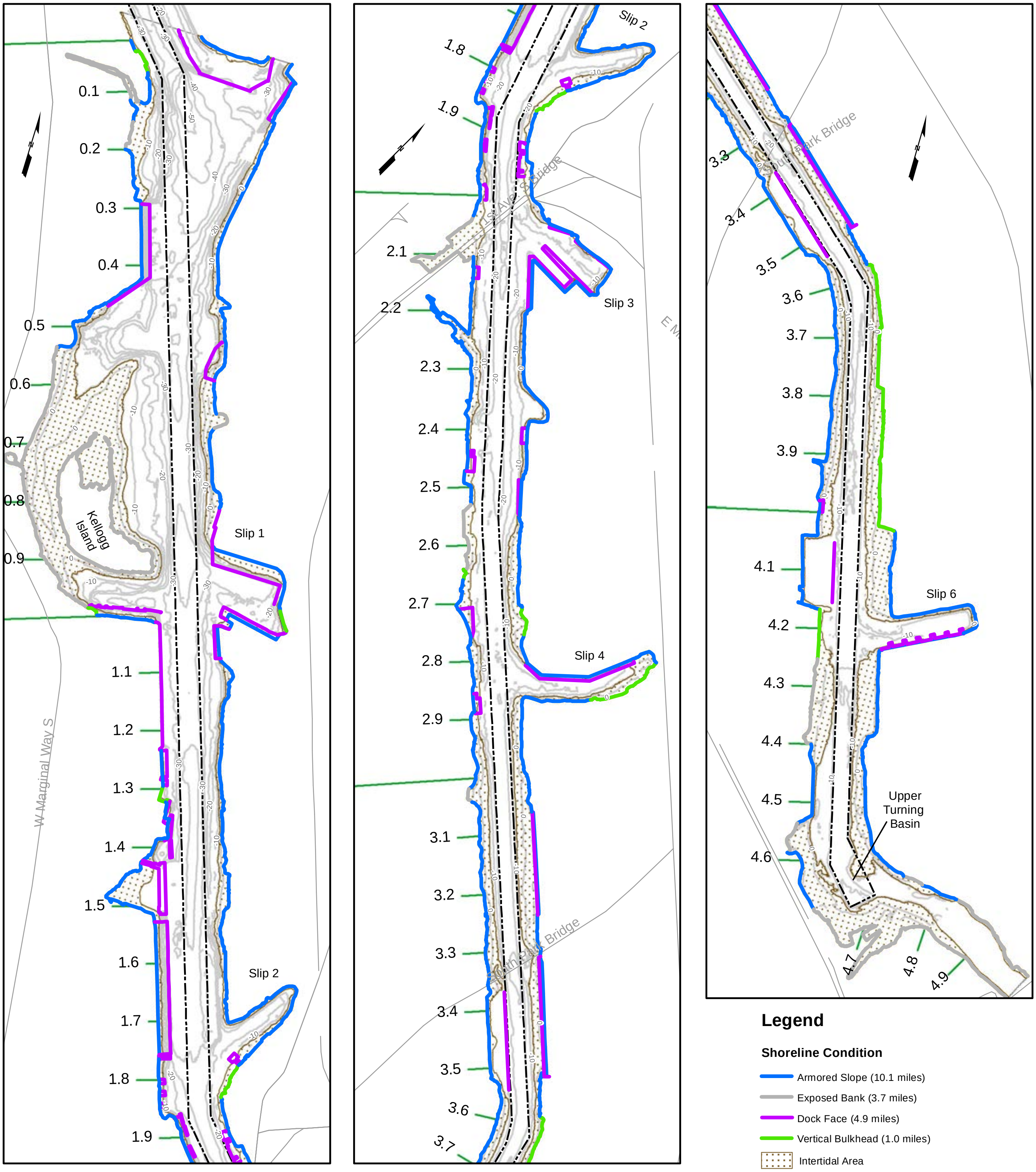
In-water and Overwater Structures and Berthing Areas

FIGURE 2-28

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 2\Fig2-28BerthingOstruct.mxd



Notes:
1. Shoreline conditions layer created using video from the 2005 Windward Environmental Human Access Survey, 2007 aerial imagery, and field investigation (2008 boat tour).
2. Mapped shoreline includes frontage of overwater structures, as well as riverbank.
3. Dock face refers to shoreline features including pile bents and a deck supporting some activity or structure. The overwater buildings at Boeing Plant 2 (RM 3.1-3.5) are mapped as dock face fronting an armored slope.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 2\Figure2-29ShorelineConditions.mxd

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.34			Shoreline Conditions
DATE: 10/31/12	DWRN: MVI/sea	Revision: 0	FIGURE 2-29

3 Risk Assessment Summary

The baseline ecological and human health risk assessments were completed for the Lower Duwamish Waterway (LDW) in 2007 (Windward 2007a, 2007b). This section summarizes the findings of both risk assessments, which are used in Section 4 of this feasibility study (FS) to aid in establishing remedial action objectives (RAOs) and preliminary remediation goals (PRGs).

The baseline ecological risk assessment (ERA) (Windward 2007a) is discussed in Section 3.1, and presents the estimated risks for the benthic invertebrate community and for crabs, fish, and wildlife species. These receptors are exposed to contaminants in the LDW primarily through contact with sediment, water, or through consumption of prey species found in the LDW.

The baseline human health risk assessment (HHRA) (Windward 2007b) is discussed in Section 3.2, and presents the estimated risks for people who may be exposed to contaminants in the LDW through consumption of resident seafood from the LDW or through direct contact with sediment or water.

Both the baseline ERA and HHRA were based on the LDW Remedial Investigation (RI) baseline conditions.[1] For the early action areas (EAAs) where sediment cleanup occurred after December 2000 when the RI/FS Administrative Order on Consent (AOC) was issued (i.e., Duwamish/Diagonal and Boeing Developmental Center south storm drain), the pre-remedy data were used to characterize baseline conditions. However, sediment removal in the vicinity of the Norfolk combined sewer overflow/storm drain (CSO/SD) was conducted in 1999, so post-remedy monitoring data from the Norfolk CSO/SD area were used to represent baseline conditions in the RI.

The risk-based threshold concentrations (RBTCs), discussed in Section 3.3, represent calculated sediment and tissue concentrations estimated to be protective of a particular receptor for a given exposure pathway and target risk level. RBTCs were derived in the RI (Windward 2010) based on the baseline ERA and HHRA (Windward 2007a, 2007b). The RBTCs are also presented in this FS because they are used, along with other site information, to establish PRGs in Section 4. Finally, this section concludes with a summary of the key findings from the risk assessments (Section 3.4).

1 Additional data have been collected since the finalization of the RI baseline dataset (i.e., since October 2006). The baseline dataset used in this FS (called the "FS baseline dataset") includes those data newer than October 2006, as well as older data that were not previously included (see Section 2, Table 2-2). In addition to the newer data, post-cleanup data in the perimeter of the Duwamish/Diagonal area were included in the FS baseline dataset. For the cap and enhanced natural recovery areas, only precleanup data were included in the FS baseline dataset. Additional details on the use of data from the Duwamish/Diagonal early action area in the FS are provided in Appendix N.

3.1 Baseline Ecological Risk Assessment

The baseline ERA (Windward 2007a) estimated risks for ecological receptors that may be exposed to contaminants in sediment, water, and through consumption of prey in the LDW.

Ten receptors of concern[2] were selected in the baseline ERA to be representative of groups of organisms in the LDW with the same exposure pathways. These receptors of concern include the benthic invertebrate community; crabs; juvenile Chinook salmon, Pacific staghorn sculpin, and English sole (collectively discussed as "fish"); and spotted sandpiper, great blue heron, osprey, river otter, and harbor seal (collectively discussed as "wildlife species").

A conservative risk-based screening process first identified contaminants of potential concern (COPCs) for the ERA (Windward 2007a). In this process, contaminant concentrations in sediment, water, and aquatic biota were compared to risk-based screening levels. Those contaminants present at concentrations above the screening levels or demonstrating the potential for unacceptable effects were identified as COPCs and underwent further risk analysis in the ERA.

Risks were estimated as follows:

- Risks for the benthic community were estimated by comparing contaminant concentrations in sediment with: 1) the numerical criteria of the Washington State Sediment Management Standards (SMS), 2) literature-derived toxicity reference values (TRVs), or 3) toxicologically based guidelines. Risks were also estimated based on site-specific sediment toxicity tests; a comparison of volatile organic compound (VOC) concentrations in porewater to toxicity data; a comparison of tributyltin (TBT) concentrations in benthic invertebrate tissues to concentrations associated with adverse effects; and a study of imposex in LDW-collected gastropods.
- Risks for crabs and fish were estimated by comparing contaminant concentrations in crab and fish tissue with tissue residues associated with effects on survival, growth, or reproduction.
- Risks for fish were also evaluated by comparing contaminant concentrations in prey to dietary concentrations that have been shown to cause adverse effects on survival, growth, or reproduction.
- For wildlife, risks were estimated based on calculations of daily doses of contaminants derived from the ingestion of sediment, water, and prey

[2] Key considerations for selecting receptors of concern were the potential for direct or indirect exposure to sediment-associated contaminants, human and ecological significance, site use, sensitivity to COPCs at the site, susceptibility to biomagnification of COPCs, and data availability.

species. Risks were then estimated by comparing those doses with doses that have been shown to cause adverse effects on survival, growth, or reproduction.

The risks estimated for each of these receptors are summarized in the following sections.

3.1.1 Benthic Invertebrate Community

Contaminant concentrations in surface sediments were compared to the sediment quality standards (SQS) and the cleanup screening level (CSL) numerical chemical values of the SMS. For those that do not have SMS criteria, concentrations were compared with Dredged Material Management Program (DMMP) sediment quality guidelines (if they were toxicologically based) or with toxicity values from the scientific literature (i.e., TRVs). A contaminant was selected as a contaminant of concern (COC) if its concentration was found to be above the SQS criteria in one or more sediment samples from the LDW. Forty-four contaminants were identified as COCs for the benthic invertebrate community (Table 3-1). The three COCs with the most frequent exceedances were total polychlorinated biphenyls (PCBs), bis(2-ethylhexyl)phthalate (BEHP), and butyl benzyl phthalate. For all other COCs, exceedances occurred in 5% or less of the sediment samples.

When contaminant concentrations in surface sediment exceed the SMS criteria, the potential exists for harmful effects on the benthic invertebrate community living in intertidal and subtidal sediment. Based on the RI dataset, the SQS were exceeded in approximately 25% (110 acres) of the LDW study area. Of these 110 acres, a higher likelihood for adverse effects was identified in 31 acres, corresponding to approximately 7% of the LDW, where contaminant concentrations or biological effects resulted in exceedances of the CSL of the SMS. The other 79 acres (18% of the LDW) had contaminant concentrations or biological effects that exceeded the SQS but not the CSL. The remaining 75% of the LDW is considered unlikely to have adverse effects on the benthic invertebrate community based on the RI dataset.

Similar results were obtained using the FS dataset;[3] contaminant concentrations in approximately 18% (80 acres) of the LDW study area exceeded SMS criteria (i.e., exceedance of either the SQS and/or CSL). A higher likelihood of adverse effects was indicated in approximately 4% (16 acres) of the LDW study area because of CSL exceedances. The remaining 82% of the LDW was considered unlikely to have adverse effects on the benthic invertebrate community, based on the FS dataset.

Risks to the benthic invertebrate community from VOCs detected in sediment porewater were very low. One VOC, cis-1,2-dichloroethene, was detected in porewater samples collected from one small area located near Great Western International at river

[3] See Section 2.2 of the FS for a discussion of the differences between the RI and FS datasets.

mile (RM) 2.4E. The concentrations for this VOC were greater than the no-observed-effect concentration (NOEC) for the marine invertebrates but were less than the lowest-observed-effect concentration (LOEC). Because this location is considered to be a worst-case exposure area with respect to the potential for adverse effects to benthic invertebrates from VOCs, and other areas where porewater data are available had much lower VOC concentrations, the likelihood of risks from VOCs is very low in the rest of the LDW.

Finally, risks to benthic invertebrates from TBT, which has no SQS criterion, were considered to be low. This finding was based on a study of imposex in LDW-collected gastropods, as well as a comparison of TBT concentrations in benthic invertebrate tissue samples to tissue effect concentrations from the scientific literature.

3.1.2 Crabs, Fish, and Wildlife Species

Risks for crabs exposed to COPCs were estimated by comparing COPC concentrations in LDW crab tissue to effects data obtained from the scientific literature, including no-observed-adverse-effect levels (NOAELs) and lowest-observed-adverse-effect levels (LOAELs). Risks were estimated by calculating hazard quotients (HQs) as the ratio of the COPC concentrations in LDW crab tissue to the selected NOAELs and LOAELs for crab tissue.

For fish receptors of concern, HQs were calculated using both a critical tissue-residue approach and estimated dietary exposures, as well as a range of effects data obtained from the scientific literature, including NOAELs and LOAELs.

For wildlife receptors of concern, HQs were calculated for estimated dietary exposures and were based on a range of effects data obtained from the scientific literature, including NOAELs and LOAELs.

COCs for crabs, fish, and wildlife species were defined as contaminants with LOAEL-based HQs greater than or equal to 1, which indicate a potential for adverse effects. One contaminant (total PCBs) was identified as a COC for crabs. Total PCB concentrations in crab tissue were equal to the lowest concentrations associated with adverse effects in crabs, indicating potential for adverse effects. Seven contaminants (total PCBs, cadmium, chromium, copper, lead, mercury, and vanadium) were identified as COCs for at least one fish or wildlife species (Table 3-2).

No quantitative risk estimates were calculated for dioxins/furans in the RI because tissue data were not available from the LDW. Therefore, risks to ecological receptors associated with tissue burdens or dietary exposure to dioxins/furans are unknown.

3.1.3 Risk Drivers for Ecological Receptors

A subset of the COCs was identified as risk drivers for ecological receptors in accordance with guidance from the U.S. Environmental Protection Agency (EPA 1998a) and the Washington State Department of Ecology (Ecology) (WAC 173-340-703). A detailed explanation of the rationale for identifying these risk drivers can be found in Section 7 of the baseline ERA (Windward 2007a) and is summarized in Tables 3-1 and 3-2. Risk drivers for ecological receptors of concern were selected by considering: 1) the uncertainty in risk estimates based on quantity and quality of exposure and effects data, 2) natural background concentrations, and 3) the likely magnitude of residual risks following planned sediment remediation in EAAs.

In the baseline ERA (Windward 2007a), 44 contaminants were selected as COCs for benthic invertebrates. Of these, 41 contaminants were selected as risk drivers for benthic invertebrates because they had concentrations greater than the SQS in at least one sediment sample (Table 3-1). The other three contaminants (nickel, dichlorodiphenyl-trichloroethanes (DDTs), and chlordane) were identified as COCs based on concentrations greater than TRVs or toxicologically based DMMP guidelines; these three contaminants were not selected as risk drivers because of uncertainties in effects data and because sediment samples with concentrations greater than the TRVs or guidelines were all (except for one) located within EAAs (Windward 2007a). In consultation with EPA and Ecology, total PCBs were identified as a risk driver for river otter because estimated dietary exposure concentrations for river otter were greater than the LOAEL by a factor of 2.9 and uncertainties in the risk estimate were relatively low (Table 3-2). Although no other COCs were identified as risk drivers for fish or wildlife species, the other COCs were evaluated to assess the potential for risk reduction following remedial actions and the results of this analysis are presented in Section 9 of this FS.

3.2 Baseline Human Health Risk Assessment

The baseline HHRA (Windward 2007b) estimated risks to people from exposure to contaminants in LDW seafood, sediments, and water. The exposures were assumed to occur through consumption of resident seafood harvested from the LDW, and through direct contact with sediments during netfishing, clamming, or beach play (the exposure pathways). Risks associated with direct contact with water (i.e., swimming) are much lower than those estimated for direct sediment contact (Windward 2007b), and are therefore not discussed further in the FS.

Direct-contact risk estimates in the HHRA (Windward 2007b) for the beach play and clamming scenarios were based on the uppermost 10 cm of sediment in the beach play and clamming areas because most of the surface sediment data collected in the LDW was collected to a depth of 10 cm. However, children and clammers may dig holes deeper than 10 cm. The most abundant clam species of harvestable size in the LDW is the Eastern soft-shell clam (*Mya arenaria*), which has been reported to burrow to depths that range from 10 cm to 20 cm based on two Pacific Northwest species guidebooks (Kozloff 1973, Harbo 2001) and from 10 to 30 cm based on studies conducted throughout the United States (e.g., Blundon and Kennedy 1982, Cohen 2005, Hansen et al. 1996, Evergreen State College 1998). To ensure protection of human health, a sediment depth of 45 cm is used as the point of compliance depth in this FS for clamming and beach play areas in the LDW. This depth accounts for the potential exposure of children and clammers who may come into direct contact with sediment when digging holes in the sediment at low tide.

Using EPA guidance, a risk-based screening was first performed to identify the COPCs to be evaluated. This screening was based on an exceedance of the screening criteria (i.e., the risk-based concentration) by either the maximum detected concentrations or analytical reporting limits (RLs) (for samples with non-detected concentrations). The risk-based screening identified the following COPCs by exposure pathway: 59 COPCs for seafood consumption pathways, 20 COPCs for netfishing, and 28 COPCs for beach play and clamming direct contact pathways. COPCs that were not detected in either sediment or tissue were still included if they had RLs above the screening criteria; however, those COPCs were evaluated only in the uncertainty analysis.

For the detailed risk analysis of the COPCs, reasonable maximum exposure (RME) estimates were calculated for the exposure pathways evaluated in the HHRA (Windward 2007b) to avoid underestimating risks. The RME is the highest exposure that is reasonably expected to occur at a site. The RME, by definition, likely overestimates exposure for many individuals.

Risks estimated for the seafood consumption and direct exposure scenarios evaluated in the HHRA (Windward 2007b) are discussed in the following subsections.

3.2.1 Risks Associated with the Seafood Consumption Pathway

No seafood consumption surveys specific to the LDW were available for use in the HHRA (Windward 2007b). Therefore, seafood consumption rates assumed for the LDW were developed by EPA based on data collected from other areas of Puget Sound for tribal consumers and from an EPA consumption study for Asian and Pacific Islanders (API) in the King County area.

Seafood consumption scenarios with different levels of exposure were evaluated in the baseline HHRA to provide a broad range of risk estimates. RME estimates, which will be used for making decisions about the need for remediation at the site, included the following seafood consumption rates:

- ♦ Tulalip tribal consumption rates for adults and children from EPA’s tribal framework document (EPA 2007b)
- ♦ Seafood consumption rates for API adults, modified by EPA based on the results of a survey of API consumers (EPA 1999a) to reflect rates by individuals that harvest seafood only within King County.

The tribal consumption rates of resident seafood are likely overestimates of current consumption. However, such rates may be achieved in the LDW at some future time. The rates used are generally similar to those for other populations who consume large quantities of seafood in the absence of seafood consumption health warnings.

Other seafood consumption scenarios were also evaluated in the baseline HHRA (Windward 2007b). These other scenarios included consumption rates estimated using: 1) Suquamish tribal consumption rates from EPA's tribal framework document (EPA 2007b), 2) "average exposure" scenarios using central tendency consumption rate estimates, and 3) a "unit risk" scenario based on an assumed one seafood meal per month. Estimates for the unit risk scenario are useful for risk communication because individuals can determine what their risk might be for various seafood consumption practices.

It is noted that there is considerable uncertainty about the applicability of seafood consumption rates in the baseline HHRA (Windward 2007b), particularly for clams, given the quality and quantity of shellfish habitat in the LDW. Nonetheless, their use in the HHRA reflects health-protective estimates of risk.

Contaminant concentrations in the tissues of several different resident seafood species (e.g., English sole, perch, crabs, clams, mussels) were used to represent a typical consumer's diet. COCs were then determined by estimating cancer and non-cancer effects for the RME scenarios. Contaminants with an estimated excess cancer risk greater than 1 in 1,000,000 (1×10^{-6}) or a non-cancer HQ greater than 1 were selected as COCs for the seafood consumption exposure pathway. Nineteen contaminants were identified as COCs for the seafood consumption exposure pathway (Table 3-3).[4]

The total risk for all carcinogenic contaminants for the various RME seafood consumption scenarios ranged from 7 in 10,000 (7×10^{-4}) to 4 in 1,000 (4×10^{-3}),[5] with the primary contributors to risk being total PCBs, arsenic, and carcinogenic polycyclic

4 As noted in Table 3-3, both total PCBs and PCB toxic equivalent (TEQ) were identified in the HHRA as COCs. Because these two COCs represent different methods of evaluating the same contaminant, they are counted as one COC in the count presented here.

5 The highest RME total excess cancer risk estimate reported here (4×10^{-3}) differs from that reported in Appendix B of the RI (the HHRA, Windward 2007b) and Section 6 of the RI (3×10^{-3}) (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish) for scenarios based on the Tulalip Tribes survey was updated in response to a correction provided by EPA. The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates are described in detail in an erratum (Windward 2009) to Appendix B of the RI (Windward 2010). This total risk estimate includes risks from total PCBs but excludes risks from PCBs from a TEQ perspective to avoid double counting dioxin-like PCB risks posed by coplanar PCB congeners that are already accounted for in the slope factor for PCBs.

aromatic hydrocarbons (cPAHs) (Table 3-4a).[6] In addition, evaluation of non-cancer HQs indicates the potential for adverse effects other than cancer associated with seafood consumption, particularly from total PCBs (Table 3-4b).

To provide additional information regarding the total excess cancer risks for the RME seafood consumption scenarios, Table 3-5 presents a summary of the excess cancer risks for COCs and includes the percentages of the total risks attributable to different COCs and seafood consumption categories (i.e., fish, crabs, and clams). The main contributors to the total excess cancer risk for the RME seafood consumption scenarios were arsenic (40 to 50% of the total risk) and total PCBs (38 to 43% of the total risk). In addition, Table 3-5 shows that the majority of the arsenic and cPAH risks (96 to 98%) are attributable to clams, while the total PCB risk is attributable to several different seafood consumption categories (primarily clams [39 to 47%], pelagic fish [23 to 25%], and whole-body crabs [15%]).

It is important to recall that the risk estimates presented in the baseline HHRA (Windward 2007b) did not include the risks associated with dioxins/furans in seafood tissues because no tissue data for dioxins/furans were available at that time from the LDW. More recently, a small dataset became available for dioxin/furan concentrations in English sole fillets collected near Kellogg Island in 2007 as part of the Puget Sound Ambient Monitoring Program (Gries 2008). It should be noted that these data were collected from only a small portion of the LDW that has relatively low concentrations of dioxins/furans in sediments. These data were not included in formal risk calculations because there are no dioxin/furan tissue data from the LDW for the other seafood categories.

However, in an attempt to put these new dioxin/furan concentration data in context, excess cancer risks were calculated assuming all seafood categories had the same dioxin/furan concentrations as the English sole fillet samples collected near Kellogg Island. Based on this assumption, the excess cancer risks associated with dioxins/furans would be an order of magnitude or more lower than the total excess cancer risks (all other contaminants combined) for all three RME seafood consumption scenarios and therefore inclusion of dioxin/furan tissue data may not have substantially changed the overall risks (Table 3-5). However, COC concentrations can vary substantially across organism, tissue type, and location. Conclusive statements about the contribution of dioxins/furans to overall risk would require collection of additional dioxin/furan data for all of the organisms and tissue types considered in the HHRA. In addition, the tissue data would have to be spatially representative, not just from limited areas of the LDW (e.g., Kellogg Island).

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6 Seafood samples from the LDW were not analyzed for dioxins and furans, so risks from these contaminants are not included in seafood consumption risk estimates, but were assumed to be unacceptable.

3.2.2 Risks Associated with Direct Sediment Contact

No LDW-specific data are available for estimating the degree to which humans may currently be directly exposed to sediment via beach play or clamming. To ensure protection of human health, RME values for the beach play and clamming scenarios were identified based on regional data and best professional judgment. These values likely overestimate current exposure but provide information to risk managers for evaluating potential increases in site use following remediation. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing practices within the LDW. Netfishing can occur throughout the LDW, while clamming and beach play would occur in specific areas of the LDW. The potential clamming areas and beach play areas are shown on Figure 3-1.

Contaminants with either an estimated excess cancer risk greater than 1 in 1,000,000 (1×10^{-6}) or a non-cancer HQ greater than 1 for at least one RME scenario were selected as COCs for the direct sediment contact exposure pathways. Five contaminants were identified as COCs for direct sediment contact exposure (Table 3-3). The primary contributors to risk included total PCBs, arsenic, cPAHs, and dioxins/furans;[7] toxaphene was also identified as a COC, but it was only a tentatively identified compound and therefore its contribution to risk is highly uncertain.

3.2.2.1 Netfishing and Clamming Scenarios

As presented in the RI (Windward 2010), total excess cancer risk estimates for the direct sediment contact RME scenarios were 3 in 100,000 (3×10^{-5}) for netfishing and 1 in 10,000 (1×10^{-4}) for tribal clamming (Table 3-6a); neither of these direct sediment contact exposure scenarios had non-cancer HQs greater than 1. Dioxins/furans were a significant contributor to total carcinogenic risk for the netfishing and tribal clamming scenarios in the HHRA (2×10^{-5} [vs. a total risk of 3×10^{-5}] and 1×10^{-4} [equal to the total risk of 1×10^{-4}], respectively). The dataset for dioxins/furans available for the HHRA was much smaller than the FS dataset (see Section 2.2.1),[8] and the exposure point concentrations for dioxins/furans for these scenarios in the HHRA were highly influenced by a few high data points. When total excess cancer risks were recalculated using the much larger FS dataset, the dioxin/furan risk associated with netfishing was 3×10^{-6}, and the dioxin/furan risk associated with clamming was 5×10^{-5}.

Since the HHRA (Windward 2007b) was finalized, additional sediment samples have been collected and are now included as part of the FS dataset. If this FS dataset were used to recalculate netfishing and clamming risk estimates for the other risk drivers

[7] Dioxins/furans were analyzed in sediments, and therefore, direct contact risk estimates are available.

[8] There were 43 sediment samples available to characterize dioxin/furan TEQ netfishing exposure in the HHRA dataset, compared to 189 sediment samples in the FS dataset. There were 11 sediment samples available to characterize dioxin/furan TEQ tribal clamming exposure in the HHRA dataset, compared to 37 sediment samples in the FS dataset.

(i.e., total PCBs, arsenic, and cPAHs), risks would be similar to or lower than those calculated in the HHRA (Windward 2007b) for arsenic and cPAHs. However, for total PCBs, risk estimates would be higher based on the inclusion of two samples with very high PCB concentrations (2,900,000 micrograms per kilogram dry weight [µg/kg dw] and 230,000 µg/kg dw) collected in May 2007 from the head of the inlet at RM 2.2W. If these two samples were excluded, the risk estimates would be slightly lower than those calculated in the HHRA.

3.2.2.2 Beach Play Scenarios

As presented in the RI (Windward 2010), total excess cancer risk estimates ranged from 5 in 1,000,000 (5×10^{-6}) to 5 in 100,000 (5×10^{-5}) for the eight individual beach play areas evaluated as part of the beach play RME exposure scenario (Table 3-6a). Non-cancer HQs were less than 1 for all of the eight beach play areas.

Since the HHRA was finalized (Windward 2007b), additional sediment samples have been collected in many of the beach play areas; the data from the analysis of those samples have been incorporated into the FS dataset (see Section 2.2). This dataset was used to update beach play risk estimates for the individual beach play areas. Details regarding how the updated risk estimates were calculated, including specific information about the calculation of exposure point concentrations, are presented in Appendix B.

Based on the FS dataset, the estimated total excess cancer risks (for all four human health risk drivers combined) ranged from 2 in 1,000,000 (2×10^{-6}) to 6 in 10,000 (6×10^{-4}) for the individual beach play areas (Table 3-6b and Figure 3-1). The estimated total excess cancer risks for beach play were lower for Areas 1, 3, 7, and 8 based on the FS dataset (Table 3-6b) compared with the estimated total excess cancer risks for those areas based on the HHRA dataset (Table 3-6a) (Windward 2007b). The other beach play areas (Areas 2, 4, 5, and 6) had higher risk estimates based on the FS dataset, with Area 4 having the greatest increase in the estimated risk. This increase was largely the result of high PCB concentrations in two post-RI samples that were collected from the head of the inlet at RM 2.2W (i.e., 2,900,000 µg/kg dw and 230,000 µg/kg dw).

To provide additional information for risk communication, excess cancer risks were estimated separately for Duwamish Waterway Park (which is part of Area 5 [Figure 3-1]). In addition, excess cancer risks for Areas 4 and 5 were also estimated based on data for subsets of each of these areas. Area 4 was divided into two parts. The first part included all sediment samples except those in the inlet at RM 2.2W (referred to as Area 4 modified – without inlet). The other part included only those samples in the inlet at RM 2.2W (referred to as Area 4 modified – inlet only). Area 5 was divided into two parts. The first part (referred to as Area 5 modified – south) included the two southernmost sections of Area 5. The other part (referred to as Area 5 modified – north) included only the northernmost section of Area 5. These modified areas were assessed to facilitate remedial decision-making (i.e., clarify which portions of these beach play areas are causing most of the risk).

The estimated excess cancer risks for Duwamish Waterway Park were presented in Section 6 of the HHRA (Windward 2007b). The total excess cancer risk for arsenic, cPAHs, and total PCBs was 4×10^{-6}. No dioxin/furan data were available for Duwamish Waterway Park when the HHRA was completed. The updated total excess cancer risk estimate for Duwamish Waterway Park using the FS dataset for arsenic, cPAHs, total PCBs, and dioxins/furans was 2×10^{-6}.

The estimated total excess cancer risk for Area 4 modified – without inlet (1×10^{-5}) was much lower than that for either Area 4 modified – inlet only (3×10^{-3}) or for the entire Area 4 (6×10^{-4}) (Table 3-6b). This result is consistent with the higher concentrations of arsenic, dioxins/furans, cPAHs, and especially total PCBs found within the inlet. The estimated total excess cancer risk for Area 5 modified - south (4×10^{-6}) was also much lower than that for either Area 5 modified – north (5×10^{-5}) or for the entire Area 5 (3×10^{-5}) (Table 3-6b, Figure 3-1). This result was also consistent with the higher concentrations of cPAHs and dioxins/furans found in the northernmost portion of Area 5.

In addition to the increased excess cancer risk estimates for some beach play areas (as presented in Table 3-6b), the highest non-cancer HQ (Area 4) for total PCBs increased from 1 (as presented in the HHRA [Windward 2007b]) to 187 based on the newer (i.e., post-RI) data (Appendix B, Table B-2). The increase in the HQ is largely a result of the two samples with very high total PCB concentrations from the head of the inlet at RM 2.2W. If those two high total PCB concentrations were omitted, the non-cancer HQ for total PCBs for Area 4 would be 2 (similarly, the excess cancer risk would decrease from 6×10^{-4} to 6×10^{-6} if these two samples were excluded). The non-cancer HQ for total PCBs for Area 4 modified – without inlet is 0.4. This analysis suggests that the area of most concern is the inlet at Area 4 (which has been prioritized for remedial action in Alternative 2; see Section 8, Figure 8-6). None of the other beach play areas had non-cancer HQs greater than 1 for any contaminant.

3.2.3 Sum of Risks Across Multiple Exposure Scenarios

Risks for multiple exposure scenarios can be summed to represent possible exposure of the same individuals to LDW contaminants during different activities. Summed risks (i.e., the sum of risks across pathways) are presented in Table 3-7 for the following multiple exposure scenarios:

- Adult Tribal RME netfishing, Adult Tribal RME seafood consumption, and swimming
- Child Tribal RME seafood consumption, beach play RME, and swimming

- ♦ Adult Tribal RME clamming, Adult Tribal RME seafood consumption, and swimming.[9]

When estimated excess cancer risks were rounded to one significant figure, the sums for two of the three scenario groups above were the same as the estimates for the seafood (or clam) consumption alone. Summing risks for the Child Beach Play RME and swimming scenarios with the Child Tribal RME seafood consumption increased the estimated risks only slightly over those for seafood consumption alone. Overall, swimming had the lowest risk estimates.

This analysis demonstrates that the contributions to the sum of risks from netfishing, clamming, beach play, and swimming are relatively small in comparison to estimated risks from seafood consumption alone. This finding highlights the significance of the seafood consumption exposure pathway for all users of the LDW. Despite the lower magnitude of direct contact risks versus seafood consumption risks, several direct contact exposure risk estimates were close to the upper end of EPA's acceptable risk range of 1 in 10,000.

3.2.4 Risk Drivers for Human Health

Four COCs were selected as risk drivers for both the seafood consumption and direct sediment exposure scenarios: total PCBs, arsenic, cPAHs, and dioxins/furans.[10] A detailed explanation of the rationale for identifying these risk drivers can be found in Section 7 of the baseline HHRA (Windward 2007b) and is summarized in Table 3-8. Briefly, the risk drivers were selected based on the magnitude of their risk estimates and the relative percentage of their contributions to the total human health risk. Other factors considered in their selection were toxicological characteristics, persistence in the environment, natural background concentrations, and detection frequency. COCs not selected as risk drivers in the baseline HHRA are evaluated in Section 9.11 to assess the potential for risk reduction following remedial actions.

3.3 Risk-based Threshold Concentrations

For the LDW, RBTCs are concentrations of risk-driver COCs in sediment or tissue that are associated with specific risk estimates and exposure pathways. Cleanup of sediment to concentrations at or below a specific RBTC is predicted to be protective for the particular risk drivers, based on the exposure assumptions of the baseline risk assessments (Windward 2007a, 2007b). RBTCs for tissue and sediment were presented

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9 Although some individuals might engage in both netfishing and clamming, risks for these two scenarios were not summed, because engaging in both at the frequency assumed for each (more than 100 days per year) is unlikely.

10 Dioxins/furans were identified as a risk driver for human seafood consumption, even though no quantitative risk estimates were made.

in Section 8 of the RI (Windward 2010), and were used in this FS along with other site information to establish PRGs (as presented in Section 4).

3.3.1 Sediment RBTCs

Risk drivers for ecological receptors include the SMS contaminants with concentrations that exceeded the SQS in one or more surface sediment samples, as well as total PCBs for river otter; the risk drivers for human health include total PCBs, arsenic, cPAHs, and dioxins/furans. Sediment RBTCs for the ecological risk drivers include the following:

- The SQS and CSL sediment criteria from the SMS for the protection of benthic invertebrates (see Table 3-1 for these SMS values).
- Total PCB concentrations in sediment necessary to achieve sufficiently low total PCB concentrations in tissue for the protection of seafood consumption by river otters (128 to 159 µg/kg dw, depending on the diet assumptions for the river otter that were used in the ERA) (Table 3-9).

Sediment RBTCs for the human health risk drivers were calculated at three different excess cancer risk levels and for HQs equal to 1 (when the non-cancer hazard was greater than 1 in the HHRA) for both the direct contact with sediment scenarios (i.e., beach play, netfishing, and tribal clamming) and the seafood consumption scenarios. The equations used to calculate the sediment RBTCs are based on the risk equations used in the baseline HHRA (Windward 2007b).

Sediment RBTCs for the human health direct sediment contact exposure scenarios were calculated for all four risk drivers (i.e., PCBs, arsenic, cPAHs, and dioxins/furans) at all three excess cancer risk levels (Table 3-10). With one exception, sediment RBTCs were not calculated for non-cancer hazards (at an HQ of 1) because all HQs were less than or equal to 1 for the RME scenarios in the HHRA (Windward 2007b). The one exception was for the beach play RME scenario, for which the HQ calculated for total PCBs using the FS dataset for Area 4 was greater than 1.0 (see Section 3.2.2.2 for details).

Sediment RBTCs for the human health seafood consumption exposure scenarios represent the sediment concentrations at which tissue concentrations equate to the targeted risk level. Thus, these RBTCs require developing a relationship between concentrations in sediment and tissue, as described below for each risk driver.

- **Total PCB sediment RBTCs:** A food web model calibrated for the LDW (see Appendix D of the RI) was used to estimate the relationship between sediment and tissue concentrations for total PCBs, and to calculate sediment RBTCs. For the 1 in 10,000 (1×10^{-4}) excess cancer risk level, the food web model-calculated sediment RBTCs ranged from 7.3 to 185 µg/kg for the three RME scenarios (Table 3-9). For the excess cancer risk levels of 1 in 1,000,000 (1×10^{-6}) (required by MTCA) and 1 in 100,000 (1×10^{-5}) and for the non-cancer HQ of 1, total PCB sediment RBTCs were estimated to be less than 1 µg/kg dw (Table 3-9). Sediment RBTCs for these lower risk levels are especially difficult to quantify for several reasons. First, the food web model was calibrated for baseline conditions (i.e., a sediment concentration of 380 µg/kg PCBs), not post-remedy conditions. The greater the difference between baseline and post-remedy conditions, the greater the uncertainty in the model application. Second, at these very low sediment total PCB concentrations, the assumed total PCB concentration in water becomes increasingly important and is also uncertain. Because contaminant concentrations in both sediment and water contribute to tissue concentrations in aquatic organisms, even if total PCB sediment concentrations were assumed to be 0 µg/kg dw, water total PCB concentrations would need to be well below upstream Green River total PCB concentrations (which are currently 0.3 nanograms per liter [ng/L] on average) to calculate concentrations in tissue that would equate to these lower risk levels (see Section 3.3.2 for tissue RBTC discussion). While sediment contaminant concentrations can be directly addressed through source control and sediment remediation, surface water contaminant concentrations can only be indirectly addressed. The indirect methods make it difficult to estimate the extent to which surface water contaminant concentrations may be reduced. Only at substantially lower hypothetical water contaminant concentrations that are very probably unachievable for the LDW would the sediment RBTCs for the 1×10^{-5} or the 1×10^{-6} risk level be greater than 0 µg/kg dw (Figure 3-2). For example, using a hypothetical water concentration of 0.01 ng/L, the sediment RBTC would be greater than 0 µg/kg dw for the 1×10^{-5} risk level (equal to approximately 3.9 µg/kg dw, as shown in Figure 3-2).

- **Dioxin/furan sediment RBTCs:** The HHRA (Windward 2007b) was conducted with the assumption that risks associated with exposure to dioxins and furans through seafood consumption were unacceptable, and that the RTBCs for those risks would be more stringent than natural background concentrations. As a result, tissue data were not collected and analyzed to calculate specific exposure estimates, except for a limited data set collected from a small area of the LDW,[11] discussed in Section 3.2.1. Consequently, sediment RBTCs for dioxins/furans for seafood ingestion scenarios could not be, and were not, calculated. Because the RBTCs were assumed to be more stringent than natural background values, natural background values are used as sediment PRGs for dioxins/furans, as required by MTCA, to address seafood consumption in this FS in lieu of RBTCs (see Section 4).

[11] A total of six composite English sole fillets were collected in May 2007 near Kellogg Island and analyzed for dioxins/furans. Data for other seafood categories were not collected.

- **Arsenic and cPAH sediment RBTCs:** For arsenic and cPAHs, 95% or more of the risk associated with seafood consumption for the RME scenarios is attributable to the consumption of clams. Therefore, a relationship between arsenic and cPAHs concentrations in clams and sediment is required to estimate sediment RBTCs. However, despite efforts to better understand these relationships, EPA and Ecology agree with the Lower Duwamish Waterway Group that the clam tissue-to-sediment relationships based on the RI data for both arsenic and cPAHs were too uncertain to develop quantitative sediment RBTCs (see Sections 8.3.2 and 8.3.3 in the RI [Windward 2010]). For example, in some areas with elevated arsenic sediment concentrations, a corresponding elevation in clam tissue was not found, and other areas with comparatively low levels of arsenic in sediments contained clams with elevated arsenic tissue concentrations. Further research will be conducted prior to sediment remediation to better understand and characterize the relationship between sediment and tissue arsenic and cPAH concentrations. The results will inform remedial actions in clam habitat areas. The efficacy of completed remedial actions in reducing cPAH and arsenic concentrations in clams will be evaluated through monitoring. Further remedial actions may be required to reduce levels of cPAHs and arsenic in aquatic biota if initial efforts are unsuccessful.

3.3.2 Tissue RBTCs

Tissue RBTCs associated with the three RME seafood consumption scenarios were calculated for all four risk drivers (i.e., total PCBs, inorganic arsenic, cPAHs, and dioxins/furans) for excess cancer risk thresholds and for total PCBs and inorganic arsenic for a non-cancer HQ of 1 (Table 3-11). The risk equations and parameters used to calculate the tissue RBTCs are the same as those used in the RI, and are presented in Table 3-12. To derive the tissue RBTCs, these equations were solved for the concentration in seafood for a given target risk level using scenario-specific parameters (e.g., ingestion rates, body weights).

The tissue RBTCs for the seafood consumption scenarios presented in Table 3-11 represent the ingestion-weighted average concentrations in tissue that correspond to a certain risk threshold for each scenario. For example, the RBTC for total PCBs for the Adult Tribal RME seafood consumption scenario based on Tulalip data was 4.2 µg/kg ww at the 1×10^{-5} excess cancer risk level. Thus, the consumption of 97.5 g/day (the daily ingestion rate for the Adult Tribal RME scenario based on Tulalip data) of any tissue type with a total PCB concentration of 4.2 µg/kg ww for 70 years would result in a 1×10^{-5} excess cancer risk. The consumption of numerous types of seafood, such as crabs, clams, and fish (as specified in the exposure parameters for the Adult Tribal RME scenario based on Tulalip data), would also result in a 1×10^{-5} excess cancer risk as long as the ingestion-weighted average of the various tissue concentrations was 4.2 µg/kg ww. As shown in Table 3-11, the tissue RBTCs for the Adult Tribal RME scenario based on Tulalip data were lower than those for the other RME scenarios for a given risk threshold for each risk driver.

Species-specific tissue RBTCs for diets with a mixture of seafood (such as those for the RME scenarios evaluated in the HHRA) can also be calculated. These RBTCs are useful for comparison with single-species data collected during long-term monitoring programs to assess improvements in residual risks following cleanup actions. To calculate these RBTCs, two assumptions are required: 1) the diets in the RME scenarios remain the same over time, and 2) the relative concentrations in various seafood types consumed co-vary (i.e., decrease by a proportional amount) in the future. Changes in either of these assumptions would result in changes to species-specific tissue RBTCs. Uncertainty in RBTCs is associated with the use of these assumptions. Variability exists in the PCB concentration relationships between different organism/tissue types based on the different sources of PCB organism/tissue type data used to characterize these relationships. Data sources that were evaluated included: 1) PCB data used for the HHRA; 2) the food web model used to characterize PCB bioaccumulation; and 3) PCB data collected in 2007. It should be noted that the dataset used for the HHRA had more samples than the 2007 dataset because it represented a combination of many years of data.[12] The equations and methods used to calculate these RBTCs and the resulting species-specific tissue RBTC concentration ranges for PCBs are presented in Section B.3 of Appendix B.

Species-specific tissue RBTCs are presented in Tables 3-13 through 3-15 for the three RME seafood consumption scenarios. For informational purposes, LDW tissue data and tissue data from non-urban locations in Puget Sound are also presented.[13] Additional details regarding the Puget Sound dataset are provided in Appendix B, Section B.4. In addition, Figures 3-3 through 3-6 present the ingestion-weighted average RBTCs along with calculated ingestion-weighted average tissue concentrations based on the non-urban Puget Sound tissue dataset and on available LDW tissue data. These figures present ingestion-weighted tissue concentrations for the LDW and Puget Sound tissue datasets because these are more directly comparable to the RBTCs (which are based on market basket consumption). These ingestion-weighted concentrations were calculated by multiplying the tissue concentration for each consumption category by its percent of the total consumption rate, and then summing the results.

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[12] The dataset used to evaluate risks in the HHRA contained 221 total PCB tissue samples from throughout the LDW between 1992 and 2005. The 2007 dataset contains a total of 86 tissue samples (including benthic fish, pelagic fish, clam, and crab samples), which were intended to characterize tissue concentrations further in the LDW.

[13] Tables 3-13 through 3-15 present the LDW tissue data used to calculate risks in the HHRA (which includes samples collected between 1992 and 2005). Additional tissue samples collected from the LDW in 2007 are not shown in these tables, but can be found in the RI (Sections 4 and 8).

3.4 Key Findings of the Baseline Risk Assessments

Key findings for the baseline ERA (Windward 2007a) and HHRA (Windward 2007b) are as follows:

- Forty-one of the 44 COCs were identified as risk drivers for benthic invertebrates because concentrations of these 41 COCs in surface sediment exceed the SQS criteria at one or more locations (Table 3-16).
- For benthic invertebrates living in intertidal and subtidal sediment, sediment contaminant concentrations and site-specific sediment toxicity test results indicated that harmful effects are not likely in approximately 75% of the LDW area based on the RI dataset (or 82% based on the FS dataset).[14] There is a higher likelihood for adverse effects in approximately 7% of the LDW area (4% based on the FS dataset), where contaminant concentrations or biological effects were found to be in excess of the CSL criteria. The remaining 18% of the LDW study area (14% based on the FS dataset) had contaminant concentrations or biological effects between the SQS and CSL, indicating that risks to benthic invertebrate communities are less certain in these areas than in areas with contaminant concentrations greater than one or more CSL values. The samples with concentrations that exceeded the SMS criteria are geospatially concentrated in multiple areas that cumulatively represent about 25% of the LDW sediment surface (18% based on the FS dataset).
- Sediment RBTCs for the benthic invertebrate community were established at the SQS and CSL criteria of the SMS.
- In consultation with EPA and Ecology, PCBs were identified as a risk driver for river otters (Tables 3-2 and 3-16). The wildlife sediment RBTCs for PCBs were calculated using the food web model based on seafood consumption by river otters. No other risk drivers were identified for crabs, fish, or other wildlife (Table 3-2).
- The highest risks to people were associated with the consumption of seafood, including resident fish, crabs, and clams (Tables 3-4a and 3-4b). Lower risks were associated with activities that involve direct contact with sediment, such as clamming, beach play, and netfishing (Tables 3-6a and 3-6b).
- Total PCBs, arsenic, cPAHs, and dioxins/furans were identified as risk drivers for human health (Tables 3-8 and 3-16).

[14] Estimated areas with exceedances were based on the RI or FS baseline surface sediment datasets (as specified in text) and Thiessen polygons.

- For total PCBs, sediment RBTCs ranged from 7.3 to 185 µg/kg dw for the 1 in 10,000 (1×10^{-4}) excess cancer risk level for the three RME scenarios (Table 3-9). RBTCs for the 10^{-5} and 10^{-6} risk levels and the non-cancer RBTC for total PCBs for the RME seafood consumption scenarios were less than 1 µg/kg dw.
- For arsenic and cPAHs, 95% or more of the risk associated with seafood consumption is attributable to the consumption of clams. Because the clam tissue-to-sediment contaminant concentration relationships in the RI/FS data were too uncertain to support developing quantitative sediment RBTCs for these risk drivers, sediment RBTCs were not derived. Clam tissue and sediment relationships for arsenic and cPAHs and methods to reduce concentrations of these contaminants in clam tissue will be subject to further study prior to sediment remediation.
- For dioxins/furans, sediment RBTCs for seafood consumption were not calculated because risks for the LDW were assumed to be unacceptable. Also, RBTCs for those risks were assumed to be more stringent than the natural background concentrations to which they would default for final cleanup decision-making under MTCA. As a result, tissue dioxin/furan data were not collected and analyzed for specific exposure estimates. Without these data, sediment RBTCs for seafood ingestion scenarios could not be calculated. Natural background values are the sediment PRGs for dioxins/furans to address seafood consumption in this FS in lieu of risk-based RBTCs (see Section 4). If RBTCs had been calculated and they were more stringent than natural background values as was assumed, these same natural background values would be the PRGs.
- Sediment RBTCs for RME direct sediment contact scenarios were calculated for all four risk drivers and all three risk levels (Table 3-10).
- Tissue RBTCs for excess cancer risks at the three risk levels were calculated for seafood consumption scenarios for all four risk drivers; non-cancer hazard RBTCs were calculated for total PCBs and arsenic (Table 3-11). Species-specific RBTCs were also calculated for comparison with LDW and non-urban Puget Sound tissue concentrations (Tables 3-13 through 3-15; Figures 3-3 through 3-6).

The risk screening process used to identify COPCs, COCs, and risk drivers for human health and ecological receptors is summarized in Table 3-16. The COCs not selected as risk drivers are evaluated in Section 9 to assess the potential for risk reduction following remedial actions.

Table 3-1 Summary of COCs and Selection of Risk Drivers for Benthic Invertebrates

COPC	SMS Criteria			No. of Detected Concentrations in Surface Sediments		Benthic COC?	Benthic Risk Driver?	Rationale for Selection/Exclusion as Risk Driver
	Unit	SQS	CSL	> SQS, < CSL	> CSL			
Metals (mg/kg dw)								
Arsenic	mg/kg dw	57	93	5	8	Yes	Yes	Detected concentration(s) > SQS
Cadmium	mg/kg dw	5.1	6.7	2	11	Yes	Yes	Detected concentration(s) > SQS
Chromium	mg/kg dw	260	270	1	8	Yes	Yes	Detected concentration(s) > SQS
Copper	mg/kg dw	390	390	0	12	Yes	Yes	Detected concentration(s) > SQS
Lead	mg/kg dw	450	530	2	19	Yes	Yes	Detected concentration(s) > SQS
Mercury	mg/kg dw	0.41	0.59	14	23	Yes	Yes	Detected concentration(s) > SQS
Nickel [a]	n/a	n/a	n/a	9 (DMMP SL)	4 (DMMP ML)	Yes	No	Moderate TRV uncertainty; areas with concentrations greater than the TRV were all in planned sediment remediation areas
Silver	mg/kg dw	6.1	6.1	0	10	Yes	Yes	Detected concentration(s) > SQS
Zinc	mg/kg dw	410	960	26	16	Yes	Yes	Detected concentration(s) > SQS
PAHs (mg/kg oc)								
2-Methylnaphthalene	mg/kg oc	38	64	0	3	Yes	Yes	Detected concentration(s) > SQS
Acenaphthene	mg/kg oc	16	57	16	3	Yes	Yes	Detected concentration(s) > SQS
Acenaphthylene	mg/kg oc	66	66	0	0	No	No	No detected concentration(s) > SQS
Anthracene	mg/kg oc	220	1,200	2	0	Yes	Yes	Detected concentration(s) > SQS
Benzo(a)anthracene	mg/kg oc	110	270	9	3	Yes	Yes	Detected concentration(s) > SQS
Benzo(a)pyrene	mg/kg oc	99	210	5	3	Yes	Yes	Detected concentration(s) > SQS
Benzo(g,h,i)perylene	mg/kg oc	31	78	9	7	Yes	Yes	Detected concentration(s) > SQS
Total benzofluoranthenes	mg/kg oc	230	450	5	4	Yes	Yes	Detected concentration(s) > SQS
Chrysene	mg/kg oc	110	460	23	1	Yes	Yes	Detected concentration(s) > SQS
Dibenzo(a,h) anthracene	mg/kg oc	12	33	15	4	Yes	Yes	Detected concentration(s) > SQS
Dibenzofuran	mg/kg oc	15	58	7	3	Yes	Yes	Detected concentration(s) > SQS

Table 3-1 Summary of COCs and Selection of Risk Drivers for Benthic Invertebrates (continued)

COPC	SMS Criteria			No. of Detected Concentrations in Surface Sediments		Benthic COC?	Benthic Risk Driver?	Rationale for Selection/Exclusion as Risk Driver
	Unit	SQS	CSL	> SQS, < CSL	> CSL			
Fluoranthene	mg/kg oc	160	1,200	31	8	Yes	Yes	Detected concentration(s) > SQS
Fluorene	mg/kg oc	23	79	11	3	Yes	Yes	Detected concentration(s) > SQS
Indeno(1,2,3-cd) pyrene	mg/kg oc	34	88	15	8	Yes	Yes	Detected concentration(s) > SQS
Naphthalene	mg/kg oc	99	170	0	2	Yes	Yes	Detected concentration(s) > SQS
Phenanthrene	mg/kg oc	100	480	24	3	Yes	Yes	Detected concentration(s) > SQS
Pyrene	mg/kg oc	1,000	1,400	1	3	Yes	Yes	Detected concentration(s) > SQS
Total HPAH	mg/kg oc	960	5,300	21	3	Yes	Yes	Detected concentration(s) > SQS
Total LPAH	mg/kg oc	370	780	3	3	Yes	Yes	Detected concentration(s) > SQS
Phthalates (mg/kg oc)								
Bis(2-ethylhexyl) phthalate	mg/kg oc	47	78	48	58	Yes	Yes	Detected concentration(s) > SQS
Butyl benzyl phthalate	mg/kg oc	4.9	64	69	8	Yes	Yes	Detected concentration(s) > SQS
Diethyl phthalate	mg/kg oc	61	110	0	0	No	No	No detected concentration(s) > SQS
Dimethyl phthalate	mg/kg oc	53	53	0	2	Yes	Yes	Detected concentration(s) > SQS
Di-n-butyl phthalate	mg/kg oc	220	1,700	0	0	No	No	No detected concentration(s) > SQS
Di-n-octyl phthalate	mg/kg oc	58	4,500	0	0	No	No	No detected concentration(s) > SQS
Other SVOCs (mg/kg oc)								
1,2,4-Trichlorobenzene	mg/kg oc	0.81	1.8	0	1	Yes	Yes	Detected concentration(s) > SQS
1,2-Dichlorobenzene	mg/kg oc	2.3	2.3	0	3	Yes	Yes	Detected concentration(s) > SQS
1,4-Dichlorobenzene	mg/kg oc	3.1	9	0	3	Yes	Yes	Detected concentration(s) > SQS
2,4-Dimethylphenol	µg/kg dw	29	29	0	1	Yes	Yes	Detected concentration(s) > SQS
2-Methylphenol	µg/kg dw	63	63	0	0	No	No	No detected concentration(s) > SQS
4-Methylphenol	µg/kg dw	670	670	0	4	Yes	Yes	Detected concentration(s) > SQS
Benzoic acid	µg/kg dw	650	650	0	7	Yes	Yes	Detected concentration(s) > SQS

Table 3-1 Summary of COCs and Selection of Risk Drivers for Benthic Invertebrates (continued)

COPC	SMS Criteria			No. of Detected Concentrations in Surface Sediments		Benthic COC?	Benthic Risk Driver?	Rationale for Selection/Exclusion as Risk Driver
	Unit	SQS	CSL	> SQS, < CSL	> CSL			
Benzyl alcohol	µg/kg dw	57	73	2	2	Yes	Yes	Detected concentration(s) > SQS
Hexachlorobenzene	mg/kg oc	0.38	2.3	4	2	Yes	Yes	Detected concentration(s) > SQS
Hexachlorobutadiene	mg/kg oc	3.9	6.2	0	0	No	No	No detected concentration(s) > SQS
n-Nitrosodiphenylamine	mg/kg oc	11	11	0	2	Yes	Yes	Detected concentration(s) > SQS
Pentachlorophenol	µg/kg dw	360	690	1	0	Yes	Yes	Detected concentration(s) > SQS
Phenol	µg/kg dw	420	1,200	18	7	Yes	Yes	Detected concentration(s) > SQS
PCBs (mg/kg oc)								
Total PCBs	mg/kg oc	12	65	301	173	Yes	Yes	Detected concentration(s) > SQS
Pesticides								
Total DDTs [a]	n/a	n/a	n/a	1 (NOAEL)	1 (LOAEL)	Yes	No	Moderate TRV uncertainty; the 1 sample with a concentration greater than the TRV is in a planned sediment remediation area
Total chlordane [a]	n/a	n/a	n/a	19 (NOAEL)	14 (LOAEL)	Yes	No	High uncertainty in exposure data and TRV; 13 of 14 samples with LOAEL exceedances were in planned sediment remediation areas

Notes:

1. This table is derived from Table 5-6 of the RI (Windward 2010).
2. Statistics in this table were calculated using the RI baseline dataset.

a. No SMS numerical criteria were available for these contaminants. Thus, the comparison is with the DMMP SL and ML for nickel or with the NOAEL or LOAEL for total DDTs and total chlordane.

COC = contaminant of concern; CSL = cleanup screening level of SMS; DDT = dichlorodiphenyltrichloroethane; DMMP = Dredged Material Management Program; HPAH = high-molecular-weight polycyclic aromatic hydrocarbon; HQ = hazard quotient; LOAEL = lowest-observed-adverse-effect level; LPAH = low-molecular-weight polycyclic aromatic hydrocarbon; mg/kg oc = milligrams per kilogram organic carbon; ML = maximum level; n/a = not applicable; NOAEL = no-observed-adverse-effect level; oc – organic carbon; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; RI = remedial investigation; SL = screening level; SMS = Washington State Sediment Management Standards; SQS = sediment quality standard of SMS; SVOC = semivolatile organic compound; TRV = toxicity reference value

Table 3-2 Summary of COCs and Selection of Risk Drivers for Crab, Fish, and Wildlife Species

COC[a]	Receptor of Concern	NOAEL-based HQ	LOAEL-based HQ	Risk Driver?	Rationale for Selection or Exclusion as Risk Driver
Total PCBs	Crabs	**10**	**1.0**	No	Low risk estimate (LOAEL HQ equal to 1.0) and high level of uncertainty associated with TRV and exposure data.
	English Sole	**4.9 – 25**	0.98 – **5.0**	No	Exposure concentrations were within the LOAEL range. A LOAEL range was used because of the high level of uncertainty associated with the TRV.
	Pacific Staghorn Sculpin	**1.5 – 19**	0.30 – **3.8**	No	
	River Otter	**5.8**	**2.9**	Yes	LOAEL-based HQ for river otter was greater than 1.0 (HQ of 2.9), and the uncertainties associated with the exposure and effects data were relatively low.
Total PCBs and PCB TEQ	Spotted Sandpiper	**1.9 – 15**	0.18 – **1.5**	No	LOAEL-based HQs for total PCBs were less than 1.0, but equal to 1.5 for PCB TEQ. The effects data used to calculate risk estimates for total PCBs were less uncertain than those for PCB TEQ.
Cadmium	Juvenile Chinook Salmon	**5.0**	**1.0**	No	High level of uncertainty associated with the selected TRV and low risk estimates.
	English Sole	**6.1**	**1.2**	No	
	Pacific Staghorn Sculpin	**3.0 – 5.2**	0.60 – **1.0**	No	
Chromium	Spotted Sandpiper	**1.3 – 8.8**	0.26 – **1.8**	No	Elevated risks were driven by a single benthic invertebrate tissue sample (and co-located sediment was not elevated).
Copper	Spotted Sandpiper	0.62 – **1.5**	0.45 – **1.1**	No	Sediment concentrations were similar to PSAMP rural Puget Sound concentrations, and HQs will be less than 1 following planned sediment remediation in EAAs.
Lead	Spotted Sandpiper	0.58 – **19**	0.17 – **5.5**	No	Elevated risks were driven by a single benthic invertebrate tissue sample (and co-located sediment was not elevated).
Mercury	Spotted Sandpiper	**1.1 – 5.3**	0.21 – **1.0**	No	HQs will be less than 1 following planned sediment remediation in EAAs.
Vanadium	English Sole	**5.9**	**1.2**	No	High uncertainty in effects data (few toxicity studies), and sediment concentrations of vanadium in exposure areas were less than the 90th percentile vanadium concentration in PSAMP rural Puget Sound sediment.
	Pacific Staghorn Sculpin	**3.2 – 5.9**	0.65 – **1.2**	No	
	Spotted Sandpiper	**2.0 – 2.7**	**1.0 – 1.4**	No	

Notes:
1. This table is derived from Table 5-16 of the RI (Windward 2010).
2. HQs for fish are highest when more than one approach was used.
3. **Bold** identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

a. A contaminant was identified as a COC if the LOAEL-based HQ was greater than or equal to 1.0.

COC = contaminant of concern; EAA = early action area; HQ = hazard quotient; LOAEL = lowest-observed-adverse-effect level; NOAEL = no-observed-adverse-effect level; PCB = polychlorinated biphenyl; PSAMP = Puget Sound Ambient Monitoring Program; RI = remedial investigation; TEQ = toxic equivalent; TRV = toxicity reference value

Table 3-3 Summary of COCs for Human Health Seafood Consumption and Direct-Contact Sediment Exposure Scenarios

COC[a]	Human Health Exposure Pathway	
	Seafood Consumption	Direct Contact
Total PCBs[b]	X	X
Arsenic	X	X
cPAHs	X	X
Dioxins/furans	X	X
Aldrin[c]	X	
BEHP	X	
Alpha-BHC[c]	X	
Beta-BHC[c]	X	
Carbazole[c]	X	
Total chlordane[c]	X	
Total DDTs[c]	X	
Dieldrin[c]	X	
Gamma-BHC[c]	X	
Heptachlor[c]	X	
Heptachlor epoxide[c]	X	
Hexachlorobenzene[c]	X	
Pentachlorophenol	X	
TBT	X	
Toxaphene[c]		X
Vanadium	X	

Notes:

a. Contaminants with an excess cancer risk greater than 1×10^{-6} or a non-cancer HQ greater than 1 for at least one RME seafood consumption scenario were identified as COCs.

b. PCB TEQ was also identified as having risks greater than 1×10^{-6} for at least one RME seafood consumption scenario and at least one RME direct contact scenario.

c. These contaminants were qualified as tentatively identified compounds at estimated concentrations (JN-qualified), indicating uncertainty regarding both their presence and concentration.

BEHP = bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; DDT = dichlorodiphenyltrichloroethane; HQ = hazard quotient; PCB = polychlorinated biphenyl; RME = reasonable maximum exposure; TBT = tributyltin; TEQ = toxic equivalent

Table 3-4a Summary of Estimated Excess Cancer Risks for the Seafood Consumption Scenarios

COC	Scenarios Evaluated in the FS			Scenarios for Informational Purposes							
								Adult One Meal per Month			
COC	Adult Tribal RME (Tulalip Data)[a]	Child Tribal RME (Tulalip Data)[a]	Adult API RME	Adult Tribal CT (Tulalip Data)[a]	Child Tribal CT (Tulalip Data)[a]	Adult Tribal (Suquamish Data)	Adult API CT	Benthic Fish	Clam	Crab	Pelagic Fish
Arsenic (inorganic)[b]	2×10^{-3}	3×10^{-4}	7×10^{-4}	7×10^{-5}	3×10^{-5}	2×10^{-2} [c]	1×10^{-5}	4×10^{-7}	1×10^{-4}	3×10^{-6}	6×10^{-6}
BEHP	6×10^{-6}	1×10^{-6}	2×10^{-6}	2×10^{-7}	7×10^{-8}	4×10^{-5}	3×10^{-8}	8×10^{-7}	8×10^{-8}	8×10^{-8} [d]	1×10^{-6}
cPAHs[e]	8×10^{-5}	8×10^{-5}	3×10^{-5}	4×10^{-6}	9×10^{-6}	8×10^{-4}	8×10^{-7}	2×10^{-7}	7×10^{-6}	2×10^{-7}	3×10^{-7}
Dioxin/furans[f]	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
PCB TEQ	1×10^{-3}	2×10^{-4}	4×10^{-4}	5×10^{-5}	2×10^{-5}	7×10^{-3}	7×10^{-6}	8×10^{-5}	2×10^{-5}	2×10^{-5}	2×10^{-4}
Total PCBs	2×10^{-3}	3×10^{-4}	5×10^{-4}	6×10^{-5}	3×10^{-5}	1×10^{-2} [c]	8×10^{-6}	1×10^{-4}	5×10^{-5}	2×10^{-5}	2×10^{-4}
Pentachlorophenol[b]	9×10^{-5} [g]	2×10^{-5} [g]	2×10^{-5}	2×10^{-6} [g]	7×10^{-7} [g]	5×10^{-4} [g]	3×10^{-7}	2×10^{-5} [d]	1×10^{-6} [d]	2×10^{-6} [d]	1×10^{-5}
Subtotal (excluding PCB TEQ)	**4×10^{-3}**	**7×10^{-4}**	**1×10^{-3}**	**1×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Subtotal (excluding total PCBs)	**3×10^{-3}**	**6×10^{-4}**	**1×10^{-3}**	**1×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Tentatively Identified Compounds (JN-qualified)											
Aldrin	5×10^{-5} [g]	8×10^{-6} [g]	1×10^{-5}	1×10^{-6} [g]	6×10^{-7} [g]	2×10^{-4}	2×10^{-7}	3×10^{-6} [d]	8×10^{-7} [d]	3×10^{-6} [d]	3×10^{-6}
alpha-BHC	2×10^{-5} [g]	3×10^{-6} [g]	3×10^{-6}	5×10^{-7} [g]	2×10^{-7} [g]	6×10^{-5}	6×10^{-8}	1×10^{-6}	1×10^{-7}	1×10^{-6} [d]	1×10^{-6}
beta-BHC	6×10^{-6} [g]	1×10^{-6} [g]	1×10^{-6}	2×10^{-7} [g]	1×10^{-7} [g]	3×10^{-5}	3×10^{-8}	3×10^{-7}	1×10^{-7}	3×10^{-7} [d]	6×10^{-7}
Carbazole	4×10^{-5}	8×10^{-6}	1×10^{-5}	9×10^{-7}	4×10^{-7}	2×10^{-4}	8×10^{-8}	1×10^{-6} [d]	9×10^{-8} [d]	1×10^{-6} [d]	1×10^{-5}
Total chlordane	6×10^{-6}	1×10^{-6}	2×10^{-6}	2×10^{-7}	8×10^{-8}	3×10^{-5}	3×10^{-8}	3×10^{-7}	7×10^{-8}	7×10^{-8}	1×10^{-6}
Total DDTs	2×10^{-5}	4×10^{-6}	6×10^{-6}	1×10^{-6}	4×10^{-7}	1×10^{-4}	1×10^{-7}	1×10^{-6}	2×10^{-7}	4×10^{-7}	4×10^{-6}
Dieldrin	1×10^{-4}	3×10^{-5}	5×10^{-5}	3×10^{-6}	1×10^{-6}	1×10^{-3}	4×10^{-7}	3×10^{-6} [d]	9×10^{-6}	3×10^{-6}	3×10^{-6} [d]
gamma-BHC	5×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-7}	5×10^{-8}	3×10^{-5}	1×10^{-8}	2×10^{-7} [d]	1×10^{-7}	2×10^{-7}	1×10^{-7}
Heptachlor	1×10^{-5} [g]	3×10^{-6} [g]	3×10^{-6}	4×10^{-7} [g]	2×10^{-7} [g]	6×10^{-5}	4×10^{-8}	7×10^{-7} [d]	1×10^{-7} [d]	7×10^{-7} [d]	2×10^{-6}

Table 3-4a Summary of Estimated Excess Cancer Risks for the Seafood Consumption Scenarios (continued)

COC	Scenarios Evaluated in the FS			Scenarios for Informational Purposes							
								Adult One Meal per Month			
	Adult Tribal RME (Tulalip Data)[a]	Child Tribal RME (Tulalip Data)[a]	Adult API RME	Adult Tribal CT (Tulalip Data)[a]	Child Tribal CT (Tulalip Data)[a]	Adult Tribal (Suquamish Data)	Adult API CT	Benthic Fish	Clam	Crab	Pelagic Fish
Heptachlor epoxide	3×10^{-5}	6×10^{-6}	9×10^{-6}	1×10^{-6}	5×10^{-7}	2×10^{-4}	1×10^{-7}	1×10^{-6} [d]	6×10^{-7}	9×10^{-7}	4×10^{-6}
Hexachlorobenzene	1×10^{-5}	2×10^{-6}	2×10^{-6}	2×10^{-7}	1×10^{-7}	4×10^{-5}	3×10^{-8}	6×10^{-7}	6×10^{-8}	6×10^{-7}	9×10^{-7}
Subtotal	**3×10^{-4}**	**7×10^{-5}**	**1×10^{-4}**	**9×10^{-6}**	**4×10^{-6}**	**2×10^{-3}**	**1×10^{-6}**	**1×10^{-5}**	**1×10^{-5}**	**1×10^{-5}**	**3×10^{-5}**
Total excess cancer risk (excluding PCB TEQ)	**4×10^{-3}**	**8×10^{-4}**	**1×10^{-3}**	**1×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**
Total excess cancer risk (excluding total PCBs)	**3×10^{-3}**	**7×10^{-4}**	**1×10^{-3}**	**1×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**

Notes:

a. The excess cancer risk estimates reported here differ slightly from those reported in Appendix B (the HHRA) (Windward 2007b) and Section 6 of the RI (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish but not the total quantity consumed) for scenarios based on the Tulalip Tribes survey was updated in response to a correction provided by EPA. The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates were described in detail in an erratum (Windward 2009) to the HHRA (Windward 2007b).

b. No mussel data were available for this COC. When the chronic daily intake and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

c. Because the excess cancer risk is greater than or equal to 0.01, risk was calculated using the exponential equation in EPA (1989).

d. There were no detected values in this seafood category. Chronic daily intake and risk estimate were based on one-half the maximum reporting limit.

e. cPAHs are presented as benzo(a)pyrene TEQs. Data used in the risk characterization were only from 2004 because of high reporting limits in historical data. All cPAH data were analyzed in the uncertainty analysis (Appendix B of the RI, Section B.6, Windward 2010). Because of the potential for the increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005a), the risk estimate for children for cPAHs was based on dose adjustments across the 0-to-6-year-old age range of children. See the HHRA in Appendix B of the RI, Section B.5.1 (Windward 2010), for more information.

f. Tissue data for dioxins/furans were not collected. Thus, the calculated total risk, which does not include risks from dioxins/furans, is underestimated to an unknown degree.

g. Greater than 50% of the risk associated with this contaminant is derived from seafood categories with no detected values.

API = Asian and Pacific Islander; BEHP = bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; CT = central tendency; DDT = dichlorodiphenyltrichloroethane; EPA = U.S. Environmental Protection Agency; FS = feasibility study; HHRA = human health risk assessment; n/a = not available; PCB = polychlorinated biphenyl; RI = remedial investigation; RME = reasonable maximum exposure; TEQ = toxic equivalent

Table 3-4b Summary of Estimated Non-cancer Hazards for the Seafood Consumption Scenarios

Contaminants and Hazard Indices	Scenarios Evaluated in the FS			Scenarios for Informational Purposes							
	Adult Tribal RME (Tulalip Data)[a]	Child Tribal RME (Tulalip Data)[a]	Adult API RME	Adult Tribal CT (Tulalip Data)[a]	Child Tribal CT (Tulalip Data)[a]	Adult Tribal (Suquamish Data)	Adult API CT	Adult One Meal per Month			
								Benthic Fish	Clam	Crab	Pelagic Fish
Hazard Quotients for COPCs with HQs > 1 for One or More Scenarios[b]											
Arsenic (inorganic)[c]	4	8	3	0.4	0.7	38	0.2	0.002	0.7	0.01	0.03
Chromium[d]	0.2	0.4	0.1	0.02	0.04	2	0.01	0.002	0.03	0.006	0.007
Mercury[e]	0.5	1	0.3	0.07	0.1	2	0.02	0.06	0.02	0.07	0.04
Total PCBs	40	87	29	4	8	274	2	6	3	1	10
TBT (as ion)	2	3	1	0.2	0.4	15	0.1	0.002	0.3	0.02	0.06
Vanadium	0.9	2	0.8	0.1	0.3	9	0.07	0.01	0.2	0.01	0.06
Hazard Indices by Effect (Endpoint)[b]											
HI for cardiovascular endpoint[f]	5	10	4	0.5	1	47	0.3	0.01	0.9	0.02	0.09
HI for developmental endpoint[g]	41	88	29	4	8	276	2	6	3	1	10
HI for hematologic endpoint[h]	0.2	0.5	0.2	0.03	0.05	2	0.01	0.006	0.03	0.01	0.009
HI for immunological endpoint[i]	42	90	30	4	8	289	2	6	3	1	10
HI for kidney endpoint[j]	0.4	0.9	0.3	0.05	0.1	2	0.02	0.03	0.03	0.04	0.04
HI for liver endpoint[k]	1	2	0.8	0.1	0.3	7	0.05	0.1	0.1	0.09	0.3
HI for neurological endpoint[l]	41	88	29	4	8	276	2	6	3	1	10
HI for dermal endpoint[m]	4	8	3	0.4	0.7	38	0.2	0.01	0.7	0.02	0.06

Notes:

a. The non-cancer HIs reported here differ slightly from those reported in the HHRA (Windward 2007b) and Section 6 of the RI (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish but not the total quantity consumed) for scenarios based on the Tulalip Tribes survey was updated in response to a correction provided by EPA. The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates are described in detail in an erratum (Windward 2009) to the HHRA (Windward 2007b).

b. Hazard indices include risks associated with all COPCs by endpoint. However, only those COPCs with an HQ greater than or equal to 1 for at least one RME scenario are listed in this table.

Table 3-4b Summary of Estimated Non-cancer Hazards for the Seafood Consumption Scenarios (continued)

c. No mussel data were available for this COPC. When calculating the risk values, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.
d. Chromium HQ did not exceed 1 for any RME scenario, so it is not a COC. It is included in this table because the HQ exceeded 1 for the adult tribal (Suquamish data) scenario.
e. Mercury HQ did not exceed 1 for any RME scenario, so it is not a COC. It is included in this table because the HQ exceeded 1 for the adult tribal (Suquamish data) scenario.
f. Cardiovascular endpoint is for arsenic and vanadium.
g. Developmental endpoint is for total PCBs and mercury.
h. Hematologic endpoint is for antimony and zinc. Individual HQs for these COPCs are not presented because none are equal to or greater than 1 for any scenario.
i. Immunological endpoint is for total PCBs and TBT.
j. Kidney endpoint is for 4-methylphenol, cadmium, copper, gamma-BHC, and pentachlorophenol. Individual HQs for these COPCs are not presented because none are equal to or greater than 1 for any scenario.
k. Liver endpoint is for 4-methylphenol, aldrin, alpha-BHC, beta-BHC, BEHP, butyl benzyl phthalate, chlordane, copper, total DDTs, dieldrin, endrin, endrin aldehyde, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, and pentachlorophenol. Individual HQs for these COPCs are not presented because none are equal to or greater than 1 for any scenario.
l. Neurological endpoint is for 4-methylphenol, mercury, and total PCBs. Individual HQs for 4-methylphenol are not presented because none are equal to or greater than 1 for any scenario.
m. Dermal endpoint is for 4-methylphenol and arsenic. Individual HQs for 4-methylphenol are not presented because none are equal to or greater than 1 for any scenario.

API = Asian and Pacific Islander; BEHP = bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; COC = contaminant of concern; COPC = contaminant of potential concern; CT = central tendency; DDT = dichlorodiphenyltrichloroethane; EPA = U.S. Environmental Protection Agency; HI = hazard index; HQ = hazard quotient; HHRA = human health risk assessment; PCB = polychlorinated biphenyl; RI = remedial investigation; RME = reasonable maximum exposure; TBT = tributyltin

Table 3-5 Summary of Estimated Excess Cancer Risks for the RME Seafood Consumption Scenarios

COC	Adult Tribal RME (Tulalip Data)[a]		Child Tribal RME (Tulalip Data)[a]		Adult API RME	
	Excess Cancer Risk (% of Total)	Percent of Risk by Seafood Consumption Category	Excess Cancer Risk (% of Total)	Percent of Risk by Seafood Consumption Category	Excess Cancer Risk (% of Total)	Percent of Risk by Seafood Consumption Category
Arsenic (inorganic)	2×10^{-3} (44%)	97% clams; 1.3% crab EM; 1.1% crab WB; 0.8% pelagic; 0.06% benthic fillet	3×10^{-4} (40%)	97% clams; 1.3% crab EM; 1.1% crab WB; 0.8% pelagic; 0.06% benthic fillet	7×10^{-4} (50%)	98% clams; 0.9% crab WB; 0.7% pelagic; 0.4% crab EM; 0.05% benthic WB; 0.02% benthic fillet
cPAHs	8×10^{-5} (2%)	96% clams; 2.1% crab EM; 0.9% crab WB; 0.8% pelagic; 0.5% benthic fillet	8×10^{-5} (11%)	96% clams; 2.1% crab EM; 0.9% crab WB; 0.8% pelagic; 0.5% benthic fillet	3×10^{-5} (3%)	98% clams; 0.8% crab WB; 0.8% pelagic; 0.6% crab EM; 0.2% benthic WB; 0.2% benthic fillet
Total PCBs	2×10^{-3} (43%)	39% clams; 23% pelagic; 15% crab WB; 14% benthic fillet; 9% crab EM, 0.05% mussels	3×10^{-4} (39%)	39% clams; 23% pelagic; 15% crab WB; 14% benthic fillet; 9% crab EM, 0.05% mussels	5×10^{-4} (38%)	47% clams; 25% pelagic; 15% crab WB; 6.5% benthic fillet; 3% crab EM; 3% benthic WB; 0.5% mussels
Other COCs (BEHP, PCP, aldrin, alpha-BHC, beta-BHC, carbazole, chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, and hexachlorobenzene)[b]	4×10^{-4} (11%)	Average contribution: 29% crab EM, 29% pelagic, 20% clam, 14% benthic fillet, 9% crab WB, 0.3% mussels	8×10^{-5} (10%)	Average contribution: 29% crab EM, 29% pelagic, 20% clam, 14% benthic fillet, 9% crab WB, 0.3% mussels	1×10^{-4} (9%)	Average contribution: 35% pelagic, 29% clam, 13% crab EM, 11% crab WB, 7% benthic fillet, 3% mussels, 2% benthic WB
Total excess cancer risk and main contributors to the total excess cancer risk[c, d, e]	**4×10^{-3}**	**42% – arsenic in clams 17% – PCBs in clams 10% – PCBs in pelagic fish 6% – PCBs in WB crab 6% – PCBs in benthic fillet 19% – other**	**8×10^{-4}**	**39% – arsenic in clams 15% – PCBs in clams 10% – cPAHs in clams 9% – PCBs in pelagic fish 6% – PCBs in WB crab 5% – PCBs in benthic fillet 16% – other**	**1×10^{-3}**	**49% – arsenic in clams 18% – PCBs in clams 10% – PCBs in pelagic fish 6% – PCBs in WB crab 17% – other**

Notes:

a. The excess cancer risk estimates reported here differ slightly from those reported in the HHRA (Windward 2007b) and Section 6 of the RI (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish but not the total quantity consumed) for scenarios based on the Tulalip Tribes survey was updated in response to a correction provided by EPA. The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates were described in detail in an erratum (Windward 2009) to the HHRA (Windward 2007b).

b. Top contributors were dieldrin (approximately 3 to 4%) and pentachlorophenol (approximately 1.5 to 2.5%). All other COCs contributed less than 1.5%.

c. Seafood consumption category-COC combinations contributing greater than 5% of the total risk are listed separately. All other combinations are included in the "other" category.

d. Tissue data for dioxins/furans were not available at the time that the HHRA was finalized. After the HHRA had been finalized, data became available for six skin-off English sole fillets from a May 2007 PSAMP sampling effort near Kellogg Island (Gries 2008). Based on these data, the risks associated with dioxins/furans would be 6×10^{-5} for the adult tribal RME scenario (Tulalip data), 1×10^{-5} for the child tribal RME scenario (Tulalip data), and 2×10^{-5} for the adult API RME scenario. These risks for dioxins/furans were calculated based on the assumption that all seafood in the market basket diet for the RME scenarios had the same dioxin/furan concentrations as those in the fillets of English sole collected in 2007 near Kellogg Island.

e. Total risks are underestimated because dioxin/furan risks are not included (see Section 3.2.1). However, because excess cancer risks are presented as one significant figure, the total risk estimate may not change because the risk estimate from dioxins/furans may be an order of magnitude or more lower than the total risk estimate based on total PCBs, arsenic, and cPAHs.

API = Asian and Pacific Islander; BEHP = bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; cPAH = carcinogenic polycyclic aromatic hydrocarbon; COC = contaminant of concern; DDT = dichlorodiphenyltrichloroethane; EM = edible meat; HHRA = human health risk assessment; PCB = polychlorinated biphenyl; PCP = pentachlorophenol; PSAMP = Puget Sound Ambient Monitoring Program; RME = reasonable maximum exposure; WB = whole body

Table 3-6a Summary of Estimated Excess Cancer Risks for the Direct Sediment Contact Scenarios using the RI Baseline Dataset

COC	Netfishing RME	Beach Play RME								Tribal Clamming RME
		Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Area 7	Area 8	
Risk Drivers										
Arsenic	6×10^{-6}	5×10^{-6}	7×10^{-6}	4×10^{-6}	4×10^{-6}	3×10^{-6}	3×10^{-6}	4×10^{-6}	3×10^{-6}	2×10^{-5}
cPAHs[a]	1×10^{-6}	1×10^{-5}	4×10^{-5}	3×10^{-5}	8×10^{-6}	5×10^{-6}	5×10^{-6}	1×10^{-6}	4×10^{-6}	5×10^{-6}
Dioxins/furans	2×10^{-5} [b]	n/a	n/a	n/a	1×10^{-5}	8×10^{-8}	n/a	6×10^{-8}	n/a	1×10^{-4} [c]
PCB TEQ	4×10^{-6}	4×10^{-9}	3×10^{-7}	n/a	9×10^{-6}	1×10^{-7}	3×10^{-7}	3×10^{-8}	8×10^{-8}	3×10^{-5}
Total PCBs	2×10^{-6} [b]	7×10^{-8}	1×10^{-7}	1×10^{-7}	6×10^{-6}	1×10^{-7}	5×10^{-7}	1×10^{-7}	1×10^{-7}	8×10^{-6} [d]
Subtotal (excluding PCB TEQ)	**3×10^{-5}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**1×10^{-4}**
Subtotal (excluding total PCBs)	**3×10^{-5}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**2×10^{-4}**
Tentatively Identified Compounds (JN-qualified)										
Toxaphene	2×10^{-6}	1×10^{-8}	7×10^{-9}	2×10^{-8}	2×10^{-7}	1×10^{-7}	1×10^{-8}	1×10^{-8}	1×10^{-8}	6×10^{-6}
Total excess cancer risk across both exposure routes[c] (excluding PCB TEQ)	**3×10^{-5}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**1×10^{-4}**
Total excess cancer risk across both exposure routes[e] (excluding total PCBs)	**3×10^{-5}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**2×10^{-4}**

Notes:

a. cPAHs are presented as benzo(a)pyrene TEQs. Because of the potential for the increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005a), the risk estimate for the beach play RME for cPAHs was based on dose adjustments across the 0-to-6-year-old age range of children. See Section B.5.1 of the HHRA (Windward 2007b) for more information.

b. When risks were recalculated using the FS dataset, the dioxin/furan risk associated with netfishing was 3×10^{-6}.

c. When risks were recalculated using the FS dataset, the dioxin/furan risk associated with clamming was 5×10^{-5}.

d. The exposure point concentration for netfishing used for this risk estimate was based on an arithmetic upper confidence limit on the mean, which is expected to overestimate exposure because of spatially biased sampling. The arithmetic mean was greater than the spatially-weighted mean (developed using Thiessen polygons) by a factor of approximately 5.

e. Total excess cancer risks include the risks associated with all COCs. However, only those COCs with an excess cancer risk greater than or equal to 1×10^{-6} for at least one scenario are listed in this table. Non-cancer effects are not expected from direct contact exposures because no thresholds were exceeded.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; COC = contaminant of concern; EPA = U.S. Environmental Protection Agency; FS = feasibility study; HHRA = human health risk assessment; n/a = not available; PCB = polychlorinated biphenyl; RI = remedial investigation; RME = reasonable maximum exposure; TEQ = toxic equivalent

Table 3-6b Summary of Estimated Excess Cancer Risks for the Direct Sediment Contact Beach Play Scenarios Using the FS Baseline Dataset

COC	Beach Play RME[a]												
	Area 1	Area 2	Area 3	Area 4	Area 4 Modified[b] without Inlet	Area 4 Modified[b] Inlet Only	Area 5	Area 5 Modified[c] North	Area 5 Modified[c] South	Area 6	Area 7	Area 8	Duwamish Waterway Park
Arsenic	5×10^{-6}	6×10^{-6}	4×10^{-6}	4×10^{-6}	3×10^{-6}	1×10^{-5}	3×10^{-6}	6×10^{-6}	3×10^{-6}	3×10^{-5}	3×10^{-6}	3×10^{-6}	1×10^{-6}
cPAHs[d]	4×10^{-6}	8×10^{-5}	1×10^{-5}	1×10^{-5}	9×10^{-6}	4×10^{-5}	3×10^{-5}	4×10^{-5}	1×10^{-6}	8×10^{-5}	1×10^{-6}	3×10^{-6}	7×10^{-7}
Dioxins/furans	1×10^{-7}	3×10^{-6}	1×10^{-7}	1×10^{-5}	6×10^{-7}	2×10^{-5}	1×10^{-6}	1×10^{-6}	2×10^{-7}	3×10^{-7}	1×10^{-7}	1×10^{-7}	2×10^{-7}
Total PCBs	3×10^{-8}	1×10^{-7}	1×10^{-7}	6×10^{-4}	1×10^{-6}	3×10^{-3}	1×10^{-7}	3×10^{-7}	1×10^{-7}	5×10^{-7}	5×10^{-8}	6×10^{-8}	1×10^{-7}
Total excess cancer risk[e]	$\mathbf{9 \times 10^{-6}}$	$\mathbf{9 \times 10^{-5}}$	$\mathbf{1 \times 10^{-5}}$	$\mathbf{6 \times 10^{-4}}$	$\mathbf{1 \times 10^{-5}}$	$\mathbf{3 \times 10^{-3}}$	$\mathbf{3 \times 10^{-5}}$	$\mathbf{5 \times 10^{-5}}$	$\mathbf{4 \times 10^{-6}}$	$\mathbf{1 \times 10^{-4}}$	$\mathbf{4 \times 10^{-6}}$	$\mathbf{6 \times 10^{-6}}$	$\mathbf{2 \times 10^{-6}}$

Notes:

a. EPCs used for risk estimates are presented in Appendix B along with details regarding the calculation of the EPCs.

b. Beach 4 was divided into two parts: Area 4 modified without inlet excludes the inlet at RM 2.2W; Area 4 modified – inlet only includes only the inlet at RM 2.2W. See Figure 3-1.

c. Beach 5 was divided into two parts: Area 5 modified – north includes only the northernmost beach area and Area 5 modified – south includes only the two southernmost beach areas and excludes the northerly section. See Figure 3-1.

d. cPAHs are presented as benzo(a)pyrene TEQs. Because of the potential for the increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005a), the risk estimate for beach play RME for cPAHs was based on dose adjustments across the 0-to-6-year-old age range of children. See Section B.5.1 of the HHRA (Windward 2007b) for more information.

e. The total excess cancer risk includes only those COCs presented in this table. In the HHRA (Windward 2007b), risks from toxaphene, the other COC, made up 1% or less of the total excess cancer risk for any given assumed beach play area, and thus if the risk estimate for this other COC was added, it is unlikely that the total risk estimates presented here would change.

COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; EPC = exposure point concentration; FS = feasibility study; HHRA = human health risk assessment; PCB = polychlorinated biphenyl; RM = river mile; RME = reasonable maximum exposure; TEQ = toxic equivalent

Table 3-7 Sum of Estimated Excess Cancer Risks across Related Scenarios as Reported in the RI

Activity	Excess Cancer Risk[a]
Adult Tribal Fishing Scenarios	
Netfishing RME[b]	3×10^{-5}
Swimming[c]	$<1 \times 10^{-6}$
Adult tribal RME seafood consumption based on Tulalip data[d]	4×10^{-3}
Sum of risk across scenarios	$\mathbf{4 \times 10^{-3}}$
Child Scenarios[e]	
Beach play RME – Area 2[f]	5×10^{-5}
Swimming[c]	$<1 \times 10^{-6}$
Subtotal for beach play RME and swimming	$\mathbf{5 \times 10^{-5}}$
Child tribal RME seafood consumption based on Tulalip data[d]	8×10^{-4}
Sum of risk across scenarios	$\mathbf{9 \times 10^{-4}}$
Adult Tribal RME Clamming Scenarios	
Tribal clamming RME – 120 days per year	1×10^{-4}
Swimming[c]	$<1 \times 10^{-6}$
Adult tribal RME seafood consumption based on Tulalip data[d]	4×10^{-3}
Sum of risk across scenarios	$\mathbf{4 \times 10^{-3}}$

Notes:

a. All non-swimming risk estimates are presented in the HHRA (Windward 2007b); for each scenario, total excess cancer risk estimates excluding PCB TEQ were used because these were equal to or higher than total excess cancer risk estimates excluding total PCBs.

b. Although EPA guidance generally discourages summing risk estimates from multiple RME scenarios, risks for the RME netfishing scenario, rather than the netfishing central tendency scenario, were added to the RME seafood consumption scenario to account for the fact that RME seafood consumption and RME netfishing may be practiced by tribal members simultaneously.

c. Adult and child swimming risk estimates as reported by King County for Elliott Bay and the Duwamish River for medium exposure assumptions (12 events per year for adults or children aged 1 to 6) (King County 1999a). Exposure pathways consisted of dermal contact and incidental sediment ingestion of water during swimming. Risks were estimated based on total PCB concentrations of 14.4 ng/L in the LDW originally modeled by King County (King County 1999a). PCB congener data from samples collected from the LDW by King County in 2005 indicate that this modeled estimate is likely an overestimate of actual total PCB concentrations, which were no greater than 3.14 ng/L during low-flow sampling conducted in August 2005 (Mickelson and Williston 2006). These results indicate that the risk estimates for the swimming scenario presented by King County in the water quality assessment (King County 1999a) are also likely overestimated.

d. The excess cancer risk estimates reported here differ slightly from those reported in the HHRA (Windward 2007b) and Section 6 of the RI (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish) for scenarios based on the Tulalip Tribes survey was updated in response to a correction provided by EPA. The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates are described in detail in an erratum (Windward 2009) to the HHRA (Windward 2007b).

e. Child scenarios include the child tribal RME seafood consumption estimate based on 40% of the total adult tribal RME seafood consumption based on Tulalip data, which is considered protective of non-tribal children.

f. Area 2 is included because it had the highest risk estimate among the individual beach play scenarios evaluated for the RI (Windward 2010) (Table 3-6a). Note that when beach play risks were calculated using the FS dataset (see Table 3-6b), risk estimates for Area 2 were no longer the highest among the assumed beach play areas.

EPA = U.S. Environmental Protection Agency; FS = feasibility study; HHRA = human health risk assessment; LDW = Lower Duwamish Waterway; PCB = polychlorinated biphenyl; RI = remedial investigation; RME = reasonable maximum exposure; TEQ = toxic equivalent

Table 3-8 Summary of COCs and Selection of Risk Drivers for Human Health Exposure Scenarios

COC	Risk Driver?	Maximum RME Risk Estimate[a]	Rationale for Selection/Exclusion as Risk Driver
Seafood Consumption Scenarios			
Inorganic arsenic	Yes	2×10^{-3}	Risk magnitude, percent contribution to the total excess cancer risk (29%), and high detection frequency in tissue samples (100%).
cPAHs	Yes	8×10^{-5}	Risk magnitude and high detection frequency in tissue samples (72%).
PCBs	Yes	2×10^{-3}	Risk magnitude, high percent contribution to the total excess cancer risk (58%), and high detection frequency in tissue samples (97%).
Dioxins/furans	Yes	nd	No dioxin/furan tissue data were available. However, because excess cancer risks were assumed to be unacceptably high, dioxins/furans were identified as a risk driver.
Bis(2-ethylhexyl) phthalate	No	6×10^{-6}	Low percent contribution to the total excess cancer risk (less than or equal to 3%) and rarely detected in tissue samples (particularly when samples were re-analyzed to evaluate the effect on RLs of analytical dilutions in the initial analysis).
Pentachlorophenol	No	9×10^{-5}	
Tributyltin	No	HQ = 3	HQs for these metals were only slightly greater than 1 (only for the child tribal RME scenario). Ingestion rates used for this scenario are uncertain.
Vanadium	No	HQ = 2	
Aldrin	No	5×10^{-5}	All organochlorine pesticides were low contributors to the total excess cancer risk (less than or equal to 3% of the total risk). In addition, because of analytical interference of these contaminants with PCBs, much of the tissue data for these contaminants were qualified JN, which indicates “the presence of an analyte that has been ‘tentatively identified,’ and the associated numerical value represents its approximate concentration” (EPA 1999c). The JN-qualified organochlorine pesticide results are highly uncertain and likely biased high.
alpha-BHC	No	2×10^{-5}	
beta-BHC	No	6×10^{-6}	
Carbazole	No	4×10^{-5}	
Total chlordane	No	6×10^{-6}	
Total DDTs	No	2×10^{-5}	
Dieldrin	No	1×10^{-4}	
gamma-BHC	No	5×10^{-6}	
Heptachlor	No	1×10^{-5}	
Heptachlor epoxide	No	3×10^{-5}	
Hexachlorobenzene	No	1×10^{-5}	
Direct Sediment Exposure Scenarios			
Inorganic arsenic	Yes	2×10^{-5}	Risk magnitude, percent contribution to total excess cancer risk (14 to 19%), and high detection frequency in surface sediment samples (92%).
cPAHs	Yes	4×10^{-5}	Risk magnitude, percent contribution to total excess cancer risk (3 to 85%), and high detection frequency in surface sediment samples (94%).
PCBs	Yes	8×10^{-6}	Lower risk magnitude and percent contribution to total excess cancer risk than the other sediment risk drivers, but selected because of importance in the seafood consumption scenarios.
Dioxins/furans	Yes	1×10^{-4}	Risk magnitude, percent contribution to total excess cancer risk (35 to 72%), and high detection frequency in surface sediment samples (100%).
Toxaphene	No	6×10^{-6}	Low percent contribution to total excess cancer risk (6% or less) and low detection frequency in surface sediment samples (1%).

Notes:

a. Only RME scenarios were used to designate COCs. The highest risk estimate for any of the RME scenarios is shown in this table. Note that the estimates reported here differ slightly from those reported in Appendix B of the RI (the HHRA) (Windward 2007b), and Section 6 of the RI (Windward 2010). The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish but not the total quantity consumed) for scenarios based on the Tulalip Tribes survey was updated in response to an EPA correction). The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates were described in detail in an erratum (Windward 2009) to the HHRA (Windward 2007b).

BHC = benzene hexachloride; cPAH = carcinogenic polycyclic aromatic hydrocarbon; COC = contaminant of concern; HHRA = human health risk assessment; DDT = dichlorodiphenyltrichloroethane; HQ = hazard quotient; J = estimated concentration; N = tentative identification; nd = no data; PCB = polychlorinated biphenyl; RI = remedial investigation; RL = reporting limit; RME = reasonable maximum exposure

Table 3-9 Sediment RBTCs for Total PCBs Based on the Human Health RME Seafood Consumption Scenarios and on Seafood Consumption by River Otters

Seafood Consumption Scenario	Sediment RBTCs for Total PCBs (µg/kg dw)			
	1 in 1,000,000 Risk Level (1×10^{-6})	1 in 100,000 Risk Level (1×10^{-5})	1 in 10,000 Risk Level (1×10^{-4})	HQ = 1
Human Health				
Adult Tribal RME (Tulalip data)	<1[a]	<1[a]	7.3	<1
Child Tribal RME (Tulalip data)	<1[a]	<1[a]	185	<1
Adult API RME	<1[a]	<1[a]	100	<1
Ecological				
River otter	n/a	n/a	n/a	128 – 159[b]

Notes:

a. Sediment RBTCs are reported as < 1 µg/kg because even if total PCB sediment concentrations were assumed to be 0 µg/kg dw, water concentrations would need to be well below upstream concentrations (which are currently 0.3 ng/L on average) to calculate concentrations in tissue that would equate to these lower risk levels. Only at hypothetical water concentrations that are not believed to be achievable for the LDW are the sediment RBTCs for the 1×10^{-5} or 1×10^{-6} risk levels greater than 0 µg/kg dw (Figure 3-2). For example, using a hypothetical water concentration of 0.01 ng/L, the sediment RBTC would be greater than 0 for the 1×10^{-5} risk level.

b. Represents best-fit estimates for two different river otter dietary scenarios as presented in the ERA (Windward 2007a).

API = Asian and Pacific Islander; dw = dry weight; ERA = ecological risk assessment; HQ = hazard quotient; kg = kilograms; L = liter; µg = micrograms; n/a = not applicable; ng = nanograms; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure

Table 3-10 Sediment RBTCs for Human Health Direct Sediment Contact RME Exposure Scenarios

Risk Driver	Target Risk	Sediment RBTC		
		Netfishing RME	Beach Play RME	Tribal Clamming RME
Arsenic (mg/kg dw)	1×10^{-6}	3.7	2.8	1.3
	1×10^{-5}	37	28	13
	1×10^{-4}	370	280	130
cPAH TEQ[a] (µg/kg dw)	1×10^{-6}	380	90	150
	1×10^{-5}	3,800	900	1,500
	1×10^{-4}	38,000	9,000	15,000
Dioxins/furan TEQ[b] (ng/kg dw)	1×10^{-6}	37	28	13
	1×10^{-5}	370	280	130
	1×10^{-4}	3,700	2,800	1,300
Total PCBs (µg/kg dw)	1×10^{-6}	1,300	1,700	500
	1×10^{-5}	13,000	17,000	5,000
	1×10^{-4}	130,000	170,000	50,000
	HQ = 1	n/a[c]	5,900	n/a[c]

Notes:

a. cPAHs are presented as benzo(a)pyrene TEQs.

b. Dioxins/furans are presented as 2,3,7,8-TCDD mammalian TEQs.

c. Sediment RBTCs were calculated for non-cancer risk (HQ of 1) only when HQs were greater than 1 for a given scenario-risk driver combination.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; HQ = hazard quotient; kg = kilograms; µg = micrograms; mg =milligrams; n/a = not applicable; ng = nanograms; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure; TCDD = tetrachlorodibenzo-p-dioxin; TEQ = toxic equivalent

Table 3-11 Ingestion-weighted Tissue RBTCs for the Human Health RME Seafood Consumption Scenarios

Risk Driver	Target Risk	Ingestion-weighted Tissue RBTC[a]			
		Excess Cancer Risk			Non-cancer Hazard
		1×10^{-6}	1×10^{-5}	1×10^{-4}	HQ = 1
Arsenic (mg/kg ww)	Adult Tribal RME (Tulalip Data)	0.00056	0.0056	0.056	0.25
	Child Tribal RME (Tulalip Data)	0.0030	0.030	0.30	0.12
	Adult API RME	0.0019	0.019	0.19	0.37
cPAH TEQ[b] (µg/kg ww)	Adult Tribal RME (Tulalip Data)	0.11	1.1	11	n/a
	Child Tribal RME (Tulalip Data)	0.12[c]	1.2[c]	12[c]	n/a
	Adult API RME	0.39	3.9	39	n/a
Dioxin/furan TEQ[d] (ng/kg ww)	Adult Tribal RME (Tulalip Data)	0.0056	0.056	0.56	n/a
	Child Tribal RME (Tulalip Data)	0.030	0.30	3.0	n/a
	Adult API RME	0.019	0.19	1.9	n/a
Total PCBs (µg/kg ww)	Adult Tribal RME (Tulalip Data)	0.42	4.2	42	17
	Child Tribal RME (Tulalip Data)	2.3	23	230	7.8
	Adult API RME	1.4	14	140	24

Notes:

a. Tissue RBTCs associated with human seafood consumption scenarios were calculated using the risk equations in the baseline HHRA (Windward 2007b). These tissue RBTCs represent the ingestion-weighted average concentration in tissue (across all seafood types), resulting in a risk threshold. For example, the RBTC for total PCBs for the adult tribal RME seafood consumption scenario based on Tulalip data was 4.2 µg/kg ww at the 1×10^{-5} excess cancer risk level. Thus, consumption of 97.5 g/day (adult tribal RME daily ingestion rate based on Tulalip data) of any tissue type with a total PCB concentration of 4.2 µg/kg ww for 70 years would result in a 1×10^{-5} excess cancer risk. Consumption of numerous types of seafood, such as crabs, clams, and fish (as specified in the adult tribal RME exposure parameters based on Tulalip data) would also result in a 1×10^{-5} excess cancer risk as long as the ingestion-weighted average of the various tissue concentrations consumed was 4.2 µg/kg ww.

b. cPAHs are presented as benzo(a)pyrene TEQs.

c. Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005a), the risk estimate for children for cPAHs is based on dose adjustments across the 0-to-6-year age range of children (see Appendix B of the RI, Section B.5.1, for more information).

d. Dioxins/furans are presented as 2,3,7,8-TCDD mammalian TEQs.

API = Asian and Pacific Islanders; cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; EPA = U.S. Environmental Protection Agency; HQ = hazard quotient; kg = kilograms; µg = micrograms; mg =milligrams; n/a = not applicable; ng = nanograms; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RI = remedial investigation; RME = reasonable maximum exposure; TCDD = tetrachlorodibenzo-*p*-dioxin; TEQ = toxic equivalent; ww = wet weight

Table 3-12 Equations and Parameter Values for the Calculation of Tissue RBTCs

RBTC equation for carcinogenic effects:

$$Tissue\ RBTC = \frac{TR}{\left(\left[\frac{1R\ \times FC\ \times\ EF\ \times\ ED\ \times CF}{BW\ \times\ AT_c}\right] \times\ SF\right)}$$

RBTC equation or non-carcinogenic effects:

$$Tissue\ RBTC = \frac{THQ}{\left(\left[\frac{1R\ \times FC\ \times\ EF\ \times\ ED\ \times CF}{BW\ \times\ AT_{nc}}\right] \times\ \frac{1}{RfD}\right)}$$

Parameter Name	Acronym	Unit	Parameter Values[a] Adult Tribal RME (Tulalip Data)	Child Tribal RME (Tulalip Data)	Adult API RME
Risk-based threshold concentration	RBTC	mg/kg ww	see Table 3-11 for calculated RBTCs		
Target excess cancer risk	TR	unitless	10^{-6}, 10^{-5}, 10^{-4}	10^{-6}, 10^{-5}, 10^{-4}	10^{-6}, 10^{-5}, 10^{-4}
Target HQ	THQ	unitless	1	1	1
Ingestion rate	IR	g/day	97.5	39.0	51.5
Fraction from contaminated site	FC	unitless	1	1	1
Exposure frequency	EF	days	365	365	365
Exposure duration	ED	years	70	6	30
Conversion factor	CF	kg to g	0.001	0.001	0.001
Body weight	BW	kg	81.8	15.2	63
Averaging time, cancer	AT_c	days	25,550	25,550	25,550
Averaging time, non-cancer	AT_{nc}	days	25,550	2,190	10,950
Slope factor	SF	$(mg/kg\text{-}day)^{-1}$	toxicity values are contaminant-specific (Total PCBs = 2; Inorganic arsenic = 1.5; cPAH TEQ = 7.3; dioxin/furan TEQ = 150,000)		
Reference dose	RfD	mg/kg-day	toxicity values are contaminant-specific (Total PCBs = 0.00002; Inorganic arsenic = 0.0003)		

Notes:

a. Parameter values are the same as those used in the LDW HHRA (Windward 2007b).

API = Asian and Pacific Islanders; cPAH = carcinogenic polycyclic aromatic hydrocarbon; g = gram; HHRA = human health risk assessment; HQ = hazard quotient; kg = kilogram; LDW = Lower Duwamish Waterway; mg = milligram; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure; TEQ = toxic equivalent

Table 3-13 Comparison of Tissue RBTCs for the Adult Tribal RME Scenario Based on Tulalip Data and Non-Urban Puget Sound Tissue Data

Species Categories	RBTCs for 10⁻⁴ Risk Level[a]		RBTCs for 10⁻⁵ Risk Level[a]		RBTCs for 10⁻⁶ Risk Level[a]		RBTCs for HQ = 1[a]		LDW HHRA Average Conc.[b]	Non-Urban Puget Sound Tissue Data[c]			
	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]		Detection Frequency	Range of Detects	Mean Value[e]	Species Types
Total PCBs (µg/kg ww)													
Benthic fish, fillet[f]		75		7.5		0.75		30	700	158 / 242	1.3 – 75.4	11	English sole, rock sole
Pelagic fish		181		18		1.8		73	1,700	n/a	n/a	n/a	n/a
Crab, edible meat	42	18	4.2	1.8	0.42	0.18	17	7.3	170	17 / 17	0.43 – 1.92	0.86	Dungeness crab
Crab, whole body		95		9.5		0.95		38	890	15 / 15	3.03 – 16[g]	7.1[g]	Dungeness crab
Clams		15		1.5		0.15		6.0	140	24 / 70	0.09 – 1.43	0.3	Butter clam, geoduck, horse clam, littleneck clam
Inorganic arsenic (mg/kg ww)													
Benthic fish, fillet[f]		0.00039		0.000039		0.0000039		0.0017	0.004	3 / 12	0.002 – 0.004 J	0.002	English sole
Pelagic fish		0.0056		0.00056		0.000056		0.025	0.057	8 / 9	0.009 J – 0.03	0.02	Shiner surfperch (whole body)
Crab, edible meat	0.056	0.0022	0.0056	0.00022	0.00056	0.000022	0.25	0.010	0.023	12 / 12	0.01 – 0.04	0.02	Dungeness crab, slender crab
Crab, whole body		0.0073		0.00073		0.000073		0.033	0.075	12 / 12	0.032 – 0.13[g]	0.075[g]	Dungeness crab, slender crab
Clams		0.12		0.012		0.0012		0.54	1.24	24 / 24	0.044 J – 0.62 J	0.21	Eastern softshell clam, composites with multiple species (butter clam, cockle, Eastern softshell clam, littleneck clam)
cPAH TEQ (µg/kg ww)													
Benthic fish, fillet[f]		0.61		0.061		0.0061		np	0.39	0 / 1	< 0.114 (no detects)	0.114 (no detects)	Starry flounder
Pelagic fish		1.2		0.12		0.012		np	0.78	n/a	n/a	n/a	n/a
Crab, edible meat	11	0.69	1.1	0.069	0.11	0.0069	np	np	0.44	0 / 8	< 1.63 (no detects)	0.406 (no detects)	Dungeness crab
Crab, whole body		1.2		0.12		0.012		np	0.75	0 / 7	< 0.923 (no detects)[g]	0.230 (no detects)[g]	Dungeness crab
Clams		24		2.4		0.24		np	15	3 / 11	0.069 – 0.171	0.088	Butter clam, geoduck, littleneck clam
Dioxin/furan TEQ (ng/kg ww)													
Benthic fish, fillet[f]		nc		nc		nc		np	n/a	4 / 4	0.166 – 0.923	0.421	Starry flounder, rock sole
Pelagic fish		nc		nc		nc		np	n/a	n/a	n/a	n/a	n/a
Crab, edible meat	0.56	nc	0.056	nc	0.0056	nc	np	np	n/a	27 / 27	0.027 – 1.37	0.24	Dungeness crab
Crab, whole body		nc		nc		nc		np	n/a	25 / 25	0.089 – 5.12[g]	0.81[g]	Dungeness crab
Clams		nc		nc		nc		np	n/a	43 / 43	0.011 – 1.63	0.26	Butter clam, geoduck, horse clam, littleneck clam

Notes:

a. RBTCs are for the adult tribal RME scenario based on Tulalip data.
b. The LDW HHRA dataset includes tissue samples collected between 1992 and 2005. Additional tissue samples were collected from the LDW in 2007 (see Section 4 of the RI).
c. Details regarding the non-urban Puget Sound tissue dataset are presented in Section B.4 of Appendix B of the FS.
d. Species-specific tissue RBTCs are based on only the HHRA dataset. Additional species-specific tissue RBTCs are available for total PCBs and are presented in Section B.3 of Appendix B (Table B-5).
e. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset.
f. Whole-body benthic fish consumption is assumed to be equal to zero for the adult tribal RME scenario based on Tulalip data, and thus this category is not shown on this table. However, for informational purposes, background whole-body benthic fish data could be compared to fillet data using the fillet-to-whole body ratio developed in the RI (fillet = 0.526 x whole body).
g. When only edible meat and hepatopancreas samples were available for a given sampling event, whole-body concentrations were calculated as in the LDW HHRA.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; FS = feasibility study; HHRA = human health risk assessment; HQ = hazard quotient; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg = milligrams; n/a = not available; nc = cannot be calculated; ng = nanograms; np = not applicable; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure; TEQ =toxic equivalent; ww = wet weight

Table 3-14 Comparison of Tissue RBTCs for the Child Tribal RME Scenario Based on Tulalip Data and Non-Urban Puget Sound Tissue Data

Species Categories	RBTCs for 10^{-4} Risk Level[a]		RBTCs for 10^{-5} Risk Level[a]		RBTCs for 10^{-6} Risk Level[a]		RBTCs for HQ = 1[a]		LDW HHRA Average Conc.[b]	Non-Urban Puget Sound Tissue Data[c]			
	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]		Detection Frequency	Range of Detects	Mean Value[e]	Species Types
Total PCBs (µg/kg ww)													
Benthic fish, fillet[f]		412		41		4.1		14	700	158 / 242	1.3 – 75.4	11	English sole, rock sole
Pelagic fish		1,000		100		10		34	1,700	n/a	n/a	n/a	n/a
Crab, edible meat	230	100	23	10	2.3	1.0	7.8	3.4	170	17 / 17	0.43 – 1.92	0.86	Dungeness crab
Crab, whole body		523		52		5.2		18	890	15 / 15	3.03 – 16[g]	7.1[g]	Dungeness crab
Clams		82		8.2		0.82		2.8	140	24 / 70	0.09 – 1.43	0.3	Butter clam, geoduck, horse clam, littleneck clam
Inorganic arsenic (mg/kg ww)													
Benthic fish, fillet[f]		0.0021		0.00021		0.000021		0.00083	0.004	3 / 12	0.002 – 0.004 J	0.002	English sole
Pelagic fish		0.030		0.0030		0.00030		0.012	0.057	8 / 9	0.009 J – 0.03	0.02	Shiner surfperch (whole body)
Crab, edible meat	0.30	0.012	0.030	0.0012	0.0030	0.00012	0.12	0.0048	0.023	12 / 12	0.01 – 0.04	0.02	Dungeness crab, slender crab
Crab, whole body		0.039		0.0039		0.00039		0.016	0.075	12 / 12	0.032 – 0.13[g]	0.075[g]	Dungeness crab, slender crab
Clams		0.65		0.065		0.0065		0.26	1.24	24 / 24	0.044 J – 0.62 J	0.21	Eastern softshell clam, composites with multiple species (butter clam, cockle, Eastern softshell clam, littleneck clam)
cPAH TEQ (µg/kg ww)													
Benthic fish, fillet[f]		0.66		0.066		0.0066		np	0.39	0 / 1	< 0.114 (no detects)	0.114 (no detects)	Starry flounder
Pelagic fish		1.3		0.13		0.013		np	0.78	n/a	n/a	n/a	n/a
Crab, edible meat	12	0.75	1.2	0.075	0.12	0.0075	np	np	0.44	0 / 8	< 1.63 (no detects)	0.406 (no detects)	Dungeness crab
Crab, whole body		1.3		0.13		0.013		np	0.75	0 / 7	< 0.923 (no detects)[g]	0.230 (no detects)[g]	Dungeness crab
Clams		26		2.6		0.26		np	15	3 / 11	0.069 – 0.171	0.088	Butter clam, geoduck, littleneck clam
Dioxin/furan TEQ (ng/kg ww)													
Benthic fish, fillet[f]		nc		nc		nc		np	n/a	4 / 4	0.166 – 0.923	0.421	Starry flounder, rock sole
Pelagic fish		nc		nc		nc		np	n/a	n/a	n/a	n/a	n/a
Crab, edible meat	3.0	nc	0.30	nc	0.030	nc	np	np	n/a	27 / 27	0.027 – 1.37	0.24	Dungeness crab
Crab, whole body		nc		nc		nc		np	n/a	25 / 25	0.089 – 5.12[g]	0.81[g]	Dungeness crab
Clams		nc		nc		nc		np	n/a	43 / 43	0.011 – 1.63	0.26	Butter clam, geoduck, horse clam, littleneck clam

Notes:

a. RBTCs are for the child tribal RME scenario based on Tulalip data.

b. The LDW HHRA dataset includes tissue samples collected between 1992 and 2005. Additional tissue samples were collected from the LDW in 2007 (see Section 4 of the RI).

c. Details regarding the non-urban Puget Sound tissue dataset are presented in Section B.4 of Appendix B of the FS.

d. Species-specific tissue RBTCs are based on only the HHRA dataset. Additional species-specific tissue RBTCs are available for total PCBs and are presented in Section B.3 of Appendix B (Table B-6).

e. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset.

f. Whole-body benthic fish consumption is assumed to be equal to zero for the child tribal RME scenario based on Tulalip data, and thus this category is not shown on this table. However, for informational purposes, background whole-body benthic fish data could be compared to fillet data using the fillet-to-whole body ratio developed in the RI (fillet = 0.526 x whole body).

g. When only edible meat and hepatopancreas samples were available for a given sampling event, whole-body concentrations were calculated as in the LDW HHRA.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; FS = feasibility study; HHRA = human health risk assessment; HQ = hazard quotient; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg = milligrams; n/a = not available; nc = cannot be calculated; ng = nanograms; np = not applicable; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure; TEQ =toxic equivalent; ww = wet weight

Table 3-15 Comparison of Tissue RBTCs for the Adult API RME Scenario and Non-Urban Puget Sound Tissue Data

Species Categories	RBTCs for 10^{-4} Risk Level[a]		RBTCs for 10^{-5} Risk Level[a]		RBTCs for 10^{-6} Risk Level[a]		RBTCs for HQ = 1[a]		LDW HHRA Average Conc.[b]	Non-Urban Puget Sound Tissue Data[c]			
	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]	Total Seafood Diet	Species-Specific[d]		Detection Frequency	Range of Detects	Mean Value[e]	Species Types
Total PCBs (µg/kg ww)													
Benthic fish, fillet[f]	140	230	14	23	1.4	2.3	24	39	700	158 / 242	1.3 – 75.4	11	English sole, rock sole
Benthic fish, whole body		723		72		7.2		124	2,200	n/a	n/a	n/a	n/a
Pelagic fish		559		56		5.6		96	1,700	n/a	n/a	n/a	n/a
Crab, edible meat		56		5.6		0.56		9.6	170	17 / 17	0.43 – 1.92	0.86	Dungeness crab
Crab, whole body		293		29		2.9		50	890	15 / 15	3.03 – 16[g]	7.1[g]	Dungeness crab
Clams		46		4.6		0.46		7.9	140	24 / 70	0.09 – 1.43	0.3	Butter clam, geoduck, horse clam, littleneck clam
Inorganic arsenic (mg/kg ww)													
Benthic fish, fillet[f]	0.19	0.00097	0.019	0.000097	0.0019	0.0000097	0.37	0.0019	0.004	3 / 12	0.002 – 0.004 J	0.002	English sole
Benthic fish, whole body		0.014		0.0014		0.00014		0.026	0.056	12 / 12	0.007 J – 0.03	0.01	English sole
Pelagic fish		0.014		0.0014		0.00014		0.027	0.057	8 / 9	0.009 J – 0.03	0.02	Shiner surfperch (whole body)
Crab, edible meat		0.0056		0.00056		0.000056		0.011	0.023	12 / 12	0.01 – 0.04	0.02	Dungeness crab, slender crab
Crab, whole body		0.018		0.0018		0.00018		0.035	0.075	12 / 12	0.032 – 0.13[g]	0.075[g]	Dungeness crab, slender crab
Clams		0.30		0.030		0.0030		0.59	1.24	24 / 24	0.044 J – 0.62 J	0.21	Eastern softshell clam, composites with multiple species (butter clam, cockle, Eastern softshell clam, littleneck clam)
cPAH TEQ (µg/kg ww)													
Benthic fish, fillet[f]	39	1.6	3.9	0.16	0.39	0.016	np	np	0.39	0 / 1	< 0.114 (no detects)	0.114 (no detects)	Starry flounder
Benthic fish, whole body		5.7		0.57		0.057		np	1.4	n/a	n/a	n/a	n/a
Pelagic fish		3.2		0.32		0.032		np	0.78	n/a	n/a	n/a	n/a
Crab, edible meat		1.8		0.18		0.018		np	0.44	0 / 8	< 1.63 (no detects)	0.406 (no detects)	Dungeness crab
Crab, whole body		3.1		0.31		0.031		np	0.75	0 / 7	< 0.923 (no detects)[g]	0.230 (no detects)[g]	Dungeness crab
Clams		61		6.1		0.61		np	15	3 / 11	0.069 – 0.171	0.088	Butter clam, geoduck, littleneck clam
Dioxin/furan TEQ (ng/kg ww)													
Benthic fish, fillet[f]	1.9	nc	0.19	nc	0.019	nc	np	np	n/a	4 / 4	0.166 – 0.923	0.421	Starry flounder, rock sole
Benthic fish, whole body		nc		nc		nc		np	n/a	7 / 7	0.152 – 0.417	0.281	English sole, rock sole
Pelagic fish		nc		nc		nc		np	n/a	n/a	n/a	n/a	n/a
Crab, edible meat		nc		nc		nc		np	n/a	27 / 27	0.027 – 1.37	0.24	Dungeness crab
Crab, whole body		nc		nc		nc		np	n/a	25 / 25	0.089 – 5.12[g]	0.81[g]	Dungeness crab
Clams		nc		nc		nc		np	n/a	43 / 43	0.011 – 1.63	0.26	Butter clam, geoduck, horse clam, littleneck clam

Notes:

a. RBTCs are for the adult API RME scenario.
b. The LDW HHRA dataset includes tissue samples collected between 1992 and 2005. Additional tissue samples were collected from the LDW in 2007 (see Section 4 of the RI).
c. Details regarding the non-urban Puget Sound tissue dataset are presented in Section B.4 of Appendix B of the FS.
d. Species-specific tissue RBTCs are based on only the HHRA dataset. Additional species-specific tissue RBTCs are available for total PCBs and are presented in Section B.3 of Appendix B (Table B-7).
e. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset.
f. For informational purposes, background whole-body benthic fish data could be compared to fillet data using the fillet-to-whole body ratio developed in the RI (fillet = 0.526 x whole body).
g. When only edible meat and hepatopancreas samples were available for a given sampling event, whole-body concentrations were calculated as in the LDW HHRA.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; FS = feasibility study; HHRA = human health risk assessment; HQ = hazard quotient; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg = milligrams; n/a = not available; nc = cannot be calculated; ng = nanograms; np = not applicable; PCB = polychlorinated biphenyl; RBTC = risk-based threshold concentration; RME = reasonable maximum exposure; TEQ =toxic equivalent; ww = wet weight

Table 3-16 Summary of Risk Screening and Identification of COCs and Risk Drivers

MTCA Terminology	CERCLA Terminology	Contaminants: Human Health Seafood Consumption	Contaminants: Human Health Direct Sediment Contact	Contaminants: Benthic Invertebrate Community	Contaminants: Other Ecological Receptors
STEP 1 – Conduct conservative risk-based screening to identify COPCs *Ecological: COPCs are contaminants with maximum exposure concentrations greater than TRVs.* *Human Health: COPCs are contaminants with maximum sediment concentrations greater than the EPA Region 9 RBCs; and/or the maximum seafood tissue concentrations greater than the adjusted EPA Region 3 RBCs.*					
Hazardous substances	COPCs	59 COPCs, including metals, PAHs, PCBs, organochlorine pesticides, and other SVOCs	Beach play and clamming – 28 COPCs Netfishing – 20 COPCs, including metals, PCBs, arsenic, cPAHs, dioxins/furans, toxaphene, and other contaminants	**Benthic invertebrates** – 41 COPCs including metals, PAHs, PCBs, phthalates, and other SVOCs based on detected exceedance of SQS in surface sediment at one or more locations; non-SMS contaminants – TBT; nickel; total DDTs; total chlordane; cis-1,2-dichloroethene	**Crabs** – zinc and PCBs **Fish** – arsenic, cadmium, copper, vanadium, PCBs, TBT, dioxins/furans **Birds** – arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, PCBs, dioxins/furans **Mammals** – arsenic, cobalt, mercury, selenium, PCBs, dioxins/furans
STEP 2 – Compare risk estimates to thresholds to identify COCs for both human health and ecological receptors *Ecological: COCs are contaminants with LOAEL-based HQs greater than or equal to 1.0.* *Human Health: COCs are contaminants with excess cancer risk estimates greater than 1×10^{-6} or an HQ greater than 1 for any RME scenario.*					
Hazardous substances	COCs	PCBs, arsenic, cPAHs, dioxins/furans, BEHP, pentachlorophenol, TBT, vanadium, and 11 tentatively identified compounds (aldrin, alpha-BHC, beta-BHC, carbazole, total chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene)[a]	PCBs, arsenic, cPAHs, dioxins/furans, toxaphene	**Benthic invertebrates** – 41 COCs above SQS; non-SMS contaminants – nickel, total DDTs, total chlordane	**Crabs** – PCBs **Fish** – cadmium, vanadium, PCBs **Birds** – chromium, copper, lead, mercury, vanadium, PCBs **Mammals** – PCBs
STEP 3 – Apply weight-of-evidence approach to identify risk drivers *Ecological: Selection based on risk estimates, uncertainties discussed in the baseline ERA, natural background concentrations and residual risk following planned early actions in the LDW.* *Human Health: Selection based on magnitude of risk and relative percentage of total human health risk posed by the COC and indicator hazardous substance criteria set forth in WAC 173-340-703.*					
Indicator hazardous substances	Risk drivers[b]	Total PCBs, arsenic, cPAHs, dioxins/furans[c]	Total PCBs, arsenic, cPAHs, dioxins/furans	**Benthic invertebrates** – 41 COCs above SQS[d]	**Mammals** (river otter) – total PCBs

Notes:

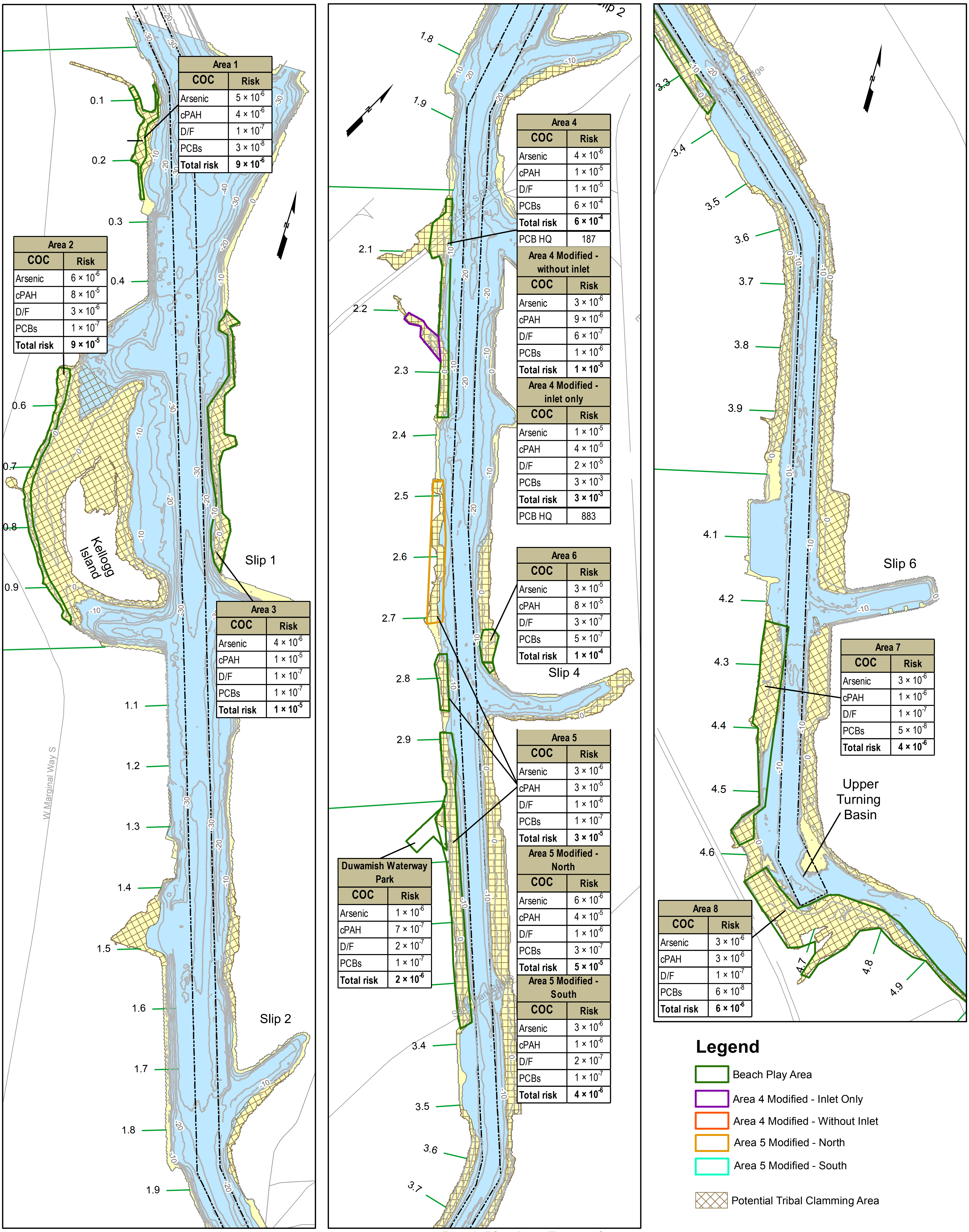
a. Organochlorine pesticides were qualified as tentatively identified compounds at estimated concentrations (JN-qualified), indicating uncertainty regarding both their presence and concentration.

b. COCs that were not selected as risk drivers are evaluated to assess the potential for risk reduction following remedial actions; this evaluation is presented in Section 9.

c. Risks were assumed to be unacceptable; no quantitative risk analysis was performed for dioxins and furans via the seafood consumption pathway.

d. The 41 risk-drivers for the benthic community are: total PCBs, BEHP, chromium, arsenic, mercury, lead, zinc, copper, cadmium, silver, fluoranthene, butyl benzyl phthalate, indeno(1,2,3-cd)pyrene, phenol, benzo(g,h,i)perylene, benzoic acid, dibenzo(a,h)anthracene, total benzofluoranthenes, 4-methylphenol, phenanthrene, total high-molecular-weight PAHs, acenaphthene, fluorene, benzo(a)anthracene, dibenzofuran, benzo(a)pyrene, total low-molecular-weight PAHs, pyrene, 1,4-dichlorobenzene, 1,2-dichlorobenzene, 2-methylnaphthalene, dimethyl phthalate, naphthalene, n-nitrosodiphenylamine, hexachlorobenzene, benzyl alcohol, chrysene, 1,2,4-trichlorobenzene, 2,4-dimethylphenol, anthracene, and pentachlorophenol.

BEHP = bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; CERCLA = Comprehensive Environmental Response, Compensation, and Liability Act; COC = contaminant of concern; COPC = contaminant of potential concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; DDT = dichlorodiphenyltrichloroethane; EPA = U.S. Environmental Protection Agency; ERA = ecological risk assessment; HQ = hazard quotient; LDW = Lower Duwamish Waterway; LOAEL = lowest-observed-adverse-effect level; MTCA = Model Toxics Control Act; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; RBC = risk-based concentration; RME = reasonable maximum exposure; SMS = Washington State Sediment Management Standards; SQS = sediment quality standard; SVOC = semivolatile organic compound; TBT = tributyltin; TRV = toxicity reference value; WAC = Washington Administrative Code



Area 1	
COC	Risk
Arsenic	5×10^{-6}
cPAH	4×10^{-6}
D/F	1×10^{-7}
PCBs	3×10^{-8}
Total risk	**9×10^{-6}**

Area 2	
COC	Risk
Arsenic	6×10^{-6}
cPAH	8×10^{-5}
D/F	3×10^{-6}
PCBs	1×10^{-7}
Total risk	**9×10^{-5}**

Area 3	
COC	Risk
Arsenic	4×10^{-6}
cPAH	1×10^{-5}
D/F	1×10^{-7}
PCBs	1×10^{-7}
Total risk	**1×10^{-5}**

Area 4	
COC	Risk
Arsenic	4×10^{-6}
cPAH	1×10^{-5}
D/F	1×10^{-5}
PCBs	6×10^{-4}
Total risk	**6×10^{-4}**
PCB HQ	187

Area 4 Modified - without inlet	
COC	Risk
Arsenic	3×10^{-6}
cPAH	9×10^{-6}
D/F	6×10^{-7}
PCBs	1×10^{-6}
Total risk	**1×10^{-5}**

Area 4 Modified - inlet only	
COC	Risk
Arsenic	1×10^{-5}
cPAH	4×10^{-5}
D/F	2×10^{-5}
PCBs	3×10^{-3}
Total risk	**3×10^{-3}**
PCB HQ	883

Area 6	
COC	Risk
Arsenic	3×10^{-5}
cPAH	8×10^{-5}
D/F	3×10^{-7}
PCBs	5×10^{-7}
Total risk	**1×10^{-4}**

Area 5	
COC	Risk
Arsenic	3×10^{-6}
cPAH	3×10^{-5}
D/F	1×10^{-6}
PCBs	1×10^{-7}
Total risk	**3×10^{-5}**

Area 5 Modified - North	
COC	Risk
Arsenic	6×10^{-6}
cPAH	4×10^{-5}
D/F	1×10^{-6}
PCBs	3×10^{-7}
Total risk	**5×10^{-5}**

Area 5 Modified - South	
COC	Risk
Arsenic	3×10^{-6}
cPAH	1×10^{-6}
D/F	2×10^{-7}
PCBs	1×10^{-7}
Total risk	**4×10^{-6}**

Duwamish Waterway Park	
COC	Risk
Arsenic	1×10^{-6}
cPAH	7×10^{-7}
D/F	2×10^{-7}
PCBs	1×10^{-7}
Total risk	**2×10^{-6}**

Area 7	
COC	Risk
Arsenic	3×10^{-6}
cPAH	1×10^{-6}
D/F	1×10^{-7}
PCBs	5×10^{-8}
Total risk	**4×10^{-6}**

Area 8	
COC	Risk
Arsenic	3×10^{-6}
cPAH	3×10^{-6}
D/F	1×10^{-7}
PCBs	6×10^{-8}
Total risk	**6×10^{-6}**

Legend

- Beach Play Area
- Area 4 Modified - Inlet Only
- Area 4 Modified - Without Inlet
- Area 5 Modified - North
- Area 5 Modified - South
- Potential Tribal Clamming Area

Intertidal and Subtidal Areas

- Intertidal Area (< -4 ft MLLW)
- Subtidal Area (> -4 ft MLLW)
- Bathymetric Contours (ft MLLW)
- Road
- Navigation Channel
- River Mile Marker

Beach Play Area Excess Cancer Risks Based on FS Baseline Dataset

Area 3	
COC	Risk
Arsenic	4×10^{-6}
cPAH	1×10^{-5}
D/F	1×10^{-7}
PCBs	1×10^{-7}
Total risk	**1×10^{-5}**

COC = contaminant of concern
Risk = excess cancer risk estimate
D/F = Dioxins/furans
cPAH = carcinogenic polycyclic aromatic hydrocarbon
PCBs = polychlorinated biphenyls

Notes:
1. Excess cancer risks and non-cancer HQs were estimated using the FS baseline dataset. Beach play areas were developed in the Human Health Risk Assessment (Map B.3-1; Windward 2007). In addition to risk estimates for these 8 areas, risk estimates for modified beach play areas 4 and 5 were also developed based on data for subsets of each of these areas, which were assessed to facilitate remedial decision-making (i.e., clarify which portions of the beach play areas are causing most of the risk). Area 4 was modified to examine the influence of higher concentrations in the inlet at RM 2.2 (risks are presented both for Area 4 modified – without inlet and Area 4 modified – inlet only). Area 5 was modified to examine the influence of higher concentrations in the northernmost section (risks are presented both for Area 5 modified – south and Area 5 modified – north). Additionally, risks were addressed separately for Duwamish Waterway Park (which is a part of Area 5).
2. Except where noted, all non-cancer HQs in all areas were less than 1.

0 200 400 800 Feet

AECOM
Wind Ward environmental LLC
Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway
Final Feasibility Study
60150279-14.35

Beach Play Areas with Excess Cancer Risks and Non-Cancer HQs for Risk Drivers and Clamming Areas

DATE: 10/31/12 | DWRN:MVI/CEH | Revision: 5

FIGURE 3-1

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 3\figure3-1beachclammingareas.mxd

Figure 3-2 FWM-Predicted Ingestion-Weighted Average Concentrations of Total PCBs in Tissue as a Function of Concentrations in Sediment at Various Water Concentrations

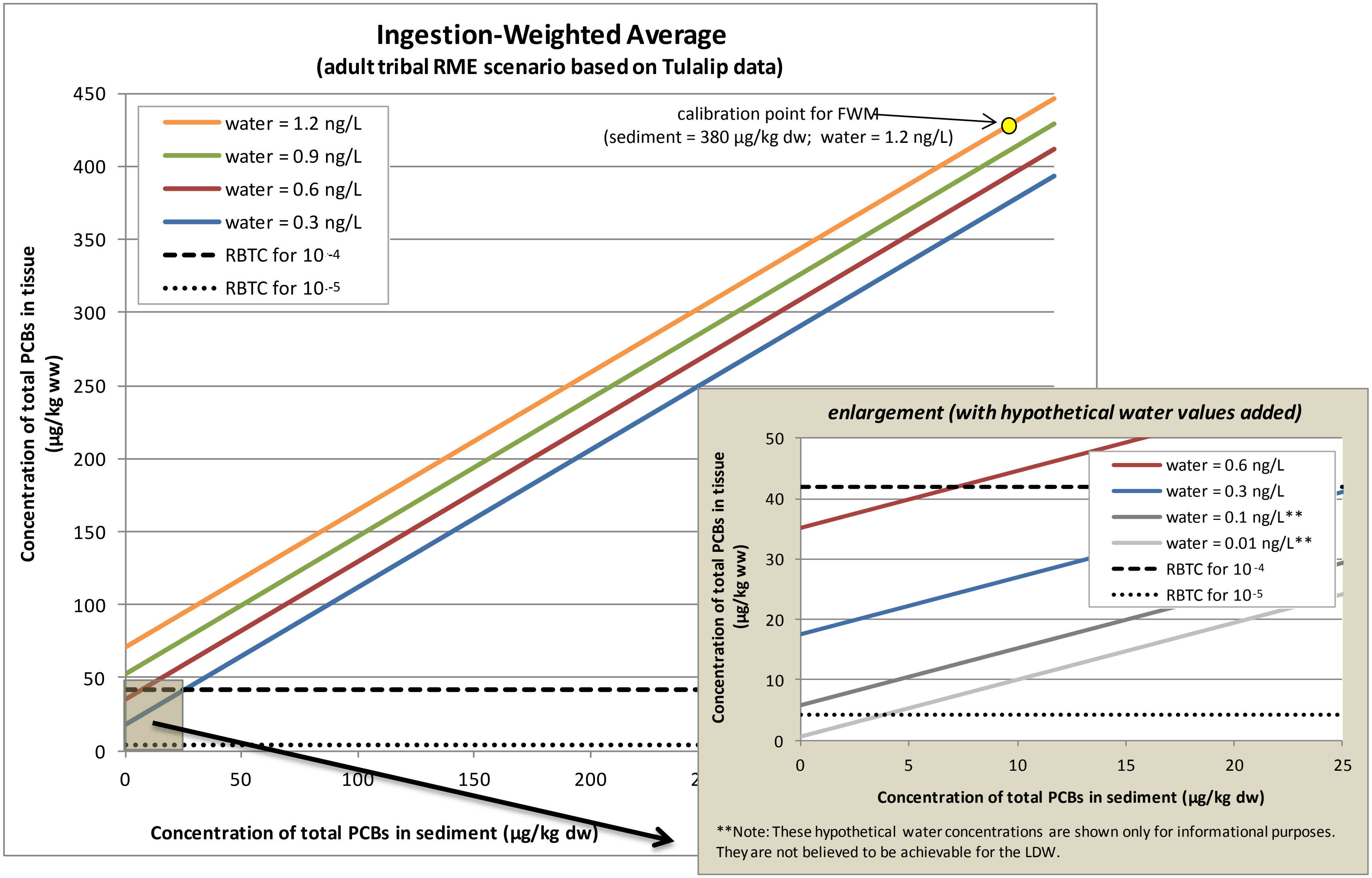
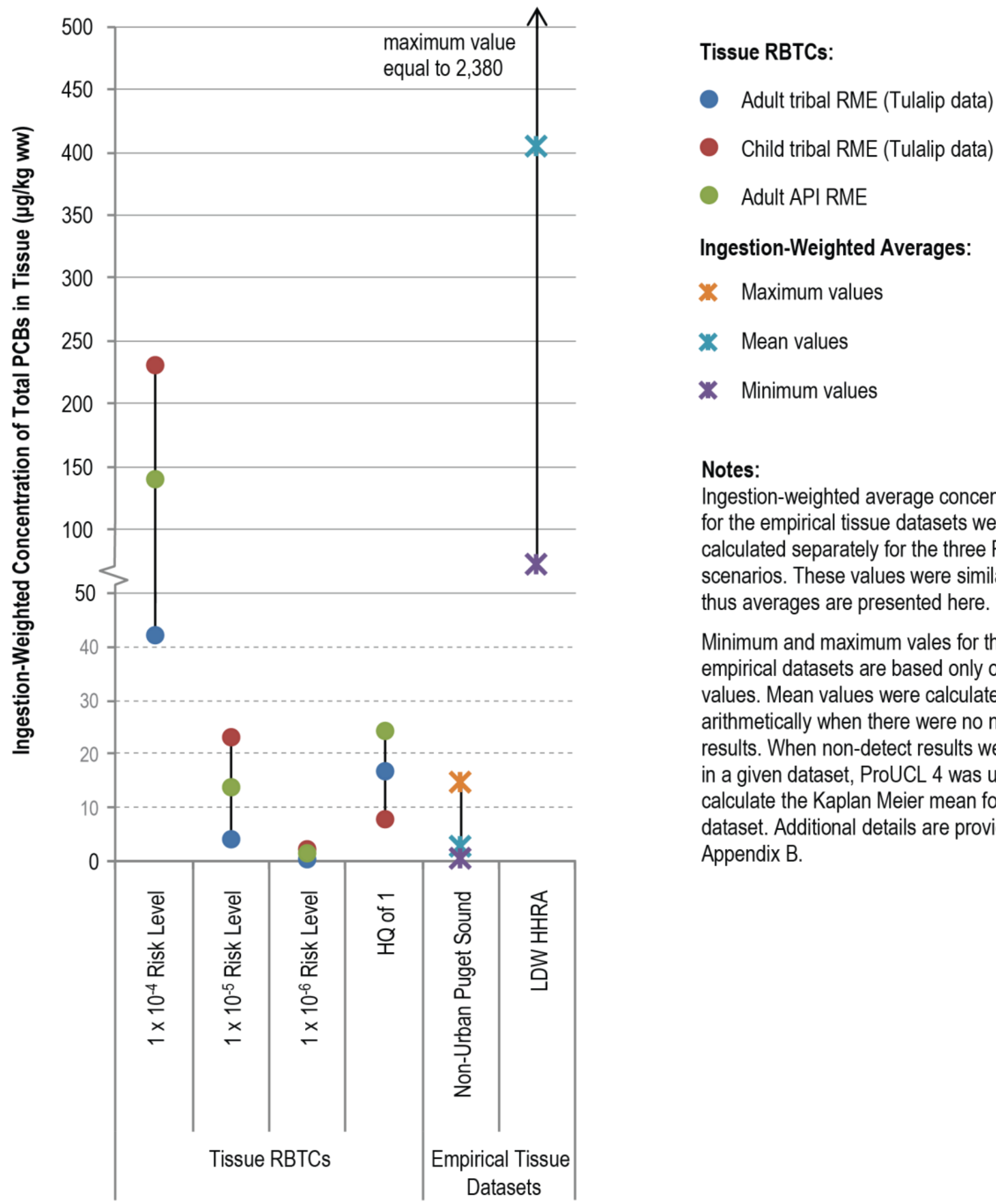


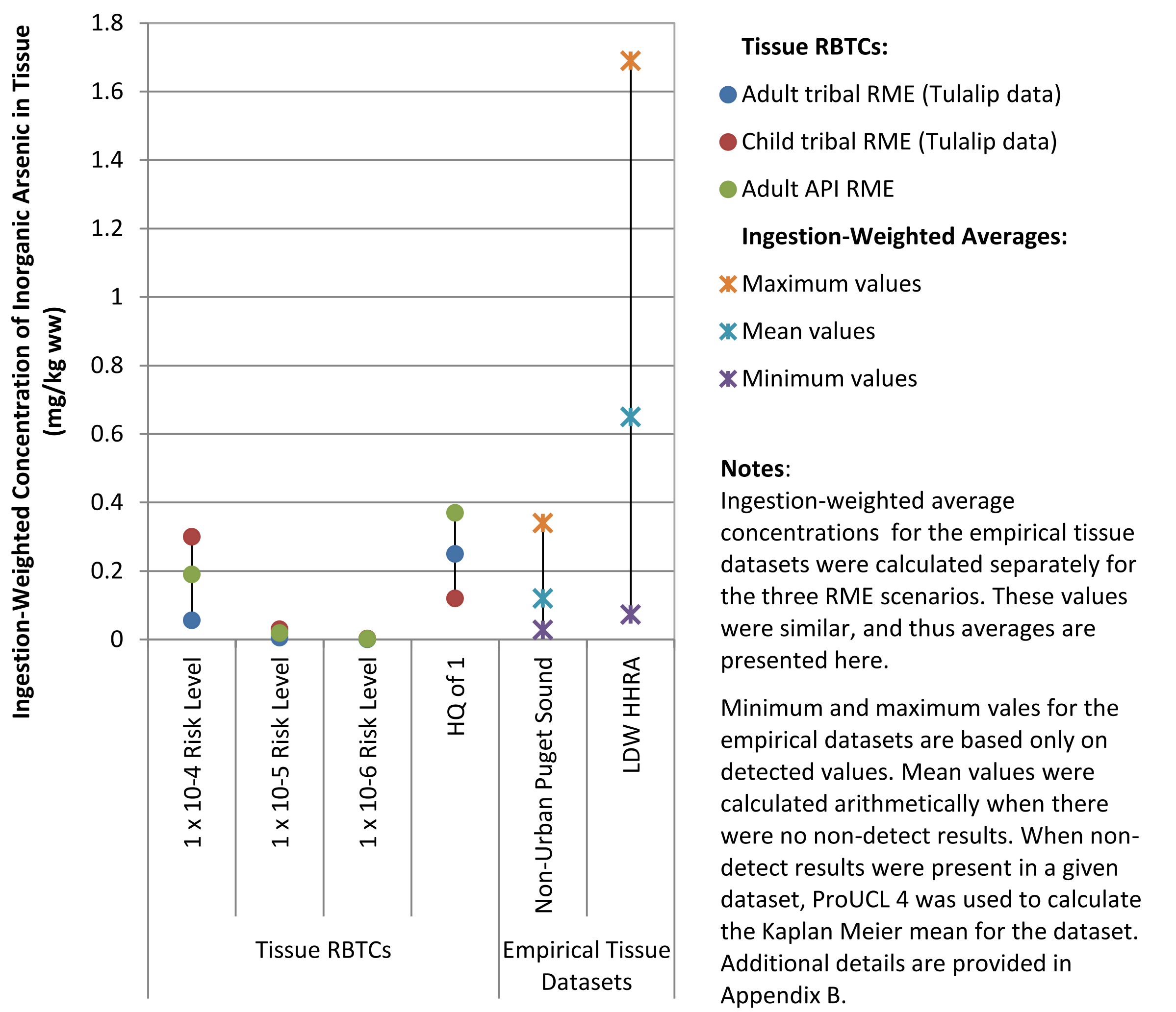
Figure 3-3 Comparison of Total PCB RBTCs with Ingestion-Weighted Average Concentrations from LDW and Non-Urban Puget Sound Tissue Datasets



Notes:
Ingestion-weighted average concentrations for the empirical tissue datasets were calculated separately for the three RME scenarios. These values were similar, and thus averages are presented here.

Minimum and maximum vales for the empirical datasets are based only on detected values. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset. Additional details are provided in Appendix B.

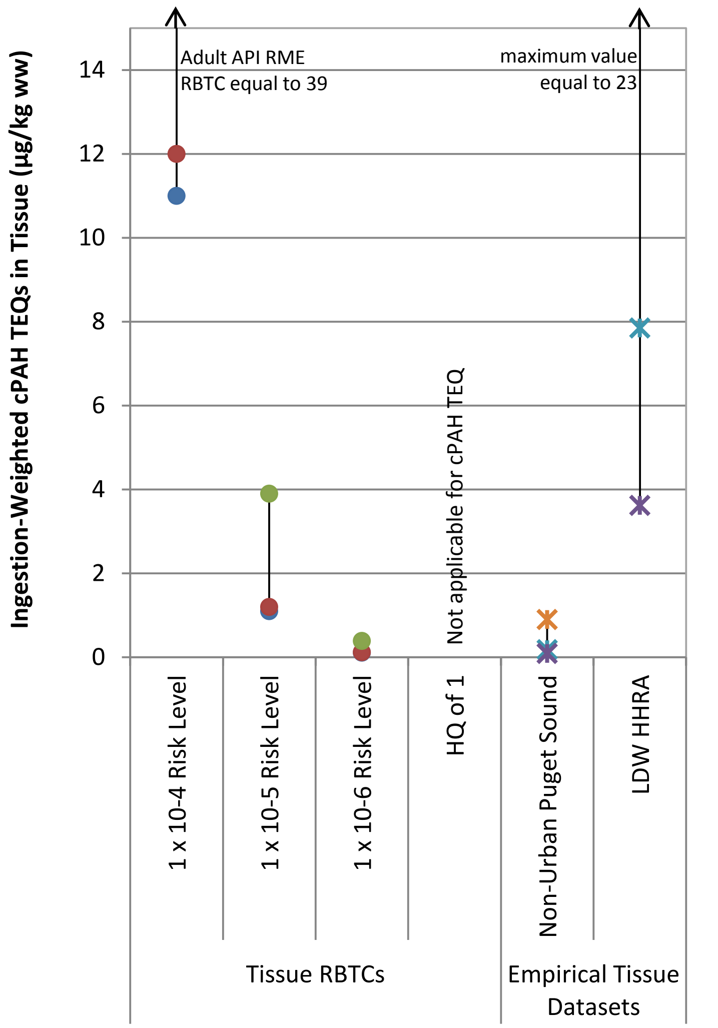
Figure 3-4 Comparison of Inorganic Arsenic RBTCs with Ingestion-Weighted Average Concentrations from LDW and Non-Urban Puget Sound Tissue Datasets



Notes:
Ingestion-weighted average concentrations for the empirical tissue datasets were calculated separately for the three RME scenarios. These values were similar, and thus averages are presented here.

Minimum and maximum vales for the empirical datasets are based only on detected values. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset. Additional details are provided in Appendix B.

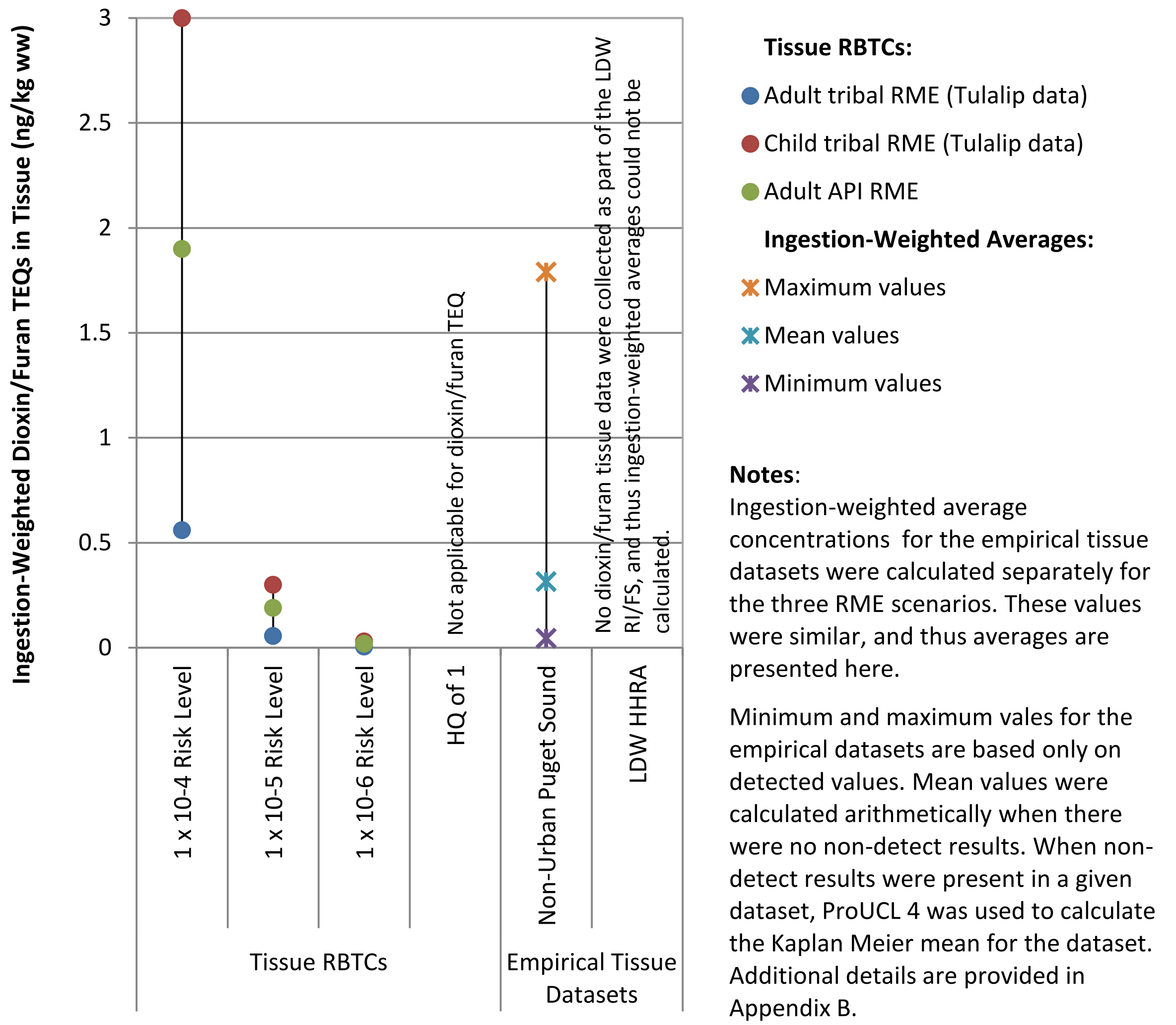
Figure 3-5 Comparison of cPAH RBTCs with Ingestion-Weighted Average Concentrations from LDW and Non-Urban Puget Sound Tissue Datasets



Notes:
Ingestion-weighted average concentrations for the empirical tissue datasets were calculated separately for the three RME scenarios. These values were similar, and thus averages are presented here.

Minimum and maximum vales for the empirical datasets are based only on detected values, or are based on the TEQ calculated using half-RLs when no detected values are available. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset. Additional details are provided in Appendix B.

Figure 3-6 Comparison of Dioxin/Furan RBTCs with Ingestion-Weighted Average Concentrations from LDW and Non-Urban Puget Sound Tissue Datasets



Notes:
Ingestion-weighted average concentrations for the empirical tissue datasets were calculated separately for the three RME scenarios. These values were similar, and thus averages are presented here.

Minimum and maximum vales for the empirical datasets are based only on detected values. Mean values were calculated arithmetically when there were no non-detect results. When non-detect results were present in a given dataset, ProUCL 4 was used to calculate the Kaplan Meier mean for the dataset. Additional details are provided in Appendix B.

4 Remedial Action Objectives and Preliminary Remediation Goals

This section of the feasibility study (FS) identifies narrative remedial action objectives (RAOs) and numerical preliminary remediation goals (PRGs) for cleanup of the Lower Duwamish Waterway (LDW). RAOs for the LDW describe what a proposed cleanup remedy is expected to accomplish to protect human health and the environment (EPA 1999b) PRGs are the contaminant endpoint concentrations or risk levels associated with each RAO that are believed to be sufficient to protect human health and the environment based on available site information (EPA 1997b).

The step of identifying narrative RAOs provides a transition between the findings of the human health and ecological risk assessments and development of remedial alternatives in the FS. The RAOs pertain to the specific exposure pathways and receptors evaluated in the risk assessments and for which unacceptable risks were identified.

RAOs are developed herein for cleanup of contaminated sediment in the LDW Superfund site. Surface water within the site is also a medium of concern. However, no active remedial measures are anticipated for the water column. Improvements in surface water quality are expected following sediment cleanup and implementation of upland source control measures. Further, water quality monitoring will be part of long-term monitoring for the site.

PRGs are intended to protect human health and the environment and to comply with applicable or relevant and appropriate requirements (ARARs) for specific contaminants (EPA 1991b). For the LDW, PRGs are numerical concentrations or ranges of concentrations in sediment that protect a particular receptor from exposure to a hazardous substance by a specific pathway. The PRGs are expressed as sediment concentrations for the identified risk drivers because the alternatives in this FS address cleanup of contaminated sediments. PRGs are not developed in this FS for surface water because actions to directly address water quality are not included among the FS alternatives. Instead, surface water quality will be discussed as water quality ARARs, which are equivalent to PRGs. The RAOs, ARARs, and PRGs presented here may be modified and will be finalized by the U.S. Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology) in the Record of Decision (ROD).

4.1 Development of Remedial Action Objectives

The RAOs are narrative statements of the medium-specific or area-specific goals for protecting human health and the environment. RAOs describe in general terms what the sediment cleanup will accomplish for the LDW. RAOs help focus the development and evaluation of remedial alternatives and form the basis for establishing PRGs.

EPA's *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988) specifies that RAOs are to be developed based on the results of the human health risk assessment (HHRA) and ecological risk assessment (ERA). Other EPA guidance (EPA 1991a, 1999a) states that RAOs should specify:

- The exposure pathways, the receptors, and the contaminants of concern (COCs)
- An acceptable concentration or range of concentrations for each exposure pathway.

Section 2 summarized the remedial investigation (RI), including the chemical and physical conceptual site model. Section 3 summarized the results of the risk assessments, which identified receptors, exposure pathways, risk drivers, and, where calculable, risk-based threshold concentrations (RBTCs). The RAOs presented here were crafted based on the RI and findings from the baseline ERA and HHRA (Windward 2010, 2007a, 2007b).

4.1.1 Remedial Action Objectives for the Lower Duwamish Waterway

The results of the baseline HHRA and ERA indicate that remedial action is warranted to reduce unacceptable human health and ecological risks posed by COCs in LDW sediments. Unacceptable risks were estimated for certain human health exposure scenarios (through seafood consumption and direct contact exposure pathways) and for certain ecological risks (for benthic organisms and for other ecological receptors).

For human health, EPA defines a generally acceptable risk range for excess cancer risks as between one in ten thousand (1×10^{-4}) and one in one million (1×10^{-6}) (i.e., the "target risk range") and for non-cancer risks a hazard index (HI)[1] of 1 or less is considered acceptable (EPA 1991a). Excess cancer risks greater than 10^{-4} or HIs greater than 1 generally warrant a response action (EPA 1997b).

To establish cleanup levels and remedial action levels (RALs), the Washington State Model Toxics Control Act (MTCA) specifies that individual excess cancer risks for identified COCs should be 1×10^{-6} or less, and total excess cancer risks (all carcinogens combined) should not exceed one in one hundred thousand (1×10^{-5}). Cleanup levels should be adjusted downward to take into account exposure to multiple hazardous substances if the total excess cancer risk exceeds 1×10^{-5}. MTCA also specifies that risks resulting from exposure to multiple hazardous substances may be apportioned among hazardous substances in any combination as long as: 1) the total excess cancer risk (all carcinogens combined) does not exceed 1×10^{-5}; and 2) the health threats resulting from exposure to two or more non-carcinogenic hazardous substances with similar types of toxic response does not exceed an HI of 1 (WAC 173-340-708).

[1] HIs are calculated as the sum of hazard quotients with similar non-cancer toxic endpoints.

Based on guidance provided by EPA under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and other requirements provided in MTCA/Sediment Management Standards (SMS), four RAOs have been identified for the cleanup of LDW sediments. These RAOs are identified below, and a discussion of each RAO follows.

RAO 1: Reduce human health risks associated with the consumption of resident LDW fish and shellfish by reducing sediment and surface water concentrations of COCs to protective levels.

Lifetime excess cancer risks from human consumption of resident LDW seafood are estimated to be greater than 1×10^{-6} for some individual carcinogens, and greater than 1×10^{-4} for carcinogens cumulatively under reasonable maximum exposure (RME) seafood consumption scenarios. In addition, the estimated non-cancer risks exceed an HI of one (see Tables 3-4a and 3-4b of Section 3). These estimated risks warrant response actions to reduce exposure.

Total polychlorinated biphenyls (PCBs), arsenic, and carcinogenic polycyclic aromatic hydrocarbons (cPAHs) are the primary risk drivers that contribute to the estimated risks based on consumption of resident seafood. As discussed in Section 3, although risks associated with consumption of dioxins/furans in resident seafood were not quantitatively assessed in the baseline HHRA, those risks were assumed to be unacceptable; thus, dioxins/furans are also considered risk drivers with respect to the consumption of resident seafood.

Achieving RAO 1 requires that site-wide average[2] concentrations of COCs in sediment be reduced, which in turn is expected to reduce tissue COC concentrations in fish and shellfish exposed to these sediments. Exposure of fish and shellfish to COCs in sediment occurs within the biologically active zone. As reported in the RI (Windward 2010), this zone is estimated to be the upper 10 cm of sediment. Deeper, undisturbed sediments contribute negligibly to the risks addressed by this RAO if contaminants in these deeper sediments do not migrate into the biologically active zone. However, deeper sediments that contain contaminants at concentrations above action levels and that are potentially subject to disturbance (e.g., erosion, propeller scour, earthquakes) or otherwise may migrate into the biologically active zone through advection or other mechanisms may warrant response actions to satisfy this RAO.

With regard to seafood consumption, bioaccumulative COCs enter the food web from both sediment and water. For example, the food web model used to predict tissue PCB concentrations (refer to Appendix D of the RI; Windward 2010) assumes that the

[2] The FS uses average concentrations to evaluate the effectiveness of alternatives in attaining RAOs. In practice, compliance with clean-up levels will be based on the 95% upper confidence limit on the mean (UCL95).

exposure of fish and shellfish to PCBs occurs through their exposure to both sediments and surface water.

Substantial reductions in the concentrations of such COCs in sediment achieved through remediation should also reduce the concentrations of those COCs in surface water, thereby contributing to reducing their concentrations in fish and shellfish tissue and ultimately reducing human health risks, as stated in RAO 1. The relationships between sediment, surface water, and tissue concentrations are complex, and will be assessed through long-term monitoring of the remedial actions.

RAO 2: Reduce human health risks associated with exposure to COCs through direct contact with sediments and incidental sediment ingestion by reducing sediment concentrations of COCs to protective levels.

Lifetime excess cancer risks from human direct contact and incidental sediment ingestion RME scenarios (netfishing, tribal clamming, and beach play) are estimated to be within EPA's 10^{-4} to 10^{-6} target risk range (Tables 3-6a and 3-6b of Section 3) for the individual risk drivers. Some individual excess cancer risks exceed 1×10^{-6}, and total risks from all risk drivers exceed 1×10^{-5}, both of which are MTCA thresholds. Therefore, the risks associated with these exposure pathways warrant response actions to reduce exposure. No HIs were greater than 1 for any of the direct contact or incidental ingestion sediment RME scenarios, with the exception of one individual beach (Beach 4). Total PCBs, arsenic, cPAHs, and dioxins/furans are the primary risk drivers that contribute to the estimated excess cancer risks, and total PCBs are also a risk driver for noncancer risks based on direct contact.

Achieving RAO 2 requires that average concentrations of COCs be reduced at locations and depths within the sediment where people have the potential to be exposed. For netfishing activities, exposure is over the entire LDW and to surface sediments (0 to 10 cm). Direct contact risks in the beach play and clamming areas are assumed to result from exposure to the upper 45 cm depth interval, which accounts for potential exposures to children and clammers, who may dig holes deeper than 10 cm. Deeper sediments in other areas do not contribute appreciably to these risks unless they could be exposed by future disturbances (e.g., erosion, propeller scour, earthquakes). Achieving and maintaining this RAO may include response actions to address deeper sediments containing concentrations of the risk drivers above action levels if such disturbances of the overlying sediments over time may potentially expose these sediments.

RAO 3: Reduce risks to benthic invertebrates by reducing sediment concentrations of COCs to comply with the Washington State SMS.

The SMS provide both chemical and biological effects-based criteria. The numerical SMS chemical criteria are available for 47 contaminants or groups of contaminants (i.e., sediment quality standards [SQS] and cleanup screening levels [CSL]). These numerical chemical criteria are based on apparent effects thresholds (AETs) developed for four different benthic endpoints by the Puget Sound Estuary Program (PSEP) (Barrick et al. 1988). An AET is the highest "no effect" sediment concentration of a specific contaminant above which a significant adverse biological effect always occurred among the several hundred samples used in its derivation. In general, the lowest of the four AETs for each contaminant was identified as the SQS; the second lowest AET was identified as the CSL. According to the SMS (WAC 173-204), locations with all contaminant concentrations less than or equal to the SQS are defined as having no acute or chronic adverse effects on biological resources, locations with any contaminant concentrations between the SQS and the CSL are defined as having minor adverse effects, and locations with any contaminant concentration greater than the CSL are defined as having more pronounced adverse effects (refer to Section 5 of the RI, Windward 2010).

The baseline ERA (Windward 2007a) reported that 41 contaminants were detected in surface sediment at one or more locations within the LDW at concentrations exceeding their respective SQS (see Table 3-1, Section 3 of this FS). Thus, the ERA determined that these 41 contaminants are COCs because they pose a risk to the benthic invertebrate community. These 41 COCs are designated as risk drivers for this pathway.

Benthic organisms reside primarily in the biologically active zone (uppermost 10 cm) of intertidal and subtidal sediments of the LDW (Section 2 of the RI, Windward 2010). Deeper sediments in areas subject to disturbance (e.g., erosion, propeller scour, earthquakes) that contain COCs at concentrations above the SQS may warrant response actions to satisfy RAO 3.

RAO 4: Reduce risks to crabs, fish, birds, and mammals from exposure to COCs by reducing concentrations of COCs in sediment and surface water to protective levels.

The ERA (Windward 2007a) determined that exposure to seven contaminants, identified as COCs, exceeded toxicity benchmarks for fish, birds, or mammals. In consultation with EPA and Ecology, total PCBs were designated as the risk driver associated with seafood consumption based on estimated risks to river otters. Thus, achievement of RAO 4 is based on addressing PCB risk to river otters (see Section 3.1.3 for discussion of other ecological COCs).

River otters are indirectly exposed to PCBs in sediment primarily through the consumption of prey. Therefore, achieving this RAO requires that site-wide average concentrations of PCBs in sediment be reduced, with the expectation that sediment cleanup will reduce PCB concentrations in fish and shellfish, and that concentrations of the remaining six COCs identified for this exposure pathway will also be reduced to acceptable levels for other receptors (Windward 2010).

The potential for exposure of prey to COCs occurs primarily within the biologically active zone (upper 10 cm of sediment). Deeper sediments, if left undisturbed, contribute negligibly to the risks addressed by this RAO. Deeper sediments in areas subject to disturbance (e.g., erosion, propeller scour, earthquakes) that contain COCs at concentrations above action levels may warrant response actions to satisfy RAO 4.

Remediation will reduce COC concentrations in the LDW sediments; this in turn should also reduce those same COC concentrations in surface water, thereby contributing to a reduction of their concentrations in the tissue of fish and shellfish (including prey species). The relationships between sediment, surface water, and tissue concentrations are complex, and will be assessed through long-term monitoring following completion of the remedial actions.

4.1.2 Role of Source Control

Controlling sources of contamination to the LDW to the maximum extent practicable is an explicit MTCA expectation when natural attenuation is part of the remedial action (WAC 173-340-370). Active sediment remediation of COCs that have accumulated in sediments over time will address a major portion of the risks addressed in each RAO; however, without continued source control to keep reducing COC inputs to the LDW, sediments will likely recontaminate and water quality may continue to be impaired. Source control must include continued involvement by the Source Control Work Group (SCWG) to protect the long-term investments in the LDW cleanup.

Contaminated media from within the LDW drainage basin can affect sediments through several pathways, which can be organized into seven general types based on the origin of contamination, pathways to sediments, and the types of source control available:

- Direct discharge into the LDW (e.g., CSOs, storm drains)
- Surface water runoff or sheet flow
- Spills and/or leaks to the ground, surface water, or directly into the LDW
- Groundwater migration/discharge
- Bank erosion/leaching
- Atmospheric deposition
- Transport of resuspended contaminated sediments.

Understanding how each of these potential sources and pathways may impact a given sediment area is a complex undertaking and beyond the scope of this FS. Whether additional localized source control actions, beyond what has already been done, are needed before in-water work can begin will be considered in remedial design. This will require a recontamination/source control assessment study that varies in scope and magnitude depending on the specific project area.

Currently, source identification and implementation of effective control efforts in the LDW watershed are supported by a cooperative interagency program with the goal of identifying sources of potential contamination and recontamination in coordination with sediment cleanups and promoting their control. Ecology, as the lead entity for implementing source controls in the LDW, formed the LDW SCWG in 2002, which conducts several source control activities within the LDW area. The SCWG is composed primarily of public agencies responsible for source control, including EPA, Seattle Public Utilities, King County, and the Port of Seattle. The LDW source control strategy (Ecology 2004) also identifies various regulatory programs at EPA and Ecology that are called upon as needed for source control as well as several *ad hoc* members of the SCWG, including the City of Tukwila, Puget Sound Clean Air Agency, and Washington State Departments of Transportation (WSDOT) and Health (WDOH). All LDW SCWG members are public agencies with various source control responsibilities; the group's collective purpose is to share information, identify issues and data gaps, develop action plans for source control tasks, coordinate implementation of various source control measures, and share progress reports on these activities. Individually, these agencies are able to use their regulatory authority to promote source control in the LDW via source tracing sampling, stormwater and combined sewer overflow (CSO) programs, permits, hazardous waste management and pollution prevention programs, inspection and maintenance programs, water quality compliance and spill response programs, and environmental and pathway assessments.

Ecology's *Lower Duwamish Waterway Source Control Strategy* (Ecology 2004) is consistent with sediment source control protocols described in EPA guidance (2002b) and the SMS (Ecology 1995). The strategy describes the process and timing for implementing source control and the roles of various regulatory agencies responsible for conducting source control (e.g., SCWG) and enforcement. The strategy also provides for tracking and documenting source control progress in the LDW.

The focus of the LDW source control strategy is to identify and manage sources of COCs to waterway sediments in coordination with sediment cleanups and to prevent post-cleanup recontamination to levels exceeding cleanup goals established in the ROD to the extent practicable (Ecology 2004). Specific goals for the source control program are:

- Minimize the potential for contaminants in sediments to exceed the SMS criteria (as stated in WAC 173-204) and the LDW sediment cleanup levels (to be established in the ROD).
- Achieve adequate source control that will allow sediment cleanups to begin.
- Increase opportunities for natural recovery of sediments.

- Support long-term suitability and success of current and future habitat restoration opportunities.

Source control started in 2002 and is an ongoing, iterative process that continually produces new information. During remedial design, the work accomplished by Ecology and other public entities will serve as a foundation for any additional source control investigations and actions necessary before implementing various components of the sediment cleanup.

4.2 Applicable or Relevant and Appropriate Requirements (ARARs)

CERCLA Section 121(d) requires remedial actions to achieve (or formally waive) ARARs, which are defined as any legally applicable or relevant and appropriate standard, requirement, criterion, or limitation under any federal environmental law, or promulgated under any state environmental or facility siting law that is more stringent than the federal law. Similarly, MTCA requires that all cleanup actions comply with all legally applicable or relevant and appropriate requirements in applicable state and federal laws, as set forth in WAC 173-340-710. Given these substantive similarities in language between CERCLA and MTCA on the role of legal requirements, the FS uses the term ARARs to identify requirements that will satisfy or comply with both statutes. This subsection identifies ARARs for cleanup of the LDW. Section 9 of this document evaluates whether the remedial alternatives developed for cleanup of the LDW comply with these ARARs.

The National Contingency Plan (40 CFR 300.5) defines applicable requirements as the more stringent among those cleanup standards, standards of control, and other substantive requirements, criteria, or limitations promulgated under federal environmental or state environmental or facility siting laws that specifically address a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstances found at a CERCLA site. A requirement may not be applicable, but nevertheless may be relevant and appropriate. Relevant and appropriate requirements address problems or situations sufficiently similar to those encountered at CERCLA and MTCA sites that their use is well-suited to the particular site. Relevant and appropriate requirements have the same effect as applicable requirements. They are not treated differently in any way.

Washington State has promulgated environmental laws and regulations to implement or co-implement several major federal laws through federally approved programs, for example, the Clean Water Act, Clean Air Act, and RCRA. The ARAR is the more stringent of either a federal requirement or a state requirement. Because this FS is being conducted under a joint CERCLA and MTCA order, applicable or relevant and appropriate provisions of MTCA and the SMS are considered to be ARARs for CERCLA, as well as governing requirements under MTCA. MTCA is a particularly important CERCLA ARAR. As will be seen, its background standards for final sediment cleanups are more stringent, and its allowable excess cancer risk standards are considerably more stringent. CERCLA permits risk-based cleanup standards within a range of 10^{-4} to 10^{-6} excess cancer risks. EPA policy and guidance recommends trying to achieve the more stringent 10^{-6} standard but accepts lesser standards within the range based on many factors. MTCA requires risk-based cleanup standards to be set at one in one million (1×10^{-6}) excess cancer risk levels for all individual carcinogens (such as PCBs) at a site, and a total excess cancer risk of one in one-hundred thousand (1×10^{-5}) for all carcinogens cumulatively at a site. Procedural requirements under state laws (e.g., MTCA disproportionate cost analysis methodology) are not CERCLA ARARs, but are required to comply with MTCA.

Table 4-1 lists and summarizes ARARs for the LDW site. Some ARARs prescribe minimum numerical requirements or standards for cleanup of specific media such as sediment, surface water, fish tissue, and groundwater. Other ARARs place requirements or limitations on actions that may be undertaken as part of a remedy. Table 4-2 lists other requirements or laws that are not considered ARARs by EPA and Ecology, generally because their primary purpose is not environmental protection (or state facility siting), but rather, for example, historical preservation of archaeological artifacts, endangered species, or workplace protection. Consideration of or compliance with requirements under these laws is anticipated for implementing most of the alternatives in this FS. While all federal, state, and local laws have to be complied with (except the need to acquire federal, state, or local permits for onsite cleanup work), it is helpful in considering remedial alternatives to list other laws or requirements alongside ARARs that will be implemented.

Some ARARs contain numerical values or methods for developing such values. These ARARs establish minimally acceptable amounts or concentrations of hazardous substances that may remain in or be discharged to the environment, or minimum standards of effectiveness and performance expectations for the remedial alternatives. RBTCs based on risks to human health or the environment may dictate setting more stringent standards for remedial action performance, but they cannot be used to relax the minimum legally prescribed standards in ARARs. The rest of this subsection focuses on ARARs containing specific minimum numerical standards.

There are no federal ARARs providing numerical standards for hazardous substances, pollutants, or contaminants in sediment. However, Washington State has promulgated numerical standards in the SMS for the protection of benthic invertebrates, and these regulations are cross-referenced in MTCA. Under CERCLA, the SMS criteria are considered ARARs and are promulgated standards for the LDW under MTCA. However, although the SMS contain narrative standards to protect human health and other biological resources, no SMS or other state numerical sediment criteria have been established to protect human health, including human consumers of seafood, or for other biological resources such as birds, fish, or mammals. Cleanup levels or standards for protection of these receptors are derived from RBTCs developed during the risk assessments performed during the LDW RI (Windward 2010).

Surface water (i.e., the water column) is also a medium of concern in the LDW. Therefore, federal water quality criteria (WQC) developed to protect ecological receptors and human consumers of fish and shellfish are relevant and appropriate requirements or minimum levels or standards for remedial action pursuant to CERCLA Section 121 (d)(2)(A)(ii) and RCW 70.105D.030(2)(e). Under CERCLA and MTCA, state water quality standards (WQS) approved by EPA are generally applicable requirements under the Clean Water Act (CWA). National recommended federal WQC established pursuant to Section 304(a)(1) of the CWA are compiled and presented on the EPA website at http://www.epa.gov/waterscience/criteria/wqctable/. Although these criteria are advisory for CWA purposes (to assist states in developing their standards), the last sentence of CERCLA Section 121(d)(2)(A)(ii) makes them minimum cleanup levels or standards, where relevant and appropriate under the circumstances, for CERCLA site remedial actions.

Consequently, the more stringent of the federal WQC and the state WQS are the cleanup levels or standards for the site. Washington State WQS for the protection of aquatic life are found at WAC 173-201A-240. The numerical criteria for aquatic life meet the federal requirements of Section 303(c)(2)(B) of the CWA and are at least as stringent as the federal WQC. Table 4-3 presents state and federal marine and freshwater values that have been developed for aquatic life and human health WQC. Specific considerations for compliance with federal and state aquatic life WQC and human health WQC are discussed in Section 4.2.2 of the RI (Windward 2010).

4.3 Process for Development of Preliminary Remediation Goals

PRGs are the COC endpoint concentrations initially identified for each RAO that are believed to be sufficient to protect human health and the environment based on available site information (EPA 1997b). The PRGs are used in the FS to guide the geographic definition of areas of potential concern (AOPCs) and the evaluation of proposed sediment remedial alternatives. PRGs are not final CERCLA/MTCA cleanup levels and standards. EPA and Ecology will select CERCLA/MTCA cleanup levels and standards in the ROD.

PRGs are developed in this subsection for each risk-driver COC, and are expressed as sediment concentrations that are intended to achieve the corresponding RAO. PRGs are based on considering the following factors:

- ARARs, including MTCA risk requirements, and SMS criteria
- RBTCs based on the human health and ecological risk assessments

- Background concentrations if protective RBTCs are below background concentrations
- Analytical practical quantitation limits (PQLs) if protective RBTCs are below concentrations that can be quantified by chemical analysis.

This section presents the numerical criteria in these categories to enable a comprehensive analysis and identification of PRGs. The pertinent information is then compiled and numerical PRGs are identified for each risk driver and each RAO.

4.3.1 Role of ARARs

Certain PRGs in this FS are set based on MTCA's more stringent (than CERCLA) excess cancer risk standards and its requirement that final cleanups achieve natural background levels when RBTCs are below background. The SMS (WAC 173-204) also contain numerical sediment contaminant concentration criteria pertinent for protecting the marine benthic invertebrate community (and hence the SMS criteria apply to PRGs for RAO 3).[3]

The SMS chemical and biological criteria are applied on a point basis to the biologically active zone of the sediments (i.e., upper 10 cm). Under the SMS, sediment cleanup standards may be established on a site-specific basis within an allowable range of contamination. The SQS, also called the sediment cleanup objective, and the CSL, also called the minimum cleanup level (MCUL), define this range. WAC 173-204-570(4) specifies that the site-specific cleanup standards shall be as close as practicable to the cleanup objective (the SQS) but in no case shall exceed the minimum cleanup level (the CSL). For this reason, in developing PRGs and analyzing alternatives, the SQS is used in this FS.[4] This WAC subsection also states that the cleanup standards shall be defined in consideration of the net environmental effects, cost, and engineering feasibility of different cleanup alternatives. The following WAC subsection (WAC 173-204-570(5)) emphasizes that all cleanup standards must ensure protection of human health (for which there are no SMS numerical criteria) and the environment (which encompasses receptors beyond the benthic invertebrate community). The SMS also require that contaminant concentrations (and toxicity) meet the cleanup standards within a reasonable time frame, as defined by a number of factors in WAC 173-204-580(3)(a).

As described in Section 4.2, surface water quality criteria are ARARs for the site because the water column is part of the site. The water column is affected by the sediment contaminant concentrations, as well as other factors, including ongoing releases, inflowing water from the Green/Duwamish River system, direct discharges to the LDW, and aerial deposition. However, the water column cannot practicably be directly

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3 The SMS are ARARs under CERCLA and promulgated numerical standards under MTCA.

4 Co-located sediment toxicity test results that "pass," (i.e., indicate no toxicity) override exceedances of the SMS numerical criteria only for determining compliance with RAO 3.

remediated. Thus, while surface water is included as a medium of concern to be addressed by RAOs 1 and 4, surface water quality ARARs have not been identified as numerical PRGs at the site. However, because the WQC are CERCLA ARARs, the quality of LDW surface water will have to meet the more stringent of the federal and state aquatic life and human health WQC (Table 4-3) or be waived at or before completion of CERCLA remedial action.

Significant water quality improvements are anticipated as a result of sediment remediation and source control. Water quality monitoring will be part of the selected remedy to help measure the efficacy of sediment remediation and source control, and to assess compliance with ARARs. The remedial alternatives developed and evaluated in this FS may not comply with all surface water quality standards, or with natural background sediment standards required under MTCA in lieu of protective human seafood consumption RBTCs, in which case surface water quality and MTCA ARAR waivers could be issued by EPA at or before the completion of the remedial action. Potential ARAR waivers are listed in Section 121(d)(4) of CERCLA. The most common waiver is for technical impracticability, the standards for which are explained in detail in comprehensive EPA guidance designed to ensure a rigorous evaluation, and that only genuine demonstrated technical impracticability will qualify.

4.3.2 Role of RBTCs

The RI developed site-specific sediment RBTCs (summarized in Section 3.3 of this document) for each of the risk-driver COCs. RBTCs for human health were calculated based on risks associated with the direct sediment contact RME scenarios and seafood consumption RME scenarios. RBTCs for wildlife receptors were calculated based on prey consumption by river otters. For the benthic invertebrate community, RBTCs were set at the SQS and CSL.

Total PCBs, cPAHs, arsenic, and dioxins/furans are the risk drivers for the human seafood consumption pathway. Sediment RBTCs for total PCBs were calculated for the 1×10^{-4} excess cancer risk level and are applied as site-wide average concentrations.[5] As discussed in Section 3.3, sediment RBTCs based on the seafood consumption pathway were not calculated for arsenic and cPAHs, because correlations between sediment contaminant concentrations and receptor tissue concentrations could not be established. Sediment RBTCs were also not calculated for dioxins/furans. Fish and shellfish tissue data were not collected for this risk driver during the RI because it was determined that sediment concentrations would exceed RBTCs, which would be more stringent than

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5 For the excess cancer risk levels of 1 in 1,000,000 (1×10^{-6}) and 1 in 100,000 (1×10^{-5}) and for the non-cancer HQ of 1, even at a total PCB concentration of 0 µg/kg dw in sediment, the food web model predicted total PCB concentrations in tissue that would result in a risk estimate greater than the risk levels for the RME seafood consumption scenarios because of the contribution of total PCBs from water alone, even at concentrations similar to those in upstream water (i.e., 0.3 ng/L). Therefore, sediment RBTCs for these risk levels were represented as "< 1" (see Table 3-9).

natural background, resulting in natural background concentrations in sediment being the PRG for dioxins/furans.

Total PCBs, cPAHs, arsenic, and dioxins/furans are also the human health risk drivers for the direct sediment contact pathway. Sediment RBTCs for these hazardous substances were presented in Table 3-10 for each of the three direct sediment contact RME scenarios (i.e., netfishing, tribal clamming, and beach play). These sediment RBTCs are average concentrations applied to the spatial area over which exposure would reasonably be expected.

A total PCB sediment RBTC was calculated to protect wildlife. It protects river otters as the most sensitive representative wildlife species from the ERA, based on their consumption of prey species (Windward 2007a). The RBTC is applied as a site-wide average concentration.

4.3.3 Role of Background Concentrations

Both CERCLA and MTCA consider background hazardous substance concentrations when formulating PRGs and cleanup levels. Both recognize that setting numerical cleanup goals at levels below background is impractical (because of the potential for recontamination to the background concentration). MTCA (WAC 173-340-200) defines natural background as the concentrations of hazardous substances that are consistently present in an environment that have not been influenced by localized human activities. Thus, under MTCA, a natural background concentration can be defined for man-made compounds even though they may not occur naturally (e.g., PCBs deposited by atmospheric deposition into an alpine lake). According to CERCLA guidance, natural background refers to substances that are naturally present in the environment in forms that have not been influenced by human activity (e.g., naturally occurring metals).

MTCA cleanup levels cannot be set at concentrations below natural background (WAC 173-340-705(6)). Similarly, CERCLA guidance states that natural background concentrations establish a limit below which a lower cleanup level cannot be achieved (EPA 2005b).

Both cleanup programs also recognize that natural and man-made hazardous substance concentrations can occur at a site in excess of natural background concentrations, not as a result of local site-related releases but caused by human activities in areas remote from the site and natural processes that transport the contaminants to the site (e.g., atmospheric uptake, transport, and deposition). CERCLA defines “anthropogenic background” as natural and human-made substances present in the environment as a result of human activities, but not related to a specific release from the CERCLA site undergoing investigation and cleanup (EPA 2002c). MTCA defines the term “area background” as media-specific concentrations that are consistently present in the environment in the vicinity of a site that are attributable to human activities unrelated to specific releases from the site. CERCLA generally does not require cleanup to concentrations below anthropogenic background concentrations. In states that have a more stringent state standard, CERCLA cleanups must try to meet state ARARs, or EPA must waive the ARAR at or before completion of the remedial action. MTCA defines natural background as the cleanup standard required for final remedies when natural background concentrations are higher than the calculated risk-based cleanup levels (i.e., RBTCs). Thus, a CERCLA remedy in Washington State that cannot achieve natural background concentrations is not final unless this MTCA requirement is achieved or waived, or residual risks are otherwise sufficiently controlled. Under MTCA, because a waiver is not available, a remedy that cannot achieve natural background concentrations remains "interim" by default (see WAC 173-340-430) unless it is technically impossible to implement a more permanent cleanup action for all or a portion of the site (see WAC 173-340-360(2)(e)(iii)), and residual risks can be sufficiently controlled with institutional controls.

As a result, PRGs have been set at natural background concentrations for hazardous substances that have risk-based concentrations below natural background concentrations. EPA and Ecology recognize that natural background concentrations are unlikely to be achieved at the site and that long-term sediment contaminant concentrations following active sediment remediation will be governed primarily by concentrations in incoming sediment from the Green/Duwamish River system and new or continuing releases from other sources subject to further source control actions (see Section 5). Long-term monitoring will be used to determine what the technically practicable lower limits are for site concentrations, as well as where source control should continue to be focused. When these lower limits are reached, as demonstrated by monitoring data, a CERCLA technical impracticability (TI) waiver of the MTCA ARAR, in conjunction with institutional controls, could be used to provide administrative closure of the LDW cleanup. The TI waiver would address the gap between the technically practicable limit and natural background concentrations. Under MTCA, sufficient institutional controls that address remaining human health risks may similarly allow a final cleanup determination, where it is technically impossible to implement a more permanent cleanup action for all or a portion of the site (see WAC 173-340-360(2)(e)(iii)).

4.3.4 Natural Background in Sediment

This section presents estimates of natural background concentrations for total PCBs, arsenic, cPAHs, and dioxins/furans in sediment.[6] To characterize natural background, marine sediment data were compiled from areas within Puget Sound that have not been influenced by localized human activities. These data represent non-urban, non-localized concentrations that exist as a result of natural processes and/or the large-scale distribution of these hazardous substances from anthropogenic sources.

6 EPA and Ecology will set natural background concentrations and remediation goals in the ROD.

The Dredged Material Management Program (DMMP) (comprised of the U.S. Army Corps of Engineers [USACE], EPA, Ecology, and the Washington State Department of Natural Resources [DNR]) collected sediment data throughout Puget Sound in the summer of 2008 and documented the results in a study called *Final Report: Puget Sound Sediment PCB and Dioxin 2008 Survey, OSV BOLD SURVEY REPORT* (EPA OSV *Bold* Survey; EPA 2008b). EPA and Ecology have determined that the 95% upper confidence limit on the mean (UCL95) of the data from the EPA OSV *Bold* Survey will be used in this FS for natural background concentrations. Data were collected from 70 sampling locations throughout Puget Sound, as well as from the area around the San Juan Islands and the Strait of Juan de Fuca. Locations for each target sampling station are displayed in Figure 4-1. A subset of these sample locations (N = 20) were located within four reference areas (Carr Inlet, Samish Bay, Holmes Harbor, and Dabob Bay) established by Ecology. In each of these reference areas, five target sediment sampling locations were located based on a stratified random sampling design. The remaining 50 sample locations were spread throughout Puget Sound and the straits of Georgia and Juan de Fuca and were intended to represent areas outside the influence of urban bays and known point sources. At five stations, a duplicate sample (or field split) was collected for quality assurance purposes. Samples were analyzed for the full suite of DMMP contaminants, including semi-volatile organic compounds, PAHs, PCB Aroclors and PCB congeners, organochlorine pesticides, and trace metals, as well as for sediment conventionals (e.g., total organic carbon [TOC], grain size, percent solids). Summary statistics (see Table 4-4) were then calculated for the EPA OSV *Bold* Survey data for each of the four human health risk drivers using the statistical software ProUCL version 4.00.04. Statistical analyses of these sediment data did not adjust for the spatial bias resulting from repeated sampling of four reference areas, or other spatial aspects of how the sample locations were distributed.

4.3.4.1 Natural Background for Arsenic in Sediment

Arsenic was detected in all of the samples from the EPA OSV *Bold* Survey (Table 4-4). Concentrations ranged from 1.1 to 21 milligrams per kilogram dry weight (mg/kg dw), with a mean concentration of 6.5 mg/kg dw, and an UCL95 of 7.3 mg/kg dw. Using the UCL95 statistic, the background concentration for arsenic is rounded to 7 mg/kg dw.

4.3.4.2 Natural Background for Total PCBs in Sediment

Total PCBs as Aroclors were below reporting limits in the majority of sediment samples from the EPA OSV *Bold* Survey (Table 4-4). The PCB congener method, with its lower reporting limits, produced a detection frequency of 100%, based on quantifying at least one PCB congener in each sample. Total PCBs in each sample were calculated by summing the concentrations of all detected PCB congeners, consistent with the protocol in the SMS for reporting total PCBs by summing the concentrations of all detected PCB Aroclors. Using the congener results, total PCB concentrations ranged from 0.01 to 10.6 micrograms per kilogram (µg/kg) dw, with a mean of 1.2 µg/kg dw and an UCL95 of 1.5 µg/kg dw. Using the UCL95 statistic, the background concentration for total PCBs is rounded to 2 µg/kg dw.

4.3.4.3 Natural Background for cPAHs in Sediment

The detection frequency for cPAHs in the EPA OSV *Bold* Survey was 87%, based on quantifying at least one cPAH compound in each sample (Table 4-4). Total cPAHs in each sample were calculated by summing the concentrations of all detected cPAH compounds multiplied by their respective benzo(a)pyrene potency equivalency factors (PEFs), along with half the reporting limits of any undetected cPAH compounds multiplied by their respective PEFs. Concentrations ranged from 1.3 to 57.7 µg toxic equivalent (TEQ)/kg dw, with a mean concentration of 7.1 µg TEQ/kg dw and an UCL95 of 8.9 µg TEQ/kg dw.[7] Using the UCL95 statistic, the background concentration for cPAHs is rounded to 9 µg TEQ/kg dw.

4.3.4.4 Natural Background for Dioxins/Furans in Sediment

The detection frequency for dioxins/furans in the EPA OSV *Bold* Survey was 100%, based on quantifying at least one congener in each sample (Table 4-4). The total TEQ of dioxins/furans (relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) in each sample was calculated by summing the concentrations of certain detected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective toxic equivalency factors (TEFs), along with half the reporting limits of undetected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective TEFs. Concentrations ranged from 0.2 to 11.6 ng TEQ/kg dw, with a mean of 1.4 ng TEQ/kg dw (Table 4-4) and an UCL95 of 1.6 ng TEQ/kg dw.[8] Using the UCL95 statistic, the background concentration for dioxins/furans is rounded to 2 ng TEQ/kg dw.

4.3.5 Role of Practical Quantitation Limits

Both CERCLA and MTCA allow consideration of PQLs when formulating PRGs to address circumstances in which a concentration determined to be protective cannot be reliably detected using state-of-the-art analytical instruments and methods. For example, if an RBTC is below the concentration at which a contaminant can be reliably quantified, then the PRG for that contaminant may default to the analytical PQL.

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7 The uncertainty associated with handling the undetected cPAH data is negligible. To determine how nondetects affected the overall statistics, a sensitivity analysis was run. For this analysis, the concentrations of the undetected cPAH compounds were set to zero. The concentrations of the individual detected cPAH compounds were multiplied by their respective PEFs and the products were summed. The results indicate a mean of 6.9 µg TEQ/kg dw and an UCL95 of 8.0 µg TEQ/kg dw.

8 The uncertainty associated with handling the undetected dioxin/furan data is negligible. To determine how nondetects affected the overall statistics, a sensitivity analysis was run. For this analysis, the concentrations of the undetected dioxin/furan congeners were set to zero. The concentrations of the individual detected dioxin/furan congeners were multiplied by their respective TEFs and the products were summed. The results indicate a mean of 1.2 ng TEQ/kg dw and an UCL95 of 1.5 ng TEQ/kg dw.

MTCA defines the PQL as:

> *…the lowest concentration that can be reliably measured within specified limits of precision, accuracy, representativeness, completeness, and comparability during routine laboratory operating conditions, using department approved methods* (WAC 173-340-200).

In simpler terms, the PQL is the minimum concentration for an analyte that can be reported with a high degree of certainty.

Tables 4-5 and 4-6 list the risk-driver specific PQLs developed for the RI sediment sampling programs and documented in the associated quality assurance project plans. These PQLs represent the lowest values that can be reliably quantified when the sample matrix (in this case, sediment) is free of interfering compounds that can reduce sensitivity and raise reporting limits. Also, these tables present the range of actual sample PQLs reported by the laboratories for the data in the RI database. These results reflect the range of what the laboratories were able to achieve given the composition of and matrix complexity associated with LDW sediment samples.

Analytical quantitation limits are generally not expected to exceed RBTCs, SQS, or natural background concentrations for samples of low matrix complexity. However, empirical evidence from the RI suggests that, on a case-by-case basis, matrix interferences have the potential to preclude quantification to concentrations below the PRGs (and ultimately the cleanup levels and standards) established for cleanup of LDW sediments.

4.4 Preliminary Remediation Goals

PRGs for sediment are derived from a comparison of ARARs, RBTCs, background concentrations, and PQLs. For each RAO and risk driver, the PRG is the higher value between the natural background concentration and the lowest RBTC.[9] PQLs were also considered and were not found to influence selection of the PRGs (i.e., all PRGs are above PQLs). The RAOs and PRGs are used in Section 6 of the FS to identify AOPCs and were considered in selecting the RALs. Section 9 compares estimated concentrations of risk drivers to PRGs as one measure of the effectiveness of the remedial alternatives.

Tables 4-7 and 4-8 summarize the analysis and selection of sediment PRGs for the risk-driver COCs. Table 4-7 focuses on the four human health risk drivers and the wildlife risk driver, and is subdivided to address the various spatial applications of the PRGs for each RAO. Table 4-8 contains the PRG analysis for the remaining SMS risk drivers (i.e.,

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9 SQS and CSL values for the 41 SMS risk-driver COCs are the RBTCs for protection of benthic organisms. Sediment RBTCs were calculated (see Section 3) for protection of ecological receptors (river otters) and humans.

the risk-driver COCs for RAO 3). PRGs were not developed for the other COCs identified in the RI. The potential for risk reduction for the other COCs following remedial action is evaluated in Section 9.

The PRGs identified in Tables 4-7 and 4-8 are derived from RBTCs, natural background, or SQS values. The PRGs are applied on either a point basis or an average basis over a given exposure area depending on the COC, exposure pathway, and receptor of concern. PRGs for RAOs 1, 2, and 4 are applied on a site-wide average basis that requires a sediment spatially-weighted average concentration (SWAC) over the applicable exposure area to be below the PRG. These SWACs have been calculated to evaluate and compare remedial alternatives; ultimate compliance for remedial actions will be based on the UCL95.

For RAO 1, the numerical PRG for total PCBs is natural background because the sediment RBTCs[10] are below natural background for the RME seafood consumption scenarios. RBTCs were not derived for dioxins/furans (see Section 3.2.4), but were presumed also to be below natural background levels for the RME seafood consumption scenarios. Therefore, natural background is the PRG for dioxins/furans for RAO 1. Arsenic and cPAH PRGs were not identified for the human health seafood consumption pathway (RAO 1). Excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. Based on data collected during the RI, there is no credible relationship between cPAH or arsenic concentrations in sediment and concentrations in clam tissue (Section 8 of the RI, Windward 2010). However, the development and evaluation of remedial alternatives in the latter sections of the FS discuss the need for future investigations of the sediment/clam tissue relationships for arsenic and cPAHs. Further, meeting the PRGs defined in Tables 4-7 and 4-8 should lead to reductions in sediment concentrations of arsenic and cPAHs (see discussion of RALs in Section 6). PRGs based on natural background are unlikely to be achieved by any of the remedial alternatives developed in this FS. This is partly because of COC concentrations in inflowing sediment from the Green/Duwamish River system, as predicted in the bed composition model used in this FS (see Section 5). In addition, the urban setting of the LDW will make it difficult to achieve natural background for PCBs and dioxins/furans. However, in accordance with MTCA, natural background concentrations were used in this FS for setting background-based PRGs.

For RAO 2, PRGs are based on the sediment RBTCs (1×10^{-6} or natural background, whichever is higher) developed for three exposure scenarios: netfishing, tribal clamming, and beach play. PRGs are applied on a spatially-weighted average basis over

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10 Sediment RBTCs were calculated only for the 1×10^{-4} risk threshold. The contribution of PCBs in water alone (even at concentrations similar to those in upstream water) was high enough to result in seafood consumption risks for Adult and Child Tribal RME and Asian and Pacific Islander RME scenarios exceeding the 1×10^{-6} and 1×10^{-5} excess cancer risk thresholds even in the absence of any contribution from sediment (Table 3-9).

a given exposure area (e.g., site-wide for netfishing). Except for arsenic, the PRGs for the RAO 2 risk drivers are based on their RBTCs. The arsenic PRG for RAO 2 is based on natural background, which may be difficult to achieve by any of the remedial alternatives developed in this FS, for the same reasons explained above for total PCBs and dioxins/furans for RAO 1.

For RAO 3, the SMS numerical criteria apply on a point basis (Table 4-6). As noted in Section 4.3.1, WAC 173-204-570(4) specifies that the site-specific cleanup standards shall be as close as practicable to the cleanup objective (the SQS) but in no case shall exceed the minimum cleanup level (the CSL). For this reason, the PRGs for RAO 3 in this FS are set to the SQS. However, where co-located toxicity test data are available, sediment toxicity results override the numerical criteria for RAO 3. (However, toxicity test results do not override PRGs for RAOs 1, 2, and 4 because toxicity test results are only relevant for an assessment of effects on benthic fauna, not on other ecological or human receptors.)

For RAO 4, the PRG for seafood consumption by ecological receptors is set to the sediment RBTC for river otter (hazard quotient less than 1).

Table 4-1 ARARs for the Lower Duwamish Waterway

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Sediment Quality	Sediment quality standards; cleanup screening levels		Sediment Management Standards (WAC 173-204)	The SMS are MTCA rules and an ARAR under CERCLA. Numerical standards for the protection of benthic marine invertebrates.
Fish Tissue Quality	Concentrations of contaminants in fish tissues	Food and Drug Administration Maximum Concentrations of Contaminants in Fish Tissue (49 CFR 10372-10442)		The Washington State Department of Health assesses the need for fish consumption advisories.
Surface Water Quality	Surface Water Quality Standards	Ambient Water Quality Criteria established under Section 304(a) of the Clean Water Act (33 USC 1251 et seq) http://www.epa.gov/ost/criteria/wqctable/	Surface Water Quality Standards (RCW 90-48; WAC 173-201A)	State surface water quality standards apply where the State has adopted, and EPA has approved, Water Quality Standards that are more stringent than Federal recommended Water Quality Criteria established under Section 304(a) of the Clean Water Act. Both chronic and acute standards, and marine and freshwater are used as appropriate.
Land Disposal of Waste	Disposal of materials containing PCBs	Toxic Substances Control Act (15 USC 2605; 40 CFR Part 761)		
	Hazardous waste	Resource Conservation and Recovery Act Land Disposal Restrictions (42 USC 7401-7642; 40 CFR 268)	Dangerous Waste Regulations Land Disposal Restrictions (RCW 70.105; WAC 173-303, 140-141)	
Waste Treatment Storage and Disposal	Disposal limitations	Resource Conservation and Recovery Act (42 USC 7401-7642;40 CFR 264 and 265)	Dangerous Waste Regulations (RCW 70.105; WAC 173-303)	
Noise	Maximum noise levels		Noise Control Act of 1974 (RCW 80.107; WAC 173-60)	
Groundwater	Groundwater quality	Safe Drinking Water Act MCLs and non-zero MCLGs (40 CFR 141)	RCW 43.20A.165 and WAC 173-290-310	For on-site potable water, if any.

Table 4-1 ARARs for the Lower Duwamish Waterway (continued)

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Dredge/Fill and Other In-water Construction Work	Discharge of dredged/fill material into navigable waters or wetlands	Clean Water Act (33 USC 401 et seq.; 33 USC 141; 33 USC 1251-1316; 40 CFR 230, 231, 404; 33 CFR 320-330) Rivers and Harbors Act (33 USC 401 et seq.)	Hydraulic Code Rules (RCW 75.20; WAC 220-110)	For in-water dredging, filling, or other construction.
	Open-water disposal of dredged sediments	Marine Protection, Research and Sanctuaries Act (33 USC 1401-1445; 40 CFR 227)	DMMP (RCW 79.90; WAC 332-30-166)	
Solid Waste Disposal	Requirements for solid waste handling management and disposal	Solid Waste Disposal Act (42 USC 215103259-6901-6991; 40 CFR 257-258)	Solid Waste Handling Standards (RCW 70.95; WAC 173-350)	
Discharge to Surface Water	Point source standards for new discharges to surface water	National Pollutant Discharge Elimination System (40 CFR 122, 125)	Discharge Permit Program (RCW 90.48; WAC 173-216, 222)	
Shoreline	Construction and development		Shoreline Management Act (RCW 90.58; WAC 173-16); King County and City of Seattle Shoreline Master Plans (KCC Title 25; SMC 23.60); City of Tukwila Shoreline Master Program (TMC 18.44)	For construction within 200 feet of the shoreline.
Floodplain Protection	Avoid adverse impacts, minimize potential harm	Executive Order 11988, Protection of Floodplains (40 CFR 6, Appendix A); FEMA National Flood Insurance Program Regulations (44 CFR 60.3Ld)(3)).		For in-water construction activities, including any dredge or fill operations. Includes local ordinances: KCC Title 9 and SMC 25.09.
Critical (or Sensitive) Area ARAR	Evaluate and mitigate impacts		Growth Management Act (RCW 36.70a); King County Critical Area Ordinance (KCC Title 21A.24); City of Seattle (SMC 25.09); City of Tukwila Sensitive Area Ordinance (TMC 18.45)	

Table 4-1 ARARs for the Lower Duwamish Waterway (continued)

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Habitat for Fish, Plants, or Birds ARAR	Evaluate and mitigate habitat impacts	Clean Water Act (Section 404 (b)(1)); U.S. Fish and Wildlife Mitigation Policy (44 CFR 7644); U.S. Fish and Wildlife Coordination Act (16 USC 661 et seq.); Migratory Bird Treaty Act (16 USC 703-712)		
Pretreatment Standards	National Pretreatment Standards		40 CFR Part 403; Metro District Wastewater Discharge Ordinance (KCC) to be considered (as is local requirement)	
Environmental Impact Review	State Environmental Policy Act		State Environmental Policy Act RCW 43.21C; WAC 197-11-790)	Applicable to MTCA cleanups. Because the LDW is under a joint EPA/Ecology Order, Ecology has determined that CERCLA requirements are the functional equivalent of NEPA and SEPA

Notes:

ARAR = applicable or relevant and appropriate requirement; CERCLA = Comprehensive, Environmental Response, Compensation, and Liability Act; CFR = Code of Federal Regulations; DMMP = Dredged Material Management Program; EPA = U.S. Environmental Protection Agency; FEMA = Federal Emergency Management Act; KCC = King County Code; MCL = maximum contaminant level; MCLG = maximum contaminant level goal; MTCA = Model Toxics Control Act; NEPA = National Environmental Policy Act; PCB = polychlorinated biphenyl; RCW = Revised Code of Washington; SEPA = State Environmental Policy Act; SMC = Seattle Municipal Code; SMS = Sediment Management Standards; TMC = Tukwila Municipal Code; USC = United States Code; WAC = Washington Administrative Code

Table 4-2 Other Legal Requirements for the Lower Duwamish Waterway

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Native American Graves and Sacred Sites	Evaluate and mitigate impacts to cultural resources	Native American Graves Protection and Repatriation Act (25 USC. 3001 et seq.; 43 CFR Pt. 10) and American Indian Religious Freedom Act (42 USC 1996 et seq.)		
Critical Habitat for Endangered Species	Conserve endangered or threatened species, consult with species listing agencies	Endangered Species Act of 1973 (16 USC 1531 et seq.; 50 CFR 200, 402); Magnuson-Stevens Fishery Conservation and Management Act (16 USC 1801-1884)	Endangered, threatened, and sensitive wildlife species classification (WAC 232-12-297)	Consult and obtain Biological Opinions.
Historic Sites or Structures	Requirement to avoid, minimize, or mitigate impacts to historic sites or structures	National Historic Preservation Act (16 USC 470f; 36 CFR Parts 60, 63, and 800)		Considered if implementation of the selected remedy involves removal of historic sites or structures.
Occupational Health and Safety	Requirements to provide for worker health and safety	Occupational Safety and Health Act (29 USC; 29 CFR)	Washington Industrial Safety and Health Act (RCW 49.17; WAC 296)	

Notes:

CFR = Code of Federal Regulations; RCW = Revised Code of Washington; USC = United States Code; WAC = Washington Administrative Code

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
Metals and Trace Elements									
Antimony	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	640
Arsenic	360	190	**69**	**36**	**340**	**150**	69	36	0.14[g,h]
Cadmium	3.7	1.0	42	9.3	**2.0**	**0.25**	**40**	**8.8**	n/a
Chromium (hexavalent)	**15**	**10**	**1,100**	**50**	16	11	1,100	50	n/a
Chromium (trivalent)	**550**	180	n/a	n/a	570	**74**	n/a	n/a	n/a
Copper	**17**	**11**	**4.8**	**3.1**	nc[j]	nc[j]	4.8	3.1	n/a
Lead	**65**	**2.5**	**210**	**8.1**	65	2.5	210	8.1	n/a
Mercury	2.1	**0.012**	**1.8**	**0.025**	**1.4**	0.77	1.8	0.94	0.15[i]
Nickel	1,400	160	**74**	**8.2**	**470**	**52**	74	8.2	4,600
Selenium	**20**	**5**	**290**	**71**	n/a	5	290	71	4,200
Silver	3.4	n/a	**1.9**	n/a	**3.2**	n/a	1.9	n/a	n/a
Thallium	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.47
Zinc	**110**	**100**	**90**	**81**	120	120	90	81	26,000
PAHs									
2-Chloronaphthalene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,600
Acenaphthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	990
Anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	40,000
Benzo(a)anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Benzo(a)pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Benzo(b)fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
***PAHs** (continued)*									
Benzo(k)fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Chrysene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Dibenzo(a,h)anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	140
Fluorene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5,300
Indeno(1,2,3-cd)pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4,000
Phthalates									
BEHP	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.2[g]
Butyl benzyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,900
Diethyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	44,000
Dimethyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,100,000
Di-n-butyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4,500
SVOCs									
1,2,4-Trichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	70
1,2-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,300
1,2-Diphenylhydrazine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.2[g]
1,3-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	960
1,4-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	190
2,4,6-Trichlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.4[g]
2,4-Dichlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	290

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
SVOCs (continued)									
2,4-Dimethylphenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	850
2,4-Dinitrophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5,300
2,4-Dinitrotoluene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.4[g]
2-Chlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	150
3,3'-Dichlorobenzidine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.028[g]
4,6-Dinitro-o-cresol (2-methyl-4,6-dinitrophenol)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	280
Benzidine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.0002
Bis(2-chloroethyl)ether	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.53[g]
Bis(2-chloroisopropyl)ether	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	65,000
Hexachlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00029[g]
Hexachlorobutadiene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	18[g]
Hexachlorocyclopentadiene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,100
Hexachloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.3[g]
Isophorone	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	960[g] (**600**[i])
Nitrobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	690
n-Nitrosodimethylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3[g]
n-Nitroso-di-n-propylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.51[g]
n-Nitrosodiphenylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	6[g]
Pentachlorophenol	20[k]	**13**[k]	**13**	**7.9**	**19**[k]	15[k]	13	7.9	3[g]
Phenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	860,000

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
PCBs									
PCBs	**2**	**0.014**	**10**	**0.03**	n/a	0.014	n/a	0.03	0.000064[g]
Pesticides									
4,4'-DDD	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00031[g]
4,4'-DDE	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00022[g]
4,4'-DDT	**1.1**	**0.001**	**0.13**	**0.001**	1.1	0.001	0.13	0.001	0.00022[g]
Aldrin	n/a	n/a	n/a	n/a	**3.0**	n/a	**1.3**	n/a	0.000050[g]
Dieldrin	n/a	n/a	n/a	n/a	**0.24**	**0.056**	**0.71**	**0.0019**	0.000054[g]
Aldrin/dieldrin (sum)[l]	**2.5**	**0.0019**	**0.71**	**0.0019**	n/a	n/a	n/a	n/a	n/a
alpha-BHC	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.0049[g]
beta-BHC	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.017[g]
gamma-BHC (Lindane)	2.0	**0.08**	**0.16**	n/a	**0.95**	n/a	0.16	n/a	1.8
alpha-Endosulfan	**0.22[m]**	**0.056[m]**	**0.034[m]**	**0.0087[m]**	0.22	0.056	0.034	0.0087	89 (**2**[i])
beta-Endosulfan	**0.22[m]**	**0.056[m]**	**0.034[m]**	**0.0087[m]**	0.22	0.056	0.034	0.0087	89 (**2**[i])
Endosulfan sulfate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	89 (**2**[i])
Endrin	0.18	**0.0023**	**0.037**	**0.0023**	**0.086**	0.036	0.037	0.0023	0.06
Endrin aldehyde	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.3
Heptachlor	**0.52**	**0.0038**	**0.053**	**0.0036**	0.52	0.0038	0.053	0.0036	0.000079[g]
Heptachlor epoxide	n/a	n/a	n/a	n/a	**0.52**	**0.0038**	**0.053**	**0.0036**	0.000039[g]
Toxaphene	**0.73**	**0.0002**	**0.21**	**0.0002**	0.73	0.0002	0.21	0.0002	0.00028[g]
Chlordane	**2.4**	**0.0043**	**0.09**	**0.004**	2.4	0.0043	0.09	0.004	0.00081[g]

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
VOCs									
1,1,2,2-Tetrachloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4[g]
1,1,2-Trichloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	16[g]
1,1-Dichloroethene (1,1-dichloroethylene)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	7,100
1,2-Dichloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	37
1,2-Dichloropropane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	15[g]
Acrolein	n/a	n/a	n/a	n/a	**3**	**3**	n/a	n/a	9
Acrylonitrile	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.25[g]
Benzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	51[g]
Bromodichloromethane (dichlorobromomethane)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	17[g]
Bromoform	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	140[g]
Bromomethane (methyl bromide)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,500
Carbon tetrachloride	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1.6[g]
Chlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,600
Chloroform	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	470
Dibromochloromethane (chlorobromomethane)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	13[g]
Dichloromethane (methylene chloride)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	590[g]
Ethylbenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2,100
Tetrachloroethene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.3[g]
Toluene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	15,000
trans-1,2-Dichloroethene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	10,000
Trichloroethene (trichloroethylene)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	30g

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
***VOCs** (continued)*									
Vinyl chloride	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.4[g]
Dioxins and Furans									
2,3,7,8 TCDD	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5.1E-09[g]

Notes:

1. Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. Existing site-specific data or site-specific data that may be collected can be used to adjust values rather than using a default hardness value of 100 mg/L. Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same). The lower of the human health criteria (when multiple criteria are available) is also bolded.

a. Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

b. Standards are from the national recommended EPA AWQC (except where noted). National recommended EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

c. Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds.

d. Human health WQC are based on dissolved concentrations for all contaminants.

e. Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time.

f. Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

g. Human health WQC are based on 1 x 10^{-6} excess cancer risk for carcinogenic contaminants.

h. Criterion represents the inorganic fraction of arsenic.

i. Standards are from 40 CFR 131.36 (NTR), as referenced in WAC 173-201A-240. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

j. Criteria based on the biotic ligand model. The acute and chronic biotic ligand model-based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

k. The freshwater aquatic life WQC for pentachlorophenol is pH-dependent; a pH of 7.8 was assumed, which is the default assumption.

l. Aldrin is metabolically converted to dieldrin. Therefore, the sum of aldrin and dieldrin concentrations is compared with the dieldrin criteria.

m. Standards are for endosulfan.

AWQC = ambient water quality criteria; BEHP = Bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; DDD – dichlorodiphenyldichloroethane; DDE = dichlorodiphenyldichloroethylene; DDT = dichlorodiphenyltrichloroethane; DOC = dissolved organic carbon; MCL= maximum contaminant level; µg/L = microgram per liter; n/a = not available; nc = not calculated; NTR = National Toxics Rule; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; SVOC = semivolatile organic compound; TCDD = tetrachlorodibenzo-*p*-dioxin; VOC = volatile organic compound; WAC = Washington Administrative Code; WQC = water quality criteria

Table 4-4 Summary of Arsenic, Total PCB, cPAH, and Dioxin/Furan Datasets for Natural Background

Human Health Risk-Driver COC	Detection Frequency	Concentration							UCL Type
		Minimum	Maximum	Mean	Median	90th Percentile[a]	UCL95	UCL95 (rounded value)[b]	
Arsenic (mg/kg dw)	70/70	1.1	21	6.5	5.9	11.0	7.3	7	Approximate Gamma UCL95
Total PCBs as Aroclors (μg/kg dw)	6/70	2.1	31	11	4.4	8.0	6.5	7	KM (Percentile Bootstrap) UCL95
Total PCBs as Congeners (μg/kg dw)	70/70	0.01	10.6	1.2	0.6	2.7	1.5	2	Approximate Gamma UCL95
cPAHs (μg TEQ/kg dw)	61/70	1.3	57.7	7.1	4.5	14.7	8.9	9	KM (BCA) UCL95
Dioxins/Furans (ng TEQ/kg dw)	70/70	0.2	11.6	1.4	1.0	2.2	1.6	2	H-UCL95

Notes:

1. Dataset collected throughout Puget Sound by EPA in 2008 and referred to as the EPA OSV *Bold* Survey.
2. Summary statistics and UCL were calculated using ProUCL 4.00.04 statistical software.
3. Total PCBs were calculated by summing the concentrations of detected PCB Aroclors or detected PCB congeners. In cases where no PCB Aroclors were detected, the highest reporting limit for an individual PCB Aroclor was used as the value of total PCBs. Total cPAHs were calculated by summing the concentrations of all detected cPAH compounds multiplied by their respective potency equivalency factors (PEFs), along with half the reporting limits of any undetected cPAH compounds multiplied by their respective PEFs.
4. The total toxic equivalent (TEQ) of dioxins/furans (relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) was calculated by summing the concentrations of detected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective toxic equivalency factors (TEFs), along with half the reporting limits of undetected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective TEFs.

a. Using MTCAStat software, instead of EPA's ProUCL, risk drivers may be slightly higher.
b. Rounded values of UCL95s are used as natural background in this FS.

BCA = bias-corrected accelerated; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; FS = feasibility study; H-UCL = UCL based on Land's H-statistic; kg = kilogram; KM = Kaplan Meier method for calculating a UCL; µg = micrograms; mg = milligram; ng = nanogram; PCB = polychlorinated biphenyl; PEF = potency equivalency factor; TEF = toxic equivalency factor; TEQ = toxic equivalent; UCL95 =95% upper confidence limit on the mean

Table 4-5 Practical Quantitation Limits, Natural Background, and Risk-Based Threshold Concentrations for the Human Health and Ecological Risk-Driver COCs

Human Health& Ecological Risk-Driver COC	Practical Quantitation Limits			Natural Background[b]	Risk-Based Threshold Concentrations				
	EPA Method	RI QAPP RLs[a]	Range of RLs from undetected values		Spatial Scale of Exposure[c]	RAO 1: Human Seafood Consumption	RAO 2: Human Direct Contact	RAO 3: Benthic Organisms	RAO 4: Ecological (River Otter)
Total PCBs (µg/kg dw)	8082	4[d]	0.56 – 50[e]	2	Site-wide	nc (7 - 185)[f]	1,300	n/a	(128 - 159)[g]
					Tribal Clamming	n/a	500	n/a	n/a
					Beach Play	n/a	1,700	n/a	n/a
					Point	n/a	n/a	12/65[h]	n/a
Arsenic (mg/kg dw)	6010B	5	3.1 – 31	7	Site-wide	n/c[i]	3.7	n/a	n/a
					Tribal Clamming	n/a	1.3	n/a	n/a
					Beach Play	n/a	2.8	n/a	n/a
					Point	n/a	n/a	57/93[h]	n/a
cPAH (µg TEQ/kg dw)	8270D	6.3 – 20 µg/kg[j]	9.0 – 130 µg/kg[j]	9	Site-wide	n/c[i]	380	n/a	n/a
					Tribal Clamming	n/a	150	n/a	n/a
					Beach Play	n/a	90	n/a	n/a
					Point	n/a	n/a	n/a[k]	n/a
Dioxins/Furans (ng TEQ/kg dw)	1613B	1 – 10 ng/kg[l]	0.12 – 7.7 ng/kg[l]	2	Site-wide	nc (bg)	37	n/a	n/a
					Tribal Clamming	n/a	13	n/a	n/a
					Beach Play	n/a	28	n/a	n/a
					Point	n/a	n/a	n/a	n/a

Notes:

a. Reporting limits from Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006) in dry weight units on untransformed data.

b. UCL95 values are calculated from the EPA OSV *Bold* Survey dataset using ProUCL.

Table 4-5 Practical Quantitation Limits, Natural Background, and Risk-Based Threshold Concentrations for the Human Health and Ecological Risk-Driver COCs (continued)

c. The spatial scale of site-wide exposure is RAO-specific: (seafood consumption for RAO 1 and RAO 4; netfishing for RAO 2).
d. PCB RLs (as Aroclors) reported in Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006). RLs for individual PCB congeners are much lower (0.5 to 1 ng/kg).
e. Range of RLs for undetected values were queried from the RI database and represent RLs for undetected total PCBs. For samples in which none of the individual Aroclors are detected, the total PCB concentration value is represented as the highest RL of an individual Aroclor, and assigned a U-qualifier, indicating no detected concentrations. Individual undetected Aroclors were not reported because they are not included in the calculation of total PCBs when other Aroclors are detected in the sample.
f. RBTC <1 µg/kg dw at risk levels of 10^{-5} and 10^{-6}, and RBTC range of 7 to 185 µg/kg dw for the three RME seafood consumption scenarios at the 10^{-4} risk level.
g. Values represent best-fit estimates for two different dietary scenarios as reported in the RI (Windward 2010).
h. Total PCB concentration units are mg/kg oc and the two values are SQS/CSL. Arsenic concentration units are mg/kg dw and the two values are SQS/CSL.
i. Arsenic and cPAH PRGs are undefined for the human health seafood consumption pathway (RAO 1). Seafood consumption excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. There is no credible relationship, based on site data, relating cPAH or arsenic concentrations in sediment to concentrations in clam tissue (Section 8 of the RI, Windward 2010). Section 8 of the FS discusses the need for future investigations of the sediment/tissue relationships for arsenic and cPAHs.
j. cPAH TEQ RLs are based on those for the individual PAH compounds used in the TEQ calculation. All individual PAH compounds used in the cPAH calculation have an RL of 20 except for dibenzo[a,h]anthracene, which has an RL of 6.3. RLs reported for undetected values are based on calculated cPAHs and can be found in Table A-1, of Round 3 Surface Sediment QAPP Addendum (Windward 2006).
k. Low- and high-molecular weight PAHs are addressed by the SMS. Criteria are set for both groupings and for individual PAH compounds.
l. Dioxin/furan TEQ RLs are based on those for the individual congeners used in the TEQ calculation. RLs for undetected values are in Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006).

bg = natural background; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; CSL = cleanup screening level; dw = dry weight; EPA = U.S. Environmental Protection Agency; LDW = Lower Duwamish Waterway; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; nc = no value calculated; nc (bg) = not calculated, RBTC value expected to be below background; ng/kg = nanograms per kilogram; oc = organic carbon; PCB = polychlorinated biphenyl; QAPP = quality assurance project plan; RAO = remedial action objective; RBTC = risk-based threshold concentration; RI = remedial investigation; RL = reporting limit; RME = reasonable maximum exposure; SQS = sediment quality standard; TEQ = toxic equivalent; UCL95 = 95% upper confidence limit on the mean

Table 4-6 Practical Quantitation Limits and Risk-Based Threshold Concentrations for Benthic Risk-Driver COCs

Benthic Risk-Driver COC	Practical Quantitation Limits			Risk-Based Threshold Concentrations RAO 3: Sediment Management Standards[c]		
	EPA Method	RI QAPP RLs[a]	Range of RLs from Undetected Values[b]	Spatial Scale of Exposure	Sediment Quality Standard (SQS)	Cleanup Screening Level (CSL)
SMS Metals	(mg/kg dw)				(mg/kg dw)	
Arsenic	6010B	5	3.1 – 31	Point	57	93
Cadmium	6010B	0.2	0.4 – 2.5		5.1	6.7
Chromium	6010B	0.5	0.25 – 1		260	270
Copper	6010B	0.2	0.5 – 1		390	390
Lead	6010B	2	1.25 – 8		450	530
Mercury	7471A	0.05	0.02 – 0.1		0.41	0.59
Silver	6010B	0.3	0.046 – 5		6.1	6.1
Zinc	6010B	2	0.5 – 2		410	960
Dry Weight Basis SMS Organic Compounds	(µg/kg dw)				(µg/kg dw)	
4-methylphenol	8270D	6.7	8.6 – 2,000	Point	670	670
2,4-dimethylphenol	8270D	6.7	6.0 – 2,000		29	29
Benzoic acid	8270-SIM	20	13 – 3,000		650	650
Benzyl alcohol	8270-SIM	2	9.2 – 690		57	73
Pentachlorophenol	8270-SIM	10	7.6 – 4,900		360	690
Phenol	8270D	20	7.3 – 790		420	1,200
oc-normalized SMS Organic Compounds[d]	(µg/kg dw)				(mg/kg oc)	
Total PCBs	8082	4	0.56 – 50	Point	12	65
Acenaphthene	8270D	20	1.8 – 2,000		16	57
Anthracene	8270D	20	13 – 2,000		220	1,200
Benzo(a)pyrene	8270D	20	6.4 – 350		99	210
Benz(a)anthracene	8270D	20	6.4 – 200		110	270
Total benzofluoranthenes	8270D	20	n/a		230	450
Benzo(g,h,i)perylene	8270D	20	13 – 2,000		31	78
Chrysene	8270D	20	18 – 170		110	460
Dibenz(a,h)anthracene	8270D	6.3	1.0 – 2,000		12	33
Indeno(1,2,3-cd)pyrene	8270D	20	6.4 – 1,600		34	88
Fluoranthene	8270D	20	19 – 340		160	1,200
Fluorene	8270D	20	1.8 – 2,000		23	79
Naphthalene	8270D	20	1.0 – 2,000		99	170
Phenanthrene	8270D	20	18 – 200		100	480
Pyrene	8270D	20	18 – 170		1,000	1,400
HPAH	8270D	n/a	n/a		960	5,300
LPAH	8270D	n/a	n/a		370	780
Bis(2-ethylhexyl)phthalate	8270D	20	15 – 1,500		47	78
Butyl benzyl phthalate	8270-SIM	2	1.8 – 2,000		4.9	64
Dimethyl phthalate	8270D	20	1.8 – 2,000		53	53
1,2-dichlorobenzene	8270-SIM	2	0.4 – 2,000		2.3	2.3
1,4-dichlorobenzene	8270-SIM	2	0.2 – 2,000		3.1	9
1,2,4-trichlorobenzene	8270-SIM	2	0.4 – 2,000		0.81	1.8
2-methylnaphthalene	8270D	20	1.0 – 2,000		38	64
Dibenzofuran	8270D	20	1.7 – 2,000		15	58
Hexachlorobenzene	8081A	1.0	0.11 – 2,000		0.38	2.3
n-Nitrosodiphenylamine	8270-SIM	10	1.8 – 2,000		11	11

Notes:
1. All QAPP-based RLs are below the SQS except for n-nitrosodiphenylamine.
2. Background concentrations were not calculated for the COCs listed in this table because benthic RBTCs are not below natural background.

a. Reporting limits from Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006) in dry weight units. Low level reporting limits for contaminants analyzed by EPA method 8279-SIM from Analytical Resources, Incorporated (www.arilabs.com).
b. Range of RLs reported in Remedial Investigation dataset in instances where constituent(s) were not detected. All RLs shown in dry weight units.
c. Under the SMS, sediment cleanup standards are established on a site-specific basis within an allowable range of contamination. The SQS and CSL define this range. However, the final cleanup level will be set in consideration of the net environmental effects, cost, and engineering feasibility of different cleanup alternatives (WAC 173-204-570(4)).
d. The tabulated SMS values are oc-normalized and are screened against the RLs using the underlying apparent effects threshold concentrations, which are dry weight-based.

COC = contaminant of concern; CSL = cleanup screening level; dw = dry weight; EPA = U.S. Environmental Protection Agency; HPAH = high-molecular-weight polycyclic aromatic hydrocarbon; LDW = Lower Duwamish Waterway; LPAH = low-molecular-weight polycyclic aromatic hydrocarbon; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; oc = organic carbon; PQL = practical quantitation limit; QAPP = quality assurance project plan; RAO = remedial action objective; RL = reporting limit; SIM = selected ion monitoring; SMS = Sediment Management Standards; SQS = sediment quality standard

Table 4-7 Preliminary Remediation Goals for Total PCBs, Arsenic, cPAHs, and Dioxins/Furans for Human Health and Ecological Risk-Driver COCs

Risk-Driver COC	Preliminary Remediation Goals					
	RAO 1: Human Seafood Consumption	RAO 2: Human Direct Contact	RAO 4: Ecological (River Otter)	Basis	Statistical Metric for Application	Spatial Scale of PRG Application
Total PCBs (µg/kg dw)	2	1,300	128-159	bg (RAO 1) RBTC (RAO 2) RBTC (RAO 4)	SWAC	Site-wide
	n/a	500	n/a	RBTC	SWAC	Clamming Areas
	n/a	1,700	n/a	RBTC	SWAC	Individual Beaches
Arsenic (mg/kg dw)	n/a	7[a]	n/a	bg	SWAC	Site-wide
	n/a	7	n/a	bg	SWAC	Clamming Areas
	n/a	7	n/a	bg	SWAC	Individual Beaches
cPAH (µg TEQ/kg dw)	n/a	380[a]	n/a	RBTC	SWAC	Site-wide
	n/a	150	n/a	RBTC	SWAC	Clamming Areas
	n/a	90	n/a	RBTC	SWAC	Individual Beaches
Dioxins/Furans (ng TEQ/kg dw)	2[b]	37	n/a	bg (RAO 1) RBTC (RAO 2)	SWAC	Site-wide
	n/a	13	n/a	RBTC	SWAC	Clamming Areas
	n/a	28	n/a	RBTC	SWAC	Individual Beaches

Notes:

1. The PRGs for RAO 3 are shown separately in Table 4-8. The PRGs were developed for the 41 COCs that have been identified as benthic risk drivers for RAO 3.

a. Arsenic and cPAH PRGs are undefined for the human health seafood consumption pathway (RAO 1). Seafood consumption excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. There is no credible relationship, based on site data, relating cPAH or arsenic concentrations in sediment to concentrations in clam tissue (Section 8 of the RI, Windward 2010). Section 8 of the FS discusses the need for future investigations of the sediment/tissue relationships for arsenic and cPAHs.

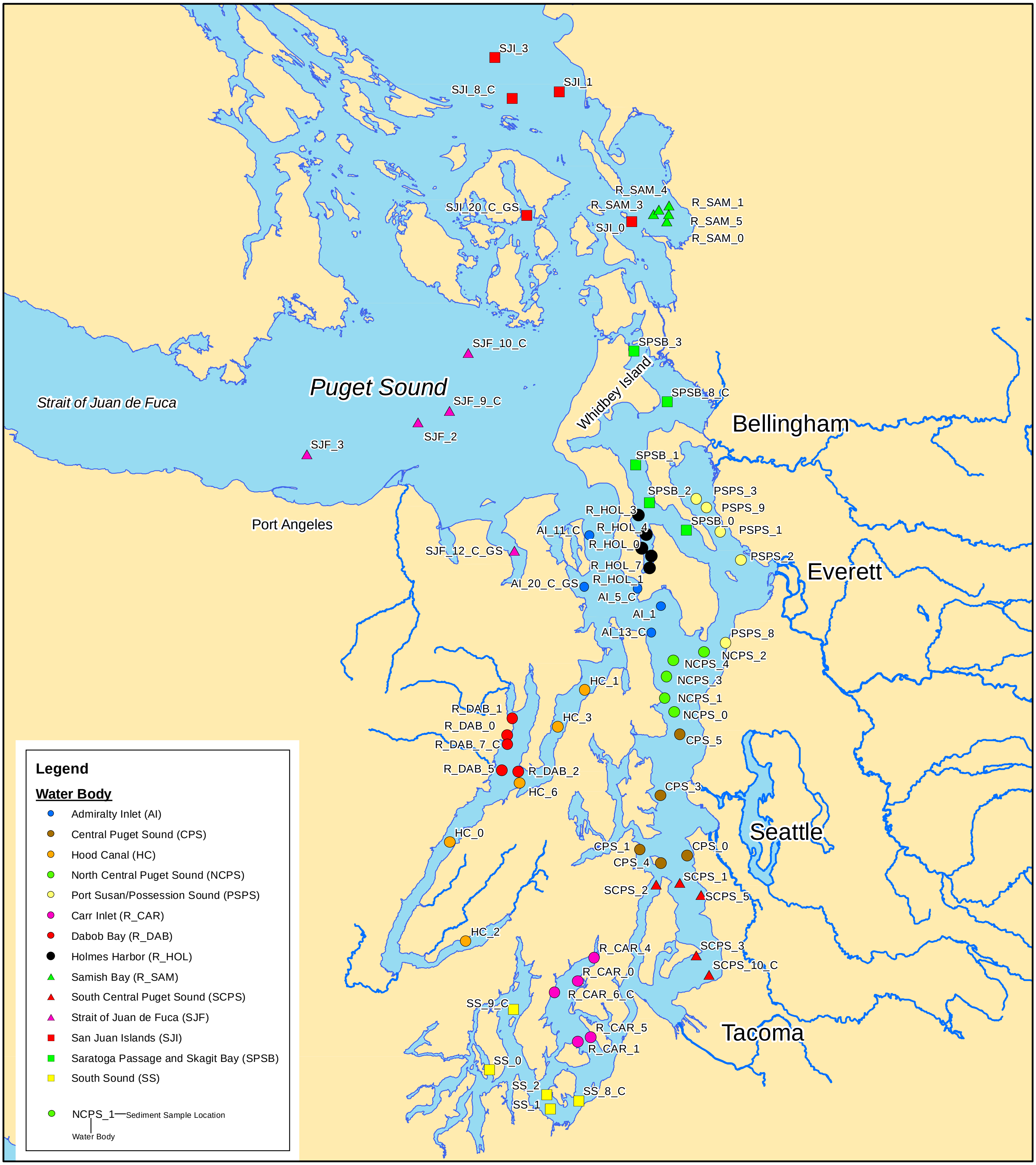
b. Although risks associated with consumption of dioxins/furans in resident seafood were not quantitatively assessed in the baseline HHRA, those risks were assumed to be unacceptable, and the associated sediment concentration was assumed to be below natural background concentrations.

bg = natural background; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; ng/kg = nanograms per kilogram; oc = organic carbon; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAO = remedial action objective; RBTC = risk-based threshold concentration; SMS = Sediment Management Standards; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent

Table 4-8 Preliminary Remediation Goals for Benthic Risk-Driver COCs

Benthic Risk-Driver COC	Preliminary Remediation Goals for RAO 3			
	Value	Basis	Statistical Metric	Spatial Scale of PRG Application
SMS metals	**(mg/kg dw)**			
Arsenic	57	SQS	Point Concentration or Toxicity Test Override	Point
Cadmium	5.1	SQS		
Chromium	260	SQS		
Copper	390	SQS		
Lead	450	SQS		
Mercury	0.41	SQS		
Silver	6.1	SQS		
Zinc	410	SQS		
Dry Weight Basis SMS Organic Compounds	**(µg/kg dw)**			
4-methylphenol	670	SQS	Point Concentration or Toxicity Test Override	Point
2,4-dimethylphenol	29	SQS		
Benzoic acid	650	SQS		
Benzyl alcohol	57	SQS		
Pentachlorophenol	360	SQS		
Phenol	420	SQS		
oc-normalized SMS Organic Compounds[d]	**(mg/kg oc)**			
Total PCBs	12	SQS	Point Concentration or Toxicity Test Override	Point
Acenaphthene	16	SQS		
Anthracene	220	SQS		
Benzo(a)pyrene	99	SQS		
Benz(a)anthracene	110	SQS		
Total benzofluoranthenes	230	SQS		
Benzo(g,h,i)perylene	31	SQS		
Chrysene	110	SQS		
Dibenz(a,h)anthracene	12	SQS		
Indeno(1,2,3-cd)pyrene	34	SQS		
Fluoranthene	160	SQS		
Fluorene	23	SQS		
Naphthalene	99	SQS		
Phenanthrene	100	SQS		
Pyrene	1,000	SQS		
HPAH	960	SQS		
LPAH	370	SQS		
Bis(2-ethylhexyl)phthalate	47	SQS		
Butyl benzyl phthalate	4.9	SQS		
Dimethyl phthalate	53	SQS		
1,2-dichlorobenzene	2.3	SQS		
1,4-dichlorobenzene	3.1	SQS		
1,2,4-trichlorobenzene	0.81	SQS		
2-methylnaphthalene	38	SQS		
Dibenzofuran	15	SQS		
Hexachlorobenzene	0.38	SQS		
n-Nitrosodiphenylamine	11	SQS		

COC = contaminant of concern; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; oc = organic carbon; PRG = preliminary remediation goal; SMS = Sediment Management Standards; SQS = sediment quality standard



Notes:
1. Surface sediment samples collected in 2008 by EPA aboard the Ocean Survey Vessel (OSV) *Bold*.

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Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.36			**2008 Puget Sound *OSV Bold* Survey Locations**
DATE: 10/31/12	DWRN:mvi/SEA	Revision: 0	**FIGURE 4-1**

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5 Evaluation of Sediment Movement and Recovery Potential

This section presents a summary of the sediment transport and related contaminant transport modeling, as well as empirical data, and develops an understanding of potential natural recovery based on the models and data. The overall modeling approaches are presented in this section. The sediment transport model (STM) is presented in the STM Report (QEA 2008). The sediment-related contaminant transport modeling is presented in Appendix C, Part 1. The data evaluations supporting the natural recovery analysis are presented in Appendix F.

One of the U.S. Environmental Protection Agency (EPA) guiding principles for managing sediments is to develop a conceptual site model (CSM) that considers sediment stability and evaluates the assumptions and uncertainties associated with site data and models (EPA 2005b). Model results are used to inform the CSM. A well-developed and calibrated model can assist in adaptively managing a site and adjusting or refining site predictions to the actual response of a system after various remedial actions and source control measures have either been completed or are under way. Sediment experts and site managers all recognize the unique challenges and difficulties in understanding the natural forces and man-made events that affect sediment movement, stability, and recovery potential, and that some uncertainty will always be present. Consistent with EPA's guiding principles, in this feasibility study (FS), the STM, the bed composition model (BCM), and the potential for sediments to be exposed at the surface are used to predict responses after applying the different remedial actions.

The hydrodynamic and sediment transport CSM for the Lower Duwamish Waterway (LDW), as described in Section 2, is largely influenced by the reduction and control of inflows through diversion of the rivers that historically flowed into the Green River and ongoing water management practices at the Howard Hanson Dam. Peak inflows have been greatly reduced, and the LDW has been widened and deepened to permit navigation. The increased cross-section acts as a natural sediment trap for incoming coarse-grained sediment. The STM simulates natural transport and bed evolution processes in this highly modified riverine/estuarine system. In addition, some effects of ships transiting the navigation channel and berthing areas under routine operating procedures are implicitly included in the STM/BCM by calibration to measured sedimentation rates. The goals of the LDW-wide modeling efforts for this FS are:

- Illustrate how contaminant concentrations vary spatially in the LDW via sediment movement, scour, and deposition processes and empirical trends.
- Predict contaminant fate and recovery potential for risk drivers over periods of time (e.g., 10 years) via the primary mechanisms of burial and source control.

- Demonstrate that model predictions and empirical measurements are comparable. Both the modeling results and empirical data have some measure of uncertainty; therefore, multiple lines of evidence are evaluated collectively to examine and reduce these uncertainties and to refine the CSM (EPA 2005b).
- Consider how navigation activities may disrupt natural recovery processes and affect BCM recovery predictions.

The four modeling process steps to address these goals are described below.

First, the STM results are used to look at general trends in an analysis of net sedimentation rates and to review agreement with the CSM with respect to the depositional environment in the absence of deep scour events. This is accomplished by comparing the estimated net sedimentation trends to empirical data. Empirical data include subsurface cores used to determine historical trends in net sedimentation rates and surface sediment locations that have been resampled over time. The STM is used to evaluate sediment movement as it relates to potential remedial areas and alternatives. This step includes an evaluation of net sedimentation rates, sediment transport into early action areas (EAAs), and other specific model runs to better understand sediment dynamics in the system (Section 5.1).

Second, the BCM, which takes output directly from the physical STM and applies contaminant concentrations to modeled sediment particles, was developed to predict future contaminant concentrations in surface sediments, and therefore recovery potential. The BCM is based on STM output, and BCM predictions assume that contaminant concentrations will be influenced only by sedimentation and resuspension due to natural processes. The BCM and associated empirical evidence are used in the FS to provide a predictive tool for evaluating whether contaminant concentrations in the surface layer/biologically active zone will decrease through natural recovery processes. The STM, BCM, and empirical evidence are used to evaluate whether the sediment bed is stable (i.e., not subject to significant scour, erosion, and transport) and whether the sedimentation rate is sufficient for burial of contaminated sediments to occur in the absence of navigation-caused disturbances. If these conditions are met in a given location, then monitored natural recovery (MNR) or enhanced natural recovery (ENR) may be appropriate response actions for evaluation in one or more remedial alternatives. Conversely, if natural processes are not effectively reducing concentrations of contaminants of concern (COCs) in surface sediments, then capping or dredging may be more appropriate choices (Section 5.2).

STM/BCM predictions of net sedimentation over much of the LDW are consistent with the CSM when ongoing navigation activities are assumed to constitute a minor influence on surface sediment contaminant concentrations (e.g., propeller wash does not expose, resuspend, and mix deeper subsurface contamination with surface sediment).

Third, smaller scale areas are analyzed to evaluate local recovery potential and assess whether empirical data and predictive models agree. MNR is a potential remedy that relies on ongoing, naturally-occurring processes (such as sediment deposition, mixing, and burial) to reduce COC concentrations in surface sediment. Several lines of evidence (e.g., isotope cores, sediment transport analysis, contaminant trends analysis, evaluation of erosion potential) are combined to assess whether contaminated subsurface sediments are stable, if they are effectively isolated, and whether surface sediment contaminant concentrations are predicted to decrease over time. The STM and BCM do not incorporate disturbances to bed sediments from propeller wash; therefore, bathymetric imaging data were used to identify these areas. These lines of evidence are used in the FS both when configuring remedial alternatives and when evaluating the long-term effectiveness of remedial alternatives (Sections 5.3 and 5.4). Local recovery potential under routine navigation procedures is discussed in Section 5.3.2.7.

Fourth, this FS considers the potential influence of contaminated subsurface sediments that may be exposed at the surface. Some effects of ships transiting the navigation channel and berthing areas under routine operating procedures are included in the STM/BCM by calibration to measured sedimentation rates. However, additional navigation and construction-related activities, as well as natural events, may result in sediment bed disturbance causing increased surface sediment contaminant concentrations that are not addressed by the STM/BCM. The STM and BCM were designed to consider only external and surface sediment sources of contamination to the LDW system. They were not set up to model deeper disturbance events, so this FS conducted a separate sensitivity analysis of deep sediment disturbance to consider the potential effects of such disturbance events on STM/BCM-predicted spatially-weighted average concentrations (SWACs; see Section 5.2.3).

This section of the FS focuses on details related to the six modeling goals:

- Providing an overview of the physical CSM and the STM relative to recovery.
- Discussing briefly the multiple lines of empirical evidence (i.e., sediment core trends, surface sediment sample trends at resampled stations, and physical features) that validate the STM and identify trends not accounted for by the predictive model.
- Developing a predictive recovery model (i.e., the BCM) and inputs to the BCM.
- Developing methods to either account for or assess the potential for scour to affect sedimentation and recovery in two ways: shallow mixing from routine

vessel operating procedures through resuspension and mixing; and episodic deep disturbances that result in subsurface contaminated sediments being exposed at the surface layer (thereby affecting the SWAC).

- Performing additional STM scenario runs to help answer FS-specific questions related to sediment movement and MNR and ENR recovery potential.
- Defining uncertainties of the STM model, including a brief overview of how it affects uncertainties in the fate and transport processes for risk drivers.

Potential application of MNR and ENR and general response actions are described in Section 7, Identification and Screening of Remedial Technologies. Additional STM runs are described in Appendix C. Empirical trends for individual areas of potential concern (AOPCs) are presented in Section 6.

5.1 Sediment Transport Modeling

Modeling of particle movement in and out of the LDW and sediment transport within the LDW was undertaken during the remedial investigation (RI) to better understand the CSM and support various FS elements.[1] The site-wide STM, which simulates the natural sediment resuspension and sedimentation processes active to varying degrees within the LDW (with the caveats noted above), has shown that the LDW is net depositional on a site-wide scale and is divided into Reaches 1, 2, and 3 based on hydrodynamic characteristics and geomorphology (see Section 2 for more details regarding the CSM). Model development and calibration are detailed in the *Final Sediment Transport Analysis Report* (STAR; Windward and QEA 2008) and the *Final Sediment Transport Modeling Report* (QEA 2008). This section reviews the resulting general trends in a site-wide analysis (Section 5.1.1) and evaluates the STM's ability, when combined with the BCM, to predict contaminant trends. This is accomplished by comparing the predicted trends to empirical data (Section 5.4).

5.1.1 Composition and Sources of Sediment Loads

The STM estimated the movement of sediment from three sources over time into and through the LDW:

- Sediment from the upstream Green/Duwamish River system

[1] The STM tracks particle movement, but it does not model contaminant transport processes or mechanical transport processes such as the effect of vessel traffic or waves on net sedimentation rates. The effect of vessel traffic was analyzed separately for moving and maneuvering tugs. The analysis of moving tugs is presented in the *Final Sediment Transport Analysis Report* (STAR; Windward and QEA 2008) and the effect of maneuvering tugs is summarized in Section 5.3.1 and in Appendix C, Part 7 of this FS.

- Sediment from lateral sources (i.e., storm drains, streams, and combined sewer overflows [CSOs]) that discharge to the LDW
- Surface sediment existing in the LDW bed at the onset of the model period.

The STM modeled both the transport of total suspended solids (TSS) and bed load. The transport of TSS is the movement of suspended particles in the water column. Bed load transport is the movement of sand and gravel in a thin layer (about 1 millimeter [mm] to 1 centimeter [cm] in thickness) located along the surface of the sediment bed. The Green/Duwamish River is the predominant source of sediment to the LDW. Figures 5-1a and 5-1b show that surface sediment (0 to 10 cm) in over 90% of the LDW model area will be comprised of over 50% upstream solids at the end of the 10-year model simulation and over 75% upstream solids at the end of the 30-year simulation. The STM quantified sediment loading from this upstream source using a flow-rating curve for the Green/Duwamish River based on discharge data gathered from 1960 to 1980 and from 1996 to 1998. The grain size characteristics of the in-flow material from both periods were also evaluated to determine the contribution from suspended material in contrast to bed load. Of the total upstream solids load, approximately 24% is bed load and 76% is suspended load in both the 10-year and 30-year simulation periods. Nearly all of the bed load and suspended load in the sand-size range settles in the LDW. Of the clay and silt suspended load, approximately 10% of the clay-size particles and 76% of the silt-size particles are predicted to settle in the LDW. All of the bed load entering the LDW from upstream is deposited within the Upper Turning Basin and the upstream portions of the navigation channel, which are periodically dredged by the U.S. Army Corps of Engineers (USACE). Approximately 50% of the total solids load entering the LDW from upstream is deposited in the LDW, with approximately 80% of this deposition occurring in the vicinity of the Upper Turning Basin in Reach 3 (QEA 2008, see Appendix B of the STM Report).

Sediment loads from lateral sources were derived from analyses conducted by the City of Seattle and King County (Nairn 2007; Seattle Public Utilities 2008). Storm drains, CSOs, and streams discharge into the LDW at over 200 locations. These were initially aggregated in the STM report into 21 discrete discharges at 16 locations to simplify modeling. In the STM, the total annual sediment load from the lateral sources was estimated to be 1,257 metric tons per year (MT/year); of this, 76% was attributed to storm drains, 3% to CSOs, and 21% to streams.

The distribution and magnitude of sediment loads from lateral sources were updated after the STM report (QEA 2008) was completed. These updated sediment loads are presented in Appendix C, Part 4, Scenario 2. The updated loads provide a more accurate distribution of the loads, reflecting better distribution of inputs and more actual outfall locations. Figure 5-2 illustrates the spatial distribution of the percentages of sediment from lateral sources at the end of the 10-year model simulation, using the updated lateral loads distribution. Updated lateral loads were used in all subsequent modeling in this FS. The areas with the greatest predicted lateral sediment contribution (i.e., the sediment bed after 10 years includes more than 10% lateral contribution) are limited to the following areas in the LDW: at the heads of Slips 4 and 6, Hamm Creek at river mile (RM) 4.3W, RM 1.8W, near Glacier at RM 1.5W, RM 1.2E, RM 0.3W, and in the Duwamish/Diagonal EAA at RM 0.5.

A third component of sediment load is the movement of surface sediment from one model grid cell to another. Bed sediment can be resuspended during a high-flow event, after which it either resettles nearby or is transported downstream. The STM tracks the movement of these particles throughout the LDW, from grid cell to grid cell. The ability of the STM to track the movement of particles within the LDW was used to evaluate the transport of sediment between Reaches 1, 2, and 3, as summarized in Figure 5-3 and in Appendix C, Part 4, Scenario 4.

The highest percentage of original bed sediments remaining in the surface layer after 10 years occurs in the grid cells east of Kellogg Island at RM 0.9 and at the Terminal 117 EAA (RM 3.0 to RM 3.5). The areas that have the highest percentage of original bed sediment remaining at the end of the 30-year simulation are consistently the highest throughout the simulation and are not the result of a short-term scour event. A higher percentage of original bed sediment indicates that much of the surface layer is not being replaced by upstream or lateral sediment (i.e., the bed surface sediments are not receiving much deposition and could be interpreted as having a more constant composition over time).

5.1.2 Solids Balance In and Out of the LDW

Figure 5-3 shows the mass of sediment moving through and within the three reaches of the LDW over 10-year and 30-year modeling periods. Year-to-year variation in sediment load occurs because of variability in river flow, with total sediment load increasing during years with relatively high flows. Over the 10-year period, more than 99% of the incoming sediment load (1,850,850 MT) originates from the Green/Duwamish River (upstream); less than 1% (12,580 MT, or an annual average of 85,000 MT/yr) enters the LDW from lateral sources. Over a 30-year period, a cumulative total of 6.2 million MT enters the LDW (for an annual average of approximately 207,000 MT/yr). The magnitude of the sediment mass movement increases, but the percent contribution from upstream and lateral sources is essentially the same as for the 10-year period. About 50% of the incoming solids (approximately 100,000 MT annually) deposit within the LDW and are not exported farther downstream into the East and West Waterways and Elliott Bay. Approximately 51% of the sediment that settles in the LDW is removed by periodic maintenance dredging, mostly in the Upper Turning Basin.[2] Thus, approximately 25% of the incoming sediment load remains in the LDW basin after dredging.

Bed load (heavier, larger particles that skip and travel along the sediment bed[3]) comprises 24% of the total incoming sediment load, on average, at the upstream boundary of the area modeled by the STM, with the remaining 76% entering the LDW as sediment suspended in the water column (QEA 2008). According to the STM, most of the bed load deposits above RM 4.0; the suspended sediment primarily deposits farther downstream or is transported through the system. The proportion of bed load to total load is inversely dependent on flow rate, decreasing from 30% to about 17% to 18% as the flow rate increases (24% on average). The estimated average annual bed load transported during the 30-year model period was 50,000 MT/year, with a range of 10,000 MT/year (1978) to 132,000 MT/year (1975) for low-flow and high-flow years, respectively (QEA 2008). This solids mass balance supports the CSM conclusion that the LDW is net depositional over long time periods and that lateral sources are important, but their effect is localized to the receiving sediments in the vicinity of these sources. The CSM and dredge records both indicate that the majority of the Upper Turning Basin dredged material is from upstream (Green/Duwamish River).

5.1.3 Scour Potential from High-flow Events and Vessel Traffic

Figure 5-4 shows potential scour areas derived from two processes: high-flow events and scour from vessel traffic. Areas of erosion from both high flows and vessel scour were considered during delineation of AOPCs (see Section 6).

Few areas in the LDW that show significant high-flow erosion potential (10 cm scour depth or more) also have subsurface contamination. These areas are identified in Appendix C (Part 4, Scenario 5) and are evaluated in Section 6 for the delineation of the AOPCs. Alternatively, most areas with significant subsurface contamination (greater than sediment quality standards [SQS]) do not show erosion potential beyond a few centimeters in depth during high-flow events. An analysis of how erosion and deposition impact surface COC concentrations over time is discussed in Section 5.2.

The STM models sedimentation and resuspension in the absence of deep sediment disturbance and exposure of contaminated subsurface sediment. No available transport model has the capacity to include anthropogenically induced resuspension and transport with confidence. Development and validation of the STM is most reliable in regions where naturally occurring sedimentation dominates transport and in areas with relatively little anthropogenic activity. The effects of such anthropogenic activity on the

2 Dredging averages 38,000 MT/yr within the navigation channel and 13,000 MT/yr in the berthing areas. The average total dredged is 51,000 MT/yr.

3 The mean percent of fines in surface sediment of Reach 3 is 34%. The mean percent of fines in surface sediment of Reaches 1 and 2 is 69%. Bed load is mostly sand and gravel-sized particles. See Appendix C, Part 3, Tables 5a and 5b for more information.

STM/BCM are separately evaluated in Section 5.3.1.2 by modifying long-term BCM SWAC estimates to include episodic disturbance of surface and subsurface sediments.

The 100-year high-flow event produces a maximum erosion depth of less than 1 foot (less than 30 cm) in limited areas (see Figure 2-9). Most of these areas do not show COC concentrations at this depth that are greater than the SQS and that are not already expressed as SQS exceedances at the surface. Subsurface COC concentrations in areas with scour greater than 10 cm are analyzed in Appendix C (Part 4, Scenario 5) and discussed in Section 5.3.2.5.

Although this FS focuses on single high-flow events, the 30-year hydrograph record used for the STM analysis included numerous high-flow events of more than 10,000 cfs. In some years, two high-flow events occurred in the same year. Therefore, the STM inherently accounts for multiple scour events in the same year (Appendices D and F in QEA 2008).

5.2 Bed Composition Model (BCM)

Output from the STM was coupled with contaminant concentrations in sediments from various sources to enable prediction of future surface sediment contaminant concentrations under various remedial action scenarios. This analysis is termed the BCM. This section of the FS describes the BCM, its applications, and its limitations.

Output from the STM is directly applied to the BCM. A basic and conservative assumption is that all contaminants are strongly bound to sediment particles. The BCM is conservative with respect to sediment concentrations because it only accounts for contaminant movement associated with particles (i.e., transport, resuspension, burial) and assumes no loss of contaminant mass via other physical, chemical, or biological degradation processes (e.g., desorption, diffusion, volatilization, biotransformation, dechlorination, etc.). Other degradation processes explored at other sites are documented at the end of this section to provide some context for understanding these processes. The BCM does not account for contaminant transfer from sediments to the water column. However, polychlorinated biphenyl (PCB) flux from sediments to the water column and to biota was estimated in the food web model (RI Appendix D; Windward 2010).

The BCM is used later in the FS as one line of evidence to evaluate recovery potential of LDW sediments (Section 6), to identify and screen remedial technologies (Section 7), and to develop and evaluate remedial alternatives (Sections 8 and 9). The sensitivity of the BCM is also investigated by looking at how changes in input parameters affect the output (Section 9). Sediment disturbance resulting from episodic emergency and high-power ship maneuvering and maintenance/construction is not included in the BCM. The potential influence of these disturbances on the sediment bed is discussed in Section 5.3.1.

5.2.1 The BCM Calculation

The BCM is a spreadsheet-based tool that predicts COC concentrations at individual model grid-cell locations[4] in the surface sediment layer (0 to 10 cm) by using a simple mass balance formula (RETEC 2007c, Appendix C):

$$C_{(time)} = C_{bed}*f_{bed\ (time)} + C_{lateral}*f_{lateral\ (time)} + C_{upstream}*f_{upstream\ (time)} \qquad \textbf{Equation 5-1}$$

Where:

- f_{bed}, $f_{lateral}$, and $f_{upstream}$ are, respectively, the fractions of surface sediment sourced from existing bed sediment, from lateral source sediment, and from upstream Green/Duwamish River sediment in each grid cell at a specific point in time. These surface sediment fractions change over time and are direct outputs of the surface sediment layer of the STM. The sum of these fractions in each grid cell is 1.

- $C_{bed,}$ $C_{lateral}$, $C_{upstream}$ are the concentrations of a COC associated with each sediment source. These concentrations are derived from existing bed contaminant concentrations, lateral source samples (i.e., stormwater and CSO discharges), and upstream (Green/Duwamish River) lines of evidence.

An example of how the BCM computation uses the STM output is shown in Figure 5-5. Additional mechanics of the BCM are provided in Appendix C.

As noted in Equation 5-1, the sediment composition fractions (f) vary with time because the STM output varies with time[5] and ongoing sediment transport changes the bed composition of each fraction. The concentration terms for the lateral source and upstream sediments ($C_{lateral}$ and $C_{upstream}$) are assumed to be constant over time for modeling purposes, representing current best estimates of the long-term average inputs over time.[6] The derivation of these values is discussed in greater detail in Section 5.2.3. The BCM assigns the same COC concentration (input value) to the lateral source and upstream sediments regardless of the variability observed over time or spatially (such as among different outfalls for the lateral sources). The bed concentration (C_{bed}) is the

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4 STM grid cells are taken directly from the STM setup, as described in the STM report (QEA 2008), and overlain with inverse distance weighting 10-ft by 10-ft chemistry grid cells in the BCM. Consequently, the BCM calculates results for 100-ft^2 areas.

5 STM output in 5-year increments is used in the BCM runs. The STM runs continuously for the entire 30-year simulation period at time steps on the order of minutes. The FS presents results in 5- or 10-year increments following the start of remedy construction. For remedial scenarios that take longer than 30 years to implement, the simulation starts over at the beginning of the 30-year hydrograph used for the STM.

6 However, high and low “sensitivity” concentrations were also used as input values to bracket the range of uncertainty in the input values and demonstrate the effects from anticipated reductions in contaminant concentrations over time.

best estimate of the COC concentration in the surface sediment bed at a given location at the start of the model period, defined by the FS surface sediment dataset. The BCM is implemented in a geographic information system (GIS) framework and MS Excel platform (described in Appendix B of RETEC 2007b).

The BCM (Equation 5-1) can be used to estimate COC concentrations in surface sediment at each grid cell location in the LDW as a function of time under various remedial alternatives. Where active remediation is assumed within an alternative, the grid cells contained within the actively remediated footprint receive a post-remedy bed sediment replacement value for C_{bed}. The new value is an estimate of the COC concentration that exists in the surface sediment at the completion of the remediation (see Section 5.2.3.4).

5.2.2 BCM Assumptions

The predictive accuracy of the BCM hinges on two important findings from the STM:

- Over time, the surface sediment that erodes, moves, and redeposits within the LDW originates primarily from the Green/Duwamish River, as shown in Figure 5-3. STM results indicate that movement of bedded sediment from within the LDW is a very minor component of overall sediment transport in the LDW. The effect of bedded sediment was further analyzed by a simulation that tracked the movement of bedded sediment. This analysis is presented in Appendix C, Part 4, Scenario 4 and Part 5, Scenario 6.

- The magnitude of high-flow bed scour is sufficiently minor such that subsurface sediments with COC concentrations that exceed the SQS are generally not exposed, eroded, or redistributed within the LDW. Even after a high-flow event, the bed height increases from deposition (see Appendix E, Figures E-19 through E-23 in QEA 2008). From the sediment mass balance analysis, the new sediment that accumulates is largely from the Green/Duwamish River. Given the limited movement of bed sediment during high-flow events, bed COC contaminant concentrations at the reach- or site-wide scale would not be predicted to change significantly during a high-flow event (Appendix C, Part 5).

Although the assumption of assigning the contaminant concentrations to resuspended bed sediment is not inherently mass conservative, it will not significantly impact model predictions, because: 1) in the LDW, the mass of bed sediment resuspended is much less than the mass of sediment from upstream; and 2) COC concentrations in resuspended sediments become similar to those in upstream solids over time and as the cleanup proceeds. Consequently, redistribution of existing sediments with COC concentrations that exceed the SQS is not a significant process, and future bed sediment chemistry can be reasonably estimated as a mass balance between present bed sediment and incoming sediment loads from the Green/Duwamish River and lateral sources.

These key findings are supported in three ways: 1) by the CSM (Section 2.3), 2) by a comparison of empirical trends to model estimates of net sedimentation and recovery rates (Section 5.4), and 3) by additional STM special scenario runs (Section 5.3.2) used to help refine the CSM for the FS.

In addition, the BCM assumes that:

- All COCs are permanently bound to sediment particles; degradation or phase transfer processes such as solubilization are assumed not to reduce COC concentrations over time. This assumption is generally consistent with the known properties of the COCs, and is inherently conservative because some degree of degradation or phase transfer likely occurs. The assumption could result in higher predicted concentrations in surface sediment with time.

- COC concentrations from drainage basins were derived from all storm drain and other solids sample data, but samples were collected from only a portion of the LDW drainage basin conveyances. These data are assumed to be representative of all lateral COC inputs. COC contributions from eroding bank material and groundwater were not included in the lateral source estimates.

- COC concentrations from drainage basins that have not been sampled are assumed to be similar to or lower than those in drainage basins sampled for source control evaluation. This is consistent with the sampling strategy of the Source Control Work Group (SCWG), which has focused first on areas with the most significant sediment contamination and associated outfalls identified as being the most likely sources of contaminated sediments to the LDW. The COC concentrations derived from the empirical data are then applied to all lateral sources in the model.

- The biologically active zone for most of the LDW is approximately 10 cm, and therefore the top 10-cm model layer represents exposure concentrations for benthic organisms. This depth is consistent with results from the sediment profile imaging (SPI) analyses conducted in the LDW for the Washington State Department of Ecology (Ecology 2007b) and King County (King County 2007a), as described in the RI (Windward 2010).[7] The 95% upper confidence limits (UCL95) on the mean of maximum sediment feeding void depths for benthic organisms (a conservative measure of the biologically active zone) used in the Ecology dataset was 11 cm with a mean

[7] The assumption of 10 cm can be reasonably applied as the biologically active zone in the LDW based on several factors: representativeness of entire benthic community, relationship with void depths, and central tendency of void depths (Windward 2010).

of 10 cm. The King County dataset was even shallower (9 cm with a mean of 8 cm). The 10-cm depth is used as the STM and BCM assumption for the active mixed layer.

5.2.3 Input Values to the BCM for Risk Drivers

Concentrations of risk drivers associated with the three sources or types of solids (i.e., upstream, lateral, and bed sediments) were estimated as inputs to the BCM. Samples from media representative of these three sources were analyzed for several COCs over a period of years, and the resulting concentrations were selected for use in the BCM based on summary statistics from compiled datasets. Some best professional judgment was incorporated into these datasets with assumptions about current and potential future conditions, including future source control efforts, the amount of solids entering the LDW, and potential biases of particular datasets. In selecting the BCM lateral input parameters, the median, the mean, and the 90th percentile of the datasets were used as the low, mid-range, and high values, respectively. High values were removed from the dataset, as described in Section 5.2.3.2, because it was assumed that they would be addressed by ongoing source control actions. For the BCM upstream input parameters, mean values of the most representative of several upstream datasets were selected for the low and mid-range input values, and the UCL95 was used for the high input value. High, medium, and low post-remedy bed sediment replacement values were derived assuming varying degrees of mixing of clean sediments in the remediated footprint with contaminated sediments remaining in the rest of the LDW, as described in Section 5.2.3.4. Selected values and ranges for the BCM input values for total PCBs, arsenic, carcinogenic polycyclic aromatic hydrocarbons (cPAHs), and dioxins/furans are provided in Tables 5-1a through 5-1c. The ranges of concentrations reported from various data sources are provided in Tables 5-2a through 5-2d.

5.2.3.1 Contaminant Concentrations Associated with Upstream Solids

Contaminant concentrations associated with Green/Duwamish River solids were compiled from various data sources, which are described in Appendix C, Part 3. These data provide multiple lines of evidence that characterize the contaminant concentrations associated with sediments entering the LDW from the Green/Duwamish River system. Data from the various studies were used to develop a range of input values for each risk driver (Table 5-1a).

The data sources evaluated included:

- Upstream whole-water samples collected by King County
- Upstream centrifuged suspended solids samples collected by Ecology
- Upstream surface sediment samples (containing fines greater than 30%) collected by Ecology between RM 5.0 and 7.0

- Upstream surface sediment samples from RM 5.0 to 7.0 included in the RI dataset
- Core data collected by the USACE to characterize sediment prior to dredging in the navigation channel from RM 4.3 to 4.75, which is assumed to represent the Green/Duwamish River combined bed load and suspended material that settles in the upper reach of the LDW.

The upstream King County whole-water concentrations were normalized to the value of the concurrently collected TSS, so that the concentration units were comparable with the sediment concentration units (i.e., both on a dry weight basis).[8]

A subset of the Ecology upstream surface sediment data was developed by excluding samples that contained less than 30% fines. This approach accommodates the systematic differences in grain size distributions between upstream (e.g., mid-channel) data and average conditions in the LDW. Both the full dataset and the subset with fines greater than 30% were used as lines of evidence to develop the range of BCM upstream input parameters.

Upstream surface sediment samples from RM 5.0 to 7.0, included in the RI dataset, were evaluated, but were not used in selecting BCM input values. The rationale for this approach is explained in Appendix C, Part 3. Instead, the more recent upstream surface sediment data collected by Ecology were used. The upstream surface sediment data had lower total PCB and cPAH concentrations than other upstream lines of evidence. This may reflect the coarser (i.e., sandier) material encountered during sampling that is characteristic of bed load[9] being transported down the Green/Duwamish River—very little of which is transported beyond the Upper Turning Basin. The surface sediments upstream of the LDW are generally coarser than those in the LDW because there is little net sedimentation upstream of the Upper Turning Basin as a result of higher stream velocities above RM 4.75.

The subsurface sediment cores collected by the USACE to characterize sediment prior to dredging in the navigation channel from RM 4.3 to 4.75 represent the Green/Duwamish River bed load and suspended material that settles in the upper reach of the LDW.[10] The Upper Turning Basin is a natural sink for incoming sediment loads from upstream, and

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8 Normalizing to TSS likely produces a high estimate of the COC concentration on sediment particles because some of the COC mass is likely dissolved or on colloidal particles that do not settle in the LDW.

9 Bed load is heavier, sandier material that travels along the bed surface; it is not suspended in the water column and thus, typically travels shorter distances than do suspended solids.

10 The RI summarized USACE cores in the Upper Turning Basin from RM 4.0 to 4.75. The FS screened this dataset to exclude the potential influence of sources (e.g., Hamm Creek) in the downstream portion between RM 4.0 and 4.3. The FS dataset also includes more recent data collected by USACE above RM 4.3.

because the navigation channel is dredged every 2 to 4 years from RM 4.0 to 4.75, this area is a good indicator of suspended solids settling in the upper reach of the LDW.

The upstream solids values selected for use in the BCM were based on these four datasets as values representing the best estimate concentrations of the risk-driver COCs entering and settling in the LDW. Each dataset contains information that represents, to a degree, the COC concentrations in sediment particles that enter and deposit within the LDW. As discussed below, these datasets are considered reasonable lines of evidence for developing incoming concentrations to the LDW from upstream, although each type of data collection tends to bias the results toward lower or higher values (e.g., low percent fines versus high percent fines; single collection events instead of seasonal collection events; potential influence of sources). In general, the value representing a mid-range of the various lines of evidence was considered for the input value, and then values representing upper and lower bounds were selected for the high and low sensitivity input values, respectively. One goal of including a range in the input values is to account for uncertainty in all the datasets representing upstream inputs and show how these data ranges affect the predictions of natural recovery for the remedial alternatives.

For total PCBs and cPAHs, the means of the LDW RM 4.3-4.75 USACE core data were selected as the upstream input values (35 microgram per kilogram dry weight [μg/kg dw] and 70 μg toxic equivalent [TEQ]/kg dw, respectively). To address sensitivity around the mid-range value for both total PCBs and cPAHs, the low upstream input values were the means of the Ecology upstream surface sediment samples containing fines greater than 30%. The high upstream input values were the UCL95s of the TSS-normalized King County whole-water datasets.

For arsenic, the selected upstream input value was the mean (9 milligrams per kilogram dry weight [mg/kg dw]) of the Ecology upstream samples containing fines greater than 30%. The mean of the LDW RM 4.3 to 4.75 USACE core data (7 mg/kg dw) was selected as the low sensitivity value. The high sensitivity value (10 mg/kg dw) was the UCL95 of the Ecology upstream sediment samples containing fines greater than 30%. King County surface water TSS-normalized data and Ecology centrifuged solids data were not used in the selection of BCM upstream values for arsenic because the UCL95 for both of these datasets would have resulted in much higher modeled surface sediment concentrations than in the LDW baseline dataset. It is likely that these two datasets, especially the surface water dataset, contain finer particulates with higher arsenic concentrations than those that deposit in the LDW. These finer particles tend not to settle in the LDW (approximately 50% of the Green/Duwamish River solids [bed load and suspended solids combined] do not settle in the LDW).

For dioxins/furans, the Ecology upstream sediment samples (containing fines greater than 30%) and the Ecology upstream centrifuged solids were the only datasets used for selecting the BCM input values; there were neither core data from RM 4.3 to 4.75 nor whole-water dioxin/furan data among the other datasets. Because of the smaller datasets and the desire to evaluate a range of input values, a slightly different approach was used to select dioxin/furan BCM input values. The midpoint between the means of the two datasets is the mid-range value (4 ng TEQ/kg dw); the low sensitivity value is the mean of the Ecology upstream sediment samples containing fines greater than 30% (2 ng TEQ/kg dw); and the high sensitivity value is the midpoint between the mean and UCL95 of the Ecology upstream centrifuged solids dataset (8 ng TEQ/kg dw).

Dry weight concentrations for COCs based on upstream surface sediment samples may be biased low and may underrepresent the concentrations associated with the fraction of solids entering the LDW that have finer grain size and higher organic carbon concentrations. Silt- and clay-sized suspended solids represent 67% of the sediment entering the LDW. As a result of the settling of most sand-sized particles in Reach 3, silt- and clay-sized particles make up only about 35% of the sediment that settles in Reach 3, but more than 90% of the sediment that settles in Reaches 1 and 2. Case study literature and LDW data exist that support the relationship between COC concentrations, organic carbon content, and particle size. The relationship between particle size and organic carbon content and the various methods to account for these relationships and their potential effect on model results is explored in Section 5.3.3.

5.2.3.2 Contaminant Concentrations Associated with Lateral Source Sediments

Contaminant concentrations associated with storm drains and CSOs were evaluated to estimate concentrations associated with lateral source sediments.[11] The storm drain solids and CSO data were collected as part of ongoing source control programs for the LDW. All available storm drain data were compiled by Seattle Public Utilities (SPU) for source samples collected in areas draining to the LDW through June 2009 by SPU, the Boeing Company, and King County. These data included storm drain solids collected from on-site and right-of-way catch basins, in-line grab samples, and in-line sediment traps. The storm drain solids data were used to generate a range of lateral input concentrations for total PCBs, arsenic, and cPAHs for use in the BCM. Storm drain solids and sediment data collected near large stormwater outfalls draining urban areas in the greater Seattle area were used to establish BCM lateral input values for dioxins/furans. The King County CSO whole-water data were also considered and found to support the ranges of BCM lateral input values estimated from the storm drain solids dataset. Consequently, the same COC concentration values were used for both storm drains and CSOs and were also assumed for the stream inputs.

The lateral input values selected for use in the BCM are estimates, based on the assumption that contaminant concentrations in storm drain solids will decrease as a

[11] Lateral source sediments include inputs from storm drains, CSOs, and streams discharging to the LDW.

result of source control efforts in the LDW drainage basin. The following assumptions were made for the BCM input values:

- ♦ The mid-range, or best-estimate, input value is a pragmatic assessment of what might be achieved in the future with anticipated levels of source control. This value is based on mean/median concentrations observed in the lateral dataset after excluding the highest concentrations in the dataset to represent control of high and medium priority sources.
- ♦ The high sensitivity value is a conservative representation of near future conditions assuming only modest success in management of high priority sources already identified by the SCWG.
- ♦ The low sensitivity value is an estimate of the best that might be achievable in 30 to 40 years with increased coverage and continued aggressive source control.

The assumed level of source control was based on best professional judgment of the SCWG and what is currently known about the distributions and current source(s) of each COC within the LDW drainage basin. The BCM input values reflect potential levels of source control that could occur over time. To simulate potential lateral inputs after implementing varying degrees of source control, the source tracing datasets were screened to remove all values above various concentrations already targeted for source control. Summary statistics were then generated for each level of assumed source control (high, medium, low). Table 5-1b presents the best-estimate BCM input values for lateral sources. The summary statistics for the four human health risk drivers (total PCBs, arsenic, cPAHs, and dioxins/furans) are provided in Tables 5-2a through 5-2d.

A general summary of the lateral input values selected for the BCM is presented below. The lateral sources memo (King County and SPU 2010) found in Appendix C, Part 3 describes the selection of the lateral input values in more detail. It should be noted that the high lateral input value is not intended to represent what sources could potentially exist throughout the drainage basins tributary to the LDW. This high value is used only to determine sensitivity of the model and the implications of inadequate source control at individual discharge locations; it is not an estimate of actual source loads or a target value for source control work. Similarly, the low sensitivity value should not be construed as a prediction of source control efficiency or as a determination of source control effectiveness or completeness. The actual effectiveness of source control can only be assessed after the fact because "complete" source control is the aggregate of many different actions applied to any given media, pathway, or source of COCs.

Total PCBs

Prior to generating summary statistics for total PCBs and to avoid skewing the summary statistics, the data were flow-weighted, including data from these targeted and known source areas: Rainier Commons, North Boeing Field/Georgetown Steam Plant, Terminal 117, and Boeing Plant 2/Jorgensen Forge. Flow-weighting takes into account the relative contribution of a specific contaminant by adjusting its concentration based on the land area and estimated annual runoff volume relative to the total contributing area in the LDW drainage basin. To reflect potential levels of source control that could occur over time, a range of screening concentrations was used to select the BCM lateral values for total PCBs. The mid-range BCM input value (300 µg/kg dw) is represented by the mean of data after excluding concentrations greater than 5,000 µg/kg dw.

Screening values of 2,000 and 10,000 µg/kg dw total PCBs were used to define the low and high BCM sensitivity values, respectively. If all samples with a total PCB concentration above a screening value of 2,000 µg/kg dw are removed from the dataset, the median of the remaining data is 100 µg/kg dw. This value was selected as the low BCM sensitivity value (100 µg/kg). When all samples with total PCB concentrations above a screening value of 10,000 µg/kg dw are removed from the dataset, the 90th percentile value of the remaining data is 1,000 µg/kg dw, which was selected as the high BCM sensitivity value.

cPAHs

Unlike total PCBs, cPAHs are expected to be difficult to control due to urbanization and major transportation routes in the LDW basin, and a multitude of current sources. Consequently, a more cautious approach was taken with the source tracing dataset by excluding cPAH concentrations above a single source control level of 25,000 µg TEQ/kg dw. Data for cPAHs were not flow-weighted because cPAH concentrations in the storm drain solids samples do not show a distinct geographic distribution, and higher concentrations of cPAHs are found throughout the LDW drainage basins, typically in drainage structures (catch basins and oil/water separators) at facilities engaged in transportation-related activities (e.g., bus and airport operations), maintenance facilities, service stations, foundries, and fast food facilities. The mean (1,400 µg TEQ/kg dw) of the data, excluding all samples with cPAH concentrations greater than 25,000 µg TEQ/kg dw, was selected as the BCM input value. The median (500 µg TEQ/kg dw) was selected as the low sensitivity value. The 90th percentile (3,400 µg TEQ/kg dw) was selected as the high BCM sensitivity value.

Arsenic

For arsenic, two different screening values (the SQS and cleanup screening level [CSL]) were used to reflect different potential levels of source control. The mid-range BCM input value of 13 mg/kg dw was selected based on the mean of the dataset, excluding all samples with arsenic concentrations above a screening value of 93 mg/kg dw (the CSL). The 90th percentile of the same dataset is 30 mg/kg dw, and this value was selected to represent the high BCM sensitivity value. If all samples with arsenic concentrations above a screening value of 57 mg/kg dw (the SQS) are removed from the dataset, the median of the remaining data is 9 mg/kg dw. This value was selected as the low BCM sensitivity value.

Dioxins/Furans

Available storm drain solids data for dioxins/furans were also used along with surface sediment sample data collected for the LDW RI in the vicinity of storm drains throughout the Greater Seattle metropolitan area to establish BCM lateral input values. By combining these two datasets (because the storm drain solids dataset was small compared to the other risk-driver datasets) and excluding one outlier, BCM lateral values were selected for dioxins/furans. The mean of 20 ng TEQ/kg dw was selected as the BCM input value; the median of 10 ng TEQ/kg dw as the low BCM sensitivity value; and the UCL95 of 40 ng TEQ/kg dw as the high BCM sensitivity value. In addition, the UCL95 rather than the 90th percentile was used to establish the high BCM sensitivity value, because it resulted in a more reasonable upper end estimate for the sensitivity analysis.

King County CSO Whole-Water Samples

In addition to the storm drain solids dataset, whole-water samples collected from CSOs by King County for analyses of PCBs, arsenic, and cPAHs were also considered when developing BCM lateral values. For both total PCBs and cPAHs, whole-water concentrations were divided by their sample-specific TSS concentrations to calculate TSS-normalized concentrations. This gives a conservative estimate that is likely biased high because it is assumed that all of the PCBs and cPAHs are on the particulate fraction and none are in the dissolved or colloidal phases. For arsenic, paired total and dissolved concentrations were used to estimate the portions of the total arsenic concentrations associated with the particulate fraction. These were then divided by the sample-specific TSS concentrations to calculate a TSS-normalized concentration for arsenic. Whole-water samples collected from CSOs in the LDW had not been analyzed for dioxins/furans at the time this document was prepared. Summary statistics for CSO data are provided in the lateral source memo (King County and SPU 2010) found in Appendix C, Part 3.

5.2.3.3 Contaminant Concentrations of Existing Bed Sediments

Existing bed sediment contaminant concentrations were developed by spatially interpolating surface sediment data from the FS baseline dataset for total PCBs, arsenic, and cPAHs. An inverse distance weighting (IDW) algorithm was used to interpolate the data. The IDW methodology is documented in Appendix A.

Existing bed sediment concentrations for dioxins/furans were developed by applying Thiessen polygons to the dioxin/furan surface sediment data from the FS baseline dataset. For Washington State Sediment Management Standards (SMS) contaminants, SQS and CSL exceedances at surface sediment stations were also spatially applied using

Thiessen polygons. In this case, dry weight or organic carbon (oc)-normalized concentrations were compared to SQS/CSL or apparent effects threshold criteria, as appropriate for each contaminant. Thiessen polygons were designated as a pass, SQS exceedance, or CSL exceedance. Sediment toxicity results trumped SMS chemistry results. For example, a Thiessen polygon with a contaminant CSL exceedance, but a toxicity pass, was coded as a pass.

Collectively, these risk drivers comprise the FS baseline dataset used to map "existing conditions" in the LDW. The FS baseline dataset spans about 18 years (1991 to 2009) of data collection efforts. It is likely that current concentrations of some COCs at stations sampled many years ago may now be lower than what is reflected in the FS baseline dataset (see Appendix F).

5.2.3.4 Post-Remedy Bed Sediment Replacement Values

In areas that would be actively remediated under different cleanup alternatives, the existing bed sediment concentration (C_{bed}) is replaced with a value representing near-term (0 to 2 years) conditions following the cleanup. The post-remedy surface sediment conditions are influenced by multiple factors. This subsection describes the assumptions used to model the post-cleanup concentrations.

Experience at other sediment remediation sites has shown that contaminant concentrations in the sediment bed shortly after the completion of dredging or capping cannot be assumed to be zero and are often above background (NRC 2007, EPA 2005b, Anchor 2003). This occurs because: 1) some degree of residual surface contamination always exists from the resettling of contaminated sediments suspended during remedial activities; 2) material used for capping of subsurface sediment exposed after dredging contains low concentrations of these COCs; and 3) existing adjacent sediments can become resuspended and then deposited in remediated areas.

Post-remedy bed sediment replacement values within a remediated area reflect an assumed combination of clean backfill material (e.g., from capping or ENR, and using or not using post-dredge residuals management) and the average concentration of surrounding unremediated sediments. To derive a replacement value based on this assumption, estimates of both values are required. The UCL95 values for the 2008 EPA Puget Sound Ocean Survey Vessel (OSV) *Bold* survey (EPA OSV *Bold* survey) data were used to estimate the contaminant concentrations in clean backfill. These data correspond to natural background estimates for Puget Sound.[12]

However, once clean material is placed, other sediments start settling on the backfill. These sediments are some combination of upstream and lateral inputs, resuspended bed sediments, and dredge residuals. For the purposes of this FS, the average concentration of bed sediments that will not be actively remediated was assumed to be

12 Data were also collected from the Strait of Juan de Fuca and Strait of Georgia.

representative of this mixture of inputs onto the clean backfill. The average concentration of unremediated sediments was derived using the SWACs outside of remediated areas. The average concentrations remaining outside of AOPC 1 and outside AOPC 2 for Alternative 6 (see Section 6 for AOPCs and Section 8 for alternative footprints) were used in this analysis. The post-remedy bed sediment replacement value was applied to the actively remediated footprint. Clean material was assumed not to be deposited outside of the active footprint.[13]

To calculate a range of post-remedy bed sediment replacement values, the following ratios of clean material to the post-remedy SWAC were assumed: 50:50 for the mid-range BCM input value, 75:25 for the low sensitivity value, and 25:75 for the high sensitivity value.

Post-remedy bed sediment replacement values for total PCBs, arsenic, cPAHs, and dioxins/furans are presented in Table 5-1c. The degree of residual contamination is dependent on several factors, including the type of remedial activity, specific design elements, construction methods, best management practices, engineering controls, and contingency measures (discussed further in Section 7.1). Therefore, post-remedy bed sediment replacement values for use as input parameters to the BCM were developed as a range using the proportioning values described above and best professional judgment. The same post-remedy bed sediment replacement value is applied to areas that are to be dredged, capped, undergo ENR, or have a thin-layer placement of sand inside the dredge footprint for residuals management.

5.2.4 Inputs and Application of the BCM for Other SMS Contaminants

The BCM can also be used to estimate future SQS and CSL exceedances for SMS contaminants. In the BCM, a particular SMS contaminant is selected for each point, and the BCM assigns that point into one of three categories in the future: below the SQS, SQS exceedance (but below the CSL), or CSL exceedance. The BCM equation (Equation 5-1) can be used to estimate future concentrations for any contaminant having available upstream and lateral input values. For the FS, these calculations were conducted on a subset of the SMS contaminants, termed “representative” contaminants. This subset was chosen from the full list of SMS contaminants because: 1) not every SMS contaminant has lateral and upstream data available; 2) several SMS contaminants had very low detection frequencies; and 3) indicator SMS contaminants within a specific class (e.g., PAHs) may well represent the behavior of that class. The representative SMS contaminants were identified by querying the database and counting the number of samples that exceeded the SQS for each contaminant. Those with the most frequently

13 The post-remedy bed sediment replacement value was not applied outside of the active remedial footprint because a thin layer of sand will be applied to manage dredge residuals where needed. It was assumed that such application would, on average, return any sediments affected by residuals outside of the dredge footprint to preconstruction concentrations.

detected exceedances were selected to represent a group/class (Table 5-3). They include bis(2-ethylhexyl)phthalate (BEHP) (phthalate group); chrysene, fluoranthene, and phenantherene (PAH group); and mercury and zinc (metal group). Arsenic and total PCBs were also included to assess the spatial distribution of these risk drivers in a manner consistent with the other SMS contaminants. Detected SQS/CSL exceedances for total PCBs were assessed using sample-by-sample oc-normalizations to ensure that detected exceedances were not missed in the interpolated IDW maps based on dry weight (see Table 5-2a).

After the initial representative SMS contaminant list was established, locations were identified that exceeded the SQS for other SMS contaminants, and additional SMS contaminants were added to the list so that at least one representative SMS contaminant was identified for each location. As a result, butylbenzyl-phthalate, phenol, acenaphthalene, and indeno(1,2,3-cd)pyrene were added. Table 5-3 lists these SMS contaminants and the upstream and lateral values established for each.

For each location that had a detected SQS exceedance in the FS baseline dataset, the maximum exceedance ratio above the SQS and the SMS contaminant responsible for that exceedance were determined. Typically, the SMS contaminant responsible for the highest exceedance was one of the representative SMS contaminants, and was usually total PCBs.[14] If the SMS contaminant with the maximum exceedance ratio was not in the representative SMS contaminant list, a representative SMS contaminant of the same chemical class that also exceeded the SQS at that location was used in the BCM. The future BEHP concentrations were also predicted by the BCM for each location because this SMS contaminant is a concern due to lateral sources.

5.2.4.1 Input Values for Representative SMS Contaminants

Lateral input values were determined by querying the City of Seattle's lateral source database (SPU 2010). Upstream input values were derived from the USACE Dredged Analysis Information System (DAIS) core database using data through 2009 (USACE 2009b, 2009c). For the City of Seattle data, all storm drain solids data were queried for each COC. The log-normal mean of the dataset was then calculated and used as the lateral inflow value for that contaminant (Table 5-3) after outliers were removed. The USACE core data from the Upper Turning Basin, RM 4.3 to 4.75, were used to represent the incoming sediment from upriver because that is the only upstream dataset analyzed for all SMS contaminants over a sufficient period of time. The data were screened to include only those collected after 1990 (prior data were excluded). The median of the dataset for each contaminant was then calculated and used as the upstream value for that contaminant. Table 5-3 lists the lateral and upstream inflow values used for each representative contaminant. No post-remedy bed sediment replacement values were used for these points. If a point was located in an actively remediated area, it was

[14] Several locations were sampled only for PCBs.

considered to be remediated below the SQS and removed from further bed composition modeling at that location.

5.2.4.2 BCM Equation Using Lateral and Upstream Input Parameters

For those locations where the detected concentration of any SMS contaminant exceeded the SQS at the start of the modeling period (and was not a toxicity pass), the BCM equation was run using Equation 5-1. The upstream and lateral input values discussed in Section 5.2.4.1 were employed for the contaminant selected to represent that location. Equation 5-1 was also used to estimate exceedances at the end of 10 years for BEHP, a contaminant that chronically exceeds the SQS and is generally associated with non-point source lateral discharges.

Because the lateral and upstream input parameters are on a dry weight basis, the BCM Equation 5-1 was run for the representative SMS contaminants using dry weight concentrations. For each SMS contaminant modeled at a location and having oc-normalized SMS criteria, the dry weight concentrations predicted for each time period modeled were compared to the baseline dry weight concentration. This process yielded a percent reduction that was then applied to the baseline oc-normalized concentration. If the resulting value exceeded the SQS, then the station was considered to be an SQS exceedance at the end of the modeling period.

5.2.5 BCM Output and Model Sensitivity

The output of the BCM is predicted contaminant concentrations for each grid cell[15] at specified time intervals (i.e., 5, 10, 15, 20, 25, 30, 35, 40, and 45 years). Summary statistics, such as site-wide and area-specific SWACs can be calculated for the distributions of surface sediment concentrations and used in assessing remedy effectiveness. Area-specific statistics can be calculated to assess beach play and potential clamming area-focused remedies.

Sensitivity runs of the BCM are used to evaluate the effect of varying contaminant concentrations associated with upstream and lateral source sediments and post-remedy bed sediment replacement values (in remediated areas) on bed sediment concentrations over time. The sensitivity of the BCM was investigated by looking at how changes in input parameters affect the output (Appendix C, Part 5).

When evaluating model uncertainty, it is important to understand that the contaminant concentration in a specific area is not as straightforward as selecting a specific cell and assuming that the concentration in that cell is accurately represented by the BCM value. For developing the initial contaminant concentration, the BCM uses a 10 ft × 10 ft cell size to capture the spatial scale of surface sediment contaminant concentrations used in IDW interpolations (see Appendix A). The BCM grid is used for computing SWACs in

[15] The BCM analysis uses grid cell sizes of 10-ft by 10-ft, the same as those used for the IDW interpolation of surface sediment concentrations.

this FS. However, it should not be construed that the 10 × 10-foot grid is appropriate for design purposes and the grid should not be used beyond this FS. Remedial design should be based on data and analysis specific to a design area.

Existing surface sediment contaminant data are more sparsely located in some areas and the initial contaminant concentration for a grid cell of interest may be represented using a data point that was collected anywhere from a few feet up to more than one hundred feet from the location of the grid cell. Nevertheless, when averaged over larger areas, model results are still relevant. However, the BCM model resolution on finer scales is limited not only by resolution of initial condition data but also by STM grid cell resolution[16] and other factors (such as representation of lateral load distribution). For example, specific "hot spots" may cover only a small part of an STM grid cell that extends from the bank to fairly deep water. The model-predicted current velocity and sedimentation rate are assumed to be spatially constant over this STM grid cell. The actual current velocity, and therefore sedimentation rate, may vary substantially over this STM grid cell, especially for cells that are near-channel or near-shore. The current velocity and sedimentation rate may be representative of the average for the area covered by the STM grid cell, but may not accurately represent these parameters within some subdomain of the STM grid cell. It will always be important to investigate and understand model input and processes (such as the scale of predicted sedimentation rates from the STM) when evaluating the appropriate size of areas where BCM-predicted contaminant concentrations are valid.

5.3 Additional Analyses Related to Natural Recovery Potential

The STM and the BCM presented above address most of the processes that affect natural recovery. However, this FS assesses several processes not explicitly addressed in the RI (Windward 2010) and the Final STM report (QEA 2008). These include:

- The effect of tugs on sediments in berthing areas (disturbance activity)
- Additional model scenario runs using the calibrated STM to answer several specific FS questions
- Influence of grain size and organic carbon on sediment contaminant concentrations.

The following sections discuss these other processes that may affect natural recovery.

5.3.1 Incorporating Effects of Disturbance Activity

The STM and BCM predict changes to the sediment bed for long time periods from natural processes and estimated contaminant loadings. However, STM and BCM

16 STM grid cells range in size from range from 0.1 to 4 acres, with the median area of a grid cell being 0.5 acre (e.g., a 100 ft-by-200 ft area is roughly 0.5 acres).

predictions do not incorporate long-term changes to the sediment bed that could be caused by deep disturbance of sediments (i.e., up to 2 ft), such as:

- Emergency and high-power (i.e., outside of routine operating procedures) tug or ship maneuvering, ship grounding, small boat activities in shallow water, and construction and maintenance-related activities in the LDW may cause deep scour (Section 5.3.1.3), which mixes subsurface sediments with surface sediments, resulting in higher contaminant concentrations at the surface.

- Seismic events (earthquakes) could result in liquefaction-induced ground movements that could damage in-water and upland infrastructures and could result in deep disturbance of subsurface contamination, resulting in higher contaminant concentrations at the surface.[17]

Such disturbances would likely be isolated and infrequent, but the cumulative effects could be of concern over the long term. Several approaches were utilized to increase our understanding of how BCM-predicted SWAC values are influenced by both natural and anthropogenic processes. This section discusses two topics:

- Influence on bed erosion of vessels maneuvering in the navigation channel and in areas deep enough to accommodate vessel drafts based on propeller shear stress modeling

- Areas where episodic, high-energy disturbance activity can expose more highly contaminated underlying sediments.

5.3.1.1 Propeller-Scour Model of Maneuvering Vessels

Propeller scour from tugs transiting the navigation channel under routine operating procedures in the LDW was evaluated in the STAR (Windward and QEA 2008). The analysis showed that the maximum scour from tugs transiting the navigation channel is less than 1 cm within the navigation channel and approximately 1 to 2 cm on the benches adjacent to the navigation channel. The higher potential scour on the benches is due to tugs traveling on the edge of the navigation channel adjacent to shallower depths on the benches.

Assuming that sediments resuspended by propellers redeposit near the resuspension site, then anthropogenic scour in the navigation channel and benches acts only as a mixing process in the surface layer, augmenting the mixing induced by bioturbation (which is typically greatest within the top 10 cm of sediment). The STM assumes a 0- to 10-cm mixed layer of sediment at the surface; hence, the effects of propeller scour

[17] Although earthquakes can also result in admixture of subsurface and surface sediments, this potential disturbance is not explicitly discussed in this section, because the range of effects is not readily modeled with the information currently available. However, see Section 8.1.3 for more information.

associated with vessels moving in the navigation channel are consistent with the STM assumptions for tugs operating in the navigation channel.

However, the propeller scour analysis presented in the STAR is not applicable to tugs or vessels maneuvering in areas shallower than the navigation channel or when emergency and high-power operations are needed. Tugs may occasionally need to use more power while maneuvering barges in and out of berths, and tugs may be stationary for longer periods of time (while still operating their propellers).

A modeling approach developed by the USACE was applied to the LDW for maneuvering vessels. This model was developed with an analysis of currents and shear stresses induced by towboats and barges on the Mississippi River (Maynord 2000). The methods and model were used for computing bottom currents and shear stresses caused by moving barges and propeller scour in the LDW. A detailed discussion of the Maynord model is presented in Appendix C, Part 7. Briefly, the model maps the velocity and the associated shear stress induced by the propellers that reaches the river bottom. The shear stress time series and the sediment characteristics at the river bottom determine the amount of scour that will occur over a period of time. The velocity is related to the amount of power applied by the tug. However, tugs may operate at higher power for short periods of time. The applied power under different operating conditions and durations was determined from interviews with tug operators. The analysis followed a similar approach as in the STAR (Windward and QEA 2008), using the same two tugs for model input parameters. The larger tug, *Sea Valiant*, operates downstream of the First Avenue South bridge (RM 2.1), while the smaller tug (*J.T. Quigg*) is able to operate in shallower water upstream of the bridge.

No precise methods are available to relate propeller-induced shear stress to sediment erosion. However, rough estimates of the scour magnitude can be developed. Based on the analysis,[18] localized deep (more than 10 cm) vessel scour may occur for tugs operating in shallow water and at higher power, as described by tug operators working under emergency conditions (see Appendix C, Part 7). Vessel scour depth is strongly affected by the distance between the propeller and the sediment bed, with substantially less scour in deeper water. Other factors influencing propeller scour are propeller angle, thrust, blade configuration, and duration of the high-power event under stationary conditions. For most berthing areas and operational conditions (in deeper water operations under normal power conditions), the depth of scour is estimated to be 10 cm or less, which would not necessarily disturb and expose subsurface contaminant concentrations (see Appendix C, Part 7). However, as described in Section 5.3.1.2, infrequent events can scour more than 10 cm. Results of this scour analysis, combined with empirical evidence of scour, have been incorporated into the FS in two ways: the

[18] This analysis was limited to the vertical depth of the Sedflume core data collected during the RI (about 30 cm).

development of recovery categories (Section 6) and in the technology assignments for individual remedial alternatives (Section 8). The following section discusses other components of scour.

5.3.1.2 Episodic Deep Disturbances Leading to Exposure of Subsurface Contamination

Potential influences on SWAC from routine vessel operations are described above. However, less frequent and episodic events in an active navigation area such as the LDW may induce disturbance of subsurface sediments, exposing subsurface contamination. In this FS, this process is called deep disturbance. Deep disturbances may involve ships operating with excessive propeller power, ship groundings, emergency maneuverings, or seismic events. Maintenance operations such as dock construction/maintenance and vessel maintenance may also cause deep disturbance.

The STM/BCM models were not set up to model deeper disturbance events, so this FS conducted a separate sensitivity analysis of deep sediment disturbance to consider the potential effects of such disturbance events on STM/BCM-predicted SWACs. This disturbance analysis introduces an additional, local source of contamination: the subsurface sediment bed. Natural processes (apart from earthquakes) and routine ship operations in the LDW will not typically mix the surface 0- to 10-cm layer with deeper subsurface sediments except in areas that were identified on the basis of known ship activity and from precision bathymetry, which suggested deeper erosion (Section 5.3.2.7). However, some lines of empirical evidence (geochronology cores and sediment concentration profiles) suggest that in some areas subsurface sediments may have been disturbed as a result of anthropogenic activity. There is evidence, based on contaminant profiles in some cores and geochronological data, that deep disturbance events may have hindered recovery at localized areas. The frequency and magnitude of these events is unknown. Influence of such events on BCM SWAC projections was analyzed in Appendix M, Part 5, and results are compared in Section 10. Changes in the long-term SWAC, based on potential exposure of contamination remaining in the subsurface sediment after dredging or capping, are estimated for each alternative as a function of the long-term SWAC, the size of the area disturbed, and the average contaminant concentration remaining in the subsurface after remediation. Because the total area of deep disturbance is unknown, results are presented as change in SWAC as a function of acreage that has experienced deep disturbance. Because the frequency of such events is also unknown, this FS assumes that disturbed areas would have to be exposed continuously to produce a measurable difference in the long-term model-predicted SWAC of 25%. This 25% threshold is considered the minimum change needed to detect a difference between two SWAC values (see Section 9.1.2.1). Results for the deep disturbance analysis (provided in Section 10) range from 11 to 43 acres (2% to 10% of the total LDW acreage).

5.3.2 Additional Special Scenario STM Runs

Six additional scenarios were run using the STM to further understand the movement of sediment particles within the LDW and the potential effects on the natural recovery analysis. The additional runs assessed:

1) Potential for recontamination of EAAs
2) Effect of more detailed distribution of discharges from lateral sources on the bed composition
3) Movement via tidal currents of resuspended sediment from reaches downstream of the Upper Turning Basin upstream into the Upper Turning Basin
4) Movement and deposition of sediment between Reaches 1, 2, and 3
5) Fate of sediment scoured from depths greater than 10 cm
6) Tracking of existing bed sediment movement
7) Natural recovery hindered in selected berthing areas.

A description of each of these scenarios and a summary of the results are presented in Table 5-4. A detailed accounting of scenarios 1 through 6 is presented in Appendix C, Parts 4 and 5. The findings of this work are generally consistent with the CSM (see Section 2) and support key assumptions and analyses inherent in the BCM and the assignment of remedial technologies (Section 8). The primary findings of the special scenario STM runs are discussed below.

5.3.2.1 Scenario 1: Potential Recontamination of EAAs

The purpose of this scenario was to assess the potential for remediated EAAs to be recontaminated over time by areas located outside of the EAA footprints that would be allowed to recover naturally. This may affect decisions concerning the timing and sequencing of remedial activities at specific EAAs.

The results of this analysis indicate it is unlikely that remediated areas will be recontaminated by unremediated areas unless the areas are adjacent to each other. Material resuspended from unremediated areas during high-flow events is estimated to account for less than 5% of the material that settles in remediated EAA footprints over a 10-yr period[19] (see Figure 5-6). The BCM analysis on this scenario indicates that recontamination of EAAs above the SQS (the SQS was used as a point of comparison for this analysis because other potential remedial action levels [RALs] vary by alternative)

[19] Only a few grid cells have been identified as having non-EAA source material in the range of 5 to 20% and most of these are in Reach 2. The average across the LDW is generally less than 5%.

is more likely to occur near outfalls as a result of lateral source inputs than to scour and settling of bed sediment from outside EAAs.

5.3.2.2 Scenario 2: Distributed Discharges from Lateral Sources

This scenario examined certain simplifying assumptions that were used in the STM for lateral discharge locations (for storm drains and streams), and refined those assumptions to better account for actual lateral discharge distribution. In the original STM (QEA 2008), all Duwamish watershed discharges were aggregated into 16 discharge points along the LDW. The discharge points consolidated total area runoff from storm drains to the major outfalls and did not include the more widely distributed smaller outfalls located along the shoreline. CSOs that discharge to the LDW were also included, but these were modeled at their actual locations.

In this distributed discharges modeling scenario, finer drainage basin delineations were used to more accurately reflect actual drainage subbasins and outfalls (pipe locations) of storm drains, resulting in 13 major storm drains, 9 CSOs, and 11 waterfront areas that discharge to the LDW through numerous small outfalls. The revised load estimates and drainage basins for storm drains, creeks, and City CSOs (SPU 2008) were presented and are summarized in Appendix C. Because the distributed load simulation more accurately represents the distribution of lateral loads along the shoreline, it was carried forward as the FS base case loading condition. The lateral loads used in the FS base case are shown in Figure 5-7.

5.3.2.3 Scenario 3: Movement of LDW Bed Sediment into the Upper Turning Basin

This scenario examined the degree to which bed sediments from elsewhere in the LDW may become resuspended, transported upstream, and deposit in the Upper Turning Basin (above RM 4.0). The Upper Turning Basin sediment composition and chemistry is only minimally affected (less than 0.01%) by sediment moving upstream with tidal currents (Figure 5-8). Figure 5-8 shows the geographic distribution of sediment settling in Reach 3 but originating from downstream of RM 4.0 (from Reaches 1 and 2). Only the area between RM 4.0 and 4.1, Slip 6, and a few other isolated grid cells in Reach 3 are estimated to have more than 0.01% sediment contribution from bed sediment downstream of RM 4.0, and even these areas are less than 0.05%. This estimate is in agreement with the 10-year sediment mass balance, which indicates that about 240 MT moves from Reaches 1 and 2 and is expected to deposit in Reach 3 (see Scenario 4). This is extremely small compared to the estimated total sedimentation in Reach 3 of 2.3 million MT over 30 years; 99.99% of this sedimentation is from upstream sediments. Based on this analysis and the contribution of sediments from lateral sources (see Section 5.3.2.2), the sediment in the Upper Turning Basin and the navigation channel above RM 4.1 should not be adversely affected by sediments transported from other portions of the LDW. The BCM analysis for this scenario shows that the predicted COC concentrations in the Upper Turning Basin are for the most part very low and negligibly affected by the amount of sediment deposited from downstream. This analysis also supports the use of Upper Turning Basin sediments in the navigation channel (RM 4.3 to RM 4.75) as representing the COC concentrations in sediments originating from the Green/Duwamish River.

5.3.2.4 Scenario 4: Movement of Bed Sediments between Reaches

This scenario examined the degree to which bed sediments in one reach of the river may be resuspended and transported to another reach. These results may be important in assessing recontamination potential between reaches and in assessing if locations would be important for sequencing the remedial alternatives. Sediment exchange (either upstream or downstream) is strongest between Reach 1 and Reach 2, while Reach 3 primarily contributes sediment to downstream reaches with very little sediment transported from downstream reaches back to Reach 3 (Figure 5-9). In addition, much of the bed sediment that is resuspended in a reach resettles in that same reach.

Reach 3 receives a large amount of sediment from the Green/Duwamish River as a combination of suspended load and bed load, the latter consisting mostly of sand. This reach is regularly dredged by the USACE, particularly in the Upper Turning Basin. Maintenance dredging, applied by the USACE on the cycles that we have seen in the past, should not change current natural recovery processes because it primarily removes sand that is not readily transported downstream and therefore is not a significant component of net sedimentation and natural recovery in Reaches 1 and 2.

5.3.2.5 Scenario 5: Sediment Scoured from Greater than 10 cm Depth

This analysis was used to evaluate whether scour and transport of deeper sediments may influence the waterway-wide SWAC during an extreme high-flow event. Scour during a 100-year high-flow event was analyzed in the STM report as a 30-day simulation (QEA 2008). Scour in excess of a 10-cm depth (up to about 22 cm) occurs in portions of the LDW from RM 2.9 to RM 3.9 and in isolated areas between RM 4.2 and RM 4.7. Most of these areas are in the navigation channel.

Sediment scoured from below 10 cm during a 100-year high-flow event was modeled over a 10-year period. In Figure 5-10a, the STM estimates that approximately 200,000 MT of sediment settles in the LDW during a 100-year high-flow event and of that amount, approximately 70,000 MT is eroded from the bed. However, as shown in Figure 5-10b, only about 6,600 MT of the sediment that settles is eroded from below 10 cm, which is only about 3% of the deposition during the 100-year high-flow event. Consequently, sediment eroded from below 10 cm during high-flow events, and mostly from Reach 2, makes a negligible contribution to sediment transport and deposition in the LDW during those high-flow events. In Reach 2, about 45% of eroded material is estimated to redeposit in the same reach (3,800 MT deposited out of 8,700 MT eroded) while deposition of upstream sediment and eroded shallow sediments from other areas of the LDW is estimated to be approximately ten times this amount. Consequently, erosion and redeposition of sediment scoured below 10 cm makes a negligible contribution to the potential for redistribution of subsurface sediment between reaches during high-flow events. In addition, very few sediment cores in these potential scour areas had SQS exceedances and those with exceedances were located in or adjacent to EAAs (see Appendix C, Part 4).

The areas estimated to have greater than 10 cm of scour total about 22 acres (Figure 5-11: and see Appendix C, Part 4, Scenario 5). Subsurface bed sediments (below the 10-cm depth) are generally more contaminated than surface sediments (0- to 10-cm depth). However, core data indicate that only a few areas have contaminant concentrations above the SQS or CSL in areas prone to natural erosion. The total area with surface exceedances above the SQS in areas with more than 10 cm of scour during high-flow events is 5.4 acres; of that, 1.5 acres are in the EAAs. In summary, empirical and modeling data indicate that the majority of subsurface sediment eroded will not have significantly higher contaminant concentrations.

5.3.2.6 Scenario 6: Movement of Existing Bed Sediment

This scenario was conducted to track the movement of sediment within the LDW. In the BCM, the initial COC concentration in the bed sediment at a given point is assumed to be unchanged through time. This means that the changes in COC concentrations at any given location are attributable only to the net sedimentation of upstream and lateral source sediments and mixing with bed sediments at that location. In actuality, bed sediments from other areas of the LDW are resuspended and settle throughout the waterway. The movement of resuspended bed sediment (distal sediment) and its effect on COC concentrations was evaluated by separately tracking the deposition of resuspended bed sediment and original bed sediment over time. This allows the COC concentration to change as a result of deposition of bed sediment as well as deposition of upstream and lateral source sediments. The STM analysis results are presented in Appendix C, Part 5 (LDW STM Bed-tracking Scenario Simulation).

The STM output was used in a BCM analysis with four contaminant inputs, one each for upstream, lateral, bed, and distal sediments. To account for the effect redeposition of sediment would have in a reach (the distal fraction), the total PCB concentration on resuspended sediment was based on a weighted average of the mass of sediment resuspended from each of the three reaches multiplied by the reach-wide SWAC for the reach where the sediment originated. For example, the PCB concentration associated with distal sediment from Reach 3 uses the SWAC from Reach 3 as the input value. This is an approximation that does not strictly conserve contaminant mass. However, it provides a check on the standard BCM analysis and shows the importance of resuspension and redeposition of bed sediment relative to other processes in the LDW on future SWACs. This analysis was conducted with the assumption that remediation of the EAAs had been completed.

This analysis indicates that accounting for bed sediment movement produces no substantial change to the total PCB SWAC at the end of 10 years, both on a site-wide and reach-wide basis (Table 5-5). The calculated total PCB SWAC, when this effect is considered, is unchanged in Reaches 1 and 3, and 6% lower in Reach 2. Site-wide, the decrease in predicted SWAC is approximately 1%. The changes are small because throughout the LDW, resuspended bed sediment that resettles in the LDW is a small component of the sediment mass balance. The resuspended bed sediment that settles in the LDW is only 5%, 12%, and 9% of the total mass of sediment depositing in Reaches 1, 2, and 3, respectively (see Appendix C, Part 5). In Reach 2, which has the highest fraction of bed sediment that resettles, most of the sediment that resettles originates in Reach 3, where total PCB concentrations are generally lower than in the other reaches. Overall, this simulation shows that redistribution of existing bed sediment by high-flow events has a minor effect on recovery predictions. The largest change is in Reach 2; however, the approach used in the BCM base case analysis likely underestimates natural recovery in Reach 2 compared to a model that actually tracks the movement of individual sediment particles.

5.3.2.7 Scenario 7: Natural Recovery Hindered in Selected Scour and Berthing Areas

In localized areas where high levels of routine ship activity occur and depths are sufficiently shallow to permit disturbance of the sediment bottom, natural recovery may still be occurring, but over longer periods.[20] Propeller scour from ordinary ship maneuvering activities temporarily resuspends surficial bed sediment, after which a portion of that material resettles in the same footprint, with the coarser material more likely to resettle and fines more likely to be transported away, depending on tides and currents. A constant source of incoming material from upstream also amends the bed sediment so that any exposed contaminant concentrations are reduced over time. Regular maintenance dredging in the navigation channel and active berthing areas indicates that net sedimentation is occurring and that sediment removal is required to maintain acceptable water depths for navigation. Empirical trends, where data are available, show that burial and sediment recovery are occurring in most of these areas (see Appendix F). Berthing areas were considered on a case-by-case basis during development of technology assumptions.

Some empirical data indicate that recovery may be hindered by normal navigation activities. These activities only rarely induce deep disturbance but, by continual resuspension of the unconsolidated surface sediment layer could reduce accumulation of layers of cleaner upstream sediments. To examine effects of such navigation activities on BCM predictions, a scenario was developed that assumes that natural recovery does not occur in areas considered prone to regular anthropogenic resuspension and transport of sediments (i.e., the berthing areas). At many of these locations, the STM

[20] This is for normal or routine operating conditions. See Section 5.3.1.2 for evaluation of extreme, episodic conditions.

indicates sedimentation during the recovery period. In this sensitivity analysis, the initial bed contaminant concentrations in potential scour areas and berthing areas are held constant for all BCM analyses throughout the 10-year period modeled in the BCM (i.e., no sedimentation and recovery). This assumption is the best available approach to bound uncertainty pertaining to effects of vessel scour on surface concentrations predicted by the BCM.

Areas held constant in this analysis were selected to include areas of potential scour from routine navigational activities: 1) berthing areas with net sedimentation rates less than 0.5 cm/yr (see Figure 2-11), and 2) vessel scour areas identified using sun-illuminated maps (Figure 5-4). This method has several limiting assumptions. Specifically, sun-illuminated maps are a snapshot in time of bed locations that have been disturbed by ship activity. The areas identified using this method may change in the future. Therefore, the selected areas for propeller scour are not intended as a robust indicator of all areas that may be influenced by propeller scour.

A BCM sensitivity was conducted over the 10 years following construction in order to compare the site-wide and reach-wide total PCB SWACs for the base case to a case with constant bed sediment total PCB concentration in potential scour areas.

Alternative 3: 10-Year Model Condition[a]	Total PCB SWAC (µg/kg dw)			
	Site-Wide	Reach 1	Reach 2	Reach 3
Base Case (includes modeled recovery in vessel scour areas)	62	68	61	39
Holding Cells Constant in vessel scour areas and berthing areas with net sedimentation rates <0.5 cm/yr	69	75	72	42

a. Exploratory test case condition at 10 years following remedy completion of Alternative 3 using mid-range BCM values, FS baseline data, and model assumptions used in the Draft Final FS.

This bounding exercise indicates that estimates of total PCB SWAC are not very sensitive to scour effects from normal operation of transiting vessel traffic. Vessel traffic can have some influence on SWACs (by hindering natural sedimentation and recovery), but this effect is less than a 25%[21] difference (considered the minimum detectable difference between SWAC estimates). For this scenario, the SWAC is about 10% higher for site-wide and reach-wide total PCB SWACs, except in Reach 2, which is 18% higher. However, scour and the resuspension of freshly deposited material may result in greater increases in localized areas and will need to be factored into remedial design in potential areas where natural recovery is hindered by vessel scour (see Section 6).

5.3.3 Influence of Grain Size and Organic Carbon on Sediment Chemistry

Hydrophobic compounds, such as PCBs, more readily adsorb to the organic substances attached to sediment particles than they do to the inorganic surface of sediment

21 A threshold of 25% is considered the minimum change needed to detect a difference between two SWAC values (see Section 9.1.2.1).

particles. As a result, the amount of organic carbon influences the potential adsorption of PCBs (and other hydrophobic COCs) to the particles. In addition, higher contaminant concentrations are generally associated with finer-grained sediment (clay/silt). This may be particularly important in the LDW as the grain-size distribution becomes finer from upstream to downstream (Figure 5-12), and the risk drivers are positively correlated with total organic carbon (TOC) and percent fines in the LDW (see Appendix C, Part 3b, Table 8).

Contaminant concentrations in the BCM were assigned equally to all grain sizes. In this evaluation, the sensitivity of the BCM is tested to determine the influence that size fractionation of COCs has on SWAC results. Total PCBs were assigned to the four STM particle size classes (Classes 1A [less than 10 microns], 1B [10 to 62 microns], 2 [62 to 250 microns], and 3 [250 to 2,000 microns]) in varying concentrations based on particle size (for additional details of this analysis, see Appendix C, Part 9). Three different partitioning approaches were used for assigning total PCB concentrations to the different particle size fractions (Table 5-6a). The results of the three analyses are shown in Table 5-6b.

Overall, this sensitivity analysis demonstrated that different approaches to assigning total PCB concentrations by size fraction did not substantially change the results for the BCM analysis unless the assumptions produced an increase in mass loading of total PCBs. For example, Approaches 2 and 3 demonstrated that the SWAC would decrease (14%) or remain approximately the same for cases where mass loading of the COC was not changed. This is because higher PCB concentrations are being assigned to Class 1A particles compared to the other size classes, but 90 percent of the Class 1A material passes through the LDW without settling. Approach 1 resulted in an increase in the site-wide SWAC by approximately 42% because the approach also increased the PCB loading from upstream and lateral sources by approximately 100%.

Preferential partitioning of contaminants to finer size fractions is well documented in the literature and can affect the distribution and bioavailability of contaminants. To account for this preferential partitioning, dry weight values are often normalized to the amount of organic carbon present in a sample (i.e., oc-normalization; Michelsen 1992). Many of the SMS contaminants have oc-normalized criteria.

5.4 Empirical Trends and STM/BCM Reliability

The reliability of the STM to estimate net sedimentation rates, and of the BCM to predict changes in contaminant concentrations, is supported by empirical trends (i.e., net sedimentation rates from time markers in cores and changes in contaminant concentrations over time). Consistency between empirically-derived net sedimentation rates and the STM and between the BCM and empirical trends in COC concentrations in surface sediments lends credibility to the STM/BCM prediction of natural recovery in the future. Contaminant trends in surface sediments were evaluated both by changes in risk-driver concentrations by depth in cores and by changes in their concentrations over time at resampled surface sediment locations. Appendix F presents these empirical data and the methods by which these data were evaluated. This section summarizes the findings presented in Appendix F.

Net sedimentation rates calculated from time markers (Pb210, Cs137, and contaminant peak dating) in cores that supported net sedimentation are in general agreement with rates estimated by the STM. Seven out of the 62 cores (11%) in the LDW provided no data on recovery rates, had low concentrations such that trends could not be determined, or indicated disruption to recovery. Chemical trends in most cores and at most resampled surface sediment stations show reductions in risk-driver concentrations over time. Both of these findings demonstrate that recovery is occurring in much of the LDW (as discussed and presented below for total PCBs, cPAHs, and other SMS contaminants). In areas either where these lines of evidence are not similar to one another or to the STM outputs, or where recovery is not predicted by the BCM, more attention is given to ascertain the reasons for these differences (see Appendix F). In some small-scale areas, these lines of evidence suggest that recovery is not occurring, and these areas are incorporated into assignment of recovery categories (see Section 6).

5.4.1 Net Sedimentation Rates

Net sedimentation rates were estimated from 74 cores for which time markers could be identified (Table F-3; Figure 5-13). These markers provide evidence of new material being deposited in the LDW, showing that burial, the dominant recovery mechanism, is occurring. The time markers were used to calibrate the net sedimentation rates estimated by the STM. STM calibration is discussed in Appendix F of the STAR report (Windward and QEA 2008). This analysis is also discussed in Appendix F of this FS. In the RI (Windward 2010), the depth of the peak total PCB concentration in each core was used to support the sedimentation rates estimated from the STM, and this analysis is discussed below in Section 5.4.1.1. Some cores indicated either no recovery or reduced recovery. The causes for these discrepancies are unclear. In some cases, the cores may not have been deep enough to show the time markers, concentrations were too low to detect trends, surface concentrations were too high from ongoing sources, or the area may have been previously dredged or otherwise disturbed. Deep disturbance may remove freshly deposited cleaner sediments or mix surface and subsurface sediments, resulting in exposure of higher contaminant concentrations at the surface.

System-wide statistical analysis suggests that the STM tends to underpredict sedimentation when compared to empirical data, and thus underpredict natural recovery potential. However, many of these sedimentation-rate underpredictions occur in Reach 3, which has very high sedimentation rates; thus, it does not influence model recovery predictions because both model and empirical data indicate rapid recovery. In Reaches 1 and 2, with less overall sedimentation compared to Reach 3, net sedimentation is sometimes underpredicted and sometimes overpredicted by the model. Several cores in these reaches did not have time markers preserved in the core profile from which to estimate sedimentation and recovery. Reaches 1 and 2 generally have lower empirically-derived net sedimentation rates compared to model predictions, as well as several cores that did not exhibit discernible recovery, and therefore the STM may somewhat over-predict recovery in these reaches. The base-case best-estimate STM predictions should be confirmed in localized areas during remedial design where MNR is being considered.

5.4.1.1 Vertical PCB Concentration Trends Compared to Net Sedimentation Rates

The PCB "peak" analysis presented in the RI (Windward 2010) combined information on depth patterns in PCB sediment chemistry (from sediment cores) with net sedimentation and erosion estimates from the STM to determine whether vertical patterns of total PCB concentrations are consistent with the STM's estimated net sedimentation rates and the CSM (Figure 5-14). Much of the sediment contamination in the LDW, and particularly PCB contamination, is believed to have originated from historical sources in the LDW.[22] In undisturbed depositional areas with no ongoing or recent sources, PCB concentrations should be higher in deeper core intervals than in shallower intervals. In areas with little or no deposition, localized disturbances, or ongoing or recent secondary sources (e.g., erosion of contaminated upland soil), this pattern may be altered, with higher PCB concentrations in the shallowest core intervals or relatively even distribution among core intervals.

Assuming that an area is depositional and has not been disturbed, the depth of the maximum total PCB concentration within a core should be a function of both the time since peak PCB use and release and the estimated rate of net sedimentation (from the STM). As a result, the expected depth of peak (or maximum) total PCB concentration was estimated for each core using Equation 5-2.

$$D = (T_c - T_m) \times S \qquad \textbf{Equation 5-2}$$

Where:

- D = expected depth of peak total PCB concentration (cm)
- T_c = year of core collection
- T_m = assumed year of maximum concentration in surface sediment, corresponding to the assumed peak in PCB use and releases to the LDW

22 Peak PCB use was recorded in Puget Sound sediment cores between 1960 and 1970 (Van Metre and Mahler 2005; Battelle 1997); the commercial production of PCBs was banned in 1978, and they were subsequently phased out. Although PCBs historically used in paints, caulking, and other products continue to be released into the LDW, it is believed these ongoing sources represent a smaller contribution to the LDW than historical releases.

- S = net sedimentation rate (cm/yr) estimated from the STM for the grid cell containing the core (or the closest grid cell for cores outside the STM domain).

General uncertainties associated with estimating the depth of the peak total PCB concentration include uncertainties in the net sedimentation rate estimated by the STM and uncertainty in the estimate of the year of the peak release of PCBs. In addition, uncertainty is associated with identifying the exact depth of the peak total PCB concentration within a core because of compositing within each core section. Uncertainty is particularly high at locations where the core intervals analyzed were 3 feet (ft) or greater and is lowest at locations where the core was sectioned into 0.5-ft intervals. Location-specific uncertainties include the possibility of sediment disturbance near berthing areas or local structures, and the potential for localized PCB releases to continue after the peak use/release date. To address the uncertainty in the year of maximum historical PCB releases to the LDW, a range of estimated depths of the peak total PCB concentration was calculated for each core (i.e., estimated depths within each core were calculated by assuming maximum PCB releases in 1960, 1965, and 1974).[23] These depth estimates were then compared to the depth of the peak total PCB concentration in each core. If the observed depth of the peak total PCB concentration was at or deeper than the estimated depth, the core was considered to be consistent with the CSM, and with the STM's estimated net sedimentation rates.

Of the 366 cores available in the RI dataset, 157 cores were used in the analysis and 209 cores were not used because the type of information needed for the analysis was not available for those cores. Cores were excluded if at least one of the following conditions were met:

- Only one core interval was analyzed for total PCBs
- No core interval was analyzed within the depth range of the expected peak
- PCBs were not detected in any core interval
- The sediment was disrupted by dredging prior to sampling.

Of the 157 cores included in the analysis, 110 cores (70%) had peak total PCB concentrations at depths equal to or greater than the estimated depths, consistent with the STM's estimated net sedimentation rates. Forty-seven cores (30%) had maximum total PCB concentrations that were shallower than the estimated depth range based on net sedimentation rates from the STM, or the concentrations were too diffuse to detect a significant peak at depth. For recovery estimates, the LDW model and field data are divided into three reaches. Reach 3 (the upper LDW) includes high rates of

[23] The analysis used both nationwide trends for PCB peak release (1960 and 1965; Van Metre and Mahler 2004; Battelle 1997), and the year of a PCB spill in Slip 1 (1974; Blazevich et al. 1977).

sedimentation and most maintenance dredging occurs in this reach. None of the cores in Reach 3 had maximum PCB concentrations at depths that were less than model predictions. Reach 2 includes both areas of high sedimentation and areas where no sedimentation was evident (net scour). Of the cores in this reach, 35% had maximum PCB concentrations at depths that were less than model predictions and 2% showed no discernible trend. Reach 1, which is near the mouth of the LDW, has lower sedimentation rates compared to Reach 3. Of the cores in this reach, 25% had maximum PCB concentrations at depths that were less than model predictions and 5% showed no recovery.

5.4.2 Chemical Trends at Resampled Surface Sediment Locations

Generally, chemical trends in resampled surface sediment locations show that recovery is occurring over much of the LDW, which supports the BCM findings of decreasing contaminant concentrations over time. Resampled surface sediment locations are surface sediment samples collected at different times from the same station (within 10 ft of one another). The contaminant concentrations in the LDW surface sediments have heterogeneous, but restricting the distance between older and newer locations to 10 ft reduces the uncertainty introduced by comparing samples from different locations. Appendix F describes the details, statistical results, and limitations associated with this type of comparison (analytical accuracy, etc.).

In the FS dataset, the data from 70 resampled stations (67 locations with 3 outliers excluded, and excluding those collected at the Norfolk Area and Duwamish/Diagonal EAAs) were grouped into two populations: older/original data and newer (FS baseline) data (see Table 5-7). The statistical difference between total PCB concentrations in these two groups was evaluated to provide evidence of general LDW-wide trends using simple data distributions. The comparisons of total PCB concentrations between the older and newer data show a 62% decrease in the mean value. As shown in Table 5-7, the 25th and 90th percentiles of these datasets also decreased by 31% and 64%, respectively, revealing that, in general, the empirical data support the STM findings that the LDW is recovering (at least for PCBs). Table 5-7 also summarizes these trends for arsenic, cPAHs, and BEHP. These data demonstrate that, on average, total PCBs, cPAHs, and BEHP concentrations are decreasing over time (more than or equal to a 50% reduction in concentration) while arsenic is in equilibrium (see Appendix F) and relatively close to urban background levels (see Appendix J).[24] For total PCBs and cPAHs, the mean for the older dataset is more than 20 times higher than their mid-range BCM upstream input values (Table 5-1a). For arsenic, the mean of the older dataset is only 4 times higher than the mid-range BCM upstream input value. This means that new sediment from upstream will have a greater effect on reducing concentrations of total PCBs and cPAHs over time than on reducing concentrations of

[24] The arsenic data have a narrower range of concentrations in the LDW than the other risk drivers, and are more similar to background conditions.

arsenic. Station-by-station results are presented in Appendix F for total PCBs, arsenic, cPAHs, BEHP, and SMS contaminants with detected exceedances in either the newer or older data.

5.5 Uncertainties Related to Predictive Modeling

The goal of an uncertainty analysis is to both qualitatively and quantitatively define the degree of confidence in site characterization data, both conceptual and predictive site models, and predictions of the results of remedial actions to the degree possible.[25] Bounding the certainty of estimates, especially in modeling, is a developing science. In accordance with an EPA guidance document (EPA 2005b), the potential areas of uncertainty to be identified and addressed in an FS include the CSM, data uncertainty, temporal uncertainty, spatial variability, and quantitative uncertainty. Several elements of uncertainty related to the predictive models (STM and BCM) are described below.

5.5.1 Net Sedimentation Uncertainty

Extensive sensitivity analyses were conducted on the STM and are described in detail in the STM report (QEA 2008). Sensitivity analyses were conducted on both high-flow event simulations and long-term, net sedimentation simulations. The net sedimentation sensitivity analysis showed that the model was most sensitive to the upstream sediment load and the settling speed of the fine-grain sediment classes, which make up the majority of the incoming sediment load. In this FS, because two, site-wide, independent datasets were not available for net sedimentation, uncertainty and sensitivity analyses both utilize the same input parameters. An appropriate measure for uncertainty in model predictions and application in this FS is the spatial scale analysis (QEA 2008; see Figure 2-13 from the STM Report). This analysis examined the accuracy of the model with respect to estimating net sedimentation rates from the large scale (LDW-wide) to the small scale (location-specific areas). This analysis found that the capability of the model was not affected by spatial scale (minimal bias), and that, on average, the model is able to estimate net sedimentation rates to within ±0.5 cm/yr on a typical net sedimentation rate of 1 cm/yr.

The incoming sediment load and depth of scour are affected by high-flow events. The STM used Green/Duwamish River flows from 1960 to 1989 as input flows. The maximum flow rate and upstream sediment loading for these years are shown on Figures 5-15a and 5-15b. The figures indicate that the upstream sediment load was

[25] Sensitivity analysis differs from uncertainty analysis in terms of goals and inputs. A sensitivity analysis looks at how the model responds to a range of input values, which may be extreme or not realistic, but are designed to stress the model and produce changes from the calibrated model results. Uncertainty analysis addresses the model's resolution, that is, its ability to replicate natural processes in light of unaccounted processes. Uncertainty analyses should be based on realistic and statistically defensible methods for developing a reasonable set of input parameters and conditions, which are then used to demonstrate a range in model results in order to inform decision-makers of potential model errors.

below average for the first 10 years of the simulation. Consequently, the STM and BCM may be conservatively predicting net sedimentation through the first 10-year modeling period.

The flow period represented in the STM (1960 to 1989) and shown on Figures 5-15a and 5-15b is representative of current conditions. Annual precipitation since 1989 and up to the present has not changed significantly. Global warming is also not expected to change average annual precipitation significantly (Mote and Salathe 2009). By the late 1990s, when the U.S. Geological Survey (USGS) sediment loading study was conducted, the Green/Duwamish River basin was already under control by the Howard Hanson Dam and heavily developed with agricultural, urban, and suburban land uses. For these reasons, Green/Duwamish River flows and sediment loads are not expected to change substantially in the future as long as the river flow continues to be dam controlled in a manner generally consistent with historical water management practices.

5.5.2 STM Uncertainty – Lower and Upper Bound Simulations

The effects of uncertainty in STM inputs on model estimates were analyzed and quantified in the STM report (QEA 2008; see Section 2.8 and Appendix D.6 of the STM). The results of the input parameter sensitivity analysis were used to generate reasonable lower- and upper-bound limits on the base-case results, which are based on the calibration parameter set. The upper- and lower-bound cases were a result of changing the upstream sediment loading and settling speed of Class 1A and 1B solids. The base-case upstream loading rates were developed from two USGS studies to provide a good estimate of the magnitude of Green/Duwamish River input to the LDW, and the Class 1A and Class 1B settling rates were selected during the STM calibration process because they were reasonable and because they best match the empirically-derived LDW net sedimentation rates. Therefore, the values for these two model input parameters in the STM base case were reliably defined by site-specific data and model calibration.

The base-case simulations provide the best estimates of net sedimentation rate, but the reasonable lower- and upper-bound simulations provide an acceptable range of net sedimentation rates resulting from uncertainty in model inputs, with the “true” value of net sedimentation rate being within this range. As noted in Section 5.4.1, field sedimentation data are sparse and variable by reach and location, and the STM predictions will need to be confirmed for areas where MNR is proposed during remedial design. The highest empirically-derived net sedimentation rates occur in Reach 3 and were higher than model predictions; therefore, the STM may under-predict recovery there. Reaches 1 and 2 generally have lower empirically-derived net sedimentation rates compared to model predictions, as well as several cores that did not exhibit discernible recovery, and therefore the STM may somewhat over-predict recovery in these reaches.

To demonstrate the effect of model parameters on long-term changes in bed composition, the upper- and lower-bound results have been analyzed and used to estimate uncertainty in the predicted half-time of bed-source content[26] in surface-layer (0 to 10 cm) sediment for the long-term, multi-year (e.g., 21-year calibration period) simulations. Half-time values of bed-source content in surface-layer sediment were estimated using relationships between net sedimentation rates and half-time values developed from model results presented in the STM report (QEA 2008). The approximate relationship between half-time of bed-source content and net sedimentation rate can be used to estimate the spatial distributions of half-time and recovery potential if the starting concentrations are known.

Generally, the half-time of bed-source content in surface-layer sediment tends to decrease as the net sedimentation rate increases, see Section F.2 and Figure F-37 of the STM report (QEA 2008). In general, most areas have a half-time of less than 10 years based on net sedimentation rates of 1.0 cm/yr or more. This analysis indicated a general trend of decreasing half-life of bed-source content with an increasing net sedimentation rate. Spatial distributions of the net sedimentation rate for the lower- and upper-bound simulations are shown in figures in Appendix C, Part 6. The best-fit model prediction from the bounding exercise is about 5 to 10 years (±5 years if the net sedimentation rate is more than 1 cm/yr and longer with lower net sedimentation rates). Because the bounding exercise does not represent the calibrated dataset, this characterization of uncertainty is more appropriate for those regions farther from the locations where the model was calibrated. Areas near calibrated locations have significantly lower levels of uncertainty. This level of uncertainty is acceptable for the FS. The uncertainty in the reasonable lower- and upper-bound STM runs and its effect on PCB concentrations are discussed in Section 5.5.4.

5.5.3 Uncertainty around the BCM Contaminant Input Values

For the BCM, uncertainty exists in the assumptions about contaminant concentrations in lateral and upstream sources (from both non-point and point sources). This uncertainty will exist well into the future based on the variable nature of these sources, but is managed by expressing BCM inputs as a range of concentrations (low, high, and best-estimate values). These input values are based on actual data collected over the past 20 years. BCM uncertainty is managed by bracketing the best-estimate BCM value with lower- and upper-bound BCM input values representing the mean, UCL95, or percentiles of the existing data. For the lateral inputs, the low and high estimates are meant to capture a range of uncertainty associated with potential future source control measures.

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[26] The half-time is defined as the time needed for 50% of material in the initial surface layer (0 to 10 cm) of the sediment bed to be replaced with depositing sediments.

These input values were estimated from summary statistics for various datasets (surface water, surface sediment, in-line sediments, catch basin solids, etc.). Each dataset has some degree of sample uncertainty associated with it, relating to aspects such as the matrix from which the sample was collected, the location from which the sample was collected, the differences in TOC and grain size among the datasets, the time (season, river flow, portion of storm event [e.g., first flush]) of sample collection, ongoing source control efforts, and other aspects that can affect contaminant concentrations in a sample. The high end of the range (high lateral, high Green/ Duwamish River, and high post-remedy bed sediment replacement values) is intended to capture variability in the source concentrations, worst-case recontamination potential, and regular, seasonal high flows from urbanized areas. The low end of the range (low lateral values, low Green/ Duwamish River, and low post-remedy bed sediment replacement values) represents a non-conservative set of assumptions that is considered likely to underestimate future contaminant concentrations. The probability that site conditions will produce a high-high-high contaminant concentration (lateral, Green/Duwamish, bed) is likely very small.[27] A similar low probability of occurrence exists for the low-low-low end of the range.

Another source of uncertainty related to lateral inputs is the fact that lateral contributions to the LDW can come from many different sources, including storm drains, CSOs, surface water runoff, and atmospheric deposition anywhere along the LDW and in its drainage basin. These sources were aggregated into 11 waterfront areas and 16 discharge points to the LDW for the purposes of sediment transport modeling. Of these, only the CSOs have measured discharge flows; runoff flows are estimated for other discharges. Some localized discharge points may not be adequately characterized by the 11 general waterfront areas. In addition, CSO control plans will result in reduced flows in the future for many CSOs.

Similar uncertainty exists for the post-remedy bed sediment replacement values used as input in the BCM. These values represent the bed sediment contaminant concentrations in the near-term (0 to 2 years) following completion of active remediation, including influence from multiple recontamination mechanisms. Evidence from other sediment sites shows that post-construction COC concentrations become higher than detection limits and natural background after this initial time frame. Limitations in the

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27 The likelihood of occurrence for the high-high-high contaminant concentration (lateral, Green/Duwamish, bed) is the product of the likelihood of each occurring independently. The likelihood of the upper bound representing the contaminant concentration for either upper, lateral, or bed source material is small. Therefore, the likelihood of all three upper bounds occurring is much smaller. It should be noted that a contaminant concentration value for any of these three variables that is higher than the medium, but less than the upper bound is not small. One can expect that the probability of occurrence of any combination is highest for medium-medium-medium and decreases moving toward either upper-upper-upper or lower-lower-lower combinations. The shape of this distribution is unknown.

dredging/capping equipment leave behind dredging residuals that resettle within the remedial footprint. Residual COC concentrations are typically proportional to the average COC concentration of the dredged material, and typically higher than the COC concentration in surrounding sediments (see Section 9 for a discussion on dredging residuals for each alternative). Post-construction surface sediments in the LDW may come into equilibrium with the sediments surrounding the remediated area. The equilibrium concentration of COCs in the sediment bed may be higher than the COC concentration in upstream sediments because of increased urbanization as one moves downstream toward downtown Seattle (more cars, vessel traffic, non-point sources, air emissions, accidental spills, and storm drain runoff). To address this uncertainty, the best-estimate for the post-remedy bed sediment replacement value is bracketed by low and high BCM input values that are a combination of clean backfill material (based on natural background concentrations) and the surrounding unremediated sediments, assuming various proportioning percentages, as described in Section 5.2.3.4. In addition, the effect of the post-remedy bed sediment replacement values on predicted total PCB concentrations for selected alternatives is presented in Appendix M.

By using many lines of evidence and a range of input values derived from these data, some quantitative analysis of the uncertainty is provided, and confidence in the model representing long-term conditions over time is increased. However, it is also uncertain how these input concentrations may change over time. In summary, these BCM input values are considered adequate for the purposes of assembling remedial alternatives (Section 8) and evaluating the short- and long-term effectiveness of the alternatives (Section 9) in the FS.

5.5.4 Combined STM and BCM Uncertainty

Both the STM and BCM have uncertainty associated with model input values, process descriptions, and discretization. Uncertainty in STM predictions that results from uncertainty in the input parameters was extensively examined in the STM report (QEA 2008). The uncertainty analysis in the STM report was used to develop reasonable and maximum upper and lower bounding simulations. The reasonable upper- and lower-bound simulations provide a realistic range of net sedimentation rates for the LDW and were used to examine the effect of STM uncertainty on BCM results. The maximum simulations were considered unrealistic and not carried forward in the BCM uncertainty analysis. The results from these bounding simulations are discussed in Section 5.5.2 and in Appendix C, Part 6. Uncertainty in the BCM chemistry input values is discussed in Section 5.5.3.

The STM base-case composition results were taken at the end of the 10-year model run for reasonable upper and lower bounding simulations as input to the BCM to compute the total PCB SWAC for each simulation following remediation of the EAAs. This analysis is presented in Appendix C, Part 6. The STM bounding simulations are presented in Section 2.8 of the STM report (QEA 2008). Reasonable upper and lower bounds are defined as net sedimentation rates that varied by ± 1 cm/yr from the STM base case. This provides a greater than 95 percent confidence interval around the data.[28] The reasonable lower to upper STM simulations produced a range in total PCB SWACs from 65 to 101 µg/kg dw or about -16% and +31% from the base case prediction, respectively (see Appendix C, Part 6, Table 5). However, the STM base case (with lower to upper BCM input values) produced a range in total PCB SWACs from 49 to 122 µg/kg or about -36% and +58% from the base case prediction, respectively. The analysis showed the total PCB SWAC is more sensitive to the range of BCM chemistry input values than it is to the range of net sedimentation rates from the reasonable upper and lower bounding STM simulations. Although the SWAC range based on BCM bounding is greater than the range based on STM bounding, both are still sufficiently large that they must be accounted for in future assessments. The range of total PCB SWAC values attributable to STM and BCM uncertainty is illustrated in Appendix C, Part 6, Figure 11.

5.5.5 BCM Input Values for Other SMS Contaminants

A total of 41 COCs with SMS criteria were identified for the protection of benthic invertebrates. It was not practical to run the BCM 41 times to evaluate recovery potential for every SMS contaminant. Therefore, a smaller subset of representative contaminants was selected because:

- Many co-occur with other SMS contaminants (e.g., PAHs)
- Groups of contaminants have similar modes of toxicity (e.g., phthalates)
- Lateral source data have not been collected, or at least compiled, for every contaminant
- Many of these COCs do not have widespread SQS exceedances in the LDW.

Application of the BCM using representative SMS contaminants is based on the fact that the representative contaminants account for the majority of the SQS exceedances and the assumption that all SMS contaminants within a group will behave/recover in a similar manner. Uncertainty exists with this simplifying assumption. In reality, each SMS contaminant may have a different starting concentration, recovery and/or recontamination potential, sediment-water partitioning dynamics, bioavailability based on organic carbon content, and lateral and upstream sources. Estimated exceedances of the SQS and CSL at the end of the 10-year modeling period may be biased high or low relative to the representative SMS contaminant predictions. This uncertainty will be managed during remedial design and by refinement of the CSM for remedial areas.

[28] The confidence interval for the reasonable upper and lower bounds was not specifically defined in the STM analysis. However, the 95 percent confidence interval was defined as a net sedimentation rate of ± 0.5 cm/yr.

5.5.6 Age and Spatial Extent of Contaminant Data

Over the past 18 years, numerous investigations have been conducted to determine the nature and extent of sediment contamination associated with past and present contaminant releases at various locations within the LDW. These investigations have included in-water investigations involving surface and subsurface sediment sampling, toxicity testing, shoreline habitat inventories, seep surveys, and porewater sampling. These data have been aggregated into the FS baseline dataset. There is uncertainty associated with these data related to detection limits that exceed the screening criteria, especially in older data; contaminant compositing with depth; and interpolation between sampling points. An additional large source of uncertainty is the age of the data. Many of the surface sediment data comprising the FS baseline dataset are over 10 years old and do not represent current conditions. Active remedial technologies are being assigned to particular areas based on surface sediment exceedances that may have improved (or worsened) over the past few years. Because the CSM and empirical data have shown that the LDW is recovering (in many areas), there is likely a high bias introduced into the assembly of alternatives. Remedial alternatives are being assembled on fairly conservative assumptions that no recovery has occurred between when the data were collected and now. This source of uncertainty is being managed in two ways: 1) the modeling is conservative and does not account for 10 years or more of potential recovery from when the sample was collected; and 2) areas with older data, but which are predicted to recover, will be subject to verification monitoring (see Sections 6 and 8) to confirm current contaminant concentrations and degree of recovery. Other sources of data uncertainty such as vertical and horizontal extent of contamination, elevated detection limits, and SMS compliance may also be refined during remedial design.

5.5.7 Chemical Degradation and Transport Processes

Many of the LDW risk drivers (total PCBs, cPAHs, BEHP, arsenic, dioxins/furans, and other SMS contaminants) have similar fate and transport properties in that they are strongly bound to sediment particles and do not readily degrade. Compounds that readily degrade or desorb from sediments are not persistent in sediments because the concentration declines naturally over time. Persistent contaminants cause long-term sediment contamination. The following discussion focuses on PCBs because a large body of research exists for this COC at many sites across the country. However, for most of the COCs, degradation and desorption processes decrease the concentrations in sediment over time. By not accounting for these processes, the analysis is conservative with respect to sediment contamination and natural recovery because it will overestimate both long-term sediment concentrations and the time required for natural recovery to occur.

PCBs, in particular, are stable compounds that do not degrade easily. Under certain conditions, they may be broken down by chemical, thermal, and biological processes (Erickson 1986). In the environment, photolysis (breakdown by light) is the only significant chemical degradation process, but it is not likely a significant means of PCB losses from sediments because of low PCB solubility and limited penetration of sunlight into the solid media (the sediment bed) (Hutzinger et al. 1974). Microbial processes are the main route of environmental degradation of PCBs in sediments. Reductions in the sediment concentrations of PCBs can happen via desorption from sediments into the overlying water column and volatilization. The breakdown of PCBs is generally discussed below, and implied for many other risk drivers; it is assumed to be occurring in the LDW, although these processes have not been modeled in the FS. Changes in PCB concentrations in the sediment bed can be translated into predicted concentrations of PCBs in fish and shellfish tissue via the PCB food web model (FWM) developed for the LDW (Appendix D of the RI, Windward 2010; see Section 9.3.5.2 of this FS for a discussion of uncertainties associated with FWM estimates). Section 3 evaluated whether varying water concentrations account for the effects of desorption and how other inputs into the water column would affect tissue concentrations (see Figure 3-2).

The King County model was used to predict contaminant concentrations in the water column; it employed containment flux from the sediment bed to estimate desorption of PCBs into the water column.

The effects of varying PCB concentrations in the water column and the site-wide sediment SWACs on predicted residual risks from seafood consumption are discussed in Section 9; results are presented in Appendix M.

5.5.7.1 Microbial Degradation

The viability of biodegradation as a natural method of sediment recovery for sediment-bound PCBs has been documented in several studies (RETEC 2002; Appendix F).

PCBs can undergo microbial degradation in natural environments under both aerobic (i.e., in the presence of oxygen) and anaerobic (i.e., in the absence of oxygen) conditions. PCBs are a class of 209 individual contaminants (PCB congeners), in which 1 to 10 chlorine atoms are attached to a biphenyl molecule. Most Aroclors (commercially produced groups of PCBs) contain 60 to 90 different PCB congeners, with varying numbers and positions of the chlorine atoms on the biphenyl rings.

Microbes degrade PCBs by breaking the carbon-to-carbon bond of PCBs, or by substituting the chlorine atoms with hydrogen atoms in the PCB molecule under aerobic and anaerobic conditions, respectively (McLaughlin 1994). The latter method results in the transformation of PCB congeners into other less chlorinated PCB congeners in a process called dechlorination (Abramowicz 1990). Aerobic degradation, on the other hand, results in a net PCB loss from a given PCB inventory. In river sediments, aerobic conditions are typically found in the top few centimeters of the sediment bed, while anaerobic conditions are found at greater depths below the sediment surface.

Aerobic Degradation

Even though laboratory studies have documented the existence of naturally occurring aerobic bacteria capable of degrading a large spectrum of PCB congeners, there is little direct evidence indicating that the aerobic degradation process is effective at reducing the PCB mass under field conditions. In laboratory studies of the Hudson River, PCB losses were highest in the less chlorinated congeners (43 to 47% reduction) and lowest in the more chlorinated congeners (17 to 5% reduction) (Harkness et al. 1993 and 1994). The in-field studies yielded similar results (less than 50% reduction). A study of PCB patterns in Green Bay sediments suggests that aerobic degradation is not a significant transformation mechanism for those sediments (McLaughlin 1994).

Anaerobic Dechlorination

Reduction through dechlorination (under anaerobic conditions) is generally viewed as a viable means of biodegradation for numerous compounds, including PCBs at higher concentrations. This process can alter the toxicity of these compounds and make them more readily degradable. The extent to which PCBs can degrade depends on several factors (Bedard and Quensen 1995), including the nature of the active microbial population, the type of chlorine substitution, the chlorine configuration, the initial PCB concentration, and the substrate conditions (temperature, redox conditions, ionic strength, amount of carbon, and presence of other oily contaminants, etc.). For example, no anaerobic dechlorination of PCBs was observed in the downstream deposits of the Fox River where the maximum PCB concentration was approximately 30 mg/kg dw (limited effectiveness at lower concentrations). Dechlorination activity was limited to sediment PCB concentrations of 30 mg/kg dw or greater (McLaughlin 1994). The overall PCB loss due to microbial degradation in several Fox River sediment deposits was estimated to be less than 10% with respect to the original inventory of PCBs deposited in the river.

A similar threshold for degradation of 50 mg/kg dw was observed in Sheboygan River sediments (David 1990). For Grasse River sediments (Minkley et al. 1999a, 1999b), some dechlorination activity was suggested at total PCB concentrations below 7 to 10 mg/kg dw, but the statistical evidence of dechlorination was less strong than at higher concentrations. Attempts in a laboratory study to further dechlorinate Fox River sediments met with limited success and similar results, up to 10% dechlorination on a total chlorine basis (Hollifield et al. 1995).

In the Fox River, physical loss through desorption from sediments (into the water column) exceeded any biodegradation in the sediment. It was estimated that 33% of the original PCB mass originally deposited in the Lower Fox River was lost due to desorption.

5.5.7.2 Volatilization and Desorption

Volatilization and desorption remove contaminants from sediment particles without changing the chemical make-up of the contaminant. In desorption, the contaminant is removed from the sediment and becomes dissolved in water. Volatilization is the process of a contaminant going into the gaseous state and being released to the atmosphere.

Both of these processes are relatively weak for the COCs in the LDW. For instance, all of the inorganic compounds (with the exception of mercury) and low molecular weight PCBs generally do not undergo volatilization. For PCBs, volatilization into the air can be important in shallow arable soils, but less so for subsurface soils (Meijer et al. 2003). Limited volatilization of some organics could occur from exposed intertidal sediments at low tides, but this transport mechanism would be further limited by the high water content of the sediments. COCs may diffuse from sediment into porewater and then into the water column and/or atmosphere, but these transport pathways occur at very slow rates. Because subtidal sediments are covered with water and are not in contact with the atmosphere, a very limited amount of volatilization occurs from dissolved PCBs in the water column, rather than directly from sediments. Consequently, volatilization is not considered a major process in the dynamics of PCBs or other COCs in LDW sediment.

Desorption is related to how strongly a contaminant binds to sediment or to organic carbon in sediment. All of the COCs in the LDW strongly bind to sediment. If the COCs did not bind strongly to sediments, they would have desorbed, become dissolved in surface water, and have been discharged downstream, effectively removing them from LDW sediments. Empirical evidence demonstrates the persistence of these contaminants with depth in the LDW. Many of the organic compounds, such as PCBs, PAHs, and dioxins/ furans, are referred to as hydrophobic compounds. That is, the compounds preferentially partition to solids rather than become dissolved in water.

By not including volatilization and desorption in the natural recovery analysis, estimated future contaminant concentrations in sediment are conservative because these processes should slightly accelerate the predicted natural recovery in surface sediments.

5.5.8 High-Flow Scour Potential

As discussed in Sections 5.3.1 and 5.3.2.5, the maximum scour depth during a 100-year high-flow event is estimated by the STM to be about 22 cm for the base case, and the upper bound of estimated scour is 36 cm, based on upper-bound erosion sensitivity simulations. Areas with subsurface sediment contamination located in potential scour areas, whether from high-flow events or propeller scour, are explored in Section 6 and are included in the AOPC footprints. Scour areas defined in Section 2.3.1.1 and illustrated in Figures 2-9 and 2-10 were used to assign recovery categories in Section 6.

Section 5.3.2.5 illustrates that potential exposure and transport of subsurface sediments during high-flow events is small compared to the incoming sediment loads. To explore the net effect of propeller scour events, Appendix F illustrates that empirical chemical trends from many of the resampled surface sediment stations and sediment cores have decreasing contaminant trends (or trends in equilibrium) in scour areas. The FS assumes that scour potential (less than 10 cm) in areas with subsurface exceedances of SMS criteria is of concern even if empirical evidence indicates that some recovery and scour areas with adequate net sedimentation rates and water depth may eventually recover. Uncertainty related to scour potential with subsurface exceedances is inherently accounted for in Section 6. Areas with subsurface exceedances in potential scour areas are included in the AOPC footprints for the FS, and these areas are given equal consideration as surface exceedances in the assembly of alternatives and assignment of remedial technologies to those areas (Section 8). Active remediation is assigned to scour areas (within the depth of scour potential, typically RAL exceedances in the upper 2 ft) in the absence of empirical trends showing recovery.

A sensitivity analysis was conducted to evaluate uncertainty in STM predictions that may have resulted from uncertainty in model input parameters, including those that control erosion rates. Uncertainty in the extent of areas estimated to have erosion was less than ±50% within the area from RM 0.0 to 4.3, relative to the base-case simulation. Uncertainty in predicted sediment mass eroded ranged from about -50 to +75% within the area from RM 0.0 to 4.3 as well as in the east bench and navigation channel, and ranged from -40 to +130% in the west bench. The analysis showed that the predicted depth of scour, area of scour, and mass of sediment scoured are not very sensitive to erosion rate parameters used in the model.

5.5.9 Anthropogenic and Natural Deep Disturbance Uncertainty

Section 5.3.1.3 introduces the potential for both anthropogenic and natural disturbance of subsurface sediments in the LDW that may result in contaminant exposure. These subsurface sediments are an additional potential source of contaminant mass to LDW surface sediments, similar to upstream and lateral loadings. The RI did not extensively characterize subsurface contaminant concentrations. In addition, deep disturbance is inferred in some geochronologic and chemical records. However, these data are sparse relative to the size of the study area and the frequency, cause, and magnitude of deep disturbances cannot be estimated with confidence. The data can, however, provide general, first-order estimates of bounds on reasonable minimum and maximum acreages of continuous disturbance (0 to 45 acres). These acreage bounds are used to bound the possible effects on the predicted total PCB SWAC. This analysis is provided in Appendix M, Part 5, and results are discussed in Sections 9 and 10.

The approach used for this analysis is based on some assumptions that will overestimate the predicted SWAC with time. Specifically: 1) the same area is assumed to be repeatedly disturbed (e.g., perhaps a tug regularly has trouble maneuvering a barge into a particular spot); 2) there is no mixing of ongoing sedimentation with deeper sediment during a deep disturbance event; and 3) the subsurface concentrations never change. These conditions were not factored into the analysis and would mitigate some of the increases in SWAC predicted in the analysis. In addition to change in the SWAC, ongoing deep disturbances could result in longer recovery times being required to achieve the cleanup objectives.

5.5.10 Bathymetric Changes and Dredging of Upper Turning Basin Sediments

A hydrodynamic model was used to generate flow velocities, which were then used in the STM. The hydrodynamic model was not revised for changes in bathymetry due to scour or net sedimentation. However, the STM does track the changes in bed elevation over time as sediment is scoured or deposited. Analysis of specific model cells in the navigation channel and on the benches shows that the change in bed elevation in the first 10 years of the simulation is on the order of 10 cm (4 inches). This change in bathymetry would not be expected to affect the hydrodynamic model because the water depth is much greater than the change in bed elevation.

In Reach 3, the Upper Turning Basin has much more net sedimentation than Reaches 1 and 2. However, the Upper Turning Basin is regularly dredged. By ignoring the changes in bathymetry due to deposition in the Upper Turning Basin, the model essentially assumes that the Upper Turning Basin is continually dredged. If the hydrodynamic model and STM were modified for bathymetric changes between dredging events, the Upper Turning Basin would become shallower and more sediment would move downstream, resulting in higher net sedimentation rates downstream of the Upper Turning Basin. However, the hydrodynamic model does not consider the hypothetical possibility of a cessation of dredging at the Upper Turning Basin, and therefore retains the present mass inputs and grain-size distribution into the future.

5.6 Modeling Summary and Conclusions

In summary, predictive modeling is a useful tool for the FS to evaluate the value or effectiveness of remedial alternatives and the recovery potential of the system. Some alternatives will include MNR and others will not (see Section 8). The STM and BCM support decision-making regardless of which remedial alternative is selected. However, it is understood that both tools have a large degree of uncertainty (see discussion following bullets). For the purposes of the FS, a bounded margin of uncertainty is acceptable, but this FS assumes that this uncertainty can be further managed during remedial design and future monitoring. The modeling presented in this section concluded that:

- The LDW is net depositional over time and its physical characteristics and natural processes are reasonably well understood through fine-scale hydrodynamic and sediment transport modeling. The STM output has been supported by several lines of evidence, including chemistry profiles in sediment. Areas where the STM output doesn't match empirical data are generally found in locations with features and activities that the STM didn't incorporate (e.g., bridges and pilings, high-powered ship maneuvering, and other berthing activities).Three key outputs from the STM are used in the FS: net sedimentation rates, areas subject to scour from high-flow events, and bed composition. The third output provides the framework for predictive contaminant modeling in the BCM.

- Sediment is continually depositing within the LDW. Almost all new sediment (99%) that enters the LDW originates in the Green/Duwamish River system. The STM estimates that, on average, over 185,000 MT of sediment per year enters the LDW, with approximately 100,000 MT depositing in the LDW. Approximately 90% of the total bed area in the LDW receives 10 cm of new sediment within 10 years or less. This sediment is mixed with the existing bed sediment through various processes, including bioturbation and propeller wash. On average, the annual volume dredged over the past 15 years is approximately 51% of the deposited sediment load. An annual average of approximately 38,000 MT has been dredged within the authorized navigation channel and 13,000 MT within the berthing areas, for a total annual dredge volume of about 51,000 MT.

- Overall, the maximum net erosion depth during a 100-year high-flow event is approximately 22 cm, with most areas experiencing less than 10 cm of scour, while 82% of the LDW experiences net deposition rather than net erosion over the 30-year model period.

- The effects of propeller-induced bed scour are incorporated into the present structure of the LDW sediment bed because ship movement has been occurring for at least the past 40 years. Propeller-induced bed scour from transiting ships and typical berthing activities is viewed as an impulsive erosion-deposition process that tends to behave like an ongoing mixing process for surficial bed sediment. Transiting ships in the navigation channel are not a major source of sediment transport or erosion in the LDW, except where slightly greater erosion depths (net erosion) are possible in shallower areas adjacent to the navigation channel. However, the analysis of scour prepared for this FS does not consider some possible irregular events. These events, outside of normal operating procedures, may include emergency and high-power maneuvering of tug boats under unexpected conditions, high-powered navigation activities, ships running aground, seismic events, and disturbance resulting from riverine structure maintenance construction/repair. Such events are likely infrequent relative to ships transiting the LDW, but could result in deep disturbances that affect long-term SWACs and hinder natural recovery. These events can disturb subsurface sediments and mix subsurface contamination with the surface layer. A series of post-STM/BCM analyses were performed to address the potential importance of both routine navigation activity and episodic, high-powered navigation and maintenance construction/repair events on long-term SWACs (Appendix M, Part 5). These analyses indicate that long-term recovery and SWACs could be influenced by navigation and riverine activities in the LDW, with the magnitude of the impact dependent upon the frequency and extent of the disturbance event.

- The BCM estimates changes in risk driver contaminant concentrations over time. Output from the BCM includes contaminant concentrations (point concentrations and area-based SWACs) at 5-year increments for 45 years.

- Empirical data show that, on average, LDW surface sediment contaminant concentrations are decreasing over time, consistent with BCM predictions of surface sediment concentrations approaching equilibrium over time. Appendix F shows specific locations where the empirical data demonstrate recovery. However, recovery can be locally hindered by vertical mixing of surface and subsurface sediments disturbed by anthropogenic and natural activities.

- Contaminant input values used in the BCM (lateral source, Green/ Duwamish River upstream, and post-remedy bed sediment replacement) were derived from actual input data (catch basin solids, sediment trap samples, upstream surface sediment and surface water data, USACE sediment cores) from the Upper Turning Basin. A range of values (high-medium-low) are used to address uncertainty and potential temporal variability in the range of contaminant inputs associated with each source type.

- Both the BCM predictions and empirical contaminant trends show that natural recovery is occurring in some areas of the LDW (see Appendix F). According to the BCM, MNR is a viable technology for many (but not all) areas of the LDW with moderate levels of contamination (below the CSL), net sedimentation rates of more than 1 cm/yr, and minimal scour potential (see Section 6).

- The BCM uses the FS baseline dataset (where the data are already more than 10 years old in some areas) and assumes no recovery or age-consideration for the older data in existing bed sediments; therefore, the initial bed contaminant concentrations at the start of construction may be lower than estimated in the BCM.

- The STM and BCM are not contaminant fate and transport models, and the numerous assumptions made throughout model development were designed to provide reasonable estimates with respect to predicted sediment concentrations based on available data for model development. Many assumptions used to develop model input and process descriptions are conservative. For example, the models assume no chemical transformation or degradation over time. Mass is not conserved in the BCM; however, additional analyses presented in Appendix C were used to investigate the significance of this on predictions of natural recovery. Changes in tissue and surface water COC concentrations are predicted as sediment concentrations change (i.e., through burial, scour, and resuspension processes). Changes in seafood consumption risks are evaluated for each remedial alternative in Section 9 via the PCB FWM developed as part of the RI (Windward 2010).

- The BCM may underestimate potential COC concentrations in localized areas near active discharges due to variation in loading estimates among the outfalls. These localized areas should be evaluated for adequate source control during remedial design.

Uncertainty in both the STM and BCM is recognized in sedimentation rates, erosion depths, scour areas, and contaminant inputs over time. Varying levels of confidence can be attached to these model predictions depending on: 1) the COC (i.e., arsenic has a higher level of certainty compared to PAHs, which may have increasing concentration trends from urbanization) and 2) the location in the LDW (areas with estimated net sedimentation greater than a few centimeters have a higher expectation that natural recovery will occur because the estimated net sedimentation is much greater than model error). By using many lines of evidence and a range of input values derived from these data, the uncertainty can be bounded. Overall, the uncertainty in BCM contaminant concentration input parameters has a slightly greater effect on predictions of natural recovery than does the uncertainty in sedimentation rates. Therefore, the ranges of STM and BCM input parameters are useful tools to bracket uncertainties in the evaluation of FS alternatives. Regardless, monitoring will be needed to confirm that recovery is occurring wherever MNR is proposed.

Finally, the BCM analysis is considered adequate for estimating future COC concentrations in LDW sediments (combined with the analysis of deep disturbances and exposure of subsurface contamination in Appendix M, Part 5), assigning a range of suitable remedial technologies (Section 8), and evaluating short-term and long-term effectiveness of remedial alternatives (Section 9). Model uncertainties and limitations do not negate the use of the model as a predictive tool in this FS, but must be accounted for when considering the predicted outcomes of the remedial alternatives, as discussed in Sections 9 and 10. Sections 9 and 10 also include additional detailed analysis of the effects of deep disturbance induced by anthropogenic and natural activities on long-term SWACs. Spatial areas where model predictions agree or do not agree with empirical trends and physical site conditions are accounted for in the FS in the designation of recovery categories (Section 6).

Table 5-1a Bed Composition Model Upstream Input Parameters for Human Health Risk Drivers

Rationale

Range of concentrations considered representative of current and potential future conditions for solids entering and settling in the LDW from upstream. Four different datasets used to establish range of parameter values for upstream sources because of potential biases inherent to each.

Contaminant	BCM Parameters			Basis for BCM Upstream Input and Sensitivity Values[a]
	Input	Low	High	
Total PCBs (µg /kg dw)	35	5	80	**Input:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data (36 rounded to 35 µg /kg dw). **Low:** The mean of Ecology upstream sediment samples containing fines >30%. **High:** UCL95 of TSS-normalized King County (whole-water) (82 rounded to 80 µg /kg dw).
Arsenic (mg/kg dw)	9	7	10	**Input:** Mean of Ecology upstream sediment samples containing fines >30%. **Low:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data. **High:** UCL95 of Ecology upstream sediment samples with fines >30%.
cPAHs (µg TEQ/kg dw)	70	40	270	**Input:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data (73 rounded to 70 µg TEQ/kg dw). **Low:** Mean of Ecology upstream sediment samples containing fines >30% (37 rounded to 40 µg TEQ/kg dw). **High:** UCL95 of TSS-normalized King County (whole-water) (269 rounded to 270 µg TEQ/kg dw).
Dioxins and Furans (ng TEQ/kg dw)	4	2	8	**Input:** Midpoint between mean of Ecology upstream centrifuged solids and mean of Ecology upstream sediment samples containing fines >30% **Low:** Mean of Ecology upstream sediment samples containing fines >30%. **High:** Midpoint between mean and UCL95 of Ecology centrifuged solids data.

Notes:

a. Upstream BCM parameter values were revised using updated datasets and statistics reflective of current conditions (i.e., material entering the LDW from the Green/Duwamish River). The four primary datasets used for BCM parameterization are as follows (see Tables 5-2a through 5-2d for statistical summaries of supporting datasets):

- Ecology's 2008 upstream bed sediment chemistry data: This dataset was screened to exclude samples with ≤30% fines in consideration of the systematic differences in grain size distributions between upstream (e.g., mid-channel) data and average conditions in the LDW.
- TSS-normalized King County data: King County surface water data were normalized to solid fractions by dividing by the TSS in the individual sample.
- Ecology 2008 centrifuged suspended solids data: The Ecology samples are representative of sediments suspended mid-channel in the Green/Duwamish River that enter the LDW.
- Upper-reach USACE DMMP core data (RM 4.3 to 4.75): This dataset is representative of Green/Duwamish River suspended material that settles in the upper section of the LDW.

BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; DMMP = Dredged Material Management Program; dw – dry weight; fines = sum of silt and clay grain size fractions; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg – milligrams; ng = nanograms; PCBs = polychlorinated biphenyls; RM = river mile; TEQ = toxic equivalent; TSS = total suspended solids; UCL95 = 95% upper confidence limit on the mean; USACE = U.S. Army Corps of Engineers

Table 5-1b Bed Composition Model Lateral Input Parameters for Human Health Risk Drivers

Rationale

1. High – Conservative representation of current conditions assuming modest level of source control (e.g., management of high priority sources).
2. Input (Mid-range) – Pragmatic assessment of what might be achieved in the next decade with anticipated levels of source control.
3. Low – Best that might be achievable in 30 to 40 years with increased coverage and continued aggressive source control.

Contaminant	BCM Parameters			Basis for BCM Lateral Input and Sensitivity Values
	Input	Low	High	
Total PCBs[a] (µg /kg dw)	300	100	1,000	Used a range of screening concentrations to reflect potential levels of source control that could occur over time. **Input:** Mean of flow-weighted dataset excluding values >5,000 µg/kg dw (315 rounded to 300 µg /kg dw). **High:** 90th percentile of flow-weighted source tracing dataset excluding values >10,000 µg/kg dw (1,009 rounded to 1,000 µg /kg dw). **Low:** Median of flow-weighted source tracing dataset excluding values >2,000 µg/kg dw (102 rounded to 100 µg/kg dw).[a]
Arsenic[a] (mg/kg dw)	13	9	30	Screened the source-tracing dataset to exclude concentrations above assumed SMS-based source control levels (93 and 57 mg/kg dw) **Input:** Mean excluding values >93 mg/kg (the CSL). **High:** 90th percentile excluding values >93 mg/kg (the CSL). **Low:** Median of all samples, excluding values >57 mg/kg (the SQS).[a]
cPAHs[a] (µg TEQ/kg dw)	1,400	500	3,400	Screened the source-tracing dataset to exclude concentrations above an assumed source control level. cPAHs are expected to be difficult to control due to the petroleum-based economy, intensity of urbanization in the LDW, and myriad ongoing sources. **Input:** Mean of source-tracing dataset excluding values >25,000 µg TEQ/kg dw (1,370 rounded to 1,400 µg TEQ/kg dw). **High:** 90th percentile of source-tracing dataset excluding values >25,000 µg TEQ/kg dw (3,366 rounded to 3,400 µg TEQ/kg dw). **Low:** Median of source tracing dataset excluding values >25,000 µg TEQ/kg dw (490 rounded to 500 µg TEQ/kg dw). [a]
Dioxins and Furans[b] (ng TEQ/kg dw)	20	10	40	Based on combined Greater Seattle metropolitan sediment and SPU catch basin solids datasets.[b] **Input:** Mean (22 rounded to 20 ng TEQ/kg dw) **High:** UCL95 (41 rounded to 40 ng TEQ/kg dw). **Low:** Median (15 rounded to 10 ng TEQ/kg dw).

Notes:

a. Used Lower Duwamish Waterway source tracing dataset (compiled by SPU) through June 2009 as the primary basis for establishing lateral BCM parameter values for arsenic, total PCBs, and cPAHs. The dataset was screened to remove concentrations using various source control practicability assumptions (best professional judgment by the Source Control Work Group). Total PCB data were flow-weighted before generating statistics because PCBs exhibit a distinct geographic distribution with hot spots identified at Terminal 117, North Boeing Field/Georgetown Steam Plant, Rainier Commons, and Boeing Plant 2/Jorgensen Forge. These four areas have been extensively sampled and make up a significant portion of the overall source tracing dataset. Therefore, the PCB source-tracing data were flow-weighted to avoid skewing the summary statistics used in the BCM. Arsenic and cPAH data were not flow-weighted prior to the statistical analysis because these contaminants lack a pronounced geographic dependency that would warrant flow-weighting. See Tables 5-2a through 5-2d for statistical summaries of supporting datasets.

b. Parameter estimation for dioxins and furans was based on the Greater Seattle metropolitan area receiving sediment dataset collected as part of the RI (Windward 2010) and sediment and SPU catch basin solids datasets (City of Seattle 2010; data collected through 2009). The summary statistics used to estimate parameter values correspond to the combined datasets, as supported by statistical analysis, and include the removal of outliers. See Tables 5-2a through 5-2d for statistical summaries of supporting datasets.

BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; CSL = cleanup screening level; dw = dry weight; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg – milligrams; ng = nanograms; PCBs = polychlorinated biphenyls; RI = remedial investigation; SPU = Seattle Public Utilities; TEQ = toxic equivalent; SQS = sediment quality standard; UCL95 = 95% upper confidence limit on the mean

Table 5-1c Bed Composition Model Post-Remedy Bed Sediment Replacement Values for Human Health Risk Drivers

Rationale

Range of concentrations considered representative of current and potential near-term (0-3 years) post-remedy surface sediment conditions influenced by multiple recontamination mechanisms. Values expected to vary spatially.[a]

Contaminant	SWAC Outside of AOPC 1[b]	Clean Fill Material[c]	Input and Sensitivity Values					
			Proportioned Values Using SWAC Outside of AOPC 1[d]			Proportioned Values Using SWAC Outside of AOPC 2[d,e]		
			Input	Low Input	High Input	Input	Low Input	High Input
Total PCBs (µg /kg dw)	120	2	60	30	90	20	10	40
Arsenic (mg/kg dw)	12	7	10	9	11	9	8	10
cPAHs (µg TEQ/kg dw)	270	9	140	70	200	100	50	140
Dioxins and Furans (ng TEQ/kg dw)	7	2	4	2[f]	6	n/a	n/a	n/a

Notes:

a. Actively remediated areas within the AOPC 1 footprint receive the higher input values. Actively remediated areas within AOPC 2 footprint would receive lower input values. See Section 6 for a definition of AOPCs.

b. The SWAC outside of AOPC 1 is assumed representative of concentrations adjacent to remediated areas for arsenic, total PCBs, and cPAH. The representative dioxins and furans concentration outside of AOPC 1 is based on the arithmetic mean of the point values located outside of AOPC 1. See Section 6 for definition of AOPC 1.

c. The contaminant composition of clean fill material is based on the UCL95 of 2008 EPA OSV *Bold* Survey data. Use of qualified maintenance dredged materials (e.g. from the Upper Turning Basin) for capping would, in practice, lead to higher range of post-remedy bed-sediment replacement values than calculated in this table.

d. Range of representative post-remedy bed sediment replacement values assumes combinations of clean backfill material (e.g., whether capping, ENR, or post-dredge residuals management) and surrounding representative bed sediment concentrations. Assumed proportioning percentages are as follows:

BCM Parameter	Post-Remedy Bed Sediment Replacement Value Proportioning Assumptions	
	% of Clean Import Material	% of SWAC Outside of AOPC 1
Input	50	50
Low	75	25
High	25	75

Shading indicates input value used in the BCM

e. As discussed in Section 6, a larger footprint referred to as AOPC 2 was developed. The remedial alternative that evaluates this footprint will use lower input values after all high to moderate PCB concentration areas have been remediated.

f. In this case, the “low” value of 2 is used to maintain a reasonable range of concentrations. The adjustment is considered reasonable because of the small dataset available for calculating the concentration outside of AOPC 1.

AOPC = area of potential concern; BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; OSV = ocean survey vessel; ENR = enhanced natural recovery; kg = kilograms; µg = micrograms; mg – milligrams; n/a = not available; ng = nanograms; PCBs = polychlorinated biphenyls; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; UCL95 = 95% upper confidence limit on the mean.

Table 5-2a BCM Parameter Line of Evidence Information for Total PCBs (µg/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*22*	50	21	107	82	Normalized to TSS; data from 2005 to 2008, provided by King County.
	Ecology Centrifuged Solids		*7*	14	8	54	36	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
	King County and Ecology Data Combined		*29*	42	11	120	127	Calculation of all upstream surface water data by AECOM; unpublished.
Upstream Surface Sediment	LDW RI Data		*37*	23	19	40	21	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	*30*	5	2	13	8	Data from 2008, downloaded from EIM database, stats calculated by AECOM, screened to exclude samples ≤ 30% fines; outlier excluded: 770 µg/kg dw; unpublished.
		All	*73*	3	3	6	3	Data from 2008, downloaded from EIM database; stats calculated by AECOM and outlier excluded: 770 µg/kg dw.
	LDW RI and Ecology Data Combined		*110*	8	3	23	13	Calculation of all upstream surface sediment data by AECOM and outlier excluded: 770 µg/kg dw; unpublished.
USACE Upper Turning Basin Cores		RM. 4.5 – 4.75 (1991-2009)	*10*	23	22	38	23	Calculation of DAIS core data by AECOM; unpublished.
		RM. 4.3 – 4.75 (1991-2009)	*20*	36	33	56	42	Calculation of DAIS core data by AECOM; unpublished.
Lateral Inflow								
City of Seattle Storm Drain Data	Minus samples >2,000 µg/kg dw		*625*	223	102	534	—	Flow-weighted average of storm drain solids data screened to exclude samples >2,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
	Minus samples >5,000 µg/kg dw		*692*	315	125	718	—	Flow-weighted average of storm drain solids data screened to exclude samples >5,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
	Minus samples >10,000 µg/kg dw		*755*	508	146	1,009	—	Flow-weighted average of storm drain solids data screened to exclude samples >10,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
King County CSO Water Quality Data			*28*	638	580	920	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010. Estimates biased high because method assumes all PCBs in whole-water sample in particulate phase.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data		0 – 2 years after dredging	*18*	120	120	—	150	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data		Thick Cap	—	Mean = 45 (yr 0.5), 84 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports (King County 2006; 2009).
		ENR	—	Mean = 6 (yr 0), 23 (yr 1), 62 (yr 2)				
Puget Sound Survey (*OSV BOLD*)			*70*	1	1	3	2	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 120				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 47				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2b BCM Parameter Line of Evidence Information for Arsenic (mg/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*100*	37	29	73	47	Normalized to TSS; data from 2001 to 2006. All detected arsenic concentrations associated with TSS were calculated as the difference between whole-water (i.e., unfiltered) and filtered sample data.
Green River Water Quality	Ecology Centrifuged Solids		*7*	17	14	24	22	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Upstream Surface Sediment	LDW RI Data		*24*	7	5	11	8	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
Upstream Surface Sediment	Ecology	Fines >30%	*31*	9	9	11	10	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 30% fines; unpublished.
Upstream Surface Sediment	Ecology	All	*74*	7	6	10	7	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Upstream Surface Sediment	LDW RI and Ecology Data Combined		*98*	7	6	10	7	Calculation of all upstream surface sediment data by AECOM; unpublished.
USACE Upper Turning Basin Cores	RM. 4.5 – 4.75 (1991-2009)		*8*	5	5	7	7	Calculation of DAIS core data by AECOM; unpublished. 1990 data excluded.
USACE Upper Turning Basin Cores	RM. 4.3 – 4.75 (1991-2009)		*18*	7	6	12	8	Calculation of DAIS core data by AECOM; unpublished. 1990 data excluded.
Lateral Inflow								
City of Seattle Storm Drain Data		Minus samples >57 mg/kg dw	*553*	12	9	29	—	Storm drain solids data screened to exclude samples >57 mg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
City of Seattle Storm Drain Data		Minus samples >93 mg/kg dw	*563*	13	10	30	—	Storm drain solids data screened to exclude samples >93 mg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
King County CSO Water Quality Data			*21*	9	11	13	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data		0 – 2 years after dredging	*8*	11	12	—	14	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data		Thick Cap	—	Mean = 3 (yr 0.5), 10 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports (King County 2006; 2009).
Duwamish/Diagonal Post-Capping Data		ENR	—	Mean = 2 (yr 0), 4 (yr 1), 8 (yr 2)				
EPA OSV *Bold* Survey			*70*	7	6	11	7	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 12				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 10				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2c BCM Parameter Line of Evidence Information for cPAHs (µg TEQ/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*18*	151	74	354	269	Normalized to TSS; data from 2008, provided by King County.
	Ecology Centrifuged Solids		*7*	138	53	400	432	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
	King County & Ecology Data Combined		*25*	135	58	330	266	Calculation of all upstream surface water data by AECOM; unpublished.
Upstream Surface Sediment	LDW RI Data		*16*	55	18	135	100	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	*31*	37	16	77	72	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤30% fines. Note: Outlier of 230 µg TEQ/kg dw was not excluded from any statistical calculations.
		Fines >50%	*18*	50	44	91	75	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 50% fines. Note: Outlier of 230 µg TEQ/kg dw was not excluded from any statistical calculations.
		All	*74*	18	9	57	43	Data from 2008, downloaded from EIM database, stats calculated by AECOM. Note: Outlier of 230 µg TEQ/kg dw was included in statistical calculations.
	LDW RI and Ecology Data Combined		*90*	25	10	73	55	Calculation of all upstream surface sediment data by AECOM; unpublished.
USACE Upper Turning Basin Cores	RM. 4.5 – 4.75 (1991-2009)		*9*	37	41	63	52	Calculation of DAIS core data by AECOM; outlier excluded: 1051.5 µg TEQ/kg dw; unpublished.
	RM. 4.3 – 4.75 (1991-2009)		*19*	73	57	180	134	Calculation of DAIS core data by AECOM; outlier excluded: 1051.5 µg TEQ/kg dw; unpublished.
Lateral Inflow								
City of Seattle Storm Drain Data (minus samples >25,000 µg TEQ/kg dw)			*533*	1,370	490	3,366	—	Storm drain solids data screened to exclude samples >25,000 µg TEQ/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer (2010).
King County CSO Water Quality Data			*26*	1,051	714	2,728	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010. Estimates biased high because method assumes all cPAHs in whole-water samples in particulate phase.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data	0 – 2 years after dredging		*8*	180	170	—	250	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data	Thick Cap		—	Mean = 63 (yr 0.5), 159 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports(King County 2006; 2009).
	ENR		—	Mean = 11 (yr 0), 43 (yr 1), 89 (yr 2)				
Puget Sound Survey (OSV BOLD)			*70*	7	4	15	9	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 270				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 190				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2d BCM Parameter Line of Evidence Information for Dioxins/Furans (ng TEQ/kg dw)

Study/Source			No. of Samples	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	Ecology Centrifuged Solids		6	6	3	13	10	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Upstream Surface Sediment	LDW RI Data		4	Range of Values (Median): 1.1 - 2.6 (1.7)				Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	31	2	2	3	2	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 30% fines; unpublished.
		Fines >50%	18	2	2	3	3	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 50% fines; unpublished.
		All	74	1	0.3	3	2	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
USACE Upper Turning Basin Cores RM 4.3 – 4. 75 (1991-2009)			2	2 and 2.8 ng TEQ/kg dw				Calculation of DAIS core data by AECOM; unpublished.
Lateral Inflow								
Greater Seattle Sediment and SPU Catch Basin Solids			23	22	15	48	41	Calculation of stats based on combined Greater Seattle sediment and SPU catch basin solids datasets by AECOM; outlier excluded: 187 ng TEQ/kg dw; unpublished.
Post-Remedy Bed Sediment Replacement Value								
Puget Sound Survey (OSV BOLD)			70	1	1	2	2	Calculation of Puget Sound Survey stats by AECOM.
Post- Maintenance Dredge Area Surface Data			3	Mean = 8.3 ng TEQ/kg dw				Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Outside AOPC 1 Footprint			18	Mean = 7 ng TEQ/kg dw				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			11	Mean = 5 ng TEQ/kg dw				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Value(s) used for central tendency BCM input value. (mid point between mean Ecology Centrifuged solids and mean upstream fines >30% used for Green/Duwamish River)

Value(s) used as basis for low-sensitivity BCM value.

Value(s) used as basis for high-sensitivity BCM value.(mid-point between mean and UCL95 Ecology Centrifuged Solids used for Green/Duwamish River)

Notes:

1. Statistics for these datasets were calculated using ProUCL 4.0, except that statistics for the City of Seattle Storm Drain Solids, King County CSO Water Quality, and Post-Remedy Bed Sediment Replacement Values datasets were calculated with Excel.
2. TEQs were calculated using one-half RL for undetected individual dioxin/furan congeners or PAH compounds.

'—' = not calculated; n/a = not available

AOPC = area of potential concern; BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; CSO = combined sewer overflow; DAIS = Dredged Analysis Information System; D/D = Duwamish/Diagonal; dw = dry weight; EIM = Ecology Information Management Database; ENR = enhanced natural recovery; fines = sum of silt and clay grain size fractions; IDW = inverse distance weighting; kg = kilogram; LDW = Lower Duwamish Waterway; µg = microgram; mg = milligram; ng = nanograms; OSV = ocean survey vessel; PCBs = polychlorinated biphenyls; RI = remedial investigation; RL = reporting limit; RM = river mile; SPU = Seattle Public Utilities; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; TSS = total suspended solids; USACE = U.S. Army Corps of Engineers; UCL95 = 95 percent upper confidence limit on the mean

Table 5-3 BCM Input Values for Representative SMS Contaminants[a]

Contaminant	Upstream Inflow (n = 22 to 23)		Lateral Inflow (n = 531 to 579)	
	BCM Input Value (µg/kg dw)[b]	Basis	BCM Input Value (µg/kg dw)[b]	Basis
BEHP	120	Median of USACE Dredged Material Characterization Core Data (RM 4.3 to 4.75; USACE 2009a, 2009b)	15,475	Log-normal mean of City of Seattle source-tracing data through July 2009 with outliers removed[c] (SPU 2010)
Chrysene	49		1,807	
Fluoranthene	190		3,989	
Phenanthrene	53		2,010	
Mercury (mg/kg dw)	0.1		0.14	
Zinc (mg/kg dw)	64		626	
Acenaphthalene	8		209	
Butylbenzyl-phthalate	11		972	
Indeno(1,2,3-cd)pyrene	31		675	
Phenol	10		237	

Notes:

a. FS dataset used to generate summary statistics.

b. Units are in µg/kg dw, unless otherwise noted. Input values are not flow-weighted.

c. Values that were at least two times the next highest value were removed from the analysis as outliers.

BCM = bed composition model; BEHP = bis(2-ethylhexyl)phthalate; dw = dry weight; kg = kilogram; µg = micrograms; mg = milligrams; n= number of; RM = river mile; SMS = Sediment Management Standards; SPU = Seattle Public Utilities; USACE = U.S. Army Corps of Engineers

Table 5-4 Results of Additional STM Special Scenario Runs

Purpose	Description	Results
1: Potential Recontamination of EAAs	An additional bed sediment class is added to differentiate sediment within EAAs from sediment outside of EAAs. This addition results in 16 sediment variables (four size classes for each of four sediment types): EAA bed sediments, non-EAA bed sediments, lateral source sediments, and upstream Green/Duwamish River source sediments). Model is run for 10-year period to predict how unremediated areas may contribute to recontamination of remediated area, assuming EAAs have been remediated.	• Contribution from non-EAA areas to remediated EAAs is less than 5% of the surface sediments at most EAAs after 10 years.
2: Distributed Discharges from Lateral Sources	The STM input is modified to have the discharges from lateral sources distributed to more closely describe actual drainage distribution among shoreline outfalls. The updates primarily affect private nearshore drainage basins. The model is run for both 10-year and 30-year periods to compare what was reported in the STM report (QEA 2008)(the lateral load distributed via 21 outfalls) with the redistributed lateral loads used in the FS.	• Lateral source sediments are more widely distributed, often at lower percent composition, along the nearshore STM grid cells. • Lateral source sediments are more widely distributed throughout the LDW, but most of the changes only result in some areas increasing from <1.0% lateral load content to 1.0 - 2.0%. • The greatest changes were observed around Hamm Creek and between RM 2 and 3. • Updated load distribution used in all subsequent analyses; it was used in all STM base-case model runs.
3: Movement of LDW Bed Sediment into the Upper Turning Basin	10-year model run that tracks bed sediment from four sources: Upper Turning Basin, navigation channel from RM 4.0 to 4.3, bench areas upstream of RM 4.0, and all sediment downstream of RM 4.0. The model run predicts whether downstream LDW sediments resuspend and settle upstream in the Upper Turning Basin area.	• Contribution of downstream sediment to the Upper Turning Basin area is negligible (<0.01%). • Only 240 MT of sediment is transported upstream to Reach 3 from downstream areas over 10 years compared to over 800,000 MT that settles in Reach 3 from upstream. • Supports use of USACE sediment cores collected from RM 4.3 to 4.75 in navigation channel as one line of evidence of upstream solids (i.e., negligible input from downstream sediments).
4: Movement of Bed Sediments between Reaches	Evaluation of the mass balance of sediment originating from each reach that moves between reaches and out of the LDW. This scenario is conducted for the 30-year model period.	• Much of the sediment resuspended in a reach that resettles in the LDW settles within the same reach. • There is more of an exchange of sediments between Reach 1 and 2, than from Reach 1 and 2 to Reach 3. • Reach 3 sediments are widely distributed throughout the LDW, while very little sediment from Reach 1 or 2 resettles in Reach 3.

Table 5-4 Results of Additional STM Special Scenario Runs (continued)

Purpose	Description	Results
5: Sediment Scoured from Greater than 10-cm Depth	Areas that are estimated to scour greater than 10-cm depth are assigned a new variable to represent a new sediment class. The 100-year high-flow simulation is used to predict where these >10 cm scoured sediments resettle.	• Sediment eroded from below 10 cm makes up a very small fraction of the total sediment mass moving over a 100-year high-flow event. • Sediment eroded from below 10 cm is greatest in Reach 2 and lowest in Reach 1. • Most of the scour >10 cm occurs in localized navigation channel above about RM 2.9.
6: Movement of Existing Bed Sediment (bed-tracking)	An additional bed sediment class is added to differentiate bed sediment that was resuspended and redeposited into another model cell from original bed sediment over a 10-year period. This scenario tracks the movement of bed sediments with the LDW and its effect on bed composition and SWACs.	• Resuspended bed sediment makes up less than 30% of the total original + resuspended bed fraction, and typically less than 5 to 10%. • The BCM construct is considered appropriate for use in the FS.
7: Holding Cells Constant in Selected Scour and Berthing Areas (no natural recovery)	The analysis was a 10-year model run that assumed no natural recovery in areas with high-flow scour, evidence of propeller scour, and berthing areas with less than 0.5 cm/yr of sedimentation. These areas were essentially ”held constant” at their FS baseline total PCB concentrations. The analysis was conducted over 10 years following construction of Alternative 3C and then compared to the site-wide and reach-wide best-estimate total PCB SWAC model predictions.	• Total PCB SWACs increased about 10% compared to best-estimate model predictions and up to 18% in Reach 2.

Note:

BCM = Bed Composition Model; cm = centimeter; EAA = early action area; FS = feasibility study; LDW = Lower Duwamish Waterway; MT = metric ton; PCBs = polychlorinated biphenyls; RM = river mile; STM = sediment transport model; SWAC = spatially-weighted average concentration; USACE = U.S. Army Corps of Engineers; yr = year

Table 5-5 Comparison of Year 10 Total PCB SWACs between the Bed Tracking Scenario and STM Base Case

Scenario	Total PCB SWACs (µg/kg dw)			
	Site-wide	Reach 1	Reach 2	Reach 3
Post-Alternative 1				
Year 0	180	190	220	56
Year 10 STM Base Case	73	84	70	40
Year 10 modified STM Bed Tracking with resuspended bed variable	72	84	66	40

Distal Sediment Concentration Input Values to the Analysis				
Distal Bed (µg/kg dw) – reach-wide post-Alternative 1 SWAC	n/a	176	117	57

Notes:

1. For a detailed discussion of the analyses supporting this table, see Part 5 of Appendix C.

dw = dry weight; kg = kilogram; µg =micrograms; n/a = not applicable; PCB = polychlorinated biphenyl; STM = sediment transport model; SWAC = spatially-weighted average concentration

Table 5-6a Total PCB Input Concentrations for the Particle Size Fractionation Analysis

Solids Source and Class	Percentage of Solids by Mass	Total PCB Concentration (µg/kg dw)			
		FS mid-range BCM Input Value	Approach 1	Approach 2	Approach 3
Green/Duwamish (Upstream) Solids					
Class 1A	70	35	80	42	38
Class 1B	18	35	80	21	38
Class 2	12	35	5	13	11
Class 3	0	35	5	3	2
Aggregate concentration on upstream solids		35	71	35	35
Lateral Source Solids					
Class 1A	55	300	1,000	422	374
Class 1B	18	300	1,000	211	374
Class 2	23	300	100	127	112
Class 3	4	300	100	25	22
Aggregate concentration on lateral solids		300	757	300	300

Notes:

1. For Green/Duwamish solids Classes 1A, 1B, and 2 are suspended load and Class 3 is bed load. However, there is very little bed load that reaches the LDW beyond river mile 4.5.
2. The Draft Final FS mid-range BCM input values are shown for reference when comparing input values for the three approaches.
3. Approach 1 essentially increases PCB mass from upstream and lateral sources by approximately 100 percent over the mid-range BCM input values, while Approaches 2 and 3 maintain the same PCB mass as in the mid-range BCM case.

Table 5-6b Effect of Particle Size Fractionation on Total PCB SWACs

LDW Reach	Total PCB SWAC (µg/kg dw) Resulting from Use of:			
	FS Mid-range BCM Input Value	Approach 1	Approach 2	Approach 3
1	84	120	78	85
2	67	100	60	66
3	40	51	23	28
Site-Wide	73	104	65	71

Notes:

BCM = bed composition model; dw = dry weight; FS = feasibility study; kg = kilogram; LDW = Lower Duwamish Waterway; µg = micrograms; PCB = polychlorinated biphenyl; SWAC = spatially-weighted average concentration

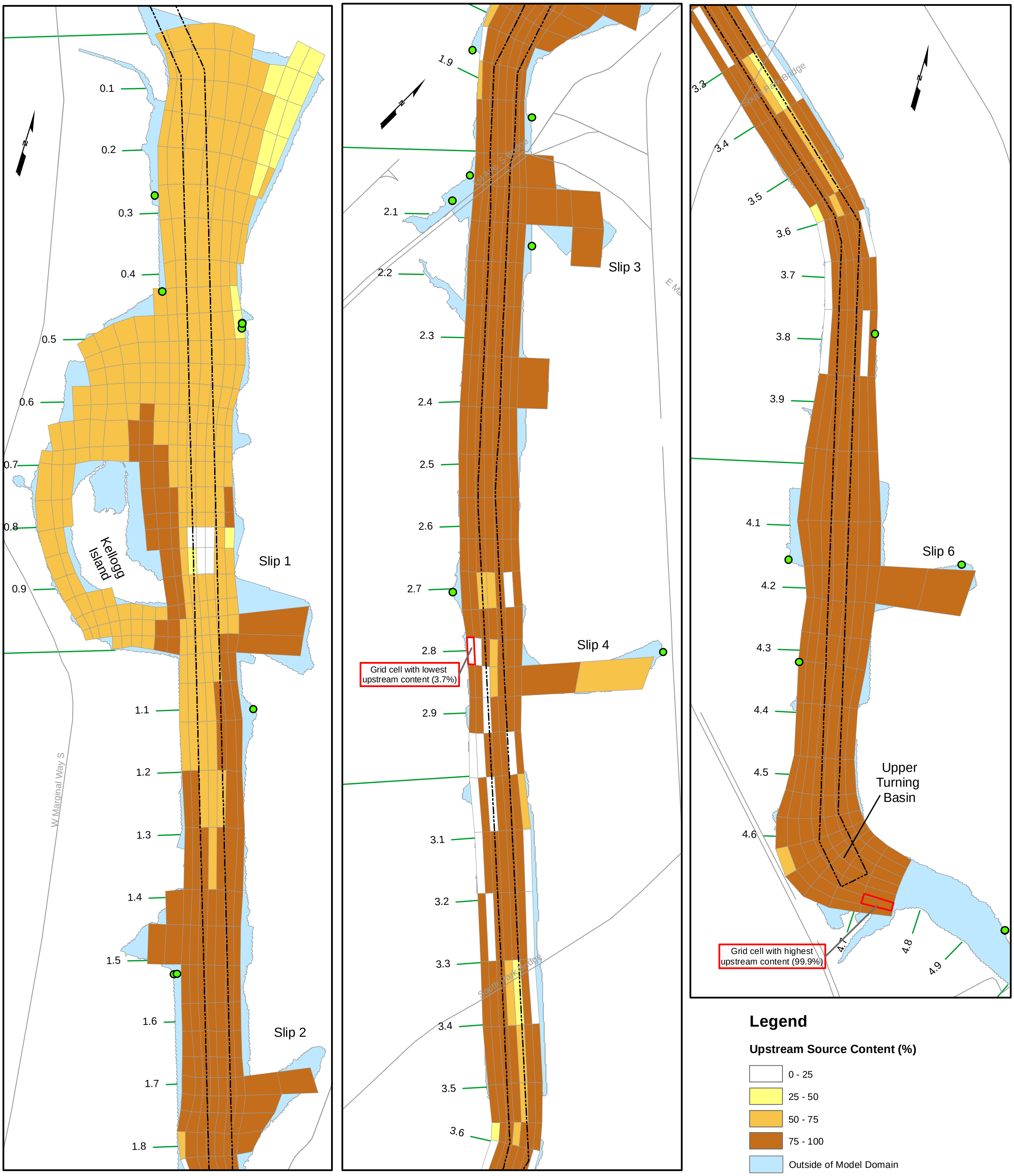
Table 5-7 Changes in Contaminant Concentrations at Resampled Surface Sediment Stations

Contaminant and Metric	Original/Older Data (1991–2006)	Newer (FS Baseline) Data (1998–2008)	Percent Decrease between Older and Newer Concentrations (%)
Total PCBs (µg/kg dw); N = 67			
25th Percentile	107	74	31
Mean	939	354	62
90th Percentile	2,141	776	64
Arsenic (mg/kg dw); N = 56			
25th Percentile	10	11	Minimal change; in equilibrium
Mean	40	35	
90th Percentile	41	40	
cPAHs (µg TEQ/kg dw); N = 53			
25th Percentile	200	145	28
Mean	1,534	437	72
90th Percentile	2,070	803	61
BEHP (µg/kg dw); N = 53			
25th Percentile	230	92	60
Mean	827	310	63
90th Percentile	1,570	606	61

Notes:

1. Newer data are co-located with older data (i.e., within 10 ft). Older data are not included in the FS baseline dataset.
2. Statistics calculated using ProUCL v.4.00.04.
3. Undetected data were set to the reporting limit.
4. Three PCB locations omitted to generate the n=67 dataset: LDW-SS110/SD-323-S; LDW-SS111/DR186; and SD-320-S/SD-DUW92. These are located within the Boeing Plant 2/Jorgensen Forge EAA.
5. Results on a station-by-station basis are provided in Appendix F.

BEHP = bis(2-ethylhexyl)phthalate; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; dw = dry weight; EAA = early action area; FS = feasibility study; kg = kilogram; LDW = Lower Duwamish Waterway; µg = micrograms; mg = milligrams; N = number of; PCB = polychlorinated biphenyl; TEQ = toxic equivalent



Legend

Upstream Source Content (%)

- 0 - 25
- 25 - 50
- 50 - 75
- 75 - 100
- Outside of Model Domain
- Discharge Location Modeled in STM
- Road
- Navigation Channel
- River Mile Marker

Feet
0 200 400 800

Notes:
1. 10-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).
2. A grid cell with 75% sediment from upstream sources has 25% of the surface sediment from lateral and original bed sediment, totaling 100% bed composition (as solids).

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			**Percentage of Bed Sediment from Upstream Sources after 10 Years**
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	**FIGURE 5-1a**

Notes:
1. 30-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).
2. A grid cell with 75% sediment from upstream sources has 25% of the surface sediment from lateral and original bed sediment, totaling 100% bed composition (as solids).

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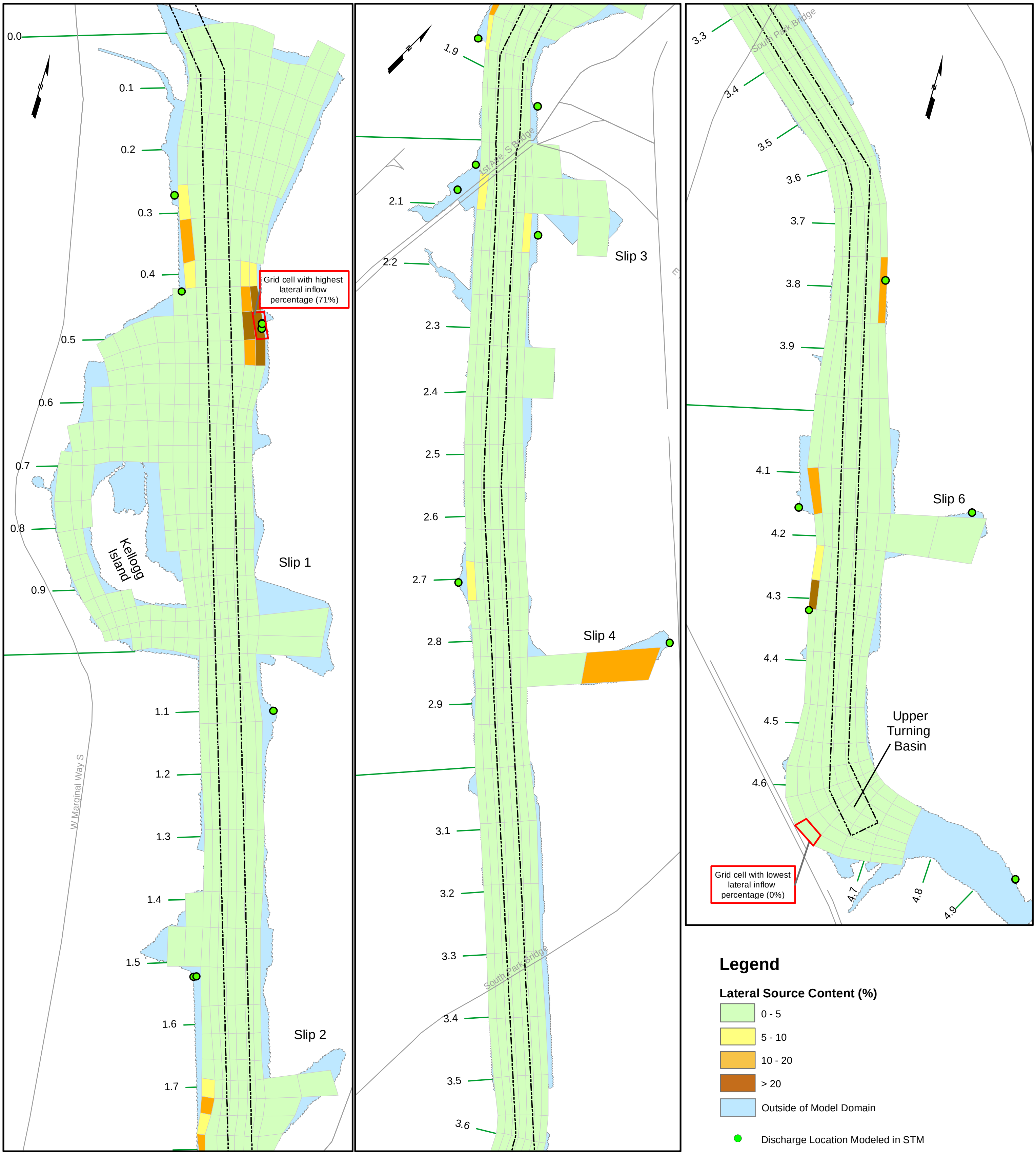
Lower Duwamish Waterway Final Feasibility Study 60150279-14.8			Percentage of Bed Sediment from Upstream Sources after 30 Years
DATE: 10/31/12	DWRN:MVI/sea	Revision: 1	FIGURE 5-1b



Notes:
1. 10-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).

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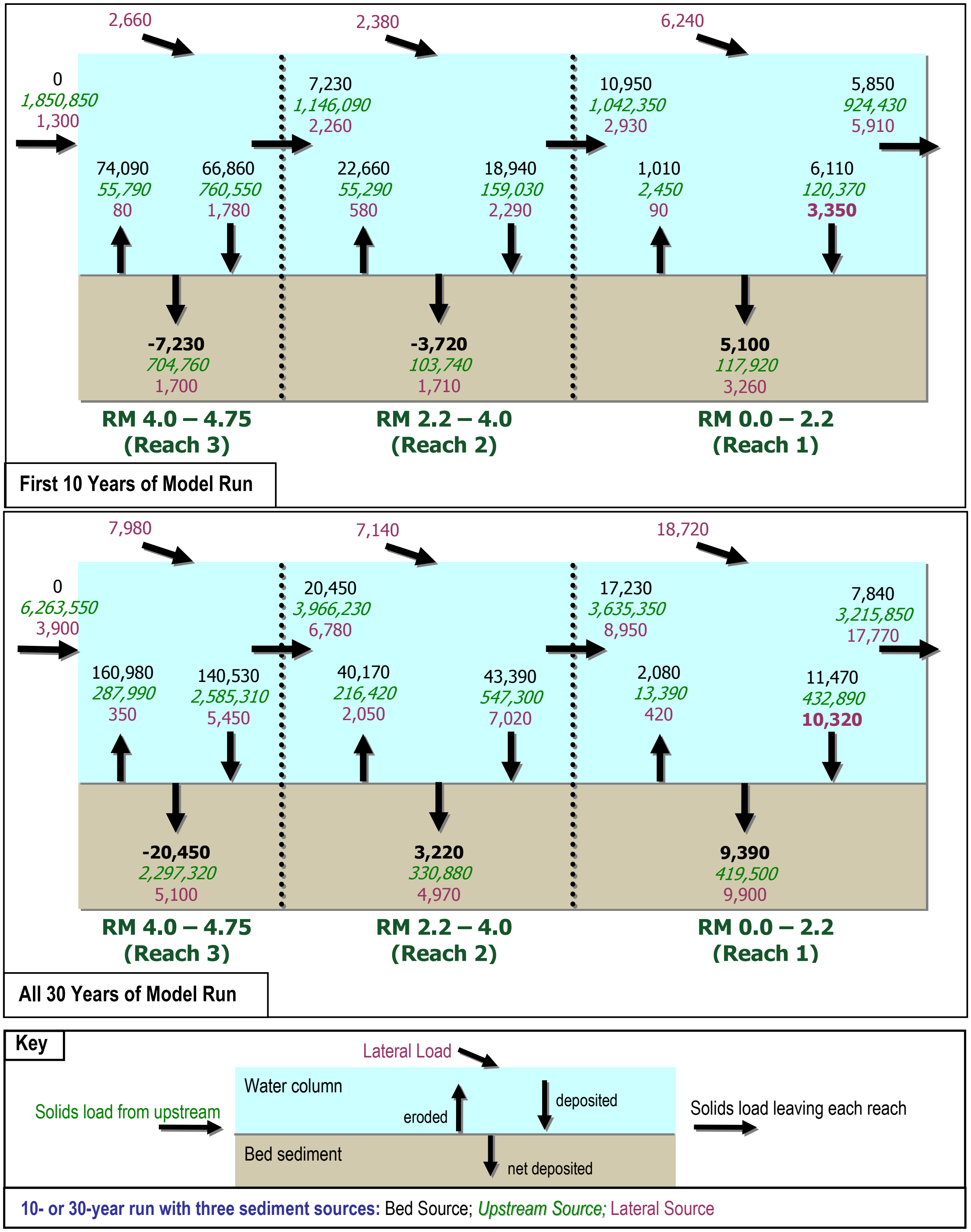
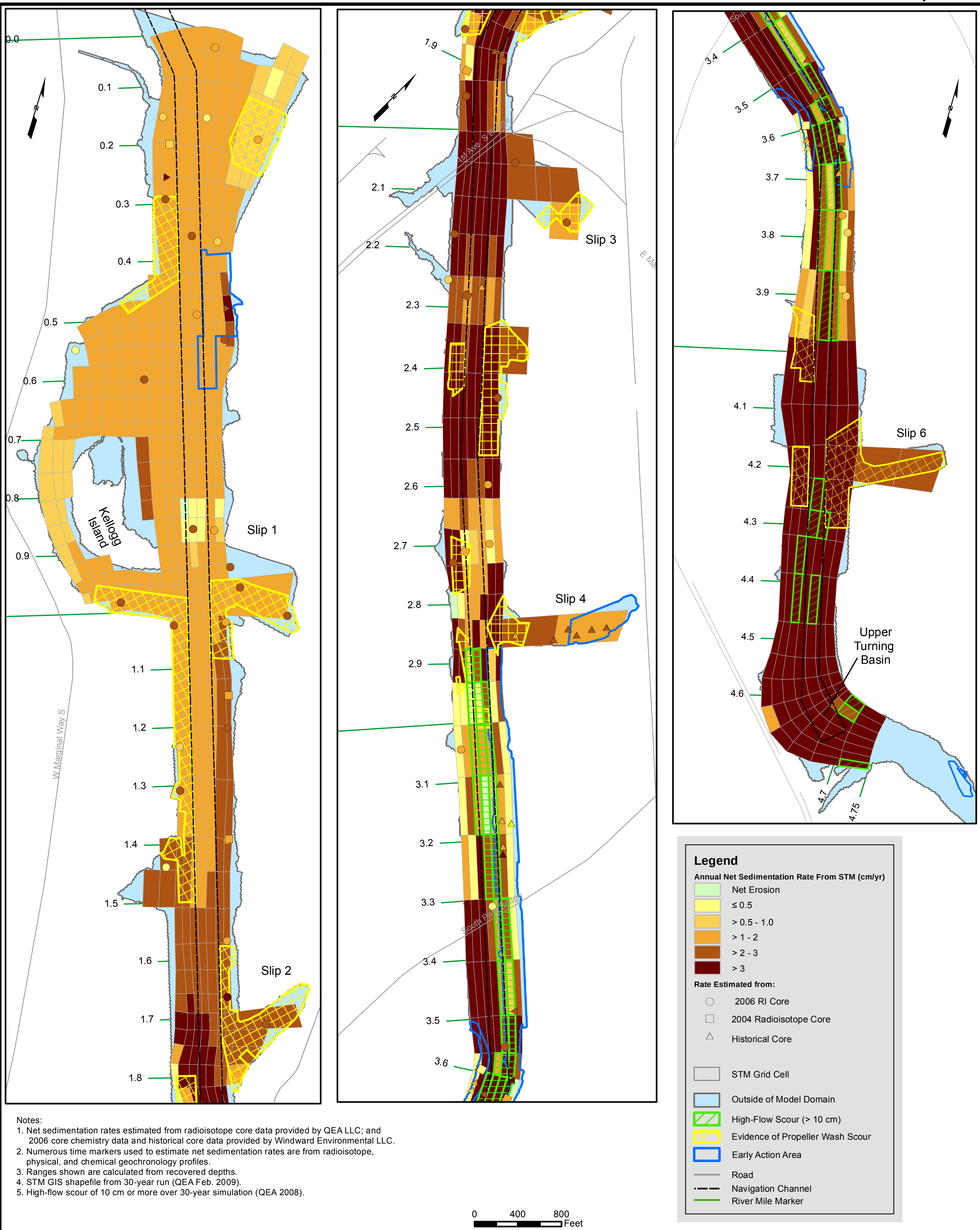
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			Percentage of Bed Sediment from Lateral Sources after 10 Years
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 5-2

Figure 5-3 Sediment Loading to, within, and through the LDW over Two STM Time Periods



Notes:
1. Sediment loads are in metric tons over a 10- or 30-year model run, and include both suspended sediment and bed load material.
2. Upper end of modeled reach extends to RM 4.75.
3. Most of the incoming bed load (sand) settles in Reach 3. Of the incoming fines fractions, 10% of clay and 76% of the silt settle in the LDW.
LDW = Lower Duwamish Waterway; RM= river mile; STM = sediment transport model.



Notes:
1. Net sedimentation rates estimated from radioisotope core data provided by QEA LLC; and 2006 core chemistry data and historical core data provided by Windward Environmental LLC.
2. Numerous time markers used to estimate net sedimentation rates are from radioisotope, physical, and chemical geochronology profiles.
3. Ranges shown are calculated from recovered depths.
4. STM GIS shapefile from 30-year run (QEA Feb. 2009).
5. High-flow scour of 10 cm or more over 30-year simulation (QEA 2008).

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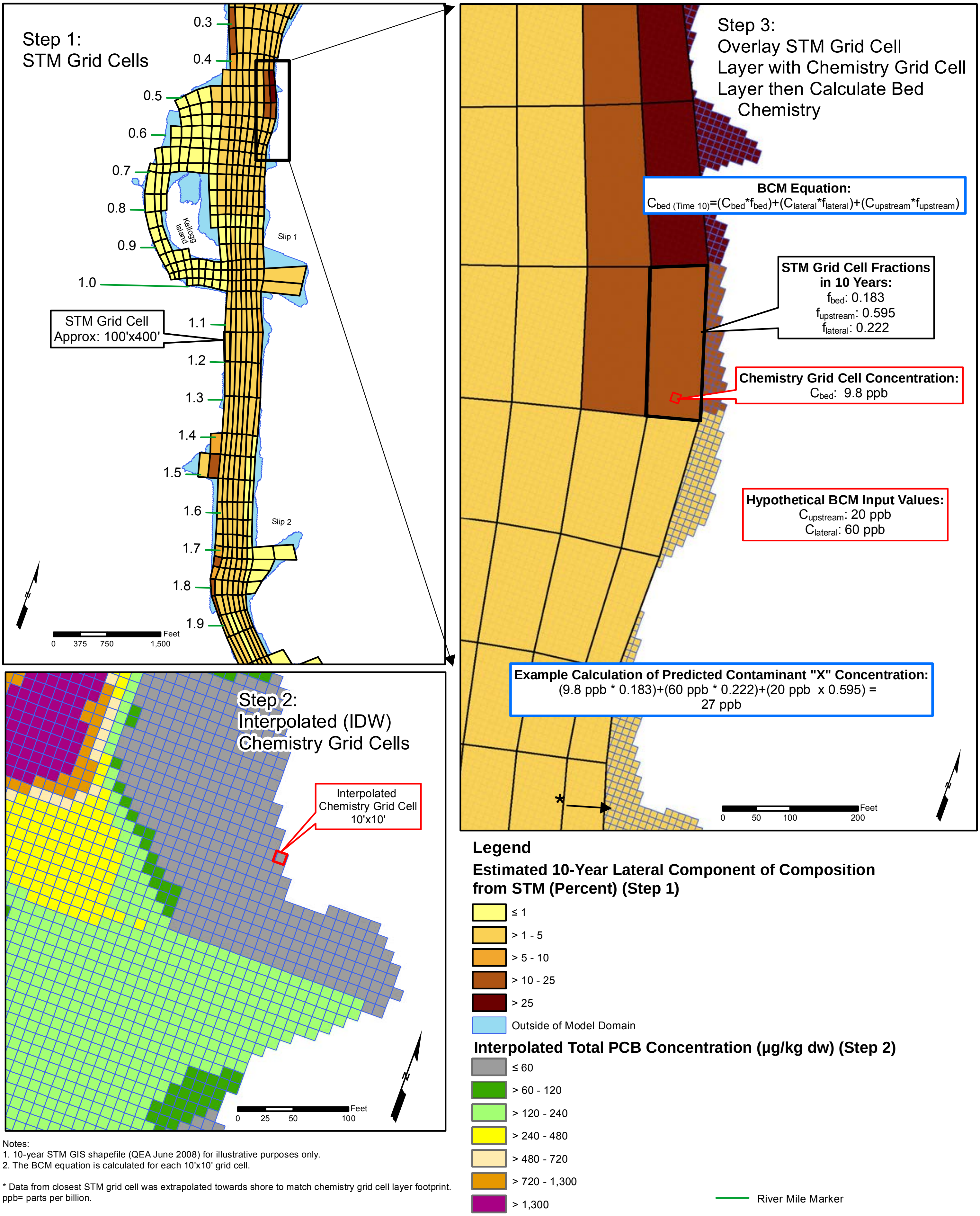
Lower Duwamish Waterway Final Feasibility Study 60150279-14.8	**Potential Scour Areas and Estimated Net Sedimentation Rates**
DATE: 10/31/12 DWN. BY: MVI/sea	**FIGURE 5-4**



Notes:
1. 10-year STM GIS shapefile (QEA June 2008) for illustrative purposes only.
2. The BCM equation is calculated for each 10'x10' grid cell.

* Data from closest STM grid cell was extrapolated towards shore to match chemistry grid cell layer footprint.
ppb= parts per billion.

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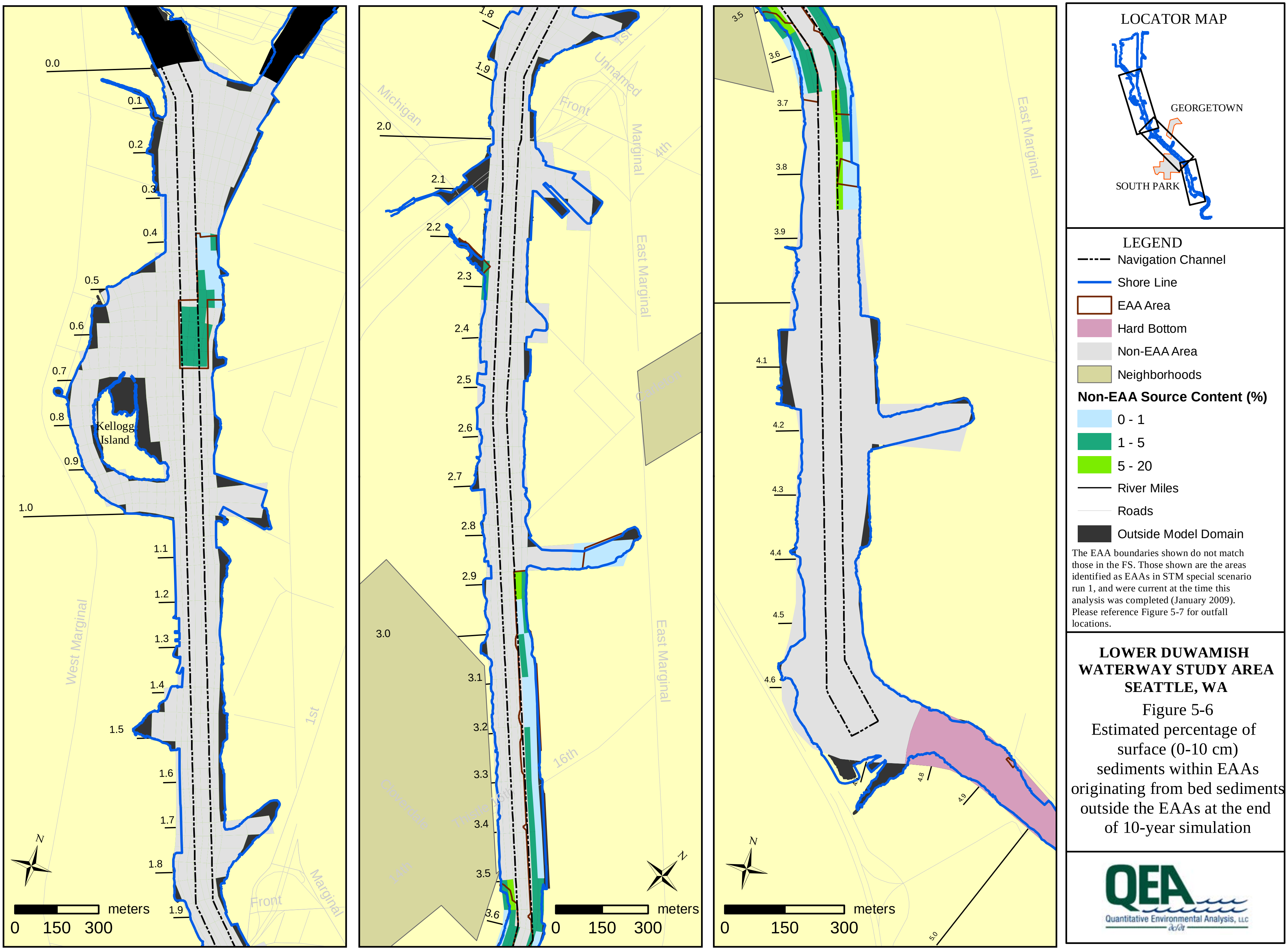
60150279-14.49

BCM Calculation in Model Grid Cells

DATE: 10/31/12 | DWRN:MVI/Sea | Revision: 0

FIGURE 5-5



LOWER DUWAMISH WATERWAY STUDY AREA SEATTLE, WA

Figure 5-6
Estimated percentage of surface (0-10 cm) sediments within EAAs originating from bed sediments outside the EAAs at the end of 10-year simulation

QEA Quantitative Environmental Analysis, LLC

FC/JC/DN- \\Jill\D_drive\Jobs\RETltw\Documents\Scenario_Memo\RETldw_tech_memo_fig_1-3_FS_fig_5-9_090320.mxd

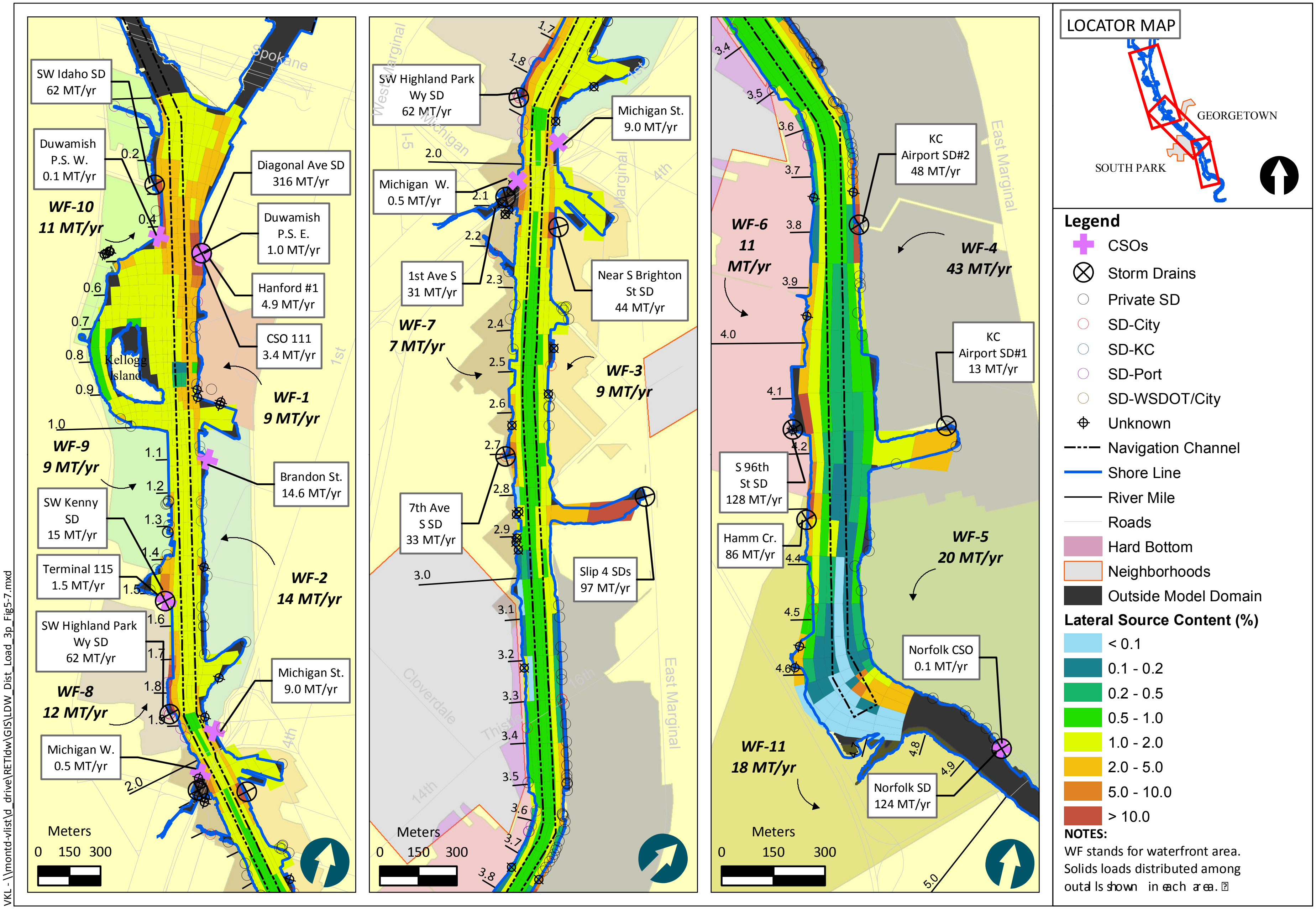
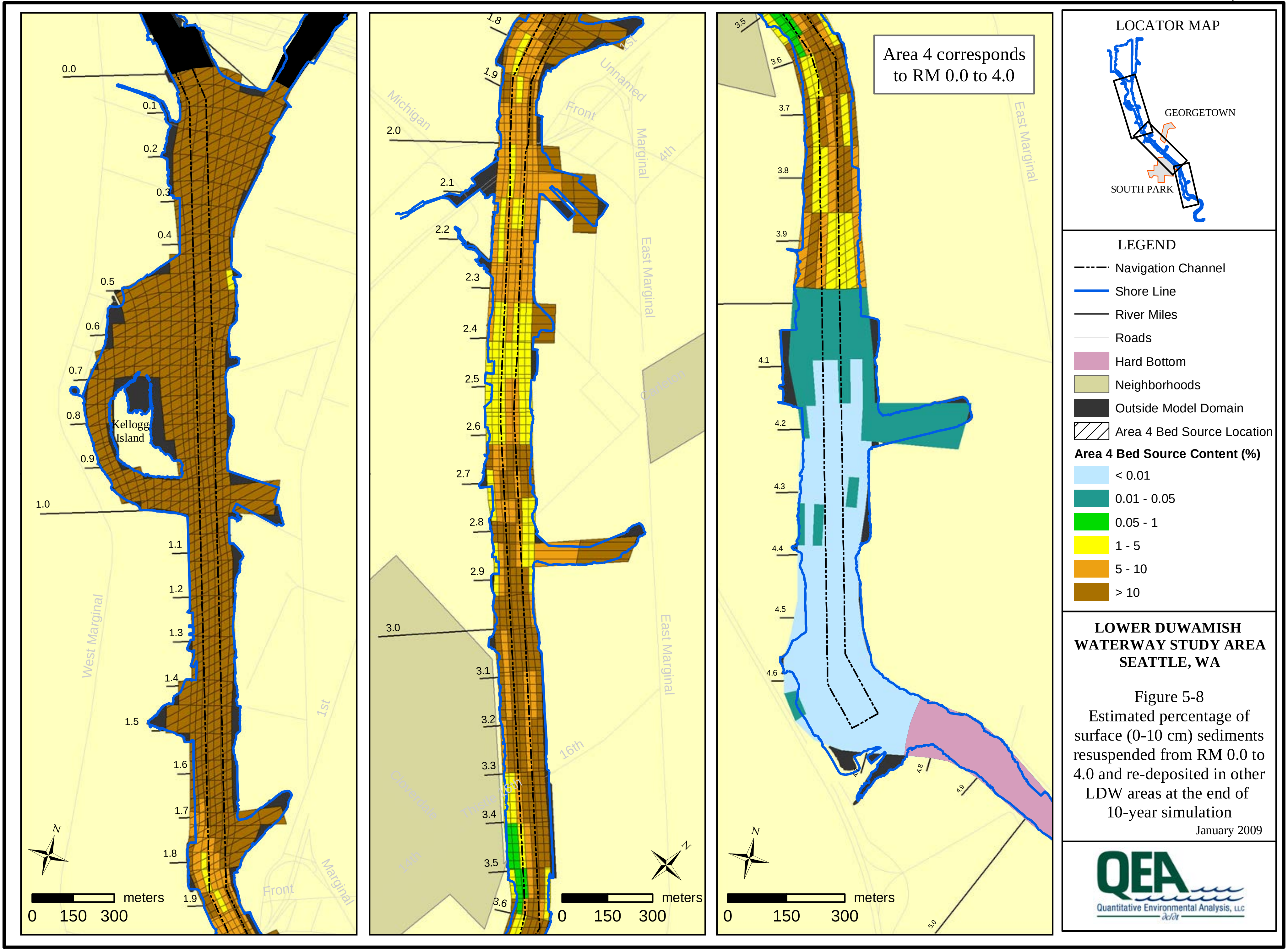


Figure 5-7

Distributed lateral annual loads and lateral source content in surface (0-10 cm) sediments at end of 10-year simulation

Quantitative Environmental Analysis, LLC

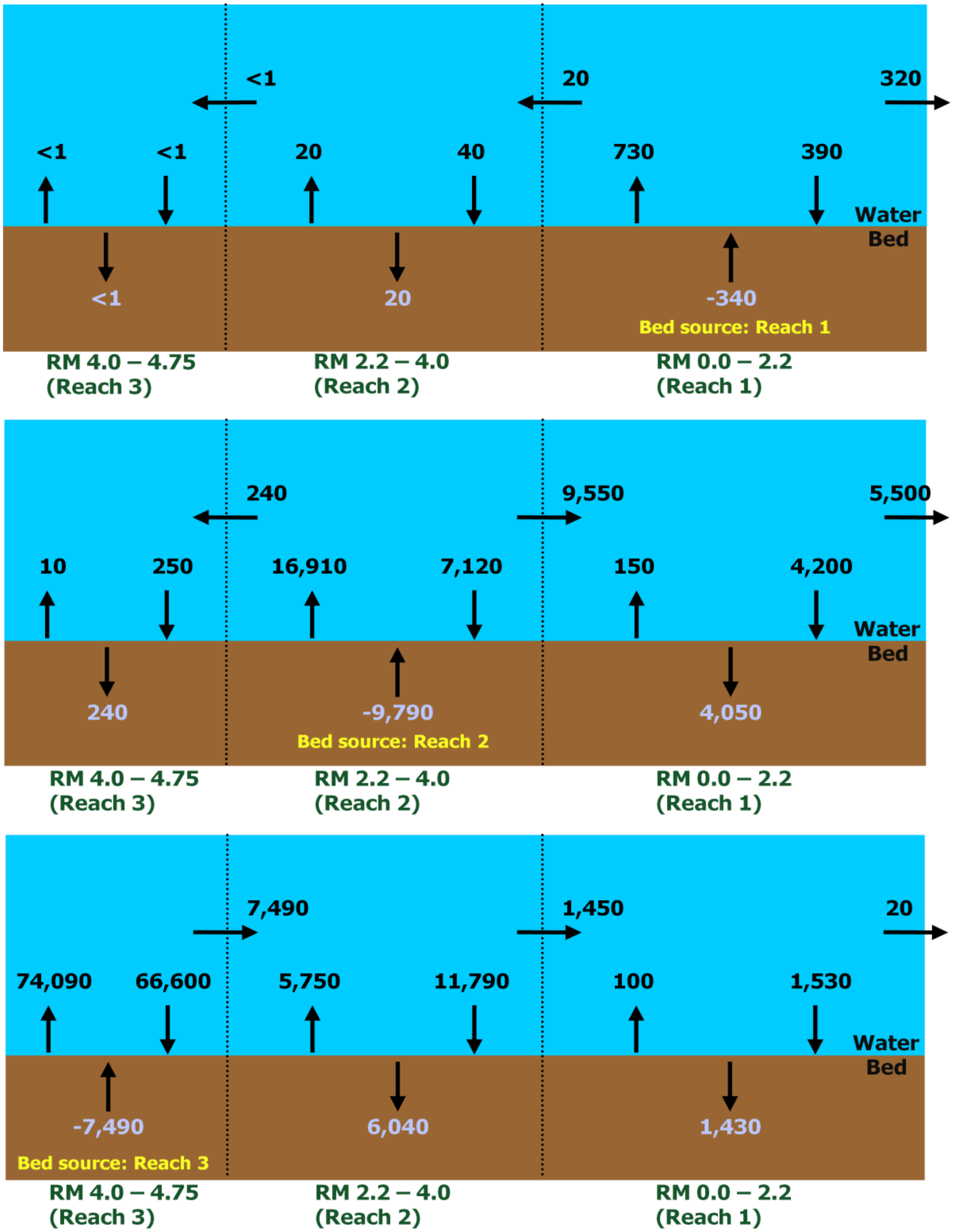


LOWER DUWAMISH WATERWAY STUDY AREA
SEATTLE, WA

Figure 5-8
Estimated percentage of surface (0-10 cm) sediments resuspended from RM 0.0 to 4.0 and re-deposited in other LDW areas at the end of 10-year simulation
January 2009

FC/JC- \\Jill\D_drive\Jobs\RETltw\Documents\Scenario_Memo\RETldw_tech_memo_fig_3-4_FS_fig_5-11.mxd

Figure 5-9 Mass Balances for Bed Sediment Originating from Reaches 1, 2, and 3 for 10-year STM Simulation



Note:
Sediment mass units are in metric tons, rounded to the nearest 10 metric tons.

Figure 5-10a Total Sediment Mass Balance for 100-year High-flow Event Simulation

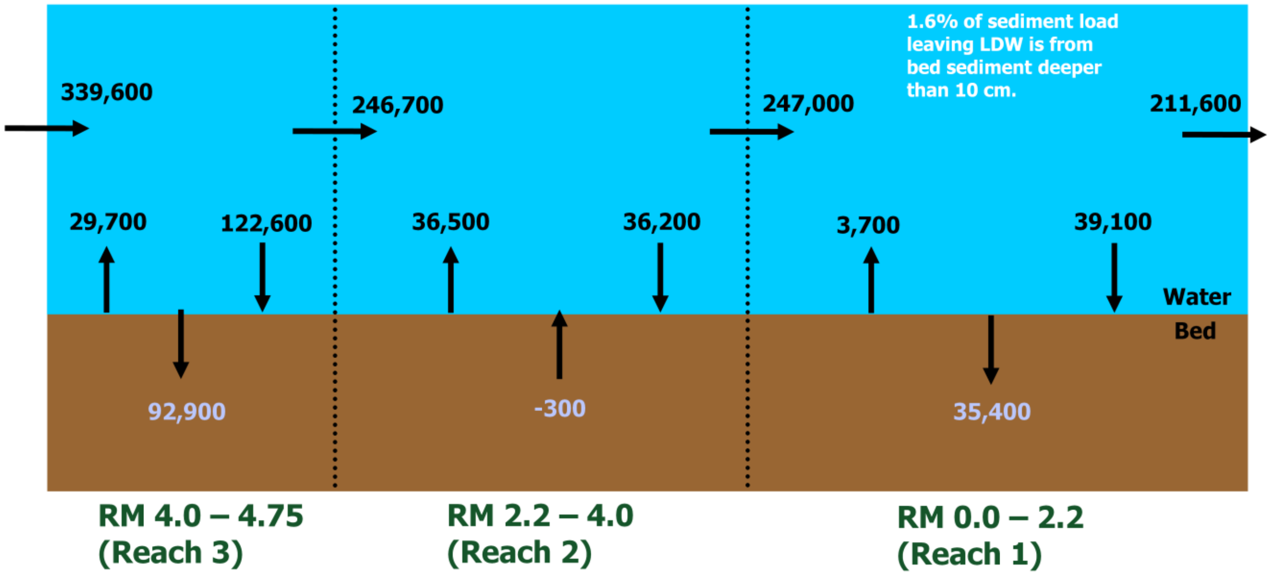
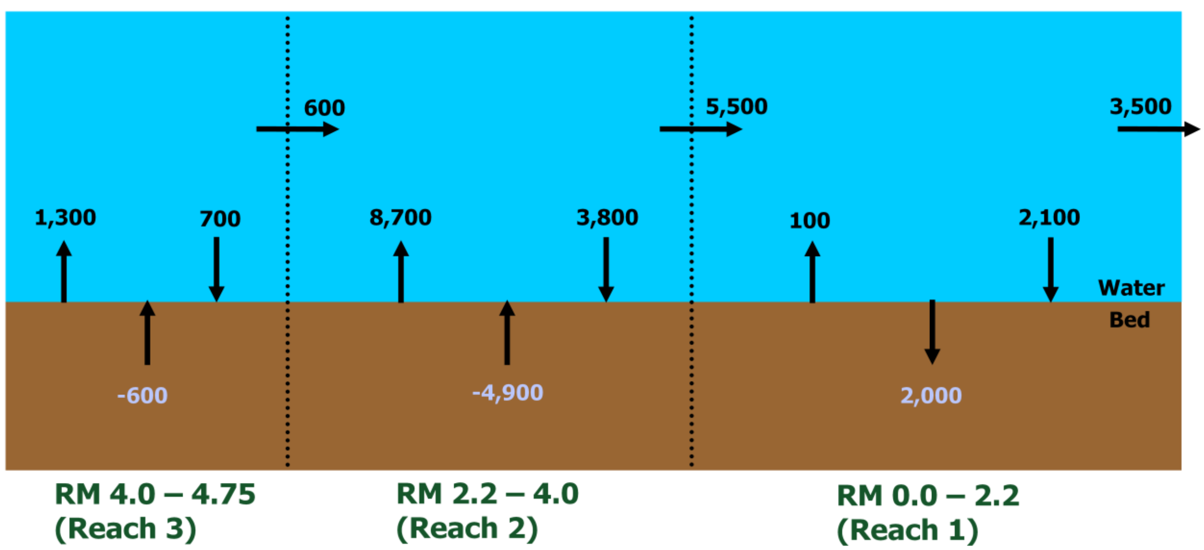
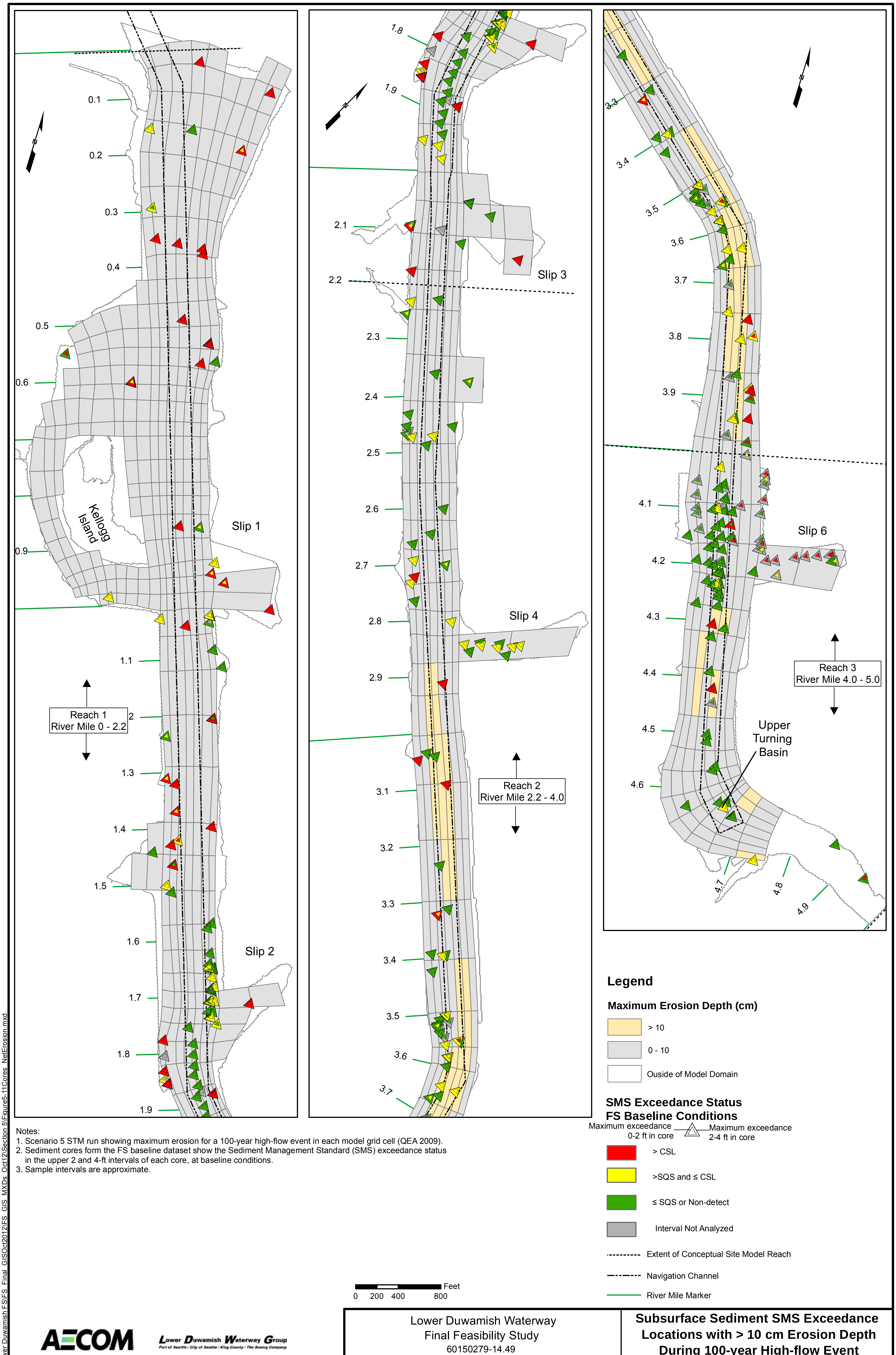


Figure 5-10b Mass Balance for Bed Sediment Originating from Deeper-than-10-cm Layer during 100-year High-flow Event Simulation





Notes:
1. Scenario 5 STM run showing maximum erosion for a 100-year high-flow event in each model grid cell (QEA 2009).
2. Sediment cores form the FS baseline dataset show the Sediment Management Standard (SMS) exceedance status in the upper 2 and 4-ft intervals of each core, at baseline conditions.
3. Sample intervals are approximate.

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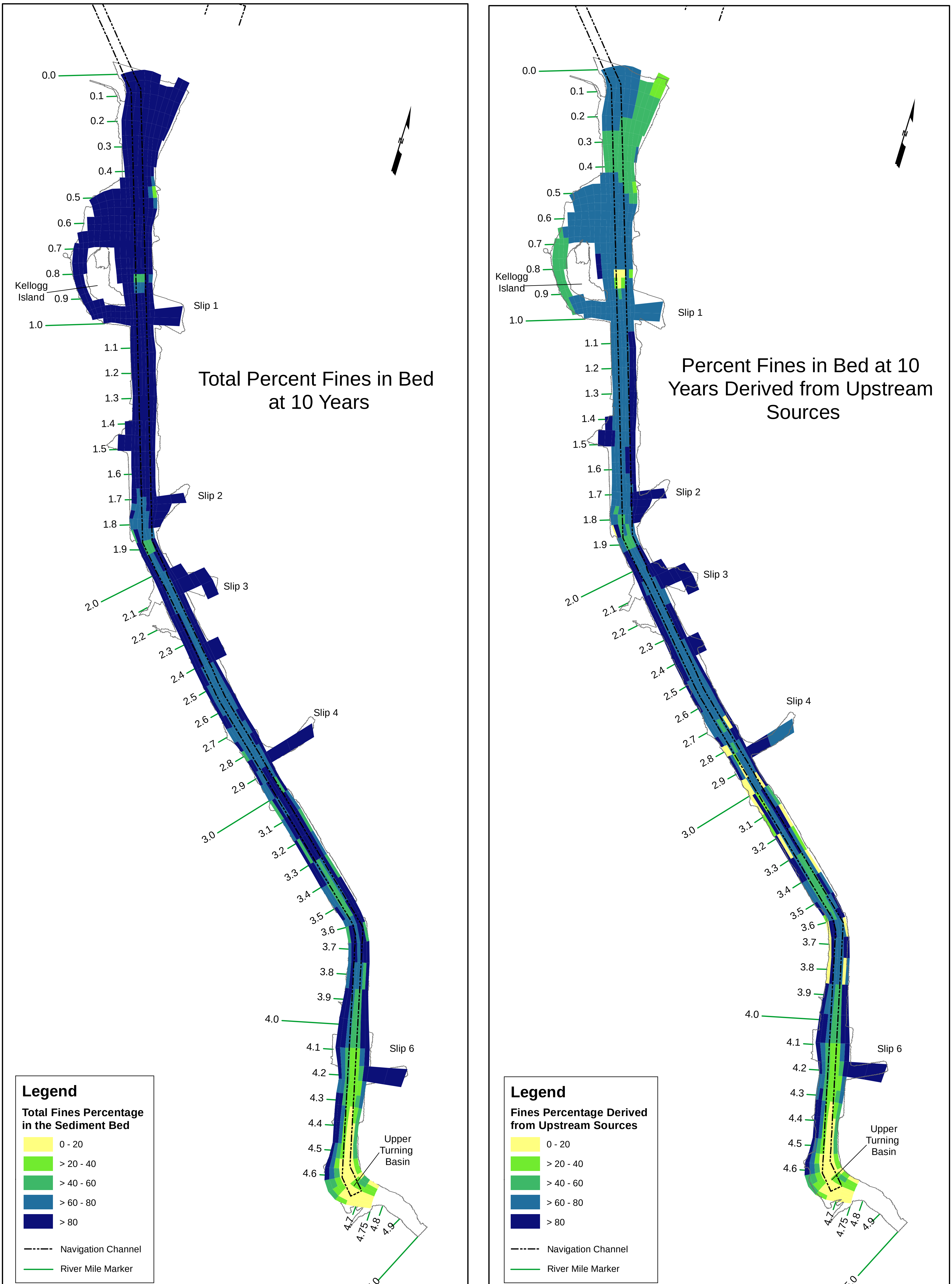
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49	Subsurface Sediment SMS Exceedance Locations with > 10 cm Erosion Depth During 100-year High-flow Event
DATE: 10/31/12 DWRN:MVI/sea Revision: 1	FIGURE 5-11



Notes:
1. 10-year STM GIS shapefile (QEA Feb. 2009).
2. Total fines represents the sum of fine-grained sediment from bed, lateral, and upstream sources in each grid cell at 10 years.
3. Upstream fines represents the percentage of fine-grained sediment from upstream sources in each grid cell at 10 years.
4. Fine-grained sediment is the sum of grain size classes 1A and 1B modeled in the STM. The fines are mostly class 1B, only about 10% of the class 1A material settles in the LDW.

0 750 1,500 3,000 Feet

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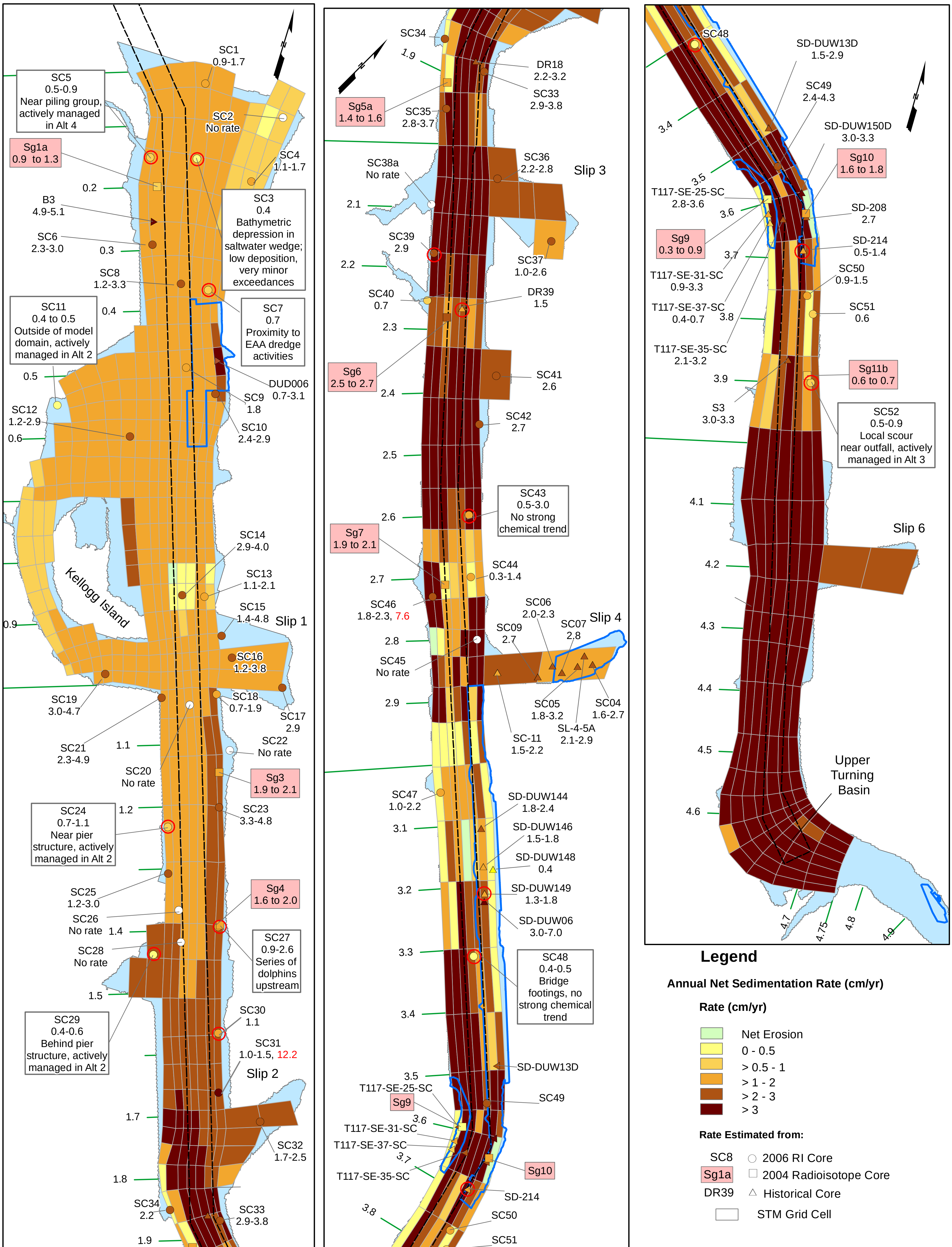
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49	**Estimated Distribution of Grain Size After 10 Years - Total Percent Fines and Fines Sourced from Upstream**
DATE: 10/31/12 DWRN: MVI/sea Revision: 1	**FIGURE 5-12**



Legend

Annual Net Sedimentation Rate (cm/yr)

Rate (cm/yr)

- Net Erosion
- 0 - 0.5
- > 0.5 - 1
- > 1 - 2
- > 2 - 3
- > 3

Rate Estimated from:

- SC8 ○ 2006 RI Core
- Sg1a □ 2004 Radioisotope Core
- DR39 △ Historical Core
- STM Grid Cell

- Net Sedimentation Rate Estimated from Core is Less than Rate from STM
- Outside Model Domain
- Early Action Area
- Navigation Channel
- River Mile Marker

0 400 800 Feet

Core Number
Net Sedimentation Rate (cm/yr)
SC48
0.4-0.5
Bridge Footings
Possible Reason for Difference

Notes:
1. Net sedimentation rates estimated from radioisotope core data, 2006 core chemistry data and historical core data in FS project database.
2. Numerous time markers used to estimate net sedimentation rates are from radioisotope, physical, and chemical geochronology profiles.
3. Ranges shown are calculated from recovered depths.
4. Red font represents rate from dredging event marker outside range of other rates.
5. STM GIS shapefile from 30-year run (QEA Feb. 2009).
6. Seven RI cores for which rates could not be calculated are displayed as white circles. Historical cores for which rates could not be calculated are not shown.
7. Cores SC11, SC40 and SC42 are outside of the model domain and therefore are not circled. Core SC46 has interference from a dredge event and was not circled. Historical core SC11 (in Slip 4) range matched model predictions. Core SC51 straddled two grid cells and therefore was not circled.

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Comparison of Net Sedimentation Rates Estimated from the STM and from Sediment Cores

DATE: 10/31/12 | DWN. BY: MVI/sea | **FIGURE 5-13**

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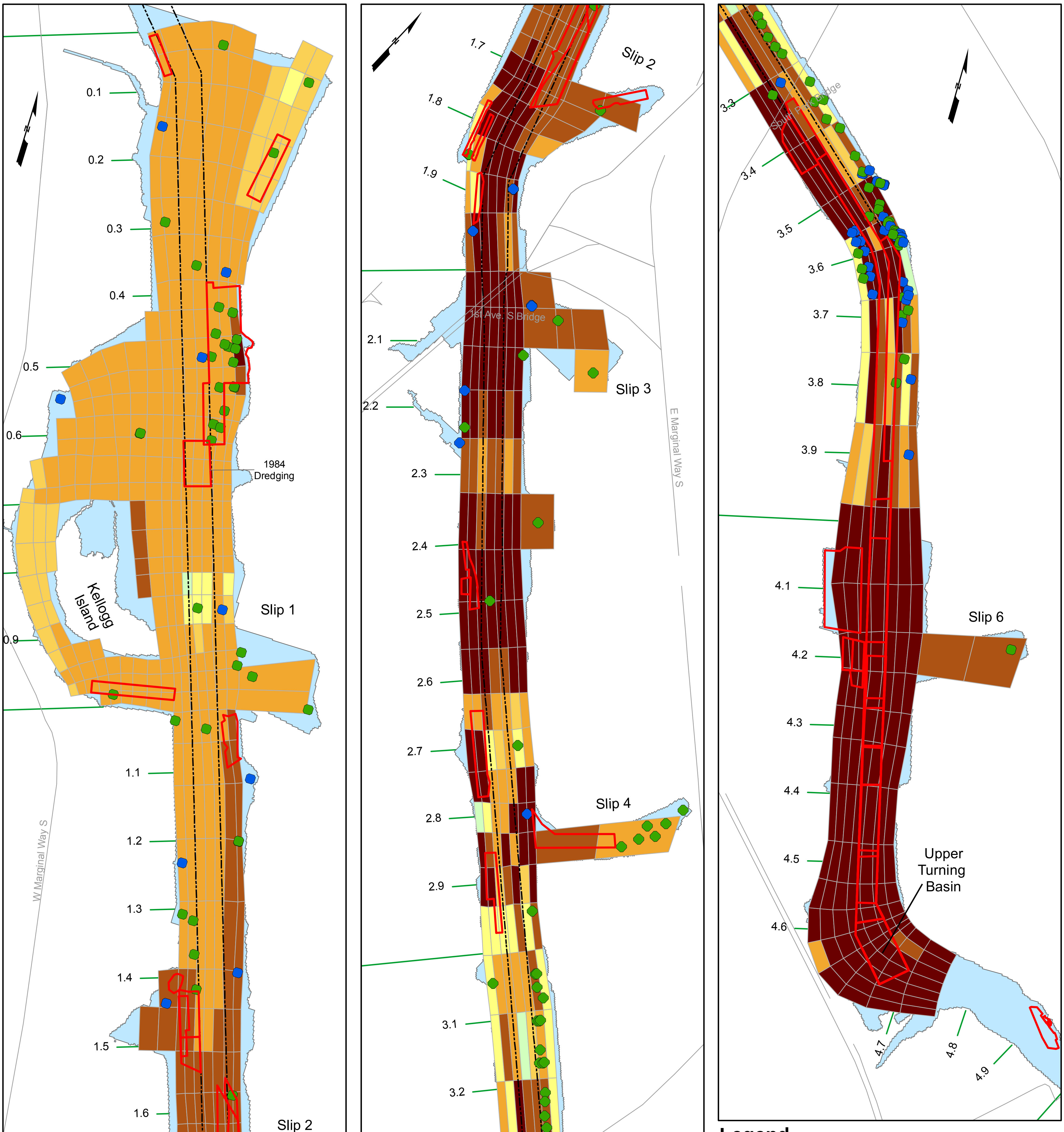

Legend

Interpretation of Subsurface Sediment PCB Profile

- Consistent with assumptions (peak total PCB concentration as deep or deeper than expected)
- Inconsistent with assumptions (peak total PCB concentration shallower than expected)

Annual Net Sedimentation Rate from STM (cm/yr)

- Net Erosion
- ≤ 0.5
- > 0.5 - 1
- > 1 - 2
- > 2 - 3
- > 3
- Outside of Model Domain
- Dredging Event within Last 30 Years
- Road
- Navigation Channel
- River Mile Marker

Notes:
1. Interpretation of total PCB profile from Final RI (Windward 2010). Interpretation identifies whether PCB peak assigned to 1965 is at a depth consistent with the net sedimentation rate from the STM annualized from a 30-year run.
2. Peak total PCB concentration at depth identified where concentration is at least two times concentration in surface interval.
3. 30-year STM GIS shapefile (QEA Feb. 2009).

0 200 400 800 Feet

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AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

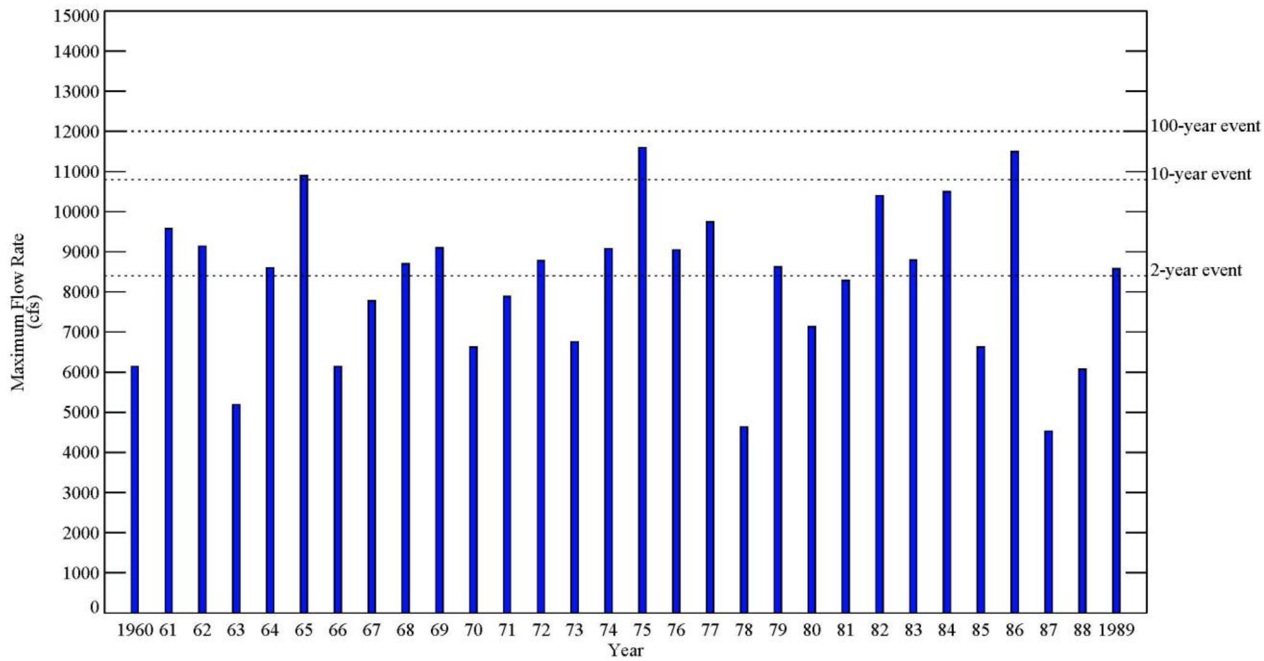
Lower Duwamish Waterway
Final Feasibility Study
60150279-14.49

DATE: 10/31/12 | DWRN:MVI/sea | Revision: 0

Annual Net Sedimentation Rates and Empirical PCB Trends in Subsurface Core Data

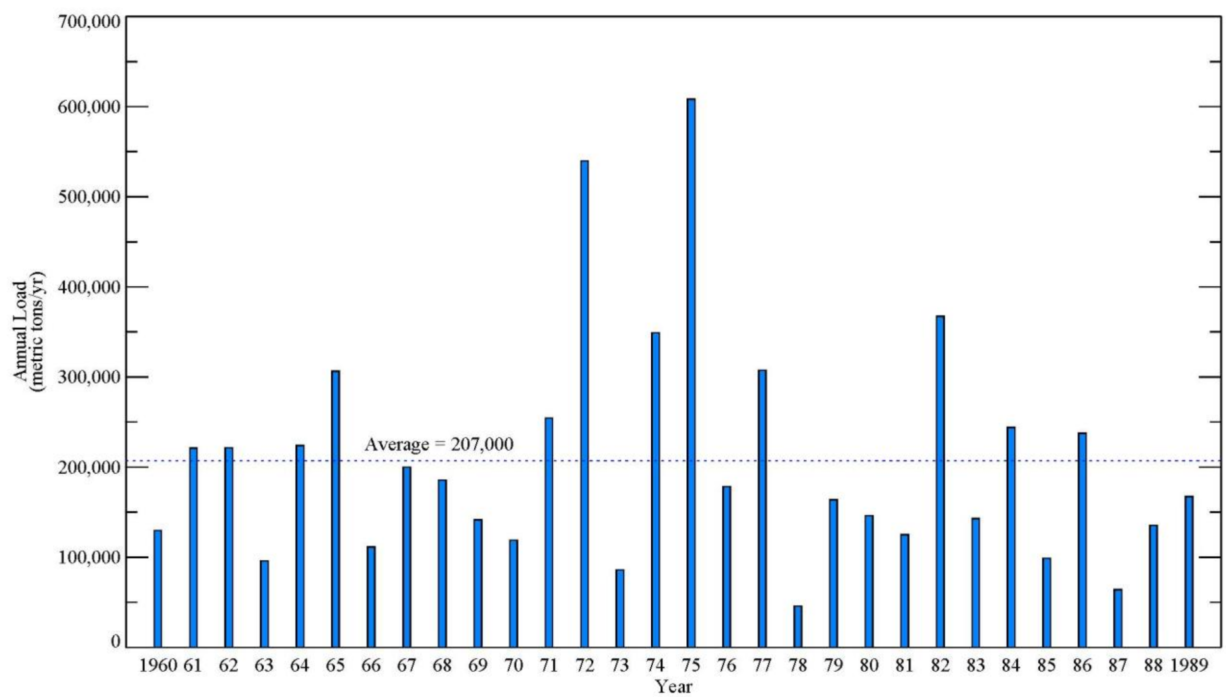
FIGURE 5-14

Figure 5-15a Maximum Flow Rate during Each Year from 1960 to 1989



Flow data: Fresh Water Discharge at USGS 12113000 (Green River).

Figure 5-15b Estimated Annual Total Sediment Load (suspended and bed load) in the Green River from 1960 through 1989



Note: 207,000 metric tons (MT) is annual average sediment load over 30-yr period. The annual average sediment load over the first 10 years (1960 – 1969) is 185,000 MT.

6 Areas of Potential Concern, Remedial Action Levels, and Recovery Potential

This section defines the areas of potential concern (AOPCs) with potentially unacceptable risks based on the findings of the baseline ecological and human health risk assessments (ERA and HHRA; Windward 2007a, 2007b). This section also presents the remedial action levels (RALs) designed to address these risks and used in developing the remedial alternatives. Lastly, this section presents categories of recovery potential for sediments in the Lower Duwamish Waterway (LDW) based on physical conditions and empirical trends in contaminant concentrations.

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Model Toxics Control Act (MTCA) require a feasibility study (FS) to identify volumes and areas of sediment where remedial action may be necessary and applied. Defining these areas requires

> *"…careful judgment and should include a consideration of not only acceptable exposure levels and exposure routes, but also site conditions and the nature and extent of contamination"* (EPA 1988).

Following U.S. Environmental Protection Agency (EPA) guidance (1988, 2005b), this section describes the relationship between location, extent, and concentrations of risk drivers relative to both hot spot areas and areas of lower level contamination. This information is used to delineate areas of sediment with potentially unacceptable risks. These areas are carried forward to Section 8, where technologies are assigned and remedial alternatives are developed. Further, the extent to which natural recovery is potentially viable is evaluated to guide the application of active and passive remedial actions in Section 8.

Hence, consistent with guidance, the steps in the FS process for mapping cleanup areas at the LDW include:

- Delineate AOPCs based on findings of unacceptable risks in the ERA and HHRA (Windward 2007a, 2007b). These areas will require consideration in this FS, and they are described in Section 6.1.
- Define a range of RALs that achieve or make progress toward achieving preliminary remediation goals (PRGs). RALs are contaminant-specific sediment concentrations that trigger the need for active remediation (e.g., dredging or capping). A RAL is equivalent to a "remediation level" under MTCA, which is defined as "…a concentration (or other method of identification) of a hazardous substance in soil, water, air, or sediment, above which a particular cleanup action component will be required as part of a cleanup action at a site" (Washington Administrative Code [WAC] 173-

340-200). A range of RALs, which trigger active remediation, is identified in Section 6.2. The remedial action objectives (RAOs; see Section 4) can be achieved through combinations of active remediation (triggered by the RALs), natural recovery, and institutional controls.

- Define areas within the AOPCs that have similar physical characteristics, engineering considerations, and recovery potential for which particular remedial technologies may be applied. These areas are referred to as recovery categories, which are discussed in Section 6.3.

Collectively, these evaluations are used in the assembly of the remedial alternatives in Section 8. Combinations of active and passive management of the AOPCs are evaluated relative to the RAOs. The AOPC boundaries and the recovery potential within those boundaries will likely need to be refined during remedial design and even, perhaps, during implementation of the remedy.

6.1 Delineating the Areas of Potential Concern (AOPCs)

The AOPCs represent the areas of sediment that have potentially unacceptable risks and will likely require application of active or passive remedial technologies. Defining the AOPC footprints requires: 1) an understanding of the types and levels of estimated risks in the LDW (see Section 3); 2) the RAOs to address those risks and associated PRGs (see Section 4); and 3) the conceptual site model, site conditions, and the data collection and analysis efforts over the past 20 years (see Section 2). The AOPC footprints defined for this FS are discussed in this section, along with a summary of the considerations used in deriving and evaluating these AOPCs. The contaminant concentrations used to develop the AOPC footprints include detected FS baseline surface sediment concentrations of risk drivers above the thresholds described below. The data used to define the AOPCs also include toxicity data and subsurface sediment data, when available (see Section 2).

The AOPCs do not include the five early action areas (EAAs; 29 acres), which are being addressed separately. However, the enhanced natural recovery (ENR) portion of the Duwamish/Diagonal EAA is included in AOPC 1. Evaluations used to define the AOPCs assume cleanup of the five EAAs will be completed prior to cleanup within the AOPCs. The two AOPC footprints developed for this FS are shown in Figure 6-1 and are described below.

Multiple thresholds were developed for each risk driver, and sediment areas were included in the AOPCs if any of the thresholds were exceeded. AOPCs are normally delineated by concentrations of contaminants of concern (COCs) or risk drivers above PRGs. For the LDW, the PRGs for total polychlorinated biphenyls (PCBs) and dioxins/furans (RAO 1) and for arsenic (RAO 2) are set at natural background for final cleanups, as required by MTCA. Model predictions indicate that natural background for these three risk drivers is unlikely to be achieved because of the concentrations of these risk drivers in incoming Green/Duwamish River suspended solids and because of practical limitations on control of lateral sources from the generally urban LDW drainage basin. For these reasons, it was not possible to use the RAO 1 PRGs for total PCBs or dioxins/furans or the RAO 2 PRG for arsenic to develop the AOPCs. Thus, a modified objective of getting those three risk-driver concentrations as close as possible to the natural background values (i.e., as low as practicable) was used to delineate the AOPCs. For the purposes of the FS, this is assumed to be the long-term model-predicted concentrations. These concentrations are believed to be the lowest technically achievable concentrations based on the available data and analyses conducted to date. These long-term model-predicted concentrations are uncertain, because future risk-driver concentrations in upstream- and lateral-source sediments are uncertain and may change in the future. The term "cleanup objective" in this FS is used to mean the PRG or as close as practicable to the PRG where the PRG is not predicted to be achievable. This FS uses long-term model-predicted concentrations as estimates of “as close as practicable to PRGs”.[1] AOPC 1 was designed to achieve this objective using a combination of active cleanup and natural recovery, and AOPC 2 was designed to achieve this objective using only active cleanup.

6.1.1 AOPC 1 Footprint

As noted above, natural background is unlikely to be achieved, and both the sediment transport model (STM) and bed composition model (BCM) predict that, in the long term, the LDW will reach concentrations similar to those incoming from the upstream Green/Duwamish River system. For these reasons, the FS has adopted an incremental approach to delineate AOPCs and to develop remedial alternatives with varying degrees of active remediation and natural recovery.

The AOPC 1 footprint is based on the PRGs that are not set at natural background (i.e., the PRGs associated with RAO 2 for risk drivers other than arsenic and with RAOs 3 and 4). Natural recovery is assumed to be required following active remediation of the AOPC 1 footprint to reduce site-wide average total PCB, dioxin/furan, and arsenic concentrations to the cleanup objective as defined above.

Interpolated surface sediment concentration maps for total PCBs, arsenic, carcinogenic polycyclic aromatic hydrocarbons (cPAHs), dioxins/furans, and contaminants that exceed the Sediment Management Standards (SMS) were the primary sources of information used to delineate the AOPC 1 footprint. In addition, shallow subsurface sediment contaminant concentrations were considered in areas prone to scour and disturbance and in intertidal areas where the point of compliance for human health direct contact risk drivers (PCBs, arsenic, cPAHs, and dioxins/furans) is the upper 45 cm of sediment. As described in Section 2, inverse distance weighting (IDW) was used for interpolating total PCBs, arsenic, and cPAHs, and Thiessen polygons were

1 For further information on cleanup objectives, see Section 9.1.2.3.

used to interpolate dioxins/furans and SMS exceedances in surface sediment. Each data layer was mapped independently. AOPC 1 was delineated where any of the layers exceeded the threshold concentrations described below.

RAO 3. AOPC 1 was first delineated for benthic community risk drivers with detected concentrations in surface sediments exceeding the sediment quality standards (SQS) (the RAO 3 PRGs). Each Thiessen polygon was classified as an SQS exceedance if one or more detected SMS contaminants exceeded this criterion. In addition, cleanup screening level (CSL) exceedances are also shown to indicate more highly contaminated areas. Toxicity test results, if available, were used in the final classification. If the Thiessen polygon exceeded the SQS, it was included in AOPC 1. Because total PCBs were spatially interpolated as dry weight concentrations (see Section 2 and Appendix A), the area with total PCB concentrations greater than 240 micrograms per kilogram dry weight (µg/kg dw; the dry weight equivalent of the 12 milligrams per kilogram organic carbon [mg/kg oc] SQS value, assuming 2% total organic carbon [TOC]) derived with IDW rather than Thiessen polygons was also used to delineate AOPC 1. Best professional judgment was used for mapping in cases where the total PCB IDW-based layer resulted in small, isolated areas exceeding 240 µg/kg dw. These small areas were not included in AOPC 1 if, using the sample-specific TOC data, they did not exceed the SQS on an organic-carbon normalized basis.

RAO 2. The AOPC 1 footprint was then evaluated for compliance with RAO 2. Active remediation of the AOPC 1 footprint achieves the total PCB PRGs (1,300 µg/kg dw for netfishing site-wide; 1,700 µg/kg dw for beach play areas; and 500 µg/kg dw for clamming areas). The footprint was expanded to achieve human health direct contact PRGs on a SWAC basis for cPAHs and dioxins/furans (380 µg toxic equivalent [TEQ]/kg dw and 37 nanograms [ng] TEQ/kg dw for netfishing [site-wide]; 90 µg TEQ/kg dw and 28 ng TEQ/kg dw for beach play; and 150 µg TEQ/kg dw and 13 ng TEQ/kg dw for clamming, respectively). The RAO 2 PRGs for arsenic are natural background over all three exposure areas (netfishing, clamming, and beach play), and therefore these PRGs are not likely to be achieved based on the model predictions. The AOPC 1 footprint was expanded to achieve site-wide and area-wide arsenic SWACs within the limits of what the long-term model predicts is achievable over time when natural recovery across the entire LDW is included. Also, to address beach play PRGs (RAO 2), individual beaches were included in AOPC 1 whenever the total direct contact excess cancer risks based on the beach play RME scenario (for all four human health risk drivers) exceeded 1×10^{-5}.

In intertidal areas, the point of compliance for human health risk drivers for clamming and beach play is assumed to be the upper 45 cm of sediment, because of potential exposures to people through direct contact with sediments during clamming or beach play activities. Average sediment concentrations from this interval[2] were considered and compared to the PRGs for direct contact tribal clamming and beach play RME scenarios. However this did not affect the designation of the AOPC footprint because the existing footprint covered these areas.

RAO 4. Active remediation of the AOPC 1 footprint achieves a site-wide spatially-weighted average concentration (SWAC) for total PCBs less than the RAO 4 PRG range of 128 to 159 µg/kg dw, and therefore no adjustment to AOPC 1 was required to meet RAO 4.

RAO 1. The AOPC 1 footprint was evaluated for compliance with RAO 1 PRGs, which are natural background concentrations on a site-wide basis for total PCBs and dioxins/furans. The footprint was not expanded for RAO 1. The FS assumes that remediation of AOPC 1 makes progress toward RAO 1 goals by achieving the long-term model-predicted sediment concentrations for total PCBs and dioxins/furans over time. Neither arsenic nor cPAHs have seafood consumption PRGs[3] for RAO 1, but remediation of AOPC 1 also reduces sediment concentrations for these risk drivers. Refer to Section 9 for predicted outcomes of the remedial alternatives.

Subsurface Contamination in Potential Scour Areas. Lastly, subsurface contamination was considered in the delineation of AOPC 1. Areas with SQS exceedances in the top 2 ft of sediment that are potentially subject to 100-year high-flow scour deeper than 10 centimeters (cm; as predicted by the STM; see Figure 2-9) or that are subject to vessel scour (see Figure 2-10) were added to the AOPC 1 footprint. In an area with an SQS exceedance in the top 2 ft of a core, the spatial extent was defined by the extent of the predicted high-flow scour area or the potential vessel scour area around that core. The spatial extent of the SQS exceedance within potential scour areas was conservatively assumed to be the entire extent of the potential scour area if there was only one core within that area (in part because there are relatively few subsurface sediment cores compared with surface sediment samples). If more than one core was located in a scour area, the spatial extent of the RAL exceedance was governed by the nearest core.

Summary. Table 6-1 lists the lowest risk-driver concentrations identified in surface sediment that were used to delineate AOPC 1 and the estimated post-construction

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2 Sediment data used to evaluate this interval included the following: surface sediment grabs in the top 10-cm, which were assumed to represent the top 45-cm; 0 to 45-cm depth samples in beaches; and where available the top 6-in or 1-ft core interval from subsurface sediment cores in intertidal areas.

3 Based on data collected during the RI, relationships between clam tissue and surface sediment concentrations of arsenic and cPAHs were too uncertain to develop quantitative risk-based threshold concentrations in sediment; therefore, no seafood consumption (RAO 1) PRGs were developed for these risk drivers.

SWACs if the entire AOPC 1 footprint was actively remediated.[4] It also compares those SWACs to the PRGs.

In summary, outside of the EAAs, the considerations used to delineate AOPC 1 were:

- Surface sediments with:
  - Areas delineated by Thiessen polygons that exceed the SQS criteria detected in surface sediment. Sediment toxicity data override chemical SQS or CSL exceedances and chemical passes, as described in Section 2.
  - Total PCB concentrations greater than 240 µg/kg dw
  - Arsenic concentrations greater than 57 mg/kg dw
  - cPAH concentrations greater than 1,000 µg TEQ/kg dw
  - Dioxin/furan concentrations greater than 25 ng TEQ/kg dw
  - Arsenic concentrations greater than 28 mg/kg dw in intertidal areas
  - cPAH concentrations greater than 900 µg TEQ/kg dw in intertidal areas.
- Areas with SQS exceedances in the top 2 ft of subsurface sediment that are predicted to be subject to 100-year high-flow scour deeper than 10 cm or are potentially subject to vessel scour based on empirical evidence.

AOPC 1 represents the maximum extent of any exceedance delineated by the layers described above. Therefore, the AOPC 1 footprint is larger than the area defined by the concentration for any one risk driver. Overall, the AOPC 1 footprint (Figure 6-1) represents about 180 acres or about 41% of the entire LDW site (441 acres).

The AOPC 1 footprint encompasses the initial area designated in the FS for remedial alternative development. Cleanup of the EAAs and all of AOPC 1, through a combination of active cleanup, verification monitoring, and natural recovery, is predicted to achieve cleanup objectives for RAOs 1, 2, 3, and 4. PRGs based on natural background for RAO 1 (total PCBs and dioxins/furans) and for RAO 2 (arsenic) are not predicted to be technically practicable, and thus, the cleanup objectives are to achieve long-term model-predicted concentrations that are as close to natural background as technically practicable.

6.1.2 AOPC 2 Footprint

In addition to AOPC 1 shown on Figure 6-1, EPA and the Washington State Department of Ecology (Ecology) required that an incrementally larger remedial footprint (outside of AOPC 1) be evaluated, called AOPC 2. The goal for final cleanup is to achieve

[4] The resulting SWACs were calculated by replacing the risk-driver concentrations in the AOPC 1 footprint with a post-remedy bed sediment replacement value, which is provided in Table 6-1. The SWACs do not assume any natural recovery.

concentrations as close to the natural background concentrations as technically practicable for total PCBs and dioxins/furans (RAO 1) and arsenic (RAO 2). Natural background for these three risk drivers is unlikely to be achieved because of incoming contaminant concentrations from the Green/Duwamish River and practical limitations on control of lateral sources. Instead, AOPC 2, when actively remediated along with AOPC 1, achieves the lowest long-term model-predicted SWACs for total PCBs, dioxins/furans, and arsenic[5] immediately after construction. AOPC 2 also addresses all areas outside AOPC 1 with subsurface contamination above the SQS. The AOPC 2 footprint is 122 acres. The AOPC 1 and AOPC 2 footprints combined encompass 302 acres (or approximately 68% of the LDW study area).

The AOPC 2 footprint was explored through a step-wise evaluation in which active remediation was first assumed for AOPC 1 plus every point with a total PCB concentration above 100 µg/kg dw. Second, site-wide SWACs for dioxins/furans and arsenic were calculated by changing the surface sediment concentrations in this larger footprint to the post-remedy bed sediment replacement values[6] and assuming no natural recovery. Based on these SWACs, the AOPC 2 footprint was then expanded to capture areas with:

- Arsenic concentrations greater than 15 mg/kg dw to achieve the long-term model-predicted site-wide SWAC.
- Dioxin/furan concentrations greater than 15 ng TEQ/kg dw to achieve the long-term model-predicted site-wide SWAC.

Finally, the footprint was again expanded to include remaining sediment cores with detected SQS exceedances at any depth (regardless of scour potential).[7]

The results of this analysis indicated that active remediation of AOPCs 1 and 2, using the post-remedy bed sediment replacement values for total PCBs, arsenic, and dioxins/furans, yields site-wide SWACs within the range of the long-term model-predicted concentrations (Table 6-1) immediately after construction. This analysis indicates:

1) Active remediation of the entire 302-acre AOPC 1 and AOPC 2 footprints would result in the lowest long-term model-predicted concentrations, and

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5 A cPAH threshold was not needed for AOPC 2 delineation because all areas where remediation is needed to meet cPAH PRGs are included in AOPC 1.

6 Post-remedy bed sediment replacement values in AOPCs 1 and 2 (respectively) for each risk driver are: total PCBs = 60 and 20 µg/kg dw; arsenic = 10 and 9 mg/kg dw; and dioxins/furans = 4 ng TEQ/kg dw (mid-range and low values, respectively, from Table 5-1c).

7 The exception is three cores collected from the Upper Turning Basin in 2009. Sediment in this area had not exceeded the SQS in previous samples, and data were not received in time to include in the AOPC 2 delineation. The sediment represented by these cores was dredged in 2010.

the model predicts that further changes over time after the cleanup through natural recovery would be minimal.

2) Any further active remediation would not yield additional sustainable SWAC reduction or risk reduction, because sediments from upstream and lateral sources would continue to deposit onto remediated areas.

It is important to recognize that, as with other input parameters, values used as post-remedy bed sediment replacement values for this analysis are uncertain. A range of replacement values was developed for each human health risk driver in this FS. The sensitivity of post-remedy sediment concentration predictions to the range of replacement values is described in Section 9. Based on this analysis, active remediation of the AOPC 1 and 2 footprints is predicted to reach long-term model-predicted concentrations. Cleanup of the EAAs and active remediation of the AOPC 1 and 2 footprints is predicted to achieve the maximum technically practicable degree of SWAC risk reduction. The areas beyond the AOPCs are not considered for active cleanup in this FS (but may be subject to sampling and verification monitoring during remedial design).

In summary, active remediation of AOPC 1 achieves the PRGs for RAOs 2 (for all human health risk drivers except arsenic), 3, and 4. The combined footprint of AOPCs 1 and 2 results in the lowest model-predicted SWACs for RAO 1 (total PCBs and dioxins/furans) and RAO 2 (arsenic) immediately after construction without consideration of natural recovery. Therefore, the AOPC 1 and 2 footprints are considered appropriate to identify alternatives that achieve the PRGs or make substantial risk reduction toward achieving the PRGs. The footprints have been defined with enough rigor to facilitate a detailed evaluation of remedial alternatives (in Section 8) for the purposes of this FS.

6.2 Remedial Action Levels

RALs are contaminant-specific sediment concentrations that trigger the need for active remediation (i.e., dredging, capping, or ENR). RALs define the active remediation footprint within the AOPCs for each remedial alternative (Section 8).

RALs are very different from PRGs. PRGs are the long-term cleanup levels and goals for the project, whereas RALs are point-based values that define where active remediation is to occur for a given alternative. PRGs are the same for all alternatives, whereas RALs vary among alternatives. RALs are also used as the compliance concentration to verify that active remediation for an alternative is complete, or successful, before equipment is demobilized from an area.

The development and use of RALs for this FS is based on the premise that once active remediation is complete (in areas where the RALs are exceeded), SWACs for human health risk drivers immediately following construction will be considerably lower than those for baseline conditions. The cleanup objectives are achieved either immediately after construction or over time through natural recovery. Higher RALs are associated with higher post-construction SWACs and larger areas that rely on natural recovery to achieve cleanup objectives. The evaluations of risk reduction over time and the time to achieve cleanup objectives are presented in Section 9.

For this FS, ranges of RALs are developed for the risk drivers (total PCBs, arsenic, cPAHs, dioxins/furans, and SMS contaminants [i.e., detected risk drivers that exceeded the SQS in surface sediments]) for which PRGs were presented in Section 4 (see Figures 6-2a through 6-2d for the human health risk drivers). RALs are developed with the understanding that remediation of these risk drivers will also address the remaining COCs (see Table 3-16) that do not have PRGs.

6.2.1 Methods Used for Development of RALs

This section briefly summarizes the methods used to develop a range of RALs that serve to define a range of active remedial footprints and a corresponding range of expected outcomes. The range of RALs allows a broad array of remedial alternatives to be defined in Section 8, each with differing:

- Areas/volumes of sediment to be actively remediated
- Levels of risk reduction immediately after construction
- Time frames for achieving cleanup objectives.

The residual risks remaining immediately after construction of each remedial alternative and additional risk reduction predicted over time through natural recovery are discussed in Sections 9 and 10 of this FS.

RAL development considers only individual COCs and does not consider the extent to which COCs are commingled. Because many of the LDW COCs have some commingling and co-occurrence, it is reasonable to expect that by remediating an area to address one risk driver exceeding a RAL, some reduction in other COCs will also occur. Thus, the remediation of sediments exceeding RALs may result in risk reduction not accounted for when only individual COCs are evaluated. Section 9.11 describes how the remedial alternatives address COCs other than the risk drivers. In addition, natural recovery is predicted to further reduce sediment concentrations over time below the reduction achieved by active remediation alone.

The approaches used to select RALs and to develop an array of remedial alternatives require best professional judgment. The RALs for this FS were selected based on the following considerations:

- **Achievement of PRGs.** Certain sediment PRGs can directly translate into RALs, such as SMS criteria applied on a point basis, which directly relate to protection of benthic receptors (RAO 3). RALs for RAO 3 were defined using two time points: at the end of construction and 10 years after construction, in accordance with SMS guidelines. Although not defined in the RAL development process, some RALs may require more than 10 years after construction to achieve PRGs. Area-based PRGs (SWACs) for certain direct contact scenarios (RAO 2) are the basis for point-based RALs for this FS.

- **Range of RALs**. By definition, the RALs are point concentrations that exceed PRGs and require active remediation. However, a direct comparison of point concentrations (at specific sample locations) to PRGs is not appropriate for RAO 1 (seafood consumption), RAO 2 (direct contact), and RAO 4 (wildlife consumption of prey) because these RAOs have SWAC-based PRGs. Therefore, each SWAC-based PRG needs to be "converted" to a not-to-exceed point concentration (RAL). To accomplish this "conversion" from SWACs to point concentrations, human health risk drivers were evaluated in an iterative fashion (called "hilltopping") by ranking their concentrations from highest to lowest (using interpolated grid cells). The highest values were sequentially replaced with a post-remedy bed sediment replacement value until the appropriate site- or area-wide PRG was achieved. The highest concentration remaining then becomes the RAL for the SWAC-based PRG.

  A range of RALs was selected for each human health risk driver by comparing the highest remaining concentration to the resulting SWAC. The RALs were selected to represent a range of acres remediated and the resulting SWACs. Figures 6-3a through 6-3d present the hilltopping curves for the four risk drivers. The RALs (point values) are identified on the curves relative to the estimated SWACs they achieve based only on active remediation and no natural recovery.

- **SWAC Reduction for PRGs Set at Natural Background.** Certain PRGs, such as those for total PCBs and dioxins/furans for RAO 1 and for arsenic for RAO 2, cannot be used directly as RALs because they are set to natural background (Table 4-7). It is not technically possible to implement a RAL set at natural background because although sediments continually entering the LDW from upstream have COC concentrations considerably lower than those in LDW sediments, these concentrations are still above natural background concentrations. For PRGs set at natural background, a range of RALs was selected to achieve the long-term model-predicted concentrations over time and immediately after construction.

  As incrementally lower RALs were considered and more acres were identified for active remediation, a point of minimal change in SWAC was predicted. The estimated curves, shown in Figures 6-3a through 6-3d,[8]

[8] Section 9 contains SWAC-over-time curves based on future site-wide SWACs predicted using the BCM.

approach a value (the asymptote) driven by continual upstream inputs from the Green/Duwamish River as well as urban inputs from lateral drainage to the LDW. The estimated rate of change (SWAC reduction per acre) is predicted to be so small that, immediately after construction, the site would be considered to have reached the lowest model-predicted post-construction SWAC. Through continued natural recovery over time, the site would reach the long-term model-predicted concentrations (shown as the asymptote on the curve). It is worth noting that predicted changes in the post-remedy SWACs (shown in Figures 6-3a through 6-3d) are largely driven by the post-remedy bed sediment replacement values, while the long-term model-predicted concentrations are largely dependent on concentrations associated with upstream sources and to a lesser extent, lateral sources (see Tables 5-1a through 5-1c).

6.2.2 Range of Selected RALs

The array of RALs and how they relate to each RAO are summarized in the following subsections and in Table 6-2.

6.2.2.1 RAO 1 (Human Health Seafood Consumption) RALs

For this FS, progress toward achievement of RAO 1 (reduction of human health risks from seafood consumption) is assessed based on estimated reductions in the site-wide SWAC of total PCBs, arsenic, cPAHs, and dioxins/furans. The RALs for each risk driver are described below.

The total PCB PRG for RAO 1 is not expected to be achieved because it is set at natural background. Therefore, the goal is to set an array of RALs that result in incrementally lower site-wide SWACs after construction and shorter model-predicted natural recovery periods to reach cleanup objectives. (However, at very low RALs, time to achieve cleanup objectives increases due to longer construction times.) A total PCB RAL of 2,200 µg/kg dw was selected to address hot spots. The remaining RALs of 1,300, 700, 240, and 100 µg/kg dw comprise a range resulting in incrementally larger areas of active remediation and corresponding reductions in the site-wide SWAC immediately after construction (Table 6-2). The SWAC reduction is in turn predicted to result in a commensurate incremental reduction in human health risks. A RAL of 1,300 µg/kg dw is based on the CSL.[9] A RAL of 700 µg/kg dw is based on providing a well-spaced range of RALs for evaluation. A RAL of 240 µg/kg dw is based on the SQS.[10] The lowest total PCB RAL (100 µg/kg dw) is predicted to yield minimal change in the average

[9] Assuming a TOC content of 2% (the site-wide average), the total PCB dry weight equivalent of the CSL (65 mg/kg oc) is 1,300 µg/kg dw. If selected, actual implementation of this RAL would be based on the organic carbon-normalized CSL.

[10] Assuming a TOC content of 2%, the total PCB dry weight equivalent of the SQS (12 mg/kg oc) is 240 µg/kg dw. If selected, actual implementation of this RAL would be based on the organic carbon-normalized SQS.

concentration immediately after construction, and to achieve the long-term model-predicted concentration range. As discussed in Section 6.1.2, further active remediation is not predicted to appreciably lower the site-wide SWAC for total PCBs.

For arsenic and cPAHs, 95% or more of the risk associated with seafood consumption is attributable to the consumption of clams. A relationship between the concentrations of arsenic and cPAHs in clam tissue and sediment would be required to estimate sediment risk-based threshold concentrations (RBTCs) for RAO 1. However, RI data showed a poor relationship between clam arsenic and cPAH concentrations and associated sediment concentrations (i.e., clam tissue-to-sediment relationships for both arsenic and cPAHs were too uncertain to develop quantitative sediment RBTCs). Because of this, neither arsenic nor cPAHs have seafood consumption PRGs. RALs were selected for each to provide for overall reductions in sediment concentrations of these two risk drivers. Co-occurrence with the other risk drivers will also reduce site-wide sediment concentrations. For arsenic, a RAL of 93 mg/kg dw (the CSL) is used to address hot spots, and two other RALs, 57 (the SQS) and 15 mg/kg dw, are used to provide a range. For cPAHs, a RAL of 5,500 µg TEQ/kg dw is used to address hot spots, and two other RALs, 3,800 and 1,000 µg TEQ/kg dw are used to provide a range.

The dioxin/furan PRG for RAO 1 is not expected to be achieved because it is set at natural background. Therefore, the goal is to set a range of RALs that result in incrementally lower site-wide SWACs following active remediation. A RAL of 50 ng TEQ/kg dw was selected to address hot spots. Other dioxin/furan RALs of 35, 25, and 15 ng TEQ/kg dw comprise the range resulting in incrementally larger areas of active remediation and corresponding reductions in the site-wide SWAC immediately after construction (Table 6-2). The lowest dioxin/furan RAL (15 ng TEQ/kg dw) is predicted to result in minimal change in the site-wide SWAC and to achieve the long-term model-predicted concentration immediately after construction is complete. Further active remediation is not predicted to appreciably lower the site-wide SWAC for dioxins/furans.

6.2.2.2 RAO 2 (Human Health Direct Contact) RALs

Achievement of RAO 2 is assessed on three spatial scales, based on the three direct contact exposure scenarios: site-wide for netfishing, area-wide within potential clamming areas, and area-wide within beach play areas. In addition, future-use scenarios for beach play are evaluated in all intertidal areas (see Figure 3-1).

Netfishing

The netfishing exposure area is site-wide (441 acres) and the point of compliance is surface sediment (0 to 10 cm). For total PCBs, cPAHs, and dioxins/furans, the netfishing site-wide PRGs are predicted to be achieved immediately following remediation of the EAAs. All arsenic direct contact PRGs are set to natural background; therefore, they are unlikely to be achieved. The goal is to achieve the long-term model-predicted concentration. An arsenic RAL of 93 mg/kg dw is used to address hot spots.

The remaining RALs (57 and 15 mg/kg dw) provide a range, with the lowest RAL set to achieve long-term model-predicted concentrations at the end of construction. Further active remediation is not predicted to appreciably lower the site-wide SWAC for arsenic.

Beach Play Areas

As described in Section 3, the LDW has eight beach play areas; note that these are not all necessarily areas where beach play currently occurs but they were identified as such because public access is possible. The beach play scenario is evaluated on an average basis at individual beaches and across all beaches combined (exposure areas). The point of compliance for the beach play scenario is 0 to 45 cm. Intertidal RALs were developed for arsenic, cPAHs, and dioxins/furans. For total PCBs, an intertidal RAL was not needed in these areas because the tribal clamming and beach play direct contact PRGs for total PCBs are predicted to be achieved following remediation of the EAAs and hot-spot areas.[11]

The PRGs for the beach play areas are the 10^{-6} RBTCs for the individual risk drivers (with the exception of arsenic where the PRG is set at natural background). Total PCB beach play PRGs are predicted to be achieved at all of the individual beach play areas using the highest RAL of 2,200 µg/kg dw. The PRG of natural background for arsenic is unlikely to be achieved. For cPAHs, the PRG falls within the range of upstream inputs and post-remedy bed sediment replacement values, and therefore may not be achieved at all beach play areas, although some of the individual beaches are predicted to achieve the PRG. A dioxin/furan intertidal RAL was set to the 10^{-6} RBTC for beach play.

The beach play RALs for both arsenic and cPAHs are set to the 10^{-5} RBTCs as points to ensure that, at a minimum, 1) the total 10^{-5} risk goals required by MTCA are achieved, and 2) progress is made toward achieving 10^{-6} RBTCs (or natural background for arsenic) on an average basis over the beaches. For arsenic, cPAHs, and dioxins/furans, RALs of 28 mg/kg dw (10^{-5} RBTC), 900 µg TEQ/kg dw (10^{-5} RBTC), and 28 ng TEQ/kg dw (10^{-6} RBTC), respectively, are applied in all intertidal areas, and hence, all potential current and future beach play areas.

Clamming Areas

The tribal clamming scenario is evaluated on an area-wide basis across the potential clamming exposure areas. The same point of compliance considerations that applied to beach play, as described above, also applied to clamming areas. The direct contact tribal clamming PRG for total PCBs is predicted to be achieved after the EAAs have been actively remediated (Figures 6-2a through 6-2d). An arsenic RAL of 93 mg/kg dw, applied on a point basis, is expected to achieve the tribal clamming 10^{-5} RBTC; the 10^{-6}

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[11] In intertidal areas, compliance for total PCBs was evaluated based on surface sediment and limited to the 10 cm depth (the biologically active zone). The site-wide RAL for total PCBs (in the top 10 cm) achieved the cleanup objectives for direct contact clamming and beach play areas.

RBTC is below natural background. The lower arsenic RALs (57, 28, 15 mg/kg dw) are designed to achieve incrementally lower SWACs and the long-term model-predicted concentrations in potential clamming areas. The RALs discussed above for cPAHs and dioxins/furans in beach play areas are also predicted to result in SWACs that achieve the PRGs in clamming areas, so no RALs based on tribal clamming were set for these two risk drivers.

6.2.2.3 RAO 3 (Protection of Benthic Invertebrates) RALs

The RALs for any risk-driver SMS contaminant for RAO 3 are:

- **CSL10 –** achieves the CSL within 10 years after construction is complete. The locations exceeding the CSL within 10 years were predicted using the recommended BCM input parameters. The BCM methods are described in Section 5, and predicted outcomes are shown in Section 9 and Appendix F.
- **CSL –** achieves the CSL by the time construction is complete.
- **SQS10 –** achieves the SQS within 10 years after construction is complete. The locations exceeding the SQS within 10 years were predicted using the recommended BCM input parameters. The BCM methods are described in Section 5, and predicted outcomes are shown in Section 9 and Appendix F.
- **SQS –** achieves the SQS by the time construction is complete.

SMS criteria for total PCBs and the other non-polar organic compounds are on an oc-normalized basis. Total PCB RALs for RAO 3 are 12 and 65 mg/kg oc for the SQS and CSL, respectively, but may be expressed as dry weight values in the FS for mapping purposes and ease of discussion (240 and 1,300 µg/kg dw for SQS and CSL, respectively, assuming 2% TOC). The SMS criteria for metals are expressed on a dry weight basis. For arsenic they are 57 and 93 mg/kg dw, for the SQS and CSL, respectively.

Implementation of the time-dependent RALs (SQS10 and CSL10) requires prediction of location-specific future concentrations using the BCM (methods are described in Section 5, and predicted outcomes are presented in Section 9 and Appendix F).

6.2.2.4 RAO 4 (Ecological Receptor Seafood Consumption) RALs

For RAO 4, total PCBs is the only risk driver. Achievement of the PRG (hazard quotient less than 1.0) is assessed on a site-wide basis. Separate RALs were not defined for RAO 4 because the total PCB range of RALs described above for RAO 1 (2,200, 1,300, 700, 240, and 100 µg/kg dw) is predicted to achieve RAO 4 immediately after construction or through a combination of active remediation and natural recovery.

6.3 Evaluating Recovery Potential of Sediments within the AOPCs

This section presents an evaluation of recovery potential intended to guide the final assembly of remedial alternatives (Section 8) within the AOPCs (outside of EAAs) and to prioritize areas that will likely require active remediation. This evaluation considers several factors, including proximity to potential contaminant sources, net sedimentation rates, scour potential, and empirical trends, that affect the ability of areas to recover through natural processes.[12]

The entire LDW was grouped into three categories with regard to recovery potential (Figures 6-4a and 6-4b). A recovery category represents areas of the LDW that share similar characteristics that could affect how well different remedial technologies would achieve the RAOs and how feasible they would be to implement. The recovery categories are:

- **Category 1** includes areas where recovery is presumed to be limited. It includes areas with observed and predicted scour, net scour, and empirical data demonstrating increasing concentrations over time.
- **Category 2** includes areas where recovery is less certain. It includes areas with net sedimentation and mixed empirical contaminant trends.
- **Category 3** includes areas where recovery is predicted. It includes areas with minimal to no scour potential, net sedimentation, and empirical trends of decreasing concentrations.

6.3.1 Mapping the Lines of Evidence for Evaluating Recovery Potential

To delineate the areas in each of these recovery categories, the following physical and chemical lines of evidence were considered (Table 6-3):

- Scour and deposition patterns:
  - Annual net sedimentation rates estimated by the STM and averaged over the 30-year STM period
  - 100-year high-flow event scour areas predicted in the STM (maximum scour depth observed over the 30-year model period)
  - Areas with empirical evidence of vessel scour, as interpreted from 2003 bathymetric survey sun-illumination maps.
- Land and water use functions:

12 When reviewing empirical trends, proximity to contaminant sources, depth of contamination, and type of contaminant exceedance were considered. When source control is complete, recovery may be viable but not yet observed empirically.

- Berthing areas, former dredging events, and potential for disturbance by future dredging
- Proximity to the toe of the slope along the navigation channel
- Shoreline land use, public access, and outfall locations
- Overwater structures
- Vessel traffic patterns, based on knowledge of navigational operations, operator interviews, and bridge opening logs
- Habitat restoration areas, recreational shoreline access areas, and historical cleanup areas.

- Empirical evidence of recovery through total PCB and other risk-driver concentration trends (excluding dioxins/furans) in:
  - Surface sediment from resampled stations
  - Subsurface sediment from the top two intervals (the shallowest 2 ft) of cores.

Table 6-3 lists the key lines of evidence and the specific criteria used to delineate each recovery category, which are discussed below. The GIS maps showing the extent of these features are presented in Section 2 and Appendix F. Other bulleted items (not listed in Table 6-3) were secondary considerations used as lines of evidence to help interpret and evaluate empirical trends and to delineate the layers in Table 6-3. For example, overwater structures and former dredging events were used to define active berthing areas. The following subsections describe how these features were overlaid to map recovery category areas. Recovery categories are defined only for the purposes of developing site-wide remedial alternatives and assigning remedial technologies (Section 8). Location-specific design considerations and new empirical data for these areas will be evaluated during remedial design.

Figure 6-4a presents the three recovery categories. Figure 6-4b includes the empirical contaminant trends with the recovery categories. A detailed analysis of this process by subarea is provided in Appendix D.

6.3.1.1 Net Sedimentation

Natural recovery processes in the LDW include the natural deposition of cleaner sediment from upstream that is expected to reduce surface sediment COC concentrations.[13] Recovery is not considered viable if the STM estimates a potential for net scour (no sedimentation under average flow conditions); such areas are considered Category 1. Any positive rate of sedimentation indicates that an area may potentially be

[13] Important mechanisms that reduce surface sediment contaminant concentrations are deposition of sediment sourced from upstream, followed by mixing and burial (see Section 2, Figure 2-11). These processes are described in greater detail in Section 5.

amenable to natural recovery, and thus this criterion places an area in Category 2 or 3 unless other lines of evidence suggest recovery is not occurring.

Additionally, changes in bathymetric data between 2003 and 2008 in the navigation channel were reviewed (Figure 6-5) as a qualitative check on STM-estimated net sedimentation rates. Pre-dredge bathymetric data collected in 2008 in the navigation channel by the USACE were paired with bathymetric data collected in August 2003 by LDWG.[14] Figure 6-5 displays the differences in elevation at points along the 2008 transects in the navigation channel. Where data from both surveys were available, differences observed over this 5-year period suggest that sedimentation had occurred in much of the navigation channel. While not used as a primary line of evidence for assigning recovery categories, many areas of empirically-estimated deposition roughly match the model predictions. However, differences in survey methods and limited documentation of the bathymetric surveys have produced some uncertainties in the data, which may inaccurately show some areas as having scour (e.g., RM 1.7 to RM 1.9 near Slip 2).

6.3.1.2 High-flow Events

High-flow events increase the rate of erosion in certain areas of the LDW, which could reduce recovery potential. Scour deeper than 10 cm, as estimated by the STM to occur any time during a 100-year high-flow event, is evidence that recovery may not be occurring (see Figure 2-9). A depth of 10 cm was selected because it is the depth of the biologically active zone and the depth of most of the surface sediment samples in the FS baseline dataset.

6.3.1.3 Vessel Scour Areas

Vessel scour areas were identified based on observed ridges and furrows (as determined using the sun-illuminated image of the 2003 bathymetric data) assumed to be caused by vessel traffic along established vessel traffic routes. These bed form areas are assigned to Category 1 because deposited sediment may be eroding or sedimentation may be restricted. The mapping of this layer was restricted to areas where active berthing (vessels and overwater structures) was observed because vessels maneuvering into these areas may be causing scour or because spud placement during vessel mooring may be disturbing the sediments. Bed forms identified outside of berthing areas could represent spud mounds (from vessels moored outside of mapped berthing areas), depressions from vessels resting on the bottom in shallow water, debris, or shallow track lines from transiting vessels. However, these bed forms outside of known vessel use areas are relatively shallow and localized and are not expected to expose buried contamination or impede recovery. Therefore, the mapping of vessel

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[14] The August 2003 data collection effort predated the January 2004 maintenance dredging in the navigation channel from RM 4.3 to 4.65 (the last navigation channel dredging event prior to the 2003 data collection was in January 2002; Table 2-9).

scour areas was restricted to higher-traffic areas based on the presence of a pier/wharf face, documented maintenance dredging events, and/or operator interviews indicating that the area supports frequent vessel traffic (see Figures 2-10 and 6-5).

6.3.1.4 Berthing Areas

Berthing areas are locations in the LDW adjacent to existing overwater structures that are not part of marinas, such as piers, wharves, pile groups, and dolphins (Figure 2-28 displays both overwater structures and berthing areas). These areas are assumed not to be viable for natural recovery if evidence of vessel scour was observed or empirical trends show increasing concentrations of risk drivers (excluding dioxins/furans). Berthing areas without evidence of vessel scour are assumed to exhibit recovery potential and thus were placed in Category 2. Berthing areas with evidence of vessel scour were placed in Category 1. Empirical contaminant trends, when available in berthing areas, were used as a final check to either confirm a recovery category designation or as an override to assign an area to another recovery category depending on the observed trend (see next subsection).

6.3.1.5 Empirical Contaminant Trends

Empirical trends in risk-driver (excluding dioxins/furans) concentrations were used as a final check to either confirm or override recovery category assignments based on physical criteria on a case-by-case basis. The identification of a sample location as belonging to an empirical trend category followed a three-step process. First, sample locations with the appropriate data (resampled surface sediment locations within 10 ft of one another or cores with two sample intervals in the top 2 ft) were identified (Table 6-4; Part 1). Second, each detected risk driver exceeding the SQS was assigned to one of three categories (Table 6-4; Part 2):

- Increase: contaminant concentration increasing more than 50% over previous or deeper concentration
- Equilibrium: a small (less than 50%) change in concentration
- Decrease: contaminant concentration decreasing more than 50% from previous or deeper concentration.

Third, the trend assignments for the risk drivers exceeding the SQS were grouped into a summary designation for each location (Table 6-4; Part 3 and Figure 6-4b). Dioxins/furans were not evaluated because of a lack of temporal data. Figure 6-4b shows two symbols per location, one for total PCBs alone and another for all other risk-driver contaminants:

- Increase (red): All contaminants evaluated increased by more than 50%. A location with two red symbols was in Category 1.

- Equilibrium or mixed (gray): A location with mixed results by contaminant (risk drivers other than total PCBs having any combination of assignments in bulleted list above) or concentration changes in equilibrium (less than 50% change) was in Category 2. If a location's trend assignment of "mixed" was based on a combination of decreasing trends and equilibrium (but no increasing trends) that location was in Category 3.
- Decrease (blue): All contaminants evaluated had concentrations decreasing by more than 50%. A location with two blue symbols was in Category 3.
- Below SQS (green): Total PCBs or all other contaminants were not detected above the SQS.

The shape of the symbol denotes whether it is a co-located surface grab sample or a sediment core. Empirical overrides of the physical criteria (Table 6-3) occurred on a case-by-case basis (described in Table D-2). The empirical data are discussed in greater detail in Appendix F.

6.4 Uncertainty Analysis of AOPCs and Recovery Potential

Uncertainties in the process of developing AOPC footprints and recovery potential categories are discussed below.

6.4.1 AOPC Uncertainty

This section examines the degree of confidence that exists with the estimate of the AOPC footprints using the criteria discussed in Section 6.1. The primary factors contributing to uncertainty in the AOPC footprints are:

- Age of the data
- Data mapping and interpolation
- Use of SWACs instead of 95% upper confidence limit (UCL95) on the SWAC.

These uncertainties are discussed below.

6.4.1.1 Age of Data

The FS baseline surface sediment dataset was used to map the AOPCs. Older data at stations that were resampled (collected within 10 ft of newer data) were excluded from the FS baseline dataset on a contaminant-by-contaminant basis. The intent was to use the most recent data available for defining the nature and extent of contamination. LDWG conducted sampling in 2005, 2006, 2009, and 2010 to expand and update the existing dataset. However, because the FS study area is large (441 acres), some data that are more than 10 years old remain in the dataset.

The FS baseline surface sediment dataset is comprised of over 1,400 surface sediment samples spanning 20 years of data collection (1990 through 2010). Between 1990 and 2004, approximately 1,200 surface sediment samples, 340 subsurface sediment cores, and 90 fish and shellfish tissue samples were collected from the LDW by parties other than LDWG. These samples and cores were analyzed for metals and organic compounds. Data that were deemed acceptable based on a review of analytical methods and quality assurance reports became part of the RI and FS baseline datasets. Additional data were collected from 2004 to 2006 by LDWG for the RI to characterize contamination and physical properties of the LDW. These data included approximately 900 samples of the following media: fish, clam, crab, and benthic invertebrate tissue; seep water (water seeping from banks along the LDW); surface sediment (the top 10 cm); subsurface sediment (below the top 10 cm); and porewater (water in spaces between sediment particles). In 2009 and 2010, LDWG collected an additional 41 surface sediment samples and 6 composite sediment samples for the FS to characterize beach play areas and to expand the dioxin/furan dataset.

Many of the sediment samples are now over 10 years old, and surface conditions may have changed in these sampled areas. In mapping the AOPCs, however, this level of uncertainty is considered to be acceptable for the FS by assuming all data points represent baseline conditions. Remedial alternatives are assembled around these predictions along with other lines of evidence described in Section 8. Sampling conducted during remedial design will be conducted to help reduce any outstanding uncertainties. To account for uncertainties associated with older data being used to evaluate RAL exceedances, areas of AOPC 1 meeting all or most of the following characteristics are assumed to be candidates for verification monitoring during remedial design:

- Relatively old data (i.e., sampled prior to 1998)[15]
- Risk-driver concentrations exceeding but close to the AOPC 1 RALs, specifically SQS exceedances less than 1.5 times the SQS or total PCB concentrations slightly over 240 µg/kg dw
- Isolated points (i.e., only 1 point with an SQS exceedance in a 0.5-acre or larger area or where a point is surrounded by passes)
- Not in Recovery Category 1
- BCM predictions of recovery within 10 to 20 years from baseline.

Verification monitoring during remedial design should confirm whether the sediments in these areas exceed the RALs. Areas designated as candidates for verification

[15] The AOPC footprint was first delineated in 2008 for the draft FS. Samples collected prior to 1998 were more than 10 years old at that time (2008).

monitoring are shown in Appendix D and are mapped separately in the remedial alternatives (Section 8). No empirical time trend data were available for these 23 acres.

6.4.1.2 Data Mapping and Interpolation

The FS baseline dataset contains data from numerous site investigations conducted over the past 20 years. These investigations have been used to determine the nature and extent of sediment contamination associated with past hazardous substance releases. This extensive dataset was used to build the conceptual site model, map the nature and extent of contamination, and understand site processes for evaluating remedial alternatives. However, as with every environmental investigation, some uncertainty remains associated with the horizontal and vertical extent of sediment contamination, as discussed in the following points:

- **Laboratory Reporting Limits:** A portion of the uncertainty is related to reporting limits that exceed the screening criteria, especially in older data. Therefore, only detected SQS exceedances (expressed spatially as Thiessen polygons) were used to delineate the AOPCs for RAO 3. Samples with only undetected data (i.e., reporting limits) exceeding the SQS criteria were not considered exceedances. In the ERA (Windward 2007a), an evaluation of the reporting limits that exceeded the SQS concluded that there was a low probability that these exceedances would be of concern.

- **Sampling Design**: Another portion of the AOPC uncertainty is related to the uneven distribution of sampling in historical datasets. Good spatial coverage exists throughout the LDW, but the sampling density is not evenly distributed. For example, some investigations targeted specific areas (e.g., Boeing Plant 2) and these areas have much denser sampling coverage than other areas of the LDW. For this reason, the spatial extent of contamination remains somewhat uncertain, which is common in the feasibility study phase of any large site. Sampling coverage and density will be refined through the addition of new data collected during remedial design.

- **Interpolation Methods:** Two interpolation methods were used to map surface sediment data (IDW and Thiessen polygons; see Appendix A). Each of these methods has inherent uncertainties, including the sampling density, influence of geomorphology on the distribution of contaminants, and influence of surrounding data. The uncertainty in these methods was minimized by conducting an extensive exploratory analysis and by optimizing the IDW parameters used for interpolating total PCBs, arsenic, and cPAHs. This parameterization simulates a "best-fit" estimate of the true concentration gradients (Appendix A). The selected mapping techniques (i.e., IDW interpolation and Thiessen polygons) are well documented and widely used in managing contaminated sediments. The spatial extent of

COC concentrations is expected to be refined during the remedial design phase when additional samples are collected.

- **Vertical Compositing**: The subsurface sediment dataset includes many sediment cores that extended down to "native sediments," where most contaminant concentrations were below the SQS. This was documented in the logs for the cores collected in 2006 for the RI. However, many cores collected for other sampling events did not have logs, were composited over broad depth intervals (e.g., 4 ft), or were too shallow to reach the native sediments and/or the bottom of contamination. For these shallower cores, the interpreted bottom of contamination may not be the true bottom. Some of the vertical core samples were composited over 2-ft or longer intervals, such that either the bottom of contamination is not completely understood within the sample interval or the depth within the core for the highest contaminant concentrations is not completely understood.

- **Vertical Extent of Contamination:** On a site-wide scale, the vertical extent of contamination (greater than SQS) has been interpolated into an isopach layer representing the bottom of this contamination (described in Appendix E). The native alluvium contact, which has also been interpolated into an isopach layer, can be used as a surrogate for the uncertainty in the extent of the bottom of contamination for this FS (see Appendix E). The top of the native alluvium isopach layer is also assumed to be the maximum depth of any subsurface sediments with total PCB concentrations greater than 100 µg/kg dw (below this contact, sediments are assumed to exhibit native, pre-industrialized conditions). Because cores are much less numerous than surface sediment samples, the interpolation of the subsurface contamination may not represent actual conditions as effectively as it does for surface sediments. These estimates will need to be refined during remedial design.

  Additionally, the cores were collected by many different parties using various sampling methods and compositing schemes. The data were also not always accompanied by field and core processing logs that could be used to adjust recovered depths to *in situ* depths or to provide other useful information. Finally, not all intervals within each core were sampled, and within those intervals sampled, not all COCs were analyzed. If a sampling interval was not analyzed and the interval immediately above was contaminated, then the bottom of the contamination is assumed to be the bottom of the skipped interval. For cores that did not reach the bottom of contamination (detected SQS exceedances), 1 ft was added to the depth of the bottom of the core, and this depth was assumed to be the bottom of contamination.

6.4.1.3 95% Upper Confidence Limits (UCL95) on SWACs

The UCL95 on the mean is a statistically derived quantity associated with a representative sample from a population (e.g., sediment or tissue chemistry results) such that 95% of the time, the true average of the population from which the sample was taken will be less than the quantity statistically derived from the sample dataset (e.g., 95% of the time, the true average sediment contaminant concentration will be less than the UCL95 based on sediment chemistry sample results). The UCL95 is used to account for uncertainty in contaminant concentration measurements and to ensure that contaminant concentrations are not underestimated.

The AOPCs were delineated in part by estimating when a post-remediation site-wide SWAC achieves a target concentration. Therefore, mean values, not UCL95s, were used to delineate the AOPCs and evaluate predicted results in Section 9. However, in accordance with EPA and Ecology policy for evaluating compliance and estimating exposure concentrations, an upper confidence limit of the true mean (UCL95 on the SWAC) will be developed for each compliance monitoring dataset and compared to the target goal to account for sampling variability. The UCL95 from a well-designed post-remediation sampling program is expected to exceed the true SWAC by some increment.

Because the delineation of the AOPCs and the evaluation of the remedial alternatives are based on SWACs instead of UCL95 values, the footprints could potentially be larger. However, remediation of incrementally larger footprints manages contaminated sediment to incrementally lower concentrations, decreasing variability in the dataset. Footprints based on achieving the long-term model-predicted concentrations (SWACs) are likely not much different than those based on UCL95 values because, over time, natural recovery, coupled with remediation of hot spots, will reduce variability such that SWACs and UCL95 values become similar.

Appendix H discusses methods for calculating the UCL95 on the SWAC using the total PCB RI baseline dataset.

Overall, the nature and extent of sediment contamination is sufficiently understood to characterize risks, and develop reasonable estimates of the AOPCs and LDW-wide remedial alternatives for the FS. Uncertainty in the horizontal and vertical extent of sediment contamination above selected RALs will be refined during remedial design.

6.4.2 Recovery Potential Uncertainty

The recovery categories synthesize a large amount of information into a simple construct that can be used for managing uncertainty in technology assignments for this FS-level analysis. However, each criterion used in this analysis contains both uncertainties and assumptions. Remedial design-level analysis will provide additional information that will supersede many of the assumptions in this analysis. A few of the major assumptions that may affect an FS and remedial design-level analysis include:

- Berthing areas, navigation channel operations, and elevations necessary for berthing and navigation may change.
- Further observations and analysis of location-specific vessel scour and its effect on recovery may change. Location-specific analysis of impacts may result in different conclusions on scour potential and may change technology selection.
- STM estimates may be combined with location-specific empirical data to refine sedimentation rates and scour potential.
- Additional data could refine location-specific contaminant trends over time.
- Source control changes could affect the rate of observed natural recovery.

A point to be considered in decision-making for source control implementation, remedy design, and remedy implementation is whether areas of AOPC 1 located near certain outfalls may be subject to recontamination. A premise of EPA's sediment remediation guidance is that active remediation should generally not be implemented until sources have been controlled to the extent necessary to reduce the risk of recontaminating the remediated area (EPA 2005b). Whether active or passive, the success of any remediation may be affected by source control. This FS analysis is consistent with these principles. The FS accounts for recontamination potential in the technology assignments (Section 8) and in the predicted outcomes (Section 9) using the range of BCM input parameters (Section 5).

Estimates of recovery potential should also include: 1) physical conditions that may preclude recovery; 2) predictive modeling that assumes lateral sources will be controlled, at least to some extent, in the future; 3) empirical trends demonstrating that recovery is underway, but that "final" recovery will require additional source control measures and time; and 4) recontamination potential from external sources (see Appendix J). All of these factors have been considered in this FS. However, remedial design-level sampling and further evaluation of source control effectiveness will be necessary in certain areas before any remedial action is initiated. These data and model predictions will be essential in reassessing future recovery or recontamination of surface sediments after source controls are in place.

Table 6-1 Lowest Point Concentrations Used to Delineate AOPCs and Associated SWACs

Risk Driver	Lowest Point Concentrations Used to Delineate AOPC[a]	Preliminary Remediation Goals (PRGs)				Long-term model-predicted concentrations (SWAC)	Estimated SWACs after Active Remediation of AOPC[b]			Are Cleanup Objectives Achieved?			
		RAO 1 (site-wide SWAC)	RAO 2[c] (site-wide netfishing; beach play; clamming SWACs)	RAO 3 (point)	RAO 4 (site-wide SWAC)		Site-wide	Beach Play	Clamming	RAO 1	RAO 2	RAO 3	RAO 4
AOPC 1 (180 acres). Active remediation of AOPC 1 would achieve PRGs for RAOs 2, 3, 4 immediately after construction (with the exception of RAO 2 for arsenic)													
Total PCBs (µg/kg dw)	240 (site-wide)	bg: 2	**1,300; 1,700; 500**	**12 mg/kg oc**	**128-159**	n/a	92	62	69	**T[e]**	✓	✓	✓
Arsenic (mg/kg dw)	57 (site-wide) 28 (intertidal)	n/c	bg: 7	**57**	n/a	n/a	11	9	9	n/a	**T[c]**	✓	n/a
cPAHs (µg TEQ/kg dw)	1,000 (site-wide) 900 (intertidal)	n/c	**380;** 90; **150**	n/a	n/a	n/a	210	150	150	n/a	✓[d]	n/a	n/a
Dioxins/furans (ng TEQ/kg dw)	25 (site-wide) 28 (intertidal)	bg: 2	**37; 28; 13**	n/a	n/a	n/a	6	4	5	**T[e]**	✓	n/a	n/a
SMS contaminants	SQS (site-wide)	n/a	n/a	**SQS**	n/a	n/a	n/a			n/a	n/a	✓	n/a
AOPCs 1& 2 (302 acres). Active remediation AOPCs 1 and 2 would achieve long-term model predicted concentrations (the lowest technically achievable SWACs) immediately after construction.													
Total PCBs (µg/kg dw)	100 (site-wide)	bg: 2	**1,300; 1,700; 500**	**12 mg/kg oc**	**128 to 159**	40-50	46	46	48	✓[e]	✓	✓	✓
Arsenic (mg/kg dw)	15 (site-wide) 28 (intertidal)	n/c	bg: 7	**57**	n/a	9-10	10	9	9	n/a	✓[c]	✓	n/a
cPAHs (µg TEQ/kg dw)	same as AOPC 1	n/c	**380;** 90; **150**	n/a	n/a	100 - 150	140	140	140	n/a	✓	n/a	n/a
Dioxins/furans (ng TEQ/kg dw)	15 (site-wide) 28 (intertidal)	bg: 2	**37; 28; 13**	n/a	n/a	4-6	4	4	4	✓[e]	✓	n/a	n/a
SMS contaminants	same as AOPC 1	n/a	n/a	**SQS**	n/a	n/a	n/a			n/a	n/a	✓	n/a

Table 6-1 Lowest Point Concentrations Used to Delineate AOPCs and Associated SWACs (continued)

Notes:

1. AOPC 1 is also delineated where cores having SQS exceedances in the top 2 ft occur in scour areas. AOPC 2 is also delineated where any core exceeds the SQS at any depth.
2. Site-wide point concentrations used to delineate AOPCs are applied to concentrations in the upper 10 cm of sediment and intertidal point concentrations used to delineate AOPCs are applied to concentrations in the upper 45 cm of sediment.

a. Site-wide point concentrations used to delineate AOPCs are applied in the upper 10 cm of sediment throughout the LDW, and in the upper 60 cm of potential scour areas (i.e., Recovery Category 1 areas; see Section 6.3). Intertidal point concentrations used to delineate AOPCs are applied in the upper 45 cm of sediment in intertidal areas (above -4 ft MLLW).

b. SWACs are estimated by replacing grid cells in AOPCs 1 and 2, respectively, with the following post-remedy bed sediment replacement values: total PCBs = 60 and 20 µg/kg dw; arsenic = 10 and 9 mg/kg dw; cPAHs = 140 and 100 µg TEQ/kg dw; and dioxin/s/furans = 4 ng TEQ/kg dw. AOPC 2 SWACs are based on replacing grid cells in both AOPCs 1 and 2. SWACs are based on the cumulative effect of removing all points/areas above the site-wide and intertidal point concentration shown for each risk driver (the entire AOPC footprint).

c. Because natural background PRG is unlikely to be achieved, this RAO is being evaluated by surface sediment reaching the long-term model-predicted arsenic concentrations. These concentrations are achieved with time after remediation of AOPC 1 and are achieved immediately after remediation of AOPCs 1 and 2.

d. Although the combined beach play area cPAH SWAC is not below 90 µg TEQ/kg dw, this PRG is considered to be achieved because most of the individual beaches achieve this PRG or a 1×10^{-6} excess cancer risk threshold.

e. Because natural background PRGs are unlikely to be achieved for total PCBs and dioxins/furans, RAO 1 is being evaluated by surface sediment reaching the long-term model-predicted concentrations for these two risk drivers. These concentrations are achieved with time after remediation of AOPC 1 and are achieved immediately after remediation of AOPCs 1 and 2.

✓ = Achieves cleanup objective (PRG or long-term model-predicted concentration) immediately following construction.

T = Achieves cleanup objective over time. Institutional controls will be required to further reduce RAO 1 risks regardless of the selected RAL. For RAOs 1 and 2 (arsenic) the goal is to reduce sediment concentrations to as close as practicable to the PRG, estimated in this FS as long-term model-predicted concentrations.

Bold = PRG achieved

AOPC = area of potential concern; bg = background; cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; kg = kilograms; µg = micrograms; mg = milligrams; n/a = not applicable; n/c = not calculated; ng = nanograms; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAL = remedial action level; RAO = remedial action objective; SMS = Sediment Management Standards; SQS = sediment quality standard; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent.

Table 6-2 Array of Remedial Action Levels

Risk-Driver Remedial Action Level[a]	Rationale	Cleanup Objective Achieved[b] (✓ = achieved immediately after construction; T = achieved with time)			
		RAO 1[c]	RAO 2	RAO 3	RAO 4
Total PCBs (µg/kg dw)					
2,200 (site-wide)	• Manage hot spots	**T**	Total PCB direct contact PRGs are achieved following remediation of EAAs and hot spots.[e]	**T** (achieves CSL with time)	**T**
1,300 (site-wide)	• Dry weight equivalent of CSL[d]; achieved immediately after construction	**T**		**T**	**T**
700 (site-wide)	• Provides a well-spaced range of RALs for evaluation	**T**		**T**	✓
240 (site-wide)	• Dry weight equivalent of SQS[d]; achieved immediately after construction	**T**		✓	✓
100 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	✓		✓	✓
Arsenic (mg/kg dw)					
93 (site-wide)	• Achieve CSL immediately after construction / Manage hot spots	n/a	**T**	**T**	n/a
57 (site-wide)	• Achieve SQS immediately after construction and part of a well-spaced range of RALs	n/a	**T**	✓	n/a
28 (intertidal)	• 10^{-5} beach play RBTC (applied as point basis; 45 cm point of compliance) and part of a well-spaced range of RALs	n/a	**T**	✓	n/a
15 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	n/a	✓	✓	n/a
cPAHs (µg TEQ/kg dw)					
5,500 (site-wide)	• Manage hot spots	n/a	**T**	n/a	n/a
3,800 (site-wide)	• 10^{-5} netfishing RBTC (applied as a point basis) and part of a well-spaced range of RALs	n/a	✓	n/a	n/a
1,000 (site-wide)	• Site-wide SWAC within range of upstream values	n/a	✓	n/a	n/a
900 (intertidal)	• Beach play 10^{-5} RBTC (applied as point basis; 45 cm point of compliance)	n/a	✓	n/a	n/a

Table 6-2 Array of Remedial Action Levels (continued)

Risk-Driver Remedial Action Level[a]	Rationale	Cleanup Objective Achieved[b] (✓ = achieved immediately after construction; T = achieved with time) RAO 1[c]	RAO 2	RAO 3	RAO 4
Dioxins/Furans (ng TEQ/kg dw)					
50 (site-wide)	• Manage hot spots	T	T	n/a	n/a
35 (site-wide)	• Provides a well-spaced range of RALs for evaluation	T	✓	n/a	n/a
28 (intertidal)	• 10^{-6} beach play RBTC (applied as point basis; 45 cm point of compliance)	T	✓	n/a	n/a
25 (site-wide)	• Provides a well-spaced range of RALs for evaluation	T	✓	n/a	n/a
15 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	✓	✓	n/a	n/a
SMS Contaminants (apply throughout the LDW)					
CSL at Year 10 (site-wide)	• Achieve CSL within 10 years after completion of construction	n/a	n/a	T	n/a
CSL at Year 0 (site-wide)	• Achieve CSL immediately after completion of construction	n/a	n/a	T	n/a
SQS at Year 10 (site-wide)	• Achieve SQS within 10 years after completion of construction	n/a	n/a	T	n/a
SQS at Year 0 (site-wide)	• Achieve SQS immediately after completion of construction	n/a	n/a	✓	n/a

Notes:

a. A remedial action level is a contaminant-specific sediment concentration that triggers the need for active remediation (i.e., dredging, capping, or ENR with or without *in situ* treatment). It is a point-based concentration that can be targeted to achieve an area-based goal (SWAC). Site-wide remedial action levels are applied to concentrations in the upper 10 cm of sediment throughout the LDW and in the upper 60 cm in Recovery Category 1 areas. Intertidal remedial action levels are applied to concentrations in the upper 45 cm of sediment in intertidal areas (above -4 ft MLLW).

b. See Section 9 for predicted outcomes and RALs by remedial alternative.

c. Risks associated with RAO 1 are reduced through a combination of active remediation, natural recovery, and institutional controls. The goal is to reach the long-term model-predicted concentration, which is as close to natural background as technically practicable (equilibrium).

d. Dry weight equivalents of the SQS and CSL SMS criteria of 12 and 65 mg/kg oc, assuming 2% TOC (average site-wide TOC value). If selected, actual implementation of this RAL would be based on organic carbon-normalized criteria defined by the SMS.

e. An intertidal RAL for PCBs in the upper 45 cm of sediment was not developed because the PRGs for direct contact scenarios are achieved after remediation of the EAAs and other hotspot areas (using the highest RAL sets shown above).

Year 0 = the point in time immediately following completion of construction.

Year 10 = the point in time 10 years after completion of construction.

✓ = Achieves cleanup objective immediately following construction. For RAO 1, institutional controls are also needed.

T = Achieves cleanup objective over time. Institutional controls will be required for RAO 1 regardless of the selected RAL. For RAOs 1 and 2 (arsenic) the goal is to reduce sediment concentrations to achieve the long-term model-predicted concentrations.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; CSL = cleanup screening level; dw = dry weight; kg = kilograms; µg = micrograms; mg = milligrams; n/a = not applicable to the RAO; ng = nanograms; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAL = remedial action level; RAO = remedial action objective; RBTC = risk-based threshold concentration; SMS = sediment management standards; SQS = sediment quality standard; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; TOC = total organic carbon.

Table 6-3 Criteria for Assigning Recovery Categories

Criteria		Recovery Categories: Category 1 Recovery Presumed to be Limited	Recovery Categories: Category 2 Recovery Less Certain	Recovery Categories: Category 3 Predicted to Recover
Physical Criteria				
Physical Conditions	Vessel scour[a]	Observed vessel scour	No observed vessel scour	
	Berthing areas[b]	Berthing areas with vessel scour	Berthing areas without vessel scour	Not in a berthing area
Sediment Transport Model	STM-predicted 100-year high-flow scour (depth in cm)[c]	> 10 cm	< 10 cm	
	STM-derived net sedimentation rate[b] (cm/yr) using average flow conditions	Net scour	Net sedimentation	
	Rules for applying criteria	*Any one criterion in Category 1 results in the area achieving a Category 1 designation.*	*Conditions achieve a mixture of Category 2 and 3 criteria*	*All conditions must achieve the Category 3 criteria.*
Empirical Contaminant Trend Criteria – used on a case-by-case basis to adjust recovery categories from the criteria above				
Empirical Contaminant Trend Criteria[d]	Resampled surface sediment locations	Increasing total PCBs or increasing concentrations of other detected risk drivers exceeding the SQS (> 50% increase)[e]	Equilibrium and mixed (increases and decreases) results (for risk drivers exceeding the SQS)	Decreasing concentrations (> 50% decrease) or mixed[f] results (decreases and equilibrium)[e]
	Sediment cores (top 2 sample intervals in upper 2 ft)			

Notes:

1. Empirical trends were evaluated for two contaminant groups: total PCBs and other risk drivers exceeding the SQS. Dioxins/furans were not evaluated because the small dioxin/furan dataset does not include resampled surface sediment locations and has very few subsurface sediment samples.

a. Observed vessel scour areas are shown on Figure 2-10.
b. Berthing areas are shown on Figure 2-28 and modeled net sedimentation rates are shown on Figure 2-11.
c. High-flow scour areas are shown on Figure 2-9.
d. Empirical trend data are described in Appendix F and summarized in Figure 6-4b. See Table 6-4 for description of empirical trend methodology.
e. ±50% decrease is reasonable considering that analytical variability alone is 25%, and the difference in co-located field replicates ranged from 8% (arsenic) to 48% (cPAHs).
f. A location with mixed results in which risk drivers exceeding the SQS have decreasing trends and concentration changes in equilibrium (but no increasing trends) can be in Recovery Category 3.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; SQS = sediment quality standard; STM = sediment transport model

Table 6-4 Empirical Data Methodology Used in Natural Recovery Trend Evaluation

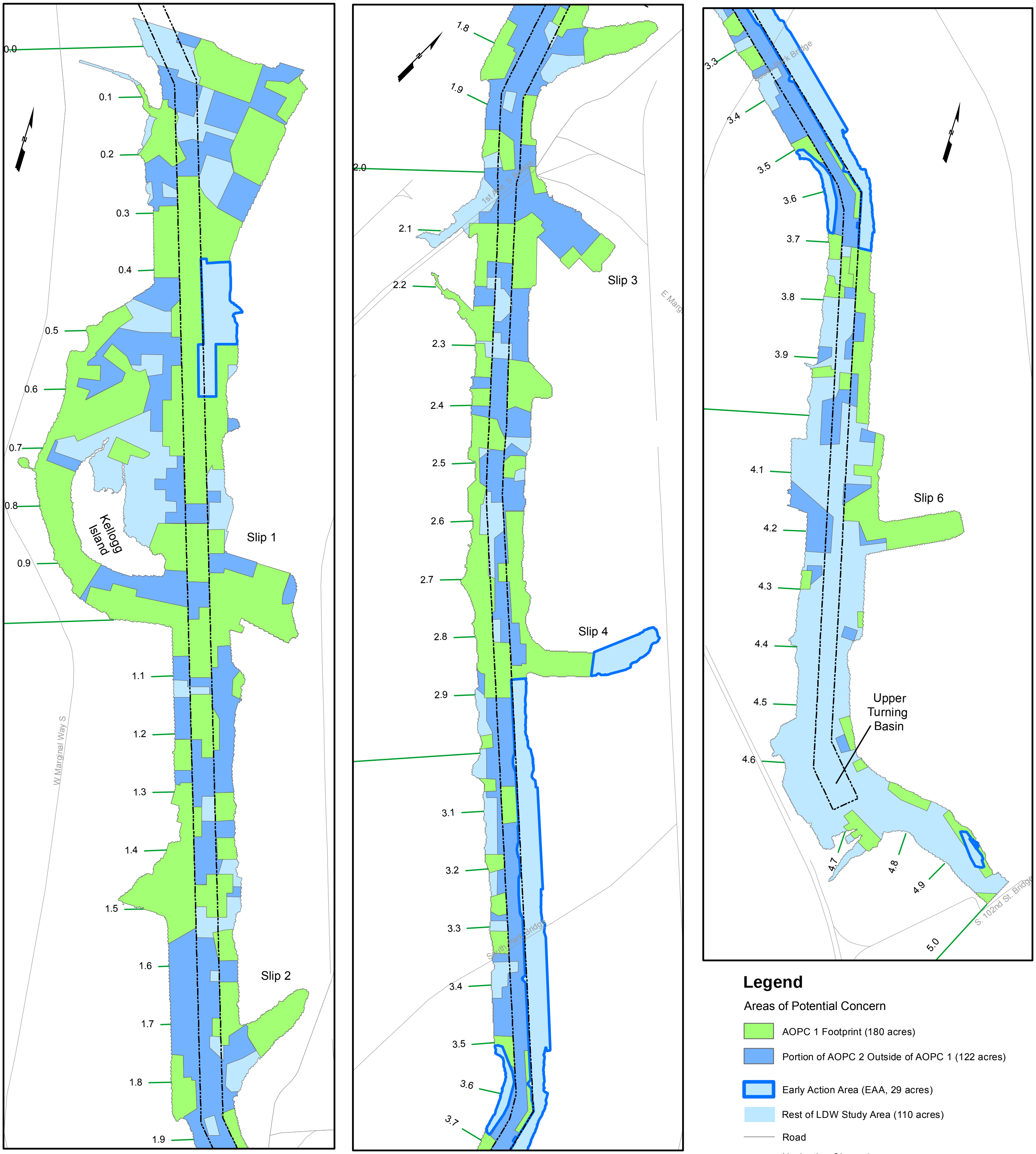
Part 1: Selection of Locations and Data	**Part 2: Trend Criteria Evaluated by Risk Driver**	**Part 3: Natural Recovery Classification by Station** (Figure 6-4b symbol colors; 1 symbol for total PCBs; 1 symbol for other risk drivers)
Resampled surface stations (not more than 10 ft apart)	Increase (> 50% change in concentration)	Increase – all evaluated risk drivers increase (**red**); all red symbols = Category 1
Top 2 sample intervals of cores within top 2 ft of core[a]	Equilibrium (less than 50% change in concentration in either direction)	Equilibrium – all evaluated risk-driver concentrations change by less than 50% (**gray**); Category 2.
Detected risk drivers exceeding the SQS evaluated for concentration changes	Decrease (>50% change in concentration)	Mixed – Risk drivers other than total PCBs have some mixture of any of 3 classifications (**gray**); Category 2 if mixture includes increases; Category 3 if mixture is decreases and equilibrium.
		Decrease – all evaluated risk drivers decrease (**blue**); Category 3.
		Detected total PCBs or other risk drivers do not exceed the SQS[b] (**green**); not specifically used for recovery assignments; area is likely below RALs.

Notes:

1. Two groups of contaminants evaluated: (a) total PCBs detected above the SQS, and (b) risk drivers other than total PCBs detected above the SQS. Figure 6-4b has one symbol for total PCBs and one symbol for other risk drivers.

2. Empirical data evaluation included: 53 to 67 resampled surface sediment locations and 165 cores with appropriate depth intervals (118 samples with an SQS exceedance for total PCBs, 58 samples with an SQS exceedance for other risk drivers). Evaluated the top two intervals of cores if both intervals were within the top 2 ft (can use co-located surface samples).

a. Core trends were also evaluated by comparing the data from the uppermost core interval to that from a co-located surface sediment location, if available.



Legend

Areas of Potential Concern

- AOPC 1 Footprint (180 acres)
- Portion of AOPC 2 Outside of AOPC 1 (122 acres)
- Early Action Area (EAA, 29 acres)
- Rest of LDW Study Area (110 acres)
- Road
- Navigation Channel
- River Mile Marker

Notes:
1. AOPC 1 was delineated using beaches with total excess cancer risk (all risk-driver contaminants combined) greater than 10^{-5}, subsurface SQS exceedances in scour areas, and surface sediments exceeding the following RALs: SQS for any contaminant, total PCBs 240 µg/kg dw, arsenic 28 mg/kg dw in intertidal areas, cPAHs 1,000 µg TEQ/kg dw sitewide and 900 in intertidal areas, and dioxins/furans 25 ng TEQ/kg dw.
2. AOPC 2 was delineated using subsurface SQS exceedances at any depth and surface sediments exceeding the following RALs: total PCBs 100 µg/kg dw, arsenic 15 mg/kg dw, and dioxins/furans 15 ng TEQ/kg dw.
3. Verification monitoring areas (23 acres) are included in AOPC 1, but are presumed to be below the RAO 3 PRGs at the time of construction.

Feet
0 200 400 800

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.38	The AOPC Footprints
DATE: 10/31/12 DWRN:mvi/sea Revision: 0	FIGURE 6-1

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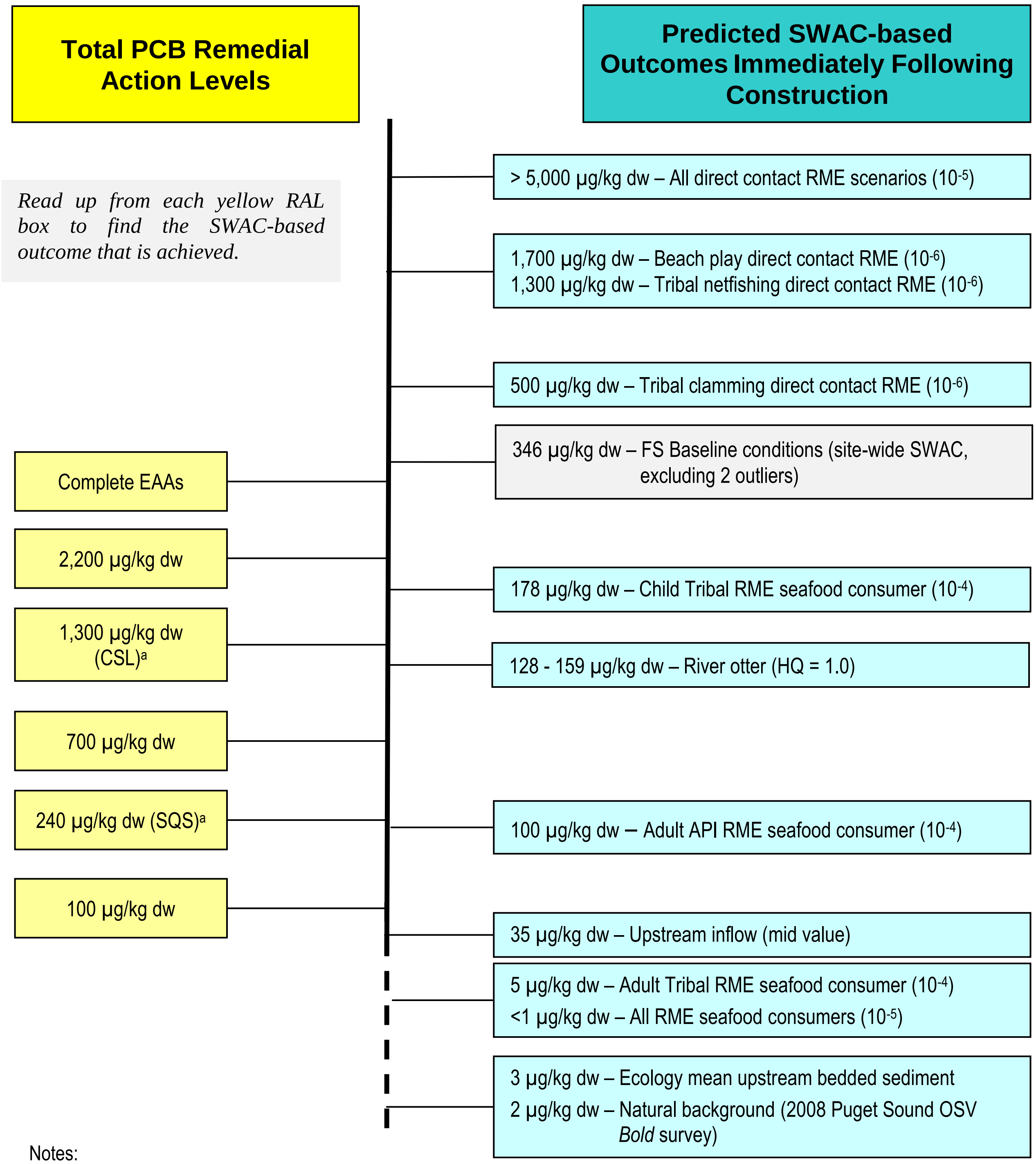
Figure 6-2a Total PCB Remedial Action Levels for Human and Ecological Health



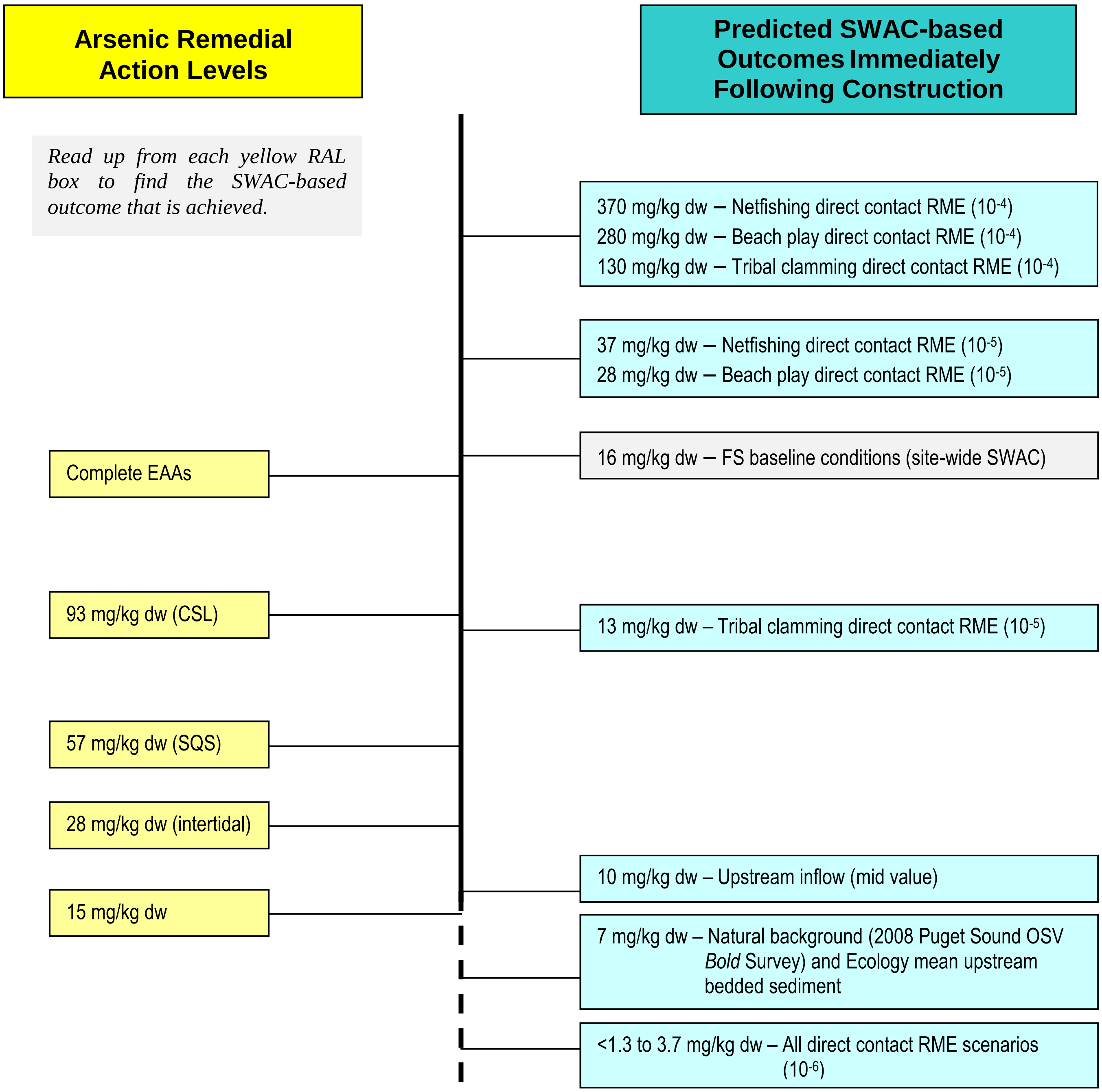
Notes:

a Dry weight equivalents of the SQS and CSL SMS criteria of 12 and 65 mg/kg oc, assuming 2% total organic carbon (average LDW-wide TOC value).

10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CSL = cleanup screening level; EAA = Early Action Area; HQ = hazard quotient; PCB = poly-chlorinated biphenyl; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; SQS = sediment quality standard; TOC = total organic carbon; UCL = upper confidence limit.

— — — — = not achievable

Figure 6-2b Arsenic Remedial Action Levels for Human and Ecological Health

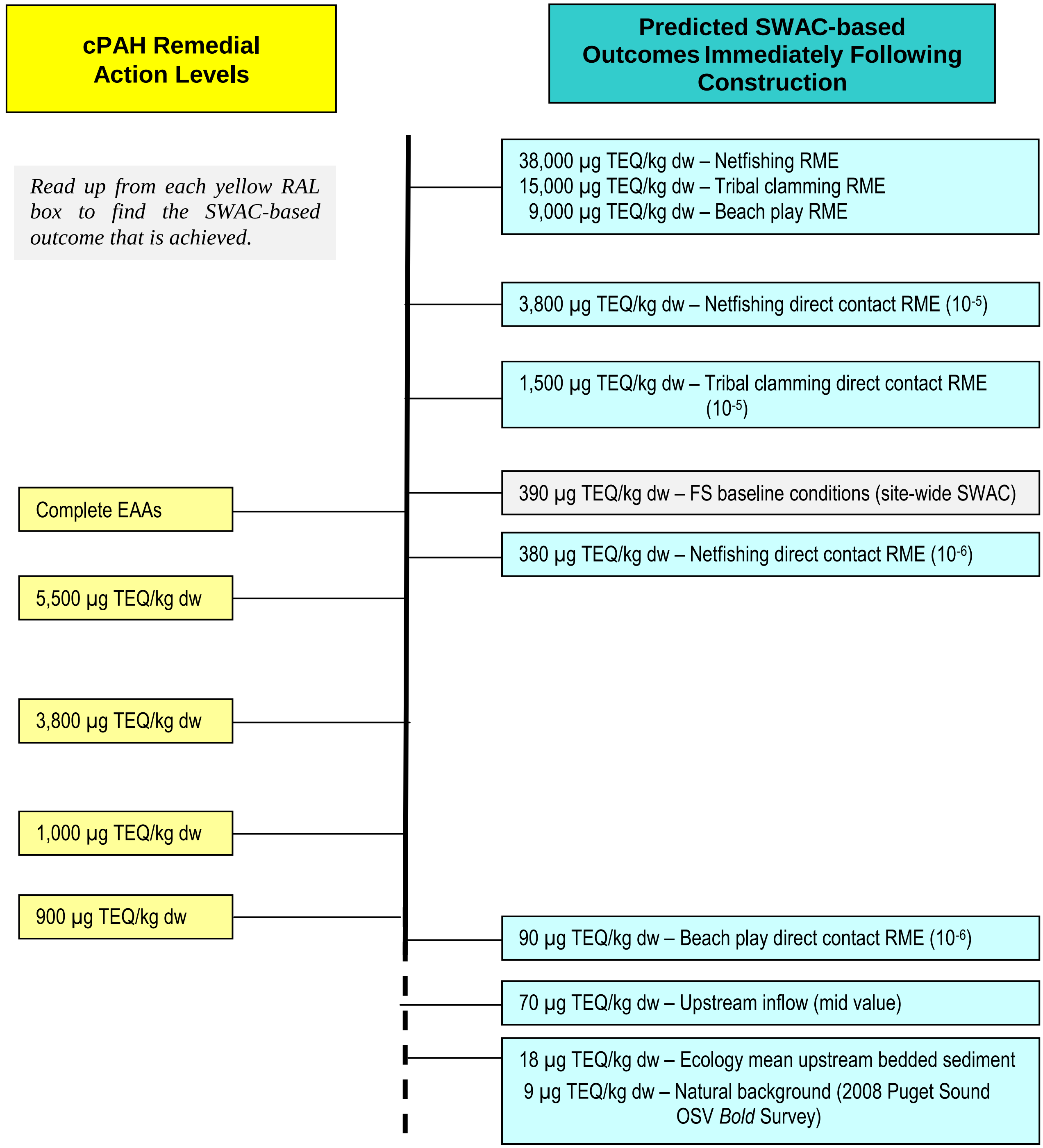


Notes:

10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CSL = cleanup screening level; EAA = Early Action Area; HQ = hazard quotient; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; SQS = sediment quality standard; UCL = upper confidence limit.

– – – – = not achievable

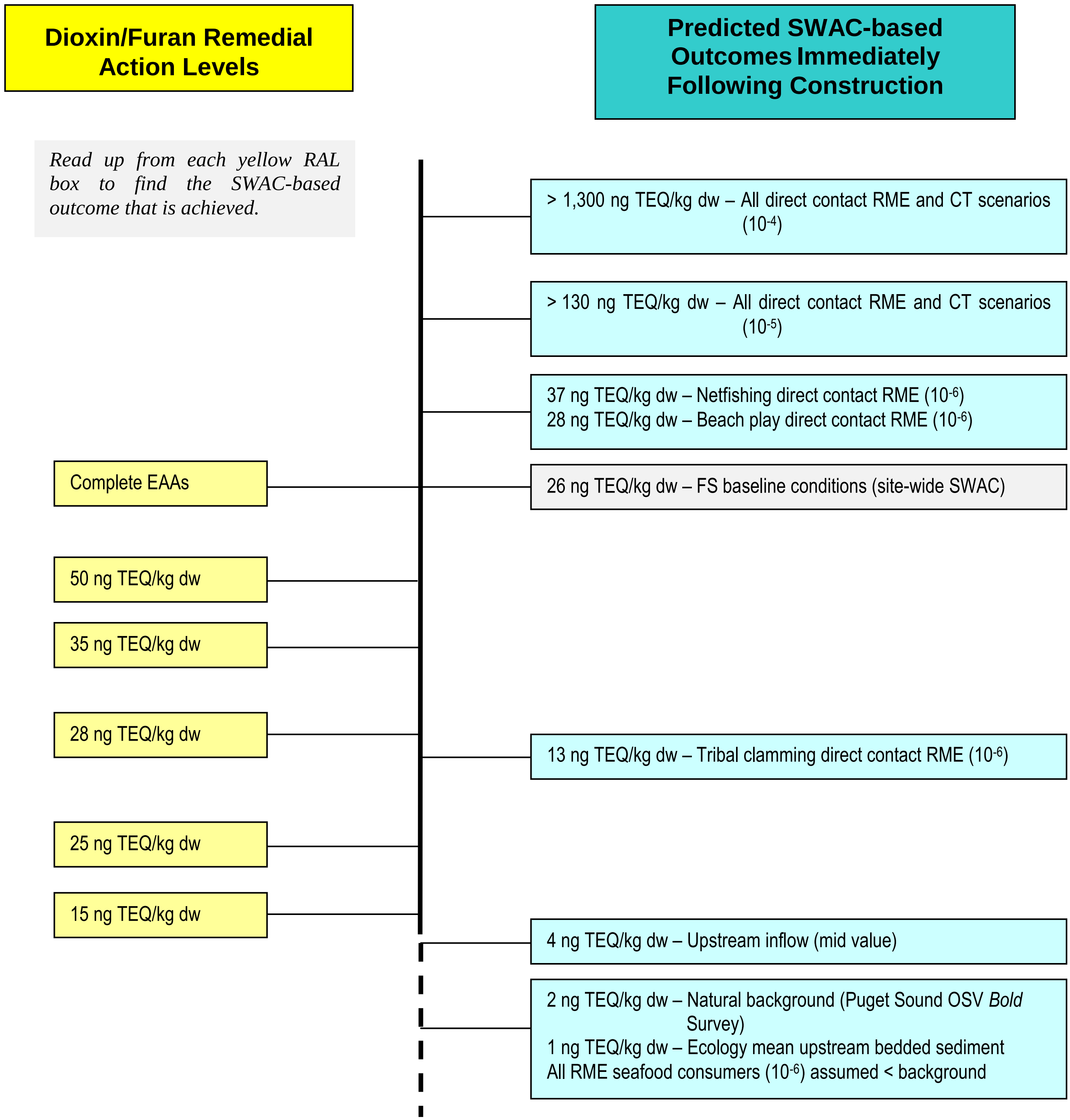
Figure 6-2c cPAH Remedial Action Levels for Human Health



Notes:
10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; cPAH = carcinogenic polycyclic aromatic hydrocarbon; EAA = Early Action Area; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; UCL = upper confidence limit.

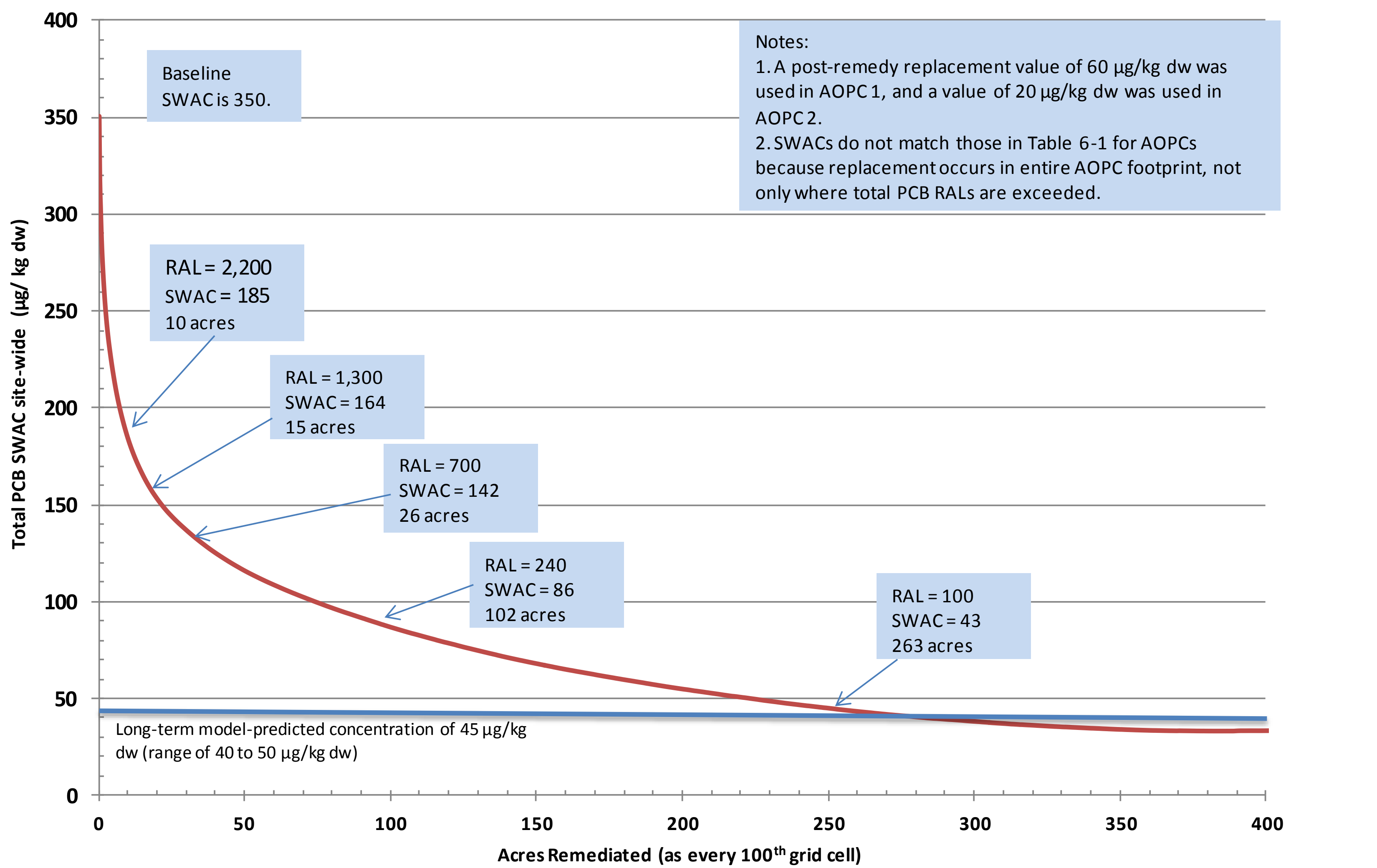
– – – – = not achievable

Figure 6-2d Dioxin/Furan Remedial Action Levels for Human Health



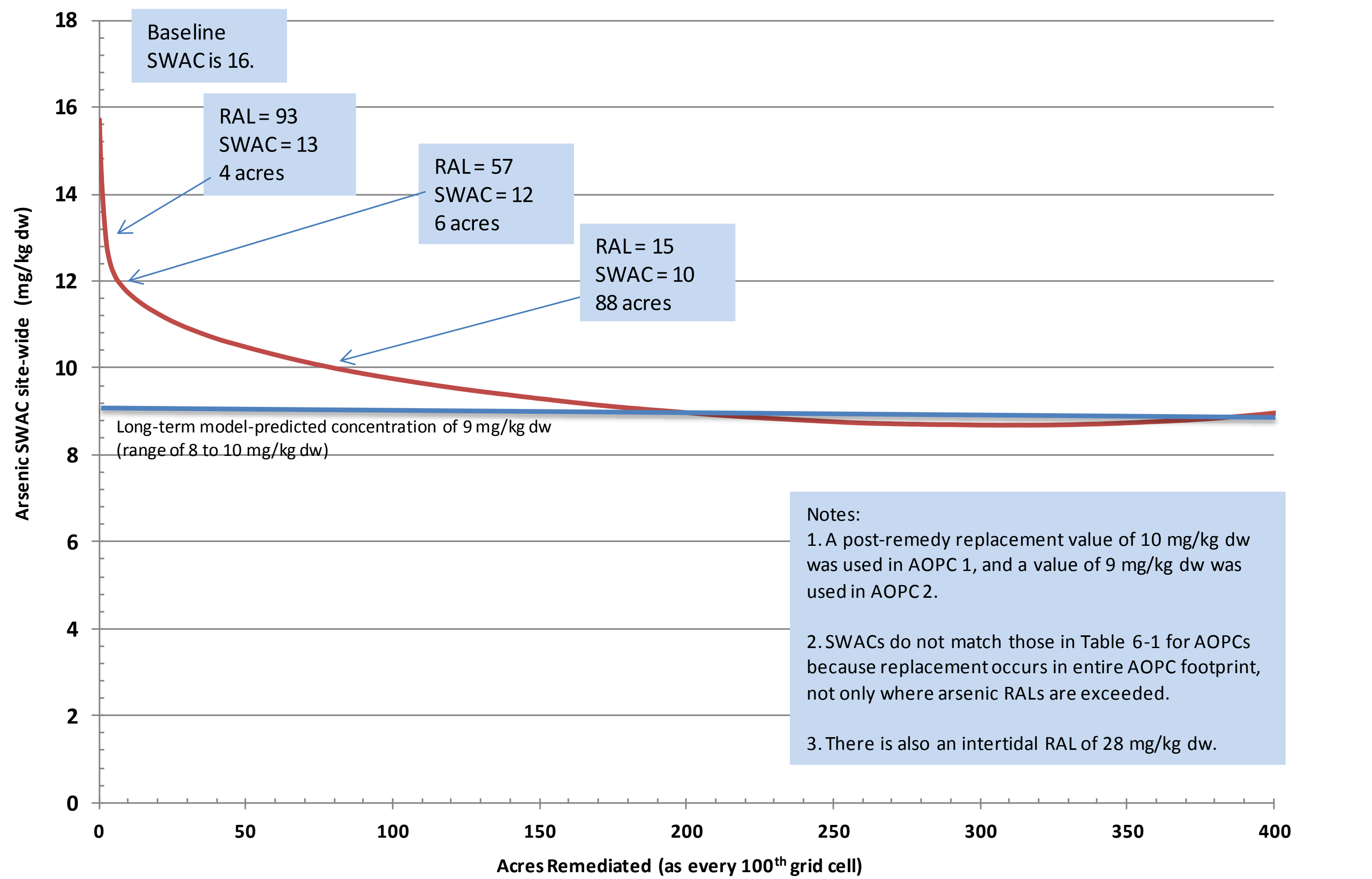
Notes:
10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CT = central tendency; EAA = Early Action Area; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent.
— — — — = not achievable

Figure 6-3a Site-wide SWACs vs. Remediated Acres – Total PCBs



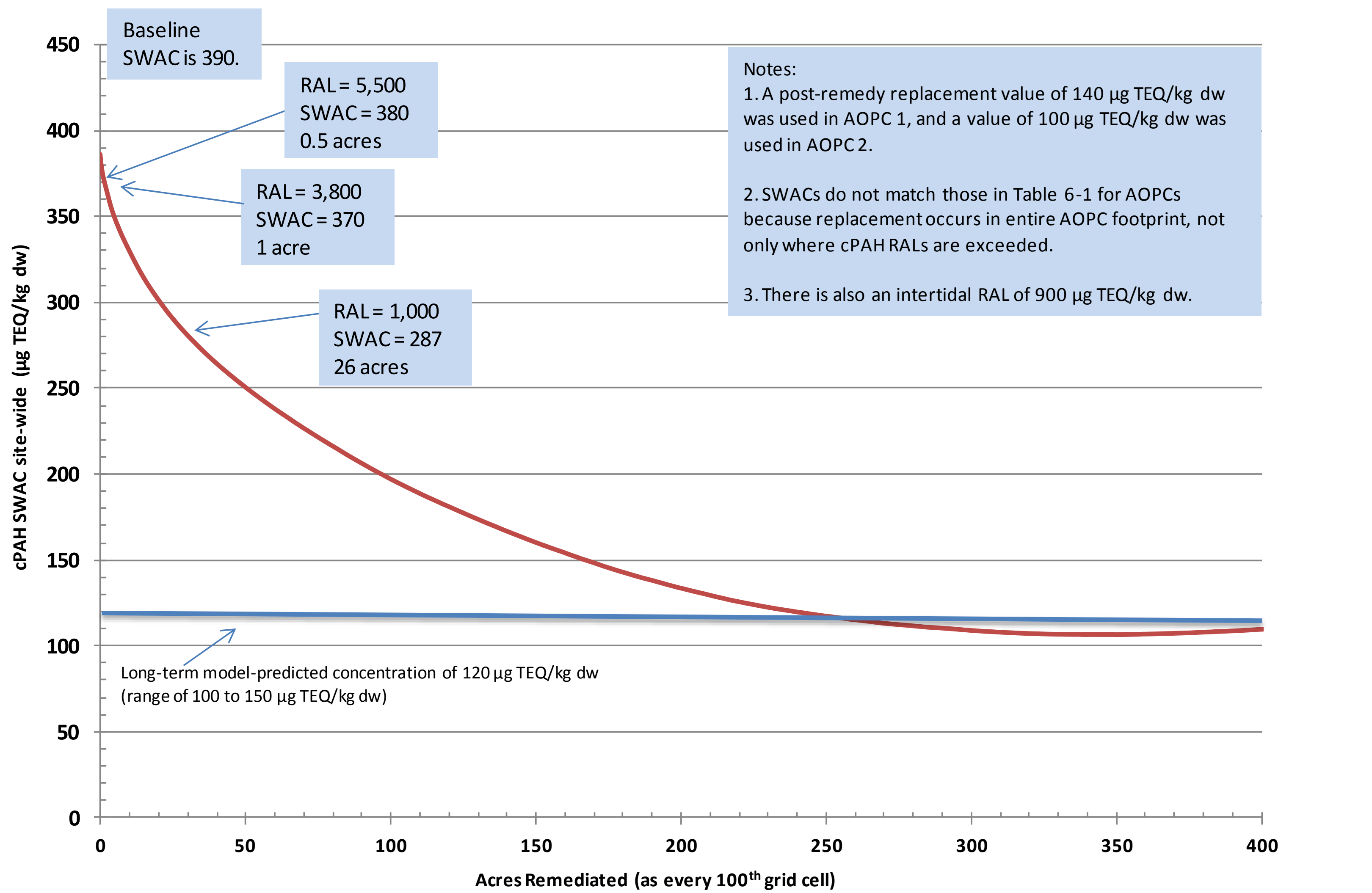
All units are µg/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3b Site-wide SWACs vs. Remediated Acres – Arsenic



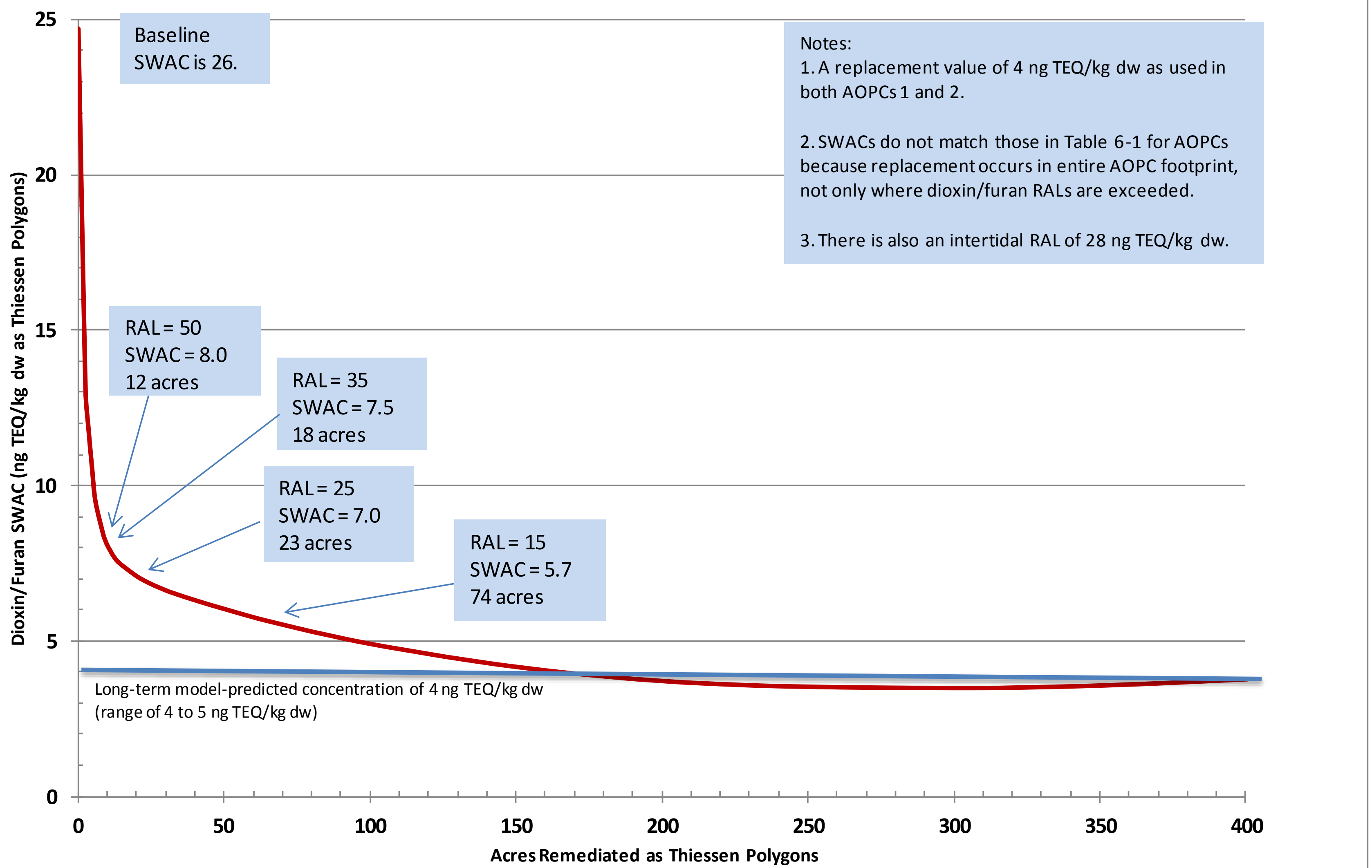
All units are mg/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3c Site-wide SWACs vs. Remediated Acres – cPAHs



All units are µg TEQ/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3d Site-wide SWACs vs. Remediated Acres – Dioxins/Furans



All units are ng TEQ/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Notes:
1. See Table 6-3 for recovery category criteria.
2. The entire FS study area downstream of RM 4.75, except the EAAs, is grouped into recovery categories (402 acres).
3. Surface sediment concentrations are evaluated separately (during technology assignments).
An area may be remediated because of elevated COCs regardless of the recovery category.

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.38	Recovery Categories
DATE: 10/31/12 DWRN:de/SEA Revision: 0	FIGURE 6-4a

Legend

Recovery Category

- Category 1: Recovery Presumed to be Limited (77 acres)
- Category 2: Recovery Less Certain (44 acres)
- Category 3: Predicted to Recover (281 acres)
- Early Action Area (29 acres)
- Outfall Location
- River Mile Marker
- Navigation Channel

Natural Recovery Empirical Data

- Resampled Surface Sediment Location Total PCBs
- Resampled Surface Sediment Location Other Risk Drivers
- Top Two Intervals in Cores Total PCBs
- Top Two Intervals in Cores Other Risk Drivers
- Below SQS
- Data that do not Support Recovery (≥50% Concentration Increase)
- Data that Support Recovery (≥50% Concentration Decrease)
- Station in Equilibrium or Mixed Result

Feet
0 200 400 800

Notes:
1. Resampled surface sample locations include data trumped from both the RI and FS datasets.
2. See Table 6-3 for recovery category criteria and Table 6-4 for the methodology used to evaluate the empirical contaminant trends.
3. Only detected risk drivers exceeding the SQS were evaluated for concentration trends.
4. Mixed results can occur at a location when the risk drivers other than total PCBs do not all have the same trend (mixture of increase, decrease, and/or equilibrium).

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.38			Recovery Categories and Empirical Contaminant Trends
DATE: 10/31/12	DWRN:de/SEA	Revision: 1	FIGURE 6-4b

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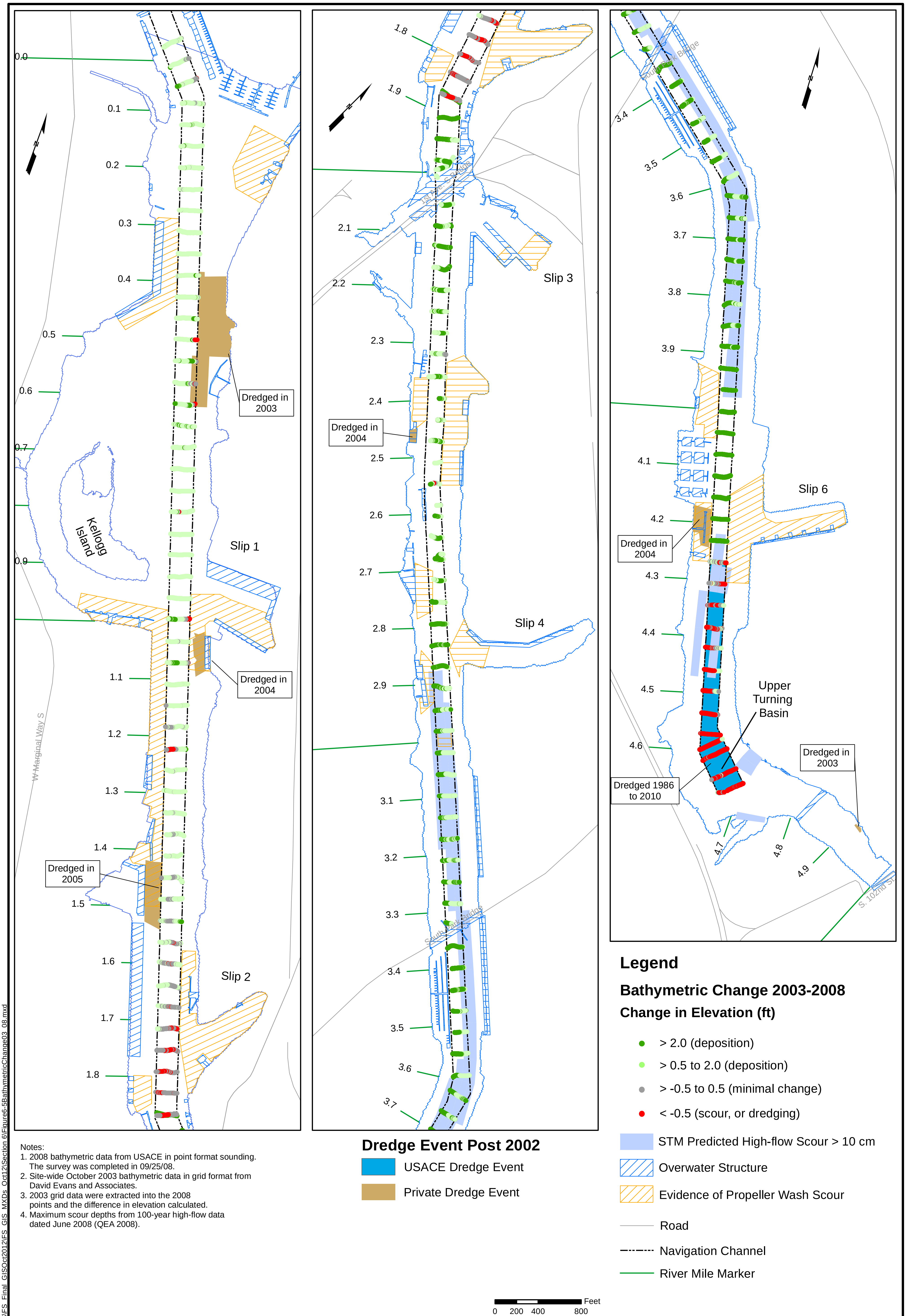

Notes:
1. 2008 bathymetric data from USACE in point format sounding. The survey was completed in 09/25/08.
2. Site-wide October 2003 bathymetric data in grid format from David Evans and Associates.
3. 2003 grid data were extracted into the 2008 points and the difference in elevation calculated.
4. Maximum scour depths from 100-year high-flow data dated June 2008 (QEA 2008).

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60150279-14.38

DATE: 10/31/12 | DWRN:de/Sea | Revision: 1

Bathymetric Change in Navigation Channel between 2003 and 2008

FIGURE 6-5

7 Identification and Screening of Remedial Technologies

This section identifies and screens remedial technologies consistent with the U.S. Environmental Protection Agency's (EPA) *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988). This step toward development of the remedial alternatives parallels and is consistent with Washington State's remedial investigation and feasibility study requirements, Washington Administrative Code (WAC) 173-340-350.

The technology screening for the Lower Duwamish Waterway (LDW) was originally completed and issued as the *Candidate Technologies Memorandum* (CTM; RETEC 2005). The CTM identified and screened a comprehensive set of general response actions, technology types, and process options that are potentially applicable to cleanup of contaminated sediments in the LDW. These three categories or tiers provide a systematic structure and method to identify and evaluate various physical, chemical, and administrative "tools" available for implementing remedial actions. General response actions describe in very broad terms the types of actions potentially applicable to cleanup of contaminated media. Each general response action may contain one or more technology type. For example, one general response action is physical removal of contaminated materials from the site, and two common technologies that can accomplish sediment removal are dredging and excavation. Process options are a further subdivision or tier in the technology screening procedure, and define the specific type of equipment used within a technology. For example, dredging may use a clamshell dredge, hydraulic dredge, or upland-based excavation equipment, such as backhoes.

The CTM evaluated remedial technologies and process options that could be carried forward for additional consideration in the FS. The screening evaluation was conducted using the effectiveness, implementability, and cost criteria consistent with EPA guidance (EPA 1988). Effectiveness refers to whether or not a technology can contain, reduce, or eliminate contaminants of concern (COCs). Implementability refers to whether a technology can be operated under the physical and chemical conditions of the LDW, is commercially available, and has been used on sites similar in scale and scope to the LDW. The CTM contains complete descriptions of remedial technologies and process options and the supporting literature considered for alternative development in the FS.

In this section, technology recommendations from the CTM (RETEC 2005) are reviewed and updated to account for any recent technology developments or relevant experience at other cleanup sites. The Superfund Innovative Technology Evaluation (SITE) Program, the EPA Hazardous Waste Clean-up Information (CLU-IN) website, and the Federal Remediation Technologies Roundtable (FRTR) were reviewed for recent and relevant information about innovative treatment technologies, including their cost and performance, results of technology development and demonstration, and technology optimization and evaluation. The complete screening process is summarized in tables as follows:

- Table 7-1 lists all of the candidate remedial technologies and process options that were evaluated in the FS process, along with an initial screening for potential applicability. Remedial technologies retained as initially feasible are shaded.
- Tables 7-2a through 7-2e provide the detailed screening of process options shown as "retained as initially feasible" in Table 7-1, which were presented previously in the CTM and were updated to account for any recent technology developments. These tables were also updated to include new technologies reviewed for the FS (e.g., spray cap).
- Table 7-3 summarizes the assessment of the effectiveness, implementability, and relative costs of the retained remedial technologies and process options.
- Table 7-4 provides the technologies and process options carried forward into alternative development as representative technologies and process options.

Finally, this section selects representative, effective, and implementable process options to carry forward for developing remedial alternatives. The selections consider information on past and current sediment remediation projects in the Puget Sound region, elsewhere in EPA Region 10, and nationally where appropriate. Selecting representative process options for the FS is consistent with the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (EPA 1988) and Model Toxics Control Act (MTCA) (Ecology 2001) guidance. Reducing the number of process options does not preclude reexamination of other options during the remedial design/remedial action (RD/RA) phase of the cleanup project. Rather, it is a means to streamline the development and evaluation of the remedial alternatives (as described in Section 8) without sacrificing engineering flexibility. Representative technologies and process options used in the development of alternatives are shaded in Table 7-4.

Section 8 of this FS provides detailed descriptions of the technology types and process options that are assumed for cost estimating purposes under each remedial alternative.

7.1 Review and Selection of Representative Process Options

7.1.1 Dredging and Excavation

Removal is a common and frequently implemented general response action for sediment remediation nationwide and in the Puget Sound region. Mechanical dredging, mechanical excavator dredging, hydraulic dredging, and excavation using upland-based equipment (dry excavation) are the four representative process options available for removing contaminated sediments.

7.1.1.1 Removal Process Options

Mechanical Dredging

A mechanical dredge typically consists of a suspended or manipulated bucket that bites the sediment and raises it to the surface via a cable, boom, or ladder. The sediment is deposited on a haul barge or other vessel for transport to a disposal site. Under suitable conditions, mechanical dredges are capable of removing sediment at near *in situ* densities, with almost no additional water entrainment in the dredged mass and little free water in the filled bucket. Low water content is important if dewatering is required for sediment treatment or upland disposal.

Clamshell buckets (open, closed, hydraulic-actuated), backhoe buckets, dragline buckets, dipper (scoop) buckets, and bucket ladders are all examples of mechanical dredges. Clamshell dredges work best in water depths less than 100 feet (ft) to maintain production efficiency. Nominal bucket capacities (i.e., when full) for environmental applications typically range from less than 1 cubic yard (cy) to 10 cy. Clamshell buckets are most effective in consolidated sediments and are the devices of choice for sediments containing debris.

Environmental buckets, or specialty level-cut buckets, offer the advantages of a large footprint, a level cut, and the capability to remove even layers of sediment. A level-cut bucket reduces the occurrence of ridges and winnows that are typically associated with conventional clamshell buckets. Environmental buckets are effective in unconsolidated sediments. They are not effective when digging in heavier sand or where a significant amount of debris may be present.

Mechanical dredging results in sediment excavation with near *in situ* density (water content), thereby reducing the need for substantial ancillary facilities and equipment to process wet dredged material. Mechanical dredging tends to minimize water entrainment by maintaining much of the *in situ* sediment structure (water entrainment ratio of approximately two parts water to one part dredged sediment). Material tends to be dewatered on the barge and then can be transloaded, transported, and managed at permitted off-site facilities that are authorized to handle wet sediments (these facilities are available to projects in this region). As a result, upland sediment processing and water treatment facilities require less acreage to handle mechanically dredged sediments.

Hydraulic Dredging

Hydraulic dredges remove and transport dredged material as a pumped sediment-water slurry. Large debris is typically removed by clamshell buckets prior to hydraulic dredging of sediments. Then, sediment is dislodged into the water column by mechanical agitation, cutterheads, augers, or high-pressure water or air jets. In very soft sediment, it may be possible to remove surface sediment by straight suction or by forcing the intake into the sediment without first mechanically dislodging the sediment. The majority of the loosened slurry is then captured by suction from pumps into an intake pipe and transported through a dredge discharge pipeline to a handling/dewatering facility.

Hydraulic dredging requires substantial ancillary facility acreage (e.g., approximately 26 acres were utilized for Fox River Operable Unit 1 remediation) and equipment to process dredged sediments (dewatering) and to treat the wastewater before discharge. Hydraulic dredging entrains tremendous volumes of water, typically at 8 to 10 parts water to 1 part dredged sediment. As a result, the upland area requirements to support sediment and water handling for hydraulic dredging are significantly greater than for mechanical dredging to handle the same volume of dredged sediment. In addition, the facilities handling the slurry need to be placed as close as possible to the dredging operations to enable pumping from the site to occur effectively.

Land available to site sediment processing equipment adjacent to the LDW is limited and consists mostly of small parcels (i.e., less than 5 to 10 acres). Areas large enough to site a facility capable of dewatering hydraulically dredged sediment with meaningful dredging production rates are not available. Hydraulic dredging may be viable for location-specific circumstances where the total volume of water generated is relatively small and controllable.

A prime example is using a diver-operated, hand-held, hydraulic dredge to remove materials under or around piers, pilings, or in other under-structure places where conventional dredging equipment is unable to reach. Using this technology, an otherwise unreachable location may be feasible to dredge, depending on circumstances. However, one must consider the diver's limited visibility, the overall safety of the diver potentially exposed to physical hazards and resuspended contaminants, and the reduced production rate compared to overall project volume requiring removal. As with other hydraulic dredges, the presence of debris limits the effectiveness of a diver-operated hydraulic dredge. Because under-pier areas typically include riprap and debris, incomplete removal of contaminated sediments can be expected even with a diver-operated hydraulic dredge, and thus capping would likely still be required following dredging.

Dry Excavation

Dry excavation using barge-mounted or upland-based precision excavators refers to the removal of sediments in the absence or limited presence (e.g., a few feet) of overlying water. This involves removing intertidal sediment under naturally-occurring low-tide (exposed) or shallow-water conditions. The fixed-arm, articulated arrangement of the precision excavators pushes the bucket into the sediment to the desired cut level without relying on the weight of the bucket for penetration. Engineered dewatering of an excavation area can also be undertaken to enable dry excavation. Dewatering methods include the use of earthen dams or sheet piling, often in combination with dewatering pump operations.

Upland-based removal of sediment using precision excavators can be employed on exposed shoreline and intertidal areas during low-tide conditions where access is feasible. To avoid the need for extensive upland dewatering treatment facilities, this FS assumes that upland-based excavation is limited to elevations above –2 ft mean lower low water (MLLW) during low-tide conditions, and where access is practicable.

7.1.1.2 Dredge Residuals

All in-water removal operations result in the release of a portion of the contaminants in the material being dredged and will leave behind some level of residual contamination in the sediment after dredging is complete (USACE 2008a). Resuspension of sediments occurs when a dredge and associated operations dislodge bedded sediment particles and disperse them into the water column. These resuspended sediments either settle back near the point of dredging (known as "residual" contamination), or are transported by currents farther afield (known as "release"). Releases also occur as a result of dissolution of contaminants into the water column and, in some cases, through volatilization. Resuspension during dredging is affected by factors such as the type and size of dredging equipment, level of operator skill, positioning of equipment used during dredging, dredge sequencing, depth of dredge cut, type and volume of debris encountered, and the substrate type and bottom topography. Resuspension, residuals, and releases can be estimated and monitored.

Resuspension, releases, and residual contamination can result from various causes that can be grouped into two categories:

- Undisturbed residuals are contaminated sediments found at the post-dredging surface that were not fully removed. The causes of undisturbed residuals include:
  - Incomplete characterization of depth-of-contamination in the remedial design, resulting in previously undocumented contaminated sediment being left in place.
  - Inaccuracies in meeting target dredge design elevation, resulting in contaminated sediment being left in place.
  - Furrows or ridges created by incomplete horizontal removal also leaving contaminated sediment in place.
- Generated residuals are contaminated post-dredging surface sediments that are dislodged or suspended by the dredging operation and subsequently redeposited on the bottom of the water body. Causes include:
  - Material resuspended by the bucket (mechanical dredging) during its bite or by the dredge cutterheads (hydraulic dredging) during its pass.
  - Material resuspended outward by the auger or cutterhead beyond the influence of the pump suction and left behind.

- Vertical positioning of the auger or cutterhead at too great of a cut depth, resulting in material riding over the dredge head.
- Material adhering to the outside of the bucket and washed off on its upward travel through the water column, then settling back down to the bottom.
- Material dripping from a partially closed or overfilled bucket on its upward travel through the water column, then settling back down to the bottom.
- Turbid flow or sloughing of material from steep cut banks spreading sediment from adjacent areas on top of areas where dredging was completed.
- Release of sediment contaminants dissolved in porewater when sediment is disturbed during dredging.

The nature and extent of dredging residuals dislodged or suspended by a dredging operation are not easily predicted. Most projects have based their post-dredging residual concentration by monitoring a specified surficial sediment thickness (e.g., 0 to 10 centimeters [cm] below mudline). By comparing the monitored thickness to the average concentration in the final production cut profile, it is possible to estimate the amount of residuals that will be generated by the project (USACE 2008a). Palermo and Patmont (2007) performed mass balance calculations for 11 project sites, estimating that generated residuals represented approximately 2 to 9% of the mass of contaminant dredged during the last production cut. The available data suggest that multiple sources contribute to generated residuals, including resuspension, sloughing, fall back, and other factors. However, on a mass basis, sediment resuspension from the dredge operations appears to explain only a portion of the observed generated residuals, suggesting that other sources such as cut slope failure and sloughing could be quantitatively more important.

The study also indicated that the presence of hardpan/bedrock, debris, and relatively low dry density sediment results in higher generated residuals.

Numerous case studies have shown that the spatial extent of dredge residuals can extend beyond the footprint of the dredge prism. For this reason, residuals monitoring and management provisions will be included in the remedial design phase that address adjacent areas as well as the dredge prism.

Dredge monitoring studies conducted over the last 13 years have estimated the rate of resuspension at 2 to 5% of polychlorinated biphenyls (PCBs) by mass downstream (or as residuals) compared to the mass of material contained in a dredge prism. Most of the release is in the bioavailable dissolved form (USACE 2008a; TetraTech 2010a, Fox River; Connolly 2010, Hudson River; Steuer 2000; Anchor QEA and ARCADIS 2010). Some loss of material is expected at all dredging sites regardless of the specific dredging process options, engineering controls (e.g., silt curtains, barriers), and best management practices used during dredging. Estimates of sediment export downstream of the LDW from resuspension during dredging are presented in Section 9.1.2.3 and Appendix M, Part 2.

7.1.1.3 Recent Developments in Dredge Positioning Technology

Recent introduction and widespread use of real-time kinematic differential global positioning systems (RTK–DGPS), coupled with radio telemetry and data logging technology, have greatly improved the accuracy and operational flexibility of mechanical dredging. The latest generation of precision dredge and bucket guidance systems integrate RTK-DGPS, excavator and bucket inclinometer sensors, vessel motion sensors, electronic heading, and tide data to enable dredging accuracy generally to within less than 6 inches. Dredge operators are now able to visualize the location of the bucket cutting edge in relation to the target elevation, the bucket open/close status, and the horizontal position of the bucket through use of these advanced positioning and monitoring systems.

7.1.1.4 Dredging and Excavation Technology Summary

Mechanical dredging and excavation, the most commonly practiced forms of sediment removal in the Puget Sound region, are adopted in this FS as the representative primary removal process options for in-water work. Dry excavation using conventional earth-moving equipment is also retained for use in intertidal and embankment areas, but it is expected to be implementable only for a low percentage of the removal volume because of access limitations.[1] Representative dredging projects in the Puget Sound region are identified in Table 7-5. As shown in Table 7-5, approximately 90% of the projects completed in the Puget Sound region adopted mechanical dredging during implementation.

Mechanical dredging and excavation were selected as the primary in-water removal technologies because several factors within the LDW favor these over hydraulic dredging:

- The LDW is a working industrial waterway and significant amounts of debris may be present in the sediments, the result of approximately 100 years of commercial and industrial activity. The presence of debris is a significant problem for hydraulic dredging. Although mechanical dredging is also adversely affected by debris, it is better suited to manage and accommodate debris removal.

[1] Details regarding the range and type of dredge equipment available within the local/regional construction community are presented in Section 8 and Appendix I. Cost estimates prepared and presented in Appendix I are based on mechanical dredging, and barge-mounted excavators.

- Two Subtitle D landfills in the region are permitted to accept wet sediment generated from mechanical dredging (see Section 7.1.3), thereby avoiding the need to dewater mechanically dredged solids.
- The environmental dredging literature contains no documented quantitative evaluations that distinguish between the resuspension and recontamination characteristics of mechanical and hydraulic dredging under other than ideal debris-free site conditions (USACE 2008a).

The assumption of mechanical dredging and excavation for development of remedial alternatives does not preclude other options from being considered during remedial design.

For the FS, partial dredging (diver-operated hydraulic dredging) and capping are assumed as the representative primary process option for under-pier work (see Section 7.1.4) because full removal of contaminated sediment is often difficult in these areas. Under-pier areas have limited access, limited maneuverability, accumulated debris, and riprap structures. This assumption does not preclude other process options from being considered during remedial design. For example, a design decision could be made to remove a pier deck to allow access for mechanical excavation or capping, or to adopt diver-operated hydraulic dredging, or to apply a spray cap.

7.1.2 Treatment Technologies

Treatment technologies can potentially be applied to in-place sediment (*in situ* treatment) or to sediment after it has been physically removed from the aquatic system (*ex situ* treatment). The CTM (RETEC 2005) presented a detailed evaluation of treatment technologies and their applicability for sediment cleanup in the LDW. This section provides updated information about innovative technology developments and relevant experience at other cleanup sites. The CTM also reviewed the extensive regulatory and industry efforts in Washington State and elsewhere to determine the viability of treatment in the context of centralized sediment management facilities. The following discussion reviews viable *in situ* and *ex situ* treatment approaches and their applicability to the LDW.

In situ treatment options with potential applicability to the LDW are physical immobilization by amendment of materials to enhance sorption capacity of the natural sediments. To date, *in situ* treatment of sediments has been mostly by amendment of activated carbon or organoclays in pilot and full-scale sediment remediation projects.

In situ treatment techniques are less energy-intensive, less expensive, and less disruptive to the environment than conventional treatment technologies, and they can reduce ecosystem exposure by binding contaminants to organic or inorganic sediment matrices. The contaminant sorption capacity of natural sediments may be modified and enhanced by adding such amendments as activated carbon for adsorption of non-polar organics and certain metals such as mercury (various activated carbon products are available as powder, granules, or pellets, each with different sediment application characteristics); natural minerals such as apatite, zeolites, or bauxite and refined minerals such as alumina/activated alumina for sequestration of metals/metalloids; ion exchange resins (organoclays) for replacement of metals/ inorganic contaminants with amines or other functional groups; zero-valent iron for dechlorination of PCBs; and lime for pH control or degradation of nitroaromatic compounds. Multifunctional amendment blends may also be used to address complex contaminant mixtures in sediments, and subsequently may enhance overall sorption capacity. Usually activated carbon serves as the backbone (for hydrophobic partitioning) and either is impregnated with the target amendment or blended in a briquette-like composite using an appropriate and non-toxic binder (e.g., clays or other binder materials; Ghosh et al. 2011). Amendments can be engineered to facilitate placement in aquatic environments, by using an aggregate core (e.g., gravel) that acts as a weighting component and resists resuspension, so that the mixture is reliably delivered to the sediment bed, where it breaks down slowly and mixes into sediment by bioturbation.

One of the most advanced *in situ* treatment technologies in terms of its state of development is amending sediment with activated carbon. This treatment has the effect of adsorbing hydrophobic contaminants, reducing porewater contaminant concentrations, and reducing their bioavailability for uptake by benthic organisms. Direct placement of activated carbon to sediments has now been demonstrated in a wide range of bench-scale and pilot studies, and successfully deployed in large field efforts with promising documented monitoring results (Ghosh et al. 2011). Activated carbon has proven effective in reducing the bioavailability of a range of sediment contaminants, including PCBs, polycyclic aromatic hydrocarbons (PAHs), dioxins, DDT, and mercury. However, while the pilot studies are starting to provide valuable information, further research is needed to understand both transient and long-term changes that take place naturally in the environment, and also demonstrate the application of activated carbon at full-scale contaminated sediment areas. Further discussion of this technology is presented in Section 7.1.2.1.

Ex situ treatment options with potential applicability to the LDW are conventional soil washing/particle separation, advanced soil washing (Biogenesis™), solidification, and thermal treatment. To date, *ex situ* treatment of sediments, while a subject of considerable interest nationwide, has been mostly limited to soil washing and air (steam injection) stripping in full-scale sediment remediation projects.

Technologies that destroy or detoxify contaminants have been accepted at very few projects (e.g., Bayou Bonfouca) for cleanup at contaminated sediment sites for two reasons. First, it is difficult to balance treatment costs with a beneficial reuse outlet for the material; and second, upland and in-water disposal alternatives are much less expensive, particularly in this region. With the exception of the addition of cement-type materials to reduce free water content and mobility prior to upland disposal, only one contaminated sediment remediation project in this region (Area 5106 at Hylebos

Waterway in Commencement Bay) has utilized treatment (see Section 7.1.2.2) or incorporated beneficial reuse of treated sediments.[2]

7.1.2.1 Direct Amendment with Activated Carbon or Organoclays

The goal of *in situ* treatment, by amending or thin capping the bioactive surface layer of sediment, is to reduce the bioavailability of hydrophobic organic contaminants. The two most common material classes for amendment are activated carbon and organoclays. The transfer of organic contaminants such as PCBs from the sediment to the strongly binding activated carbon particles not only reduces contaminant concentration and the bioavailability to benthic organisms but also reduces contaminant flux into the water column, and thus accumulation of contaminants in the aquatic food-chain (Ghosh et al. 2011). Of the two amendments, activated carbon has received more testing and evaluation than organoclays, particularly with respect to sediment remediation, because the sorption capacities for PCBs and PAHs in activated carbon are at least an order of magnitude higher than in the other sorbents (Ghosh et al. 2011). Organoclays have received attention largely in the context of addressing localized deposits of dense non-aqueous phase liquids (DNAPLs; Bullock 2007, Reible and Lampert 2008).

Extensive bench-scale studies have confirmed the effectiveness of activated carbon for *in situ* treatment. For example, average doses of 2 to 4% (by dry sediment weight) of activated carbon applied to surface sediments have resulted in reductions greater than 95% in PCB bioavailability and sorption capacities of the activated carbon have been retained for as long as the bench-scale studies were continued (up to 10 years in some studies). Based on promising laboratory results, beginning in 2006, several pilot-scale field demonstrations of activated carbon placement were implemented in the United States and Norway (see Figure 7-1). These projects show how various engineering challenges were met for applying activated carbon and monitoring of its long-term effectiveness:

- Hunter's Point Naval Shipyard (San Francisco, CA), conducted in 2006, in estuarine application to address PCBs and PAHs
- Lower Grasse River (Massena, NY), conducted in 2006, in freshwater application to address PCBs
- Trondheim/Grenlandsfjords Harbors (Norway), conducted in 2006, in estuarine application to address PCBs, PAHs, and dioxins
- Grenlandsfjords Harbors (Norway), conducted in 2009, in estuarine application to address dioxins and furans

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[2] Treatment to eliminate free liquids from dredged sediment is no longer required by two regional landfills servicing the Puget Sound area (see Section 7.1.3.2).

- ♦ Bailey Creek, U.S. Army Installation (VA), conducted in 2009, in freshwater wetland application to address PCBs
- ♦ Canal Creek (Aberdeen Proving Grounds, MD), conducted in 2010, in freshwater application to address mercury, PCBs, and DDT.

The primary objective of these demonstration projects was to verify that the bioavailability of PCBs, PAHs, DDT, dioxins/furans, and/or mercury can be effectively reduced at the field scale by placing activated carbon into surface sediments. While the specific approaches varied for each pilot project listed above, most of the projects focused on the following:

1) Evaluate efficient, low-impact delivery systems of activated carbon for amendments into in-place sediments (using large-scale equipment and a range of application methods).

2) Determine the extent of sediment resuspension and contaminant release during application.

3) Assess persistence, binding potential, and small-scale spatial variations of the activated carbon after application to sediments in the natural environment, and also assess mixing of activated carbon over time as a result of bioturbation processes.

4) Evaluate short- and longer-term changes in contaminant porewater concentrations, sediment-to-water fluxes, desorption kinetics, and/or equilibrium partitioning from sediments that result from activated carbon amendment.

5) Measure short- and long-term changes in contaminant bioavailability by biomonitoring deposit-feeding benthic organisms after applying the activated carbon amendment.

6) Evaluate activated carbon-sediment stability and erosion potential over time.

7) Evaluate contaminant bioavailability for uptake, transfer, or any changes to the benthic and/or submerged aquatic plant communities, as a result of activated carbon amendment.

Several types of activated carbon applications were evaluated at these sites, including slurry amendment (water and/or native clay mixtures) on top of the sediment surface, mixing or injection of slurry amendments into surface sediments, and pelletized applications (e.g., SediMite®, AquaBlok®).

The period over which ENR/*in situ* treatment remains effective will be an important consideration during remedial design. Design life will need to be evaluated at the location-specific level and will likely influence decisions on the type (e.g., source and type of carbon), amount of amendment used (i.e., design safety factor), and the potential need for replenishment. Physical stability and chemical activity (e.g., adsorption capacity) over the long term are the most important design life factors. Activated carbon and other charcoals created under high-temperature conditions are known to persist for thousands of years in soils and sediments, and both laboratory studies and modeling evaluations indicate promising long-term physical stability of the amendment material and chemical permanence of the remedy (Ghosh et al. 2011). Empirically-derived contaminant concentration data and modeling simulations show that *in situ* treatment can reduce bioavailability over the long term where contaminant loading (mass transfer) from groundwater, surface water, and newly deposited sediments is low.

The FS assumes that half of the ENR footprint would warrant amendment with a material such as activated carbon for *in situ* treatment. This assumption provides a basis for estimating costs and comparing the remedial alternatives; however, during remedial design, the emphasis on ENR or *in situ* treatment will depend on location-specific factors and additional testing of the implementability of these technologies. The composition of ENR/*in situ* treatment will depend on additional evaluation during remedial design; it may include carbon amendments, habitat mix, and/or scour mitigation specifications to increase stability and enhance habitat.

The following sections provide synopses of two of the most relevant field demonstrations.

Hunter's Point Naval Shipyard (San Francisco, California) – Carbon Amendment

Beginning in January 2006, a large field demonstration of activated carbon via direct amendment was conducted in a shallow tidal flat of the South Basin adjacent to the former Naval Shipyard at Hunters Point, in San Francisco Bay (CA) (Luthy 2005, Luthy et al. 2009, Cho et al. 2009). The former Navy installation was predominantly used for ship repair and maintenance, which resulted in the release of PCBs to the environment. The activated carbon was applied to two test plots (D and F) with a surface area of 34.4 m^2 each, located within the intertidal region of the former shipyard, and away from the shoreline. Two more plots (C and E) served as control and reference plots. A barge-mounted rotovator system (for plot D) and a crawler-mounted slurry injector system (for plot F) were used to mix activated carbon directly into the surface sediments at a target mixing depth of 30 cm below the mudline, to include the biologically active zone.

Baseline and post-amendment monitoring field assessments were conducted in December 2005, July 2006, July 2007, and January 2008, respectively. These assessments were performed to characterize surficial sediment concentrations, analyze the water column, test uptake, and study bioaccumulation. Prior to treatment, the PCB concentration in sediment among the plots varied between 1,350 and 1,620 micrograms per kilogram dry weight (µg/kg dw). Mixing of activated carbon into surface sediments was assessed using black carbon measurements. The measured activated carbon dose averaged 2.0 to 3.2% by dry sediment weight and exhibited small-scale spatial variability. The uneven activated carbon distribution was possibly induced by the unidirectional mixing motion of the large mechanical mixing devices, the relatively small dimensions of the test plots, and insufficient mixing time. In terms of variability, Plot F showed higher variability than Plot D, indicating that activated carbon-mixing via the slurry injection device on Plot F was less homogeneous than the rotovator device employed at Plot D. Ineffective homogenization of the activated carbon into the sediment would influence the short- and long-term performance of the technology.

No adverse impacts, such as sediment resuspension and PCB release, were observed in the water column over the treatment plots as a result of applying the activated carbon and mechanically mixing it into the sediments. In addition, the activated carbon amendment did not impact the structure of the macro benthic community (composition, richness, or diversity) (Luthy et al. 2009, Cho et al. 2009).

Both *in situ* clam bioassay and *ex situ* bioavailability for uptake studies confirmed that PCB bioaccumulation was reduced; an approximate 78% tissue concentration reduction in bioavailability was achieved when clams were exposed to sediment treated with an average 3.4% activated carbon. Although the *in situ* bioassay results were sometimes influenced by field conditions resulting from newly deposited sediment, heat stress, and shallow burrowing depth, the reduction in bioavailability was consistent with the results of earlier laboratory studies (Millward et al. 2005; McLeod et al. 2007, 2008). Reductions in congener bioaccumulation with activated carbon were inversely related to the congener octanol-water partitioning coefficient (K_{ow}), suggesting that the efficacy of activated carbon is controlled by the mass-transfer rate of PCBs from sediment into activated carbon (Millward et al. 2005). The semi-permeable membrane devices (passive samplers) were used to show that PCB uptake in activated carbon-treated sediment was reduced by 50%, with similar results in porewater. This reduction was evident 13 months post-treatment and even after a subsequent 7 months of continuous exposure, indicating activated carbon treatment efficacy was retained for an extended period (Cho et al. 2009). Although reductions in aqueous PCB concentrations in equilibrium with the sediment following activated carbon-amendment often correlate with reduced PCB bioaccumulation, the reduced availability of contaminants from ingestion of sediments appeared to be the actual cause of lower tissue concentrations (Janssen et al. 2010, 2011).

The two activated carbon-treated plots showed decreases in the fraction of PCBs desorbed with an increasing dose of activated carbon, which supports the finding of reduced PCB availability after activated carbon application. After 18 months, the field-exposed activated carbon demonstrated a strong stabilization capability to reduce aqueous equilibrium PCB concentrations by almost 90%. These results are promising and suggest the long-term effectiveness of activated carbon in the field (Luthy et al.

2009, Cho et al. 2009). Finally, based on the absence of significant differences between the 6-month and 18-month total organic carbon (TOC) values measured in cross sections of sediment cores taken from Plots D and F for sediment stability testing and based on hydrodynamic modeling, it was concluded that mixing activated carbon into cohesive sediment at selected locations within the South Basin at Hunter's Point neither reduced surface sediment stability nor resulted in significant erosion of treated sediments (Zimmerman et al. 2008). Surficial sediment of the two activated carbon-treated plots contained less black carbon/TOC 24 months after treatment, which was explained by continued sediment deposition.

Lower Grasse River (Massena, New York) – Carbon Amendment

Similar pilot field studies were initiated in September 2006 to evaluate the ability to deliver activated carbon slurries to in-place sediments and assess the effectiveness of this approach in reducing the bioavailability of PCBs in sediments and biota in the Lower Grasse River in Massena (NY). Alcoa Inc., with oversight from EPA, implemented the pilot demonstration project, which began with laboratory studies and land-based equipment testing, continued with field-scale testing of alternative placement methods, and culminated in a field demonstration of the most promising activated carbon application and mixing methods in a 0.5-acre pilot area within the Lower Grasse River (Alcoa 2007, EPA 2007b).

Based on the results of initial laboratory studies that evaluated bioavailability reductions achieved at different activated carbon doses, a target application concentration of 2.5% activated carbon (dry-weight basis) in the top 15 cm of sediment after treatment was used in the Lower Grasse River field demonstration. Three application techniques were implemented within the pilot study area as follows:

- A 7-ft by 12-ft enclosed device first applied (sprayed) the activated carbon slurry onto the sediment surface. The material was then mixed into near-surface (0 to 15 cm) sediments using a rototiller type mechanical mixing unit (tiller).
- A 7-ft by 10-ft tine sled device (tine sled) used direct injection of activated carbon into the upper 15 cm of the sediments.
- Application of activated carbon to the sediment surface using the tiller, but with the mixing devices removed. Monitoring of this "unmixed" treatment area allowed for an evaluation of the rate and extent of incorporation of the surficial layer of placed activated carbon into near-surface sediments over time through natural processes (e.g., bioturbation).

Baseline (summer 2006), construction (fall 2006), and post-construction (2007, 2008, and 2009) monitoring were conducted (Alcoa 2010). Water quality action levels for PCBs (0.065 micrograms per liter [µg/L]) were not exceeded adjacent to or downstream of the pilot project area during activated carbon application. Similarly, turbidity levels during construction never approached the action level of 25 nephelometric turbidity units (NTUs) above background. Turbidity measured downstream of the pilot project area was only slightly higher than that measured upstream, with average turbidity and total suspended solids (TSS) increases of roughly 0.2 NTU and 0.8 milligrams (mg)/L, respectively. The water column monitoring data indicated that construction activities did not have a significant impact on water quality in the river, and further suggested that silt curtains are not needed for either the tine sled or tiller equipment.

Sediment cores were collected immediately following the fall 2006 application and in the three post-construction monitoring years (2007, 2008, and 2009) and were analyzed for black carbon to verify the applied dose. The target dose of 2.5% activated carbon (dry weight basis) in the top 15 cm of sediment was achieved in nearly all test plots. Compared with the tine sled, application of activated carbon using the tiller (with or without mixing) resulted in greater small-scale spatial variability in activated carbon levels.

A detailed 3-year post-implementation physical, chemical, and biological monitoring program (i.e., 2007 through 2009) was completed to evaluate the long-term effectiveness of the activated carbon treatment. Monitoring results are summarized below:

- Measurements of activated carbon levels in the treated sediments (i.e., based on black carbon analysis and microscopy results) confirm that the applied carbon has continued to remain in place. Levels are based on mass balance calculations of activated carbon applied in 2006.
- Most of the activated carbon in the treatment areas was applied within the upper 10 cm of the sediment, declining to background levels at approximately 20 cm below the mudline. The 2008 and 2009 monitoring revealed that the activated carbon was slightly deeper in the sediment profile than observed in 2006 post-construction and 2007 sampling, due to natural sedimentation occurring on top of the activated carbon-treated sediments since 2006.
- PCB bioaccumulation in the tissue of test organisms (whole body worms; wet weight basis) in the activated carbon treatment areas was reduced in excess of 80% for the *in situ* tests and in excess of 90% for the *ex situ* tests. Greater than 90% reductions in porewater PCB concentrations were also observed in the test plots. PCB bioavailability was reduced even further over the 3-year post-construction monitoring period due to a combination of improved mixing (bioturbation) of activated carbon in surface sediments, and site-wide natural recovery over time.
- Batch equilibrium testing to evaluate the effect of activated carbon on PCB partitioning between the sediment and water phases showed reductions in the range of 93 to 99%.

- Two trends were observed in the results from *in situ* passive samplers deployed on top of the treatment areas:
  - Ambient PCB sediment levels declined from 2006 to 2009 (as a result of site-wide natural recovery); and
  - Aqueous PCB concentrations at the sediment surface in the treatment areas decreased by 90% (similar to reductions observed from biological testing), and even in 2009, the treated sediments continued to act as a "sink" for water column PCBs in the river (i.e., net flux of PCBs from surface water to sediments).
- Results of ecological monitoring activities show a benthic community adapted to fine-grained sediments both pre- and post-carbon application. Benthic habitat and community composition measures were similar (not statistically different) between the treatment areas and upstream background locations, suggesting that activated carbon application did not affect the benthic community. Additional studies of potential impacts to submerged aquatic vegetation at high activated carbon doses are ongoing.
- Erosion potential testing indicated that treated sediments had a slightly higher erosion potential than pretreatment sediments, but nevertheless were within the range of historic data for native sediments.

In summary, the Lower Grasse River pilot project demonstrated that activated carbon can be successfully applied to river sediments with minimal impact to water quality within the river. Post-construction monitoring revealed that the activated carbon is stable in the fine sediments and has significantly reduced PCB bioavailability. Batch equilibrium experiments showed that aqueous phase PCB concentrations in surface sediments have been reduced on average by more than 95% at activated carbon doses of 2% or greater. *In situ* and *ex situ* biological uptake studies showed 80 to 90% reductions with an activated carbon dose greater than 2%.

7.1.2.2 Soil Washing with Air Stripping

Soil washing can be classified as conventional or advanced form of *ex situ* treatment. Conventional soil washing is a form of primary treatment that uses conventional and readily-available material handling unit processes to separate sediment particles, typically into coarse (sand and gravel) and fines (silt and clay) fractions (Figure 7-2). This treatment process separates the sediment particles using conventional equipment. These equipment systems have been derived largely from the mining and mineral processing industries, and include screening, gravity settling, flotation, and hydraulic classification (e.g., using hydrocyclones) (USACE-DOER 2000). Advanced soil washing, such as Biogenesis™, combines the physical separation aspects of conventional soil washing with additional treatment such as agitation, or the addition of surfactants, chemical oxidants, or chelating agents to the finer fraction of material.

Soil washing is a wet process and therefore generates wastewater that requires treatment and discharge. Depending on site conditions, the washed coarse fraction may be suitable for in-water placement (see Section 7.1.3.4 for beneficial uses of sediment) as a cap, enhanced natural recovery (ENR), or habitat creation/restoration medium. The finer fraction, which has higher concentrations of contaminants, is typically dewatered, transported, and disposed of in a permitted upland landfill. Ideally, the net outcome of soil washing is a reusable coarse fraction and a reduced volume of contaminated material requiring additional treatment or direct disposal.

Sediments in portions of the LDW may be sufficiently coarse-grained to consider soil washing as a potentially viable treatment. One vendor has indicated that soil washing has the potential to be economical where the sediment contains greater than 30% sand (Boskalis-Dolman 2006). When the sediment contains less than 30% sand, treatment performance and economics deteriorate. Other factors affecting the economics and implementability of soil washing are:

- Physical and chemical properties of the sediment.
- Availability of an upland location for transloading sediment from barges.
- Availability of an upland location for sediment containment, storage, and operation of the soil washing facility. Although this facility may or may not be located at the transloading facility, this FS assumes that it will be located within the transloading facility footprint for the purpose of cost estimating.
- Disposal costs for the fines fraction.
- Ability to commit to long-term (and continuous) high-volume sediment throughput (economies of scale).
- Ability to reuse washed coarse fraction beneficially and at low cost.

The last two factors are the most difficult to reconcile in a manner that promotes economic viability.

The following sections describe conventional and advanced soil washing techniques recently used at several sites.

Area 5106, Hylebos Waterway, Commencement Bay (Washington) – Soil Washing

Unlike other parts of the Hylebos Waterway cleanup, the sediments at Area 5106 were treated before confined disposal (EPA 2004). The non-time critical removal action was conducted by Occidental Chemical Corporation at its former chlor-alkali plant facility along the Hylebos Waterway. About 36,000 cy of contaminated sediments containing volatile organic compounds and semivolatile organic compounds were hydraulically dredged and pumped to an upland treatment system. Treatment consisted of aeration and air stripping to separate out the volatile organic compounds (VOCs), which were, in turn, adsorbed onto activated carbon. The treated slurry was dewatered and the dewatered sediments were disposed of in the Blair Slip 1 confined disposal facility, because treated materials still contained relatively high concentrations of semivolatile organic compounds and metals.

Raritan River, Arthur Kill, and Passaic River (New Jersey) – Soil Washing
Biogenesis™ is an advanced soil washing process that was used in a recently completed full-scale demonstration, which treated approximately 15,000 cy of contaminated sediments from the Raritan River, Arthur Kill, and Passaic River, New Jersey (Biogenesis 2009, Malcolm Pirnie 2007). The Biogenesis™ process combines the physical separation aspects of conventional soil washing with high-pressure agitation, surfactants, chemical oxidants (e.g., hydrogen peroxide), and chelating agents. This process uses equipment including but not limited to: truck-mounted washing units, sediment processor, sediment washing unit, hydrocyclones, shaker screens, water treatment equipment, tanks, water blasters, compressors, and earth moving equipment.

Important Biogenesis™ process steps include:

1) Dredged sediment is screened to remove oversized material and debris before transfer to the holding tanks.

2) High-pressure water, proprietary solvent, and physical agitation are combined to separate contaminants from the solids.

3) Treated sediment is then dewatered using a hydrocyclone and centrifuge. Some effluent water may be recycled through the system, but significant quantities of wastewater are generated that require treatment and disposal.

The process results in residual waste products, including sludge and organic material, which require disposal at a regulated landfill. Depending on the nature of the sediment and cleanup levels required, the sediment washing process may need to be repeated for multiple cycles.

The Biogenesis™ proprietary process is designed to separate and to destroy organic contaminants partially (through oxidation); metals are conserved but concentrated in the fines fraction. Results for treated sediment from the three different dredged material sites demonstrated reductions in dioxin concentrations (from 517 nanograms toxic equivalent (ng TEQ)/kg dw prior to treatment to 71 ng TEQ/kg dw post treatment). While this washing technology achieved some measure of contaminant reduction, this appears to have been attributable primarily to solubilization of contaminants and separation of fine solids, rather than because of contaminant destruction through the cavitation/oxidation process. The mass of fine solids lost to the wastewater stream (centrate solids) ranged from approximately 9 to 18% of the incoming sediment mass, although dissolved concentrations were not evaluated (USACE 2011). Only slight decreases in PCB concentrations were documented (450 µg/kg dw prior to treatment and 380 µg/kg dw post treatment) (Biogenesis 2009). PAHs were not effectively removed or destroyed because of adsorption to, or sequestration within, the organic material mixed with the sediment. PAH concentrations in the treated sediment were approximately 52% of concentrations in the incoming sediment for the bench tests. Total PAH mass presumed destroyed or unaccounted for in the overall process ranged from zero to 49.9% (USACE 2011). Approximately 13,000 tons of processed dredged material was loaded onto trucks and transported off site for beneficial reuse as fill material.

Fox River (Wisconsin) – Soil Washing/Sediment Processing

In 2009, approximately 540,000 cy of PCB-contaminated sediments at Fox River (Operable Unit 1) were hydraulically dredged and pumped through a pipeline to a sediment processing facility equipped with particle-size separation, dewatering, and water treatment equipment (i.e., equivalent unit operations used in conventional soil washing). The sediment slurry passed over a vibrating screen enabling <0.5-inch material to pass through. The sand fraction of the slurry was then separated from the silt and clay fractions using a 150-micrometer (μm) coarse sand separation unit. The sand was polished in an up-flow clarifier, gravity dewatered, and temporarily stored on site for potential reuse. Average PCB concentration of dredged material was approximately 1,900 μg/kg dw (EPA 2009c). Total PCB concentrations in the treated sand fraction were on the order of 300 μg/kg dw.

The remaining fine grained sediment (<60 μm) was mechanically filter-pressed to dewater it. The resulting filter cake, typically containing between 1,000 and 10,000 μg/kg dw total PCBs was then land-filled. Process wastewater was treated by sand-filtration and granular activated carbon adsorption. Treated water was returned to the Fox River. Discharge water was monitored for PCBs, mercury, lead, pH, ammonia, biochemical oxygen demand, and TSS.

It is important to note that the process used at Fox River does not destroy organic contaminants. Further, while one of the project goals was beneficial reuse of the processed sand fraction, the sole beneficial reuse to date for this material was using a portion of the sand fraction as fill material (spread in the upland portion of the project site) and as a fill behind the sheetpile bulkhead wall constructed at the site. No beneficial uses outside of the project have been identified (TetraTech 2010a).

Hudson River (New York) – Soil Washing/Sediment Processing

Phase 1 of the dredging operations was conducted at the Hudson River during 2009 (Anchor QEA and ARCADIS 2010). Mechanical dredges with environmental clamshell buckets were used to remove approximately 278,000 cy of river sediments. Dredged material was transported by barges to a shore-based processing and transportation facility. Approximately 370,000 tons of PCB-contaminated sediments were processed to separate size fractions and dewater the solids in a similar fashion to that described above for the Fox River project. As a first step in processing the dredged material, debris and rock were removed and dredged sediments were processed through trammel screens and hydrocyclones to separate the material by size.

Approximately 40% of the sorted materials were fines and 60% were coarse material and wood. After coarse material separation, the slurry of fine sediments was mixed with a polymer in a gravity thickener and filter-pressed. Segregated debris and coarse solids and filter cake removed from the filter presses were temporarily stored in staging areas prior to rail transport and disposal at a permitted facility in Texas. Residual contaminant concentration in the coarse material precluded beneficial reuse of this material. All fractions of dredged material (debris, coarse, and fine) were therefore transported to and disposed of at a permitted facility in Texas. The fine fraction was separated from the coarse fraction and processed through mechanical dewatering to decrease the water content, thereby reducing the transport and disposal costs. A water treatment plant with the capacity to handle 2 million gallons of water per day was built to treat the water collected during the dewatering process. Treated water (approximately 88 million gallons per season) was discharged to the Champlain Canal.

Potential Environmental Review and Permitting Requirements

Permitting requirements for a prospective soil washing operation are currently undetermined and are dependent on the extent of the CERCLA and MTCA LDW site jurisdictional area. If the soil-washing location was determined to be on site, all substantive permitting requirements would be overseen by EPA and complied with as applicable or relevant and appropriate requirements (ARARs), and all procedural and environmental review requirements would be waived. The LDW site includes the upland areas (beyond the scope of this FS) that contributed contamination to the waterway; such upland areas would be considered "on site" for the purposes of siting a treatment facility. All necessary permits would need to be secured if the treatment location is not on site. Permits would also be required for any off-site disposition of treated CERCLA materials and waste streams, such as placement of treated material as off-site fill or off-site discharge of wastewaters to the King County sanitary sewer system.

7.1.2.3 Solidification

Solidification is a proven and effective *ex situ* technology that reduces the moisture content of dredged sediments and reduces the leachability (mobility) of metals. The process involves mechanical blending of the contaminated medium, in this case sediment, with an agent such as cement, cement kiln dust, or super-absorbent polymers. These agents react with moisture in the contaminated media and may produce a material that is much improved structurally (i.e., compressive strength) and can effectively reduce the leachability of contaminants. However, contaminants are not destroyed by solidification.

The major regional landfills (Allied Waste of Roosevelt, Washington, and Waste Management of Columbia Ridge, Oregon) are able to receive contaminated wet sediment at their sites in truck and rail containers (without requiring material to pass a Paint Filter Test [EPA 2008a]). These containers are lined to prevent loss of material (e.g., drainage) during transport.

Solidification does not adequately treat the COCs and solidified sediment would still require transport to a landfill for disposal. For this reason, solidification is not carried forward for alternative development in this FS, but it may be reconsidered during remedial design if moisture or leachability reduction is needed to comply with landfill operating permits.

7.1.2.4 Thermal Treatment

Thermal treatment involves the *ex situ* elevation of the temperature of dredged sediment to levels that either volatilize the organic contaminants (for later destruction in an afterburner) or directly combust the contaminants (e.g., incineration). A number of different system configurations and operating principles have been developed and are available in the marketplace, as described in the CTM. Thermal treatment systems are generally effective for destroying a broad range of organic compounds. Metals are not destroyed by thermal treatment systems.

Thermal treatment facilities are not available either locally or regionally. Therefore, dredged sediment would need to be transported out of state (either to Idaho or Utah) to utilize an existing facility. Alternatively, a temporary on-site (i.e., adjacent to the LDW) facility is technically feasible to consider. Implementability considerations include general siting considerations and obtaining local permits (e.g., air).

The primary drawback to thermal treatment is that treated sediment is unlikely to achieve metal concentration limits for beneficial reuse and may thus still require upland landfill disposal. Studies (e.g., toxicity testing) would also be needed to ascertain whether treated sediment would have properties suitable for supporting benthic productivity before in-water placement of the treated material would be allowed. Thermal destruction processes also require monitoring and management of air releases of hazardous constituents, such as dioxins/furans. Dioxins/furans can be created and released in air emissions from some thermal treatment processes, and fulfilling all substantive permit requirements for managing these air emissions can be difficult and can affect implementability of on-site thermal treatment.

Cement-Lock® Technology is a thermo-chemical manufacturing process that decontaminates dredged material and converts it into Ecomelt®, a pozzolanic material, which when dried and finely ground can be used as a partial replacement for Portland cement in the production of concrete. In the Cement-Lock® process, a mixture of material and modifiers is charged to a rotary kiln at high temperatures, which yields a homogeneous melt with a manageable viscosity. All nonvolatile heavy metals originally present in the sediment are incorporated into the melt matrix via an ionic replacement mechanism. The melt then falls by gravity into water, which immediately quenches and granulates it. The resulting material, Ecomelt®, is removed from the quench granulator by a drag conveyor.

Preliminary pilot-scale results have shown that organic contaminants are partially destroyed, and inorganics (e.g., metals) are encapsulated within the Cement-Lock®

matrix (i.e., Ecomelt®). Although the thermal technology is effective at destroying organic contaminants and immobilizing metals, some metals remain leachable (USACE 2011). The Cement-Lock® cement product passed the Toxicity Characteristic Leaching Procedure test for priority metals. The technology was recently demonstrated at a pilot-scale level for sediments dredged from the Stratus Petroleum site in upper Newark Bay (NJ) in 2006 and from the Passaic River (NJ) in 2006 and 2007. However, both demonstrations experienced equipment-related problems and were terminated (GTI 2008). In these studies, the Ecomelt product samples showed an average reduction in PCB concentrations from 2,800 µg/kg dw (pretreatment) to 0.2 µg/kg dw (post-treatment), with a PCB mass found in the off-gas stream of 0.01% of the incoming sediment PCB mass, for an overall 99.9% (not including the 30% of input mass adsorbed by the carbon bed) unaccounted for and presumed destroyed. The average reduction for 2,3,7,8-tetrachlorodibenzodioxin (TCDD) was from 0.17 µg/kg dw (pretreatment) to 0.008 µg/kg dw (post-treatment) (GTI 2008), with approximately 0.1% of the incoming total dioxin/furan mass being measurable in the Cement-Lock® product (USACE 2011). The fraction of metals leachable in the Ecomelt (Toxicity Characteristic Leaching Procedure [TCLP] mass/total metals in aggregate) ranged from zero to 20%, with average and median values of 3.0 and 0.28%, respectively. The fraction of metals leachable as a fraction of the total metals in the raw feed ranged from zero to 8.8%, with average and median values of 1.1 and 0.24%, respectively (USACE 2011).

Thermal treatment is not carried forward for further consideration in the FS because the process is unlikely to achieve the total metal concentration limits for beneficial reuse although a reduction in leaching potential could perhaps be achieved through use of one of the available technologies (e.g., Cement-Lock® technology).

7.1.2.5 Treatment Technology Summary

Application of activated carbon to sediments to reduce bioavailability is retained as a viable *in situ* treatment technology for the LDW. The technology can be considered in various ways from stand-alone applications to enhancements of other technologies (e.g., amending cap materials or incorporating into media used for ENR). Activated carbon amendment could also prove to be an essential tool of adaptive management (e.g., as a contingency action for underperforming remedial action areas).

Conventional soil washing/particle separation and advanced soil washing have sufficient merit to carry these processes forward in developing the LDW remedial alternatives. Soil washing is retained as an *ex situ* treatment process option because it has been applied at other contaminated sites in the United States and Europe, results in volume reduction of treated dredged material, and may result in a sand fraction suitable for beneficial use in the LDW, or possibly reduce or eliminate the cost of disposal for the sand fraction. Significant engineering design would be required to specify soil washing site location(s), special equipment needs (e.g., cyclones, filters, water treatment systems, etc.), operational procedures, and environmental review and permitting requirements to implement soil-washing treatment.

This FS assumes that soil-washing treatment would be located entirely within the transloading/dewatering facility and would consist of the following elements:

1) Physically wash the dredged material and separate the coarser grained (clean) sediment from the fine particle (contaminated) sediment.

2) Treat the wash water and discharge it to the LDW. Assume use of the following treatment train: collect and settle wastewater, flocculate, filter, analyze, and discharge.

3) Collect and stockpile the cleaned sediment in an on-site location separated from the soil-washing and wastewater treatment operations. Chemically analyze the sediment for COCs to confirm that remnant COC concentrations are less than sediment quality standards (SQS) or other applicable criteria and thereby are determined suitable for beneficial reuse.

4) Transfer the treated sands (processed material achieving target levels established for the project) off site and stockpile for assumed reuse as capping and ENR material for the project. Stockpile requirements need to address logistics and timelines for sand reuse. Specific requirements for sand quality and use need to be defined, including regulatory approvals.

5) Chemically analyze all remaining sediment to determine if treatment has magnified COC concentrations to be greater than landfill-designated hazardous waste concentrations.

6) Based on the chemical analytical results, load railcars with remaining sediment, transfer to the landfill, treat any excess wastewater, and dispose of the remaining sediment appropriately in either a Subtitle C or D landfill.

More advanced soil-washing technologies are not carried forward into the FS as the representative process option in the FS because conventional soil-washing techniques would likely produce the most value in terms of volume reduction for the cost. The expected post-treatment concentrations may preclude the material from beneficial reuse in Puget Sound.

Solidification and treatment technologies were screened out for full-scale consideration in the FS as described above.

7.1.3 Disposal/Reuse of Contaminated Sediment

Several disposal options for dredged sediment were identified in the CTM and are reconsidered here for their applicability to cleanup of the LDW:

- On-site disposal
  - Contained aquatic disposal (CAD)

    - Confined disposal facility (CDF)
- Off-site disposal
    - Existing Subtitle C landfill (40 CFR Part 265, Subtitle C of RCRA)
    - Existing Subtitle D landfill (40 CFR Part 258, Subtitle D of RCRA)
- Open water disposal
    - Dredged Material Management Program (DMMP) site
- Beneficial reuse.

The on-site disposal options retain the contaminated material in or very near the site in new, engineered facilities. The off-site disposal options pertain to upland disposal in existing regional landfills. Open water disposal is also a process option for dredged material that meets the DMMP's criteria for open water disposal. All of these disposal alternatives have demonstrated effectiveness and have been successfully used in the Puget Sound region.

Beneficial reuse is often preferred to disposal, when feasible, although application can be limited by physical characteristics or contaminant concentrations.

7.1.3.1 On-Site Disposal

CAD and CDF are two potential on-site process options for disposal of dredged sediment. As discussed in the CTM (RETEC 2005), both disposal options confine contaminated sediment within an engineered structure. These options differ primarily in location or setting: CAD facilities are located within a water body, and CDFs are located nearshore or upland.

CAD Sites

CAD implementation, although a proven technology, is constrained in the LDW. Material is typically placed in horizontal layers, which requires locating the CAD site in a relatively flat area or depression to minimize excavation quantities during construction, and to prevent spread of contaminated sediment downslope. Potential CAD sites in the LDW are located within or near the defined navigation channel. To ensure that the authorized channel depths are maintained, the top surface of the CAD must be positioned below the authorized channel depth to allow for maintenance dredging. The federally-authorized navigation channel requires maintenance of a specified depth; remedial alternatives within the channel cannot interfere with the authorized channel depth. Two locations in the LDW best satisfy these requirements:

- The deep area at the north end of the LDW directly south of Harbor Island, where the existing depth is well below the authorized navigation channel depth

- The southernmost portion of the LDW, defined by the Upper Turning Basin and adjacent navigation channel.

An advantage of CAD over upland disposal is that the overall project dredging production rate can be significantly accelerated because dredged sediment can be placed directly into bottom-dump barges for rapid movement to and placement into the CAD. Dredging would not be subject to the production rate constraints associated with transloading and transportation to a landfill. As result, the overall period of short-term dredging impacts could be reduced through use of CAD.

Numerous implementability issues would have to be addressed to implement CAD including:

- Logistical and timing considerations need to be planned for, including: 1) CAD construction (e.g., dredging and disposal of excavated sediment), 2) sequencing and timing to dredge and place contaminated sediment in the CAD, and 3) identification and coordination to secure and place capping material. In addition, capping (either interim or final) must be completed by the end of each in-water construction window to protect fish runs from disturbance by construction during migration.

- Barge dumping of contaminated sediment into a CAD site involves some dispersion of material as it falls through the water column and lands on the mudline. Unless care is taken, the dumped sediment can cause a "mud wave" when it strikes the bottom. This can cause contaminated sediment to move out of the CAD area and migrate onto adjacent surfaces. Models are available (e.g., STFATE) to assess this factor and engineering controls would need to be incorporated into the design to minimize or mitigate this factor. These engineering controls can include designing the CAD with features to limit mud waves, monitoring adjacent areas, and capping or implementing ENR for any affected adjacent areas.

- Propeller scour in the navigation channel as well as movement by tugs and other vessels accessing adjacent berthing areas could stir up exposed contaminants and move them into other areas before the cap is installed. Modeling of propeller wash, along with appropriate navigation controls during the construction season can be used to minimize this potential.

A CAD could also potentially be located outside of the LDW (e.g., elsewhere in Puget Sound). However, this would likely be an off-site disposal action subject to permitting requirements. Because the administrative implementability of an off-site CAD is considered low, these possibilities are not explored in this FS.

CAD is being carried forward, and will be evaluated as a disposal alternative with the understanding that CAD capacity may not match the total volume of contaminated dredged sediment under some alternatives. However, regardless of which remedial alternative is selected, CAD may be considered during remedial design on a smaller-scale, location-specific basis, subject to agency approval.

CDF Sites

A nearshore or upland CDF (e.g., construction of a CDF in a slip) is a technically feasible option for the disposal of LDW dredged material, but is not carried forward as a primary in-water disposal technology for the FS. During engineering design, if a small-scale CDF potentially could be applicable, numerous hurdles would need to be overcome. Some of these hurdles include: identifying suitable available land/water sites for acquisition, providing compensatory habitat mitigation for lost aquatic habitat, and demonstrating appropriate economic development purposes for the upland facility in accordance with the Clean Water Act Section 404(b)(1) guidelines.

7.1.3.2 Off-Site Landfill Disposal

Sediments removed from the LDW are not expected to require disposal in a landfill permitted to receive Resource Conservation and Recovery Act (RCRA) hazardous waste or Toxic Substances Control Act (TSCA) waste (i.e., Subtitle C landfill). Nevertheless, a regional Subtitle C landfill (Waste Management, Inc. located at Arlington, Oregon) is available to receive material that exceeds the relevant RCRA or TSCA limits should such material be encountered during remediation.

Two regional Subtitle D landfills (Waste Management, Inc. located at Columbia Ridge, Oregon, and Allied Waste, Inc. located at Roosevelt, Washington) receive both municipal waste and solid nonhazardous contaminated media. Both facilities have been used for the majority of contaminated sediment projects in the Puget Sound region, including several projects in the LDW (Table 7-5). Further, both facilities are permitted to receive wet sediment (i.e., sediment that does not pass the paint filter test and therefore contains free liquid). These existing Subtitle D landfills are retained as representative disposal process options for remedial alternatives that call for sediment removal with disposal in an upland landfill.

7.1.3.3 Open Water Disposal

In Puget Sound, the open water disposal of sediments is managed and monitored under the DMMP, which is jointly administered by the U.S. Army Corps of Engineers (USACE), the EPA, the Washington State Department of Natural Resources (WDNR), and Ecology. The DMMP User's Manual (USACE 2008b) details the sediment evaluation, testing, and disposal procedures for open water disposal of dredged material at DMMP-designated disposal sites in Puget Sound. The DMMP non-dispersive deep water disposal site nearest to the LDW is in Elliott Bay. This facility has approximately 6.6 million cy of remaining capacity.[3]

[3] Approximately 2.4 million cy of dredged material have been placed at the Elliott Bay disposal site between 1989 and 2007. With a capacity of 9.0 million cy, the site will be operational for about 50 more years, assuming about 130,000 cy of placement per year (USACE 2007a).

Some of the LDW sediments that have been dredged from the navigation channel between river mile (RM) 3.8 and RM 4.8 and from private berthing areas outside of the navigation channel have previously been tested and accepted for open water disposal. This suggests that at least some of the sediment removed during remediation may meet DMMP criteria. However, the FS assumes that dredged sediments requiring remediation would not be clean enough to meet DMMP requirements, although they are not necessarily precluded from DMMP open water disposal.

Open water disposal may be considered in the remedial design phase for the following material if the sediment is demonstrated to achieve the DMMP criteria for open water disposal:

- The clean sand fraction from conventional soil washing
- Suitable material dredged from areas during construction of a CAD facility
- Suitable material, if any, dredged under some alternatives in this FS.

7.1.3.4 Beneficial Use of Sediment (Clean and Treated)

Beneficial use of dredged sediment is preferred to its disposal, when feasible. However, contaminated and untreated sediment is not suitable for direct beneficial use applications. This subsection examines the potential beneficial use of:

- Clean dredged material generated by local navigation channel maintenance dredging projects
- Treated sand fraction of dredged contaminated sediments from the LDW.

Any potential in-water beneficial use application would need to meet associated material specifications to ensure an appropriate match between physical, chemical, and biological material properties and functionality in the aquatic environment.

Beneficial Use of Dredged Material from Navigation Projects

Regular USACE maintenance dredging of regional navigation channels in the LDW, Snohomish River, Swinomish Channel, and other rivers generates large volumes of sandy and silty sediments. In the Puget Sound region, this dredged material has been used beneficially for both remediation and habitat enhancement projects. Examples of projects in Elliott Bay that have used sediment from LDW Upper Turning Basin maintenance dredging activities include:

- **Denny Way Combined Sewer Overflow (CSO) Capping –** In 1990, King County and the USACE sponsored the Denny Way CSO capping project to test the feasibility of capping contaminated sediments in Elliott Bay. A 3-ft layer of sediment dredged from the LDW Upper Turning Basin was placed over a 3-acre area at the Denny Way CSO. Monitoring results over the last 15 years demonstrate that the cap is stable, is not eroding, and has successfully isolated the underlying contaminated sediments (King County 2007b).

- **Pier 53-55 Capping –** In March 1992, about 22,000 cy of sediment dredged from the LDW Upper Turning Basin was placed offshore of Piers 53, 54, and 55 in Elliott Bay, to cap approximately 2.9 acres and ENR approximately 1.6 acres of contaminated sediments. Monitoring results indicate that the 3-ft cap and ENR areas are stable, and contaminants are not migrating from the underlying sediments up into the 3-ft cap or ENR area (King County 2010b).

- **Bell Harbor Capping –** In March 1994, the Port of Seattle placed a thin-layer cap of sediment dredged from the LDW Upper Turning Basin over 3.9 acres of contaminated sediments at the former site of Pier 64/65 in Seattle. The site was also designed to incorporate habitat enhancement components, including rock corridors on top of the cap and gravel below the slope and between corridors. These substrata were specifically designed to serve as habitat for brown algae and juvenile rockfish. Subsequent monitoring has demonstrated the success of both actions (Erickson et al. 2005a, 2005b).

- **Pacific Sound Resources Superfund Site in West Seattle –** Approximately 66,000 cy of sediment dredged from the LDW Upper Turning Basin, along with over 200,000 cy of sediment dredged from the Snohomish River, was placed as a cap at the Pacific Sound Resources contaminated sediment site in West Seattle in 2004 (USACE 2007b).

This FS assumes that upland-sourced materials (sand, gravel, and rock) will be purchased for use as cap materials and for ENR. However, the design process should consider the use of navigation channel and berthing area dredged materials determined suitable for beneficial use application as an alternative to upland-sourced materials. The EPA's Contaminated Sediment Technical Advisory Group (CSTAG) has recommended that the navigation channel and berthing area dredged material be considered for these uses in the remediation of the LDW (CSTAG 2006). However, significant administrative issues (including timing, contracting, and administrative approvals) are associated with procuring USACE and private party dredged materials.

Beneficial Use of Treated Contaminated Sediments

For contaminated sediments dredged as part of a cleanup action, treatment would be required before possible beneficial use. Treatment by soil washing followed by beneficial use of the sand fraction may be more cost-effective than treatment followed by disposal. The coarser (sand) product (processed material achieving target levels established for the project) from a soil washing process could potentially be reused within the LDW for capping, habitat restoration, or grade restoration (i.e., to meet final bathymetry requirements) as part of the remedial action. However, a review of existing literature and local knowledge did not identify any examples of treated sediments being used beneficially in the Puget Sound region.

The sand produced from a soil washing process could also be reused in the uplands as construction fill or as material feedstock for other industrial or manufacturing applications (e.g., concrete or asphalt manufacture). Depending on the end use and associated exposure potential, it is not known whether the treated sand fraction would achieve appropriate chemical criteria for all LDW contaminants. Upland beneficial use would also require resolution of legal issues related to material classification, antidegradation, and potential liability.

Remedial alternatives that include soil washing assume that the disposition of the washed material could result in a range of outcomes: 1) achieve the applicable chemical and physical requirements for in-water use and hence be used as on-site cap or ENR material with potential material cost savings; 2) be suitable for upland use as fill with no associated value or disposal cost; 3) be suitable for open water disposal with a comparatively low disposal cost; or 4) require landfill disposal at significant cost.

7.1.4 Capping

In the CTM (RETEC 2005), capping was evaluated and retained as a containment technology that is considered both effective and implementable in the LDW. Capping is a well-developed and documented *in situ* remedial technology for sediment that isolates contaminants from the overlying water column and prevents direct contact with aquatic biota (Figures 7-3 and 7-4). Depending on the contaminants and sediment conditions present, a cap reduces risks through the following primary mechanisms (EPA 2005b):

- Physical isolation of the contaminated sediment sufficient to reduce exposure through direct contact and to reduce the ability of burrowing organisms to move contaminants to the cap surface
- Stabilization of contaminated sediment and erosion protection of the sediment and cap sufficient to reduce resuspension and transport of contaminants into the water column
- Chemical isolation sufficient to prevent unacceptable risks of exposure to sediment contaminants that are solubilized and transported through the cap material and into the water column (e.g., via diffusion or groundwater advection).

7.1.4.1 Conventional Sand and Armored Caps

A large number of sediment caps have been successfully implemented in the Puget Sound region: One Tree Island Marina, Olympia 1987; St. Paul Waterway, Tacoma 1988; Georgia Pacific Log Pond, Bellingham 2000; East and West Eagle Harbor/Wyckoff, Bainbridge Island, 1993-2002; Middle Waterway, Tacoma 2003; General Metals, Tacoma late 1990s; and others (RETEC 2002).

Within the LDW, a sand cap was constructed in 2005 in conjunction with the Duwamish/Diagonal early action area (EAA) sediment remediation project (Anchor and EcoChem 2005a) (Figure 7-5). Preliminary monitoring results from 2007 to 2009 show trends indicating that the cap has successfully isolated underlying contamination. Following cap construction, total PCB concentrations in surface sediment have fluctuated around the SQS. However, because the Duwamish/Diagonal cap is located near an active storm drain and a CSO outfall, and is adjacent to other contaminated sediments, some degree of increase in contaminant concentrations on the cap surface has been noted, highlighting the importance of source control.

The ability to implement capping technology is influenced greatly by physical constraints and engineering design. Capping may be suitable where navigation or other public uses would not be physically impeded, or in areas where it is impractical to remove all of the contaminated material because of slope or nearby structure stability concerns. If capping is chosen as part of the selected remedial alternative for the LDW, then bathymetric, hydrodynamic, slope stability, and biological conditions, as well as commercial/public land use would need to be considered. An engineered cap design specifies material types, gradation, thickness, armoring requirements, design elevation ranges, placement requirements, and other design parameters. For example, the cap design for deep depositional waters would be different from designs for intertidal and shallow subtidal areas of high habitat importance and areas that have the potential for appreciable episodic erosion.

7.1.4.2 Composite and Reactive Caps

A composite or reactive cap may be an appropriate design solution in situations where:

- A reduced cap thickness is needed in navigation-constrained areas to avoid dredging.
- Standard sand capping would require excessive thickness for containment of a specific COC.
- Contaminant migration necessitates reducing contaminant flux over what is achievable with native capping materials.

Reactive cap technology refers to including reactive amendments in the granular cap material or in manufactured mats. The additives are selected based on their ability to adsorb or react with contaminants migrating through the cap strata. Activated carbon, bentonite, apatite, AquaBlok™ (a commercial product designed to enhance contaminant sequestering through organic carbon amendments to the cap, and to reduce permeability at the sediment-water interface), and coke are examples of reactive amendment materials that have been investigated at the demonstration level or in full-scale applications. The need for and type of amendment will be evaluated for specific project areas during remedial design; design data requirements may be different between conventional and thin-layer caps. Section 7.1.4.4 summarizes preliminary modeling results that indicate amendments may not be necessary as a component of cap design for reducing migration of hydrophobic organics through the cap (e.g., PCBs and cPAHs).

The following paragraphs describe examples of composite or reactive cap demonstration level or full-scale application projects.

Carbon Amendment of Cap Materials (Various sites, Washington)
Sand with a carbon amendment was used in caps at the Upriver Dam PCB Sediments Site, Spokane, WA (Anchor 2006a), Olympic View Resource Area, Tacoma, WA (Hart Crowser 2003), and Slip 4 EAA, Seattle, WA (Integral 2010).

Activated Carbon – Reactive Core Mat (Tukwila, Washington)
After sediment dredging and capping was conducted in 1999 by King County offshore of the Norfolk combined sewer overflow (CSO) outfall within what later became the LDW study area, surface sediment monitoring showed that additional sediment removal was needed in the vicinity of the nearby south storm drain outfall of the Boeing Developmental Center to prevent recontamination (PPC 2003). Approximately 60 cy of contaminated sediment were removed and backfilled in September 2003 by Boeing to eliminate the potential source of recontamination to the adjacent cap. The sediment removal was completed during low tide cycles over a one-week period; all work was completed above the water level (at low tide). Following each day’s excavation work, a geotextile fabric layer (Mirafi filter fabric) was installed as a temporary cover to contain and limit any potential migration of silts and the associated contaminants from the excavation area. Turbidity was monitored daily as well as visual monitoring throughout the construction period. Based on turbidity measurements and visible appearance, the daily geotextile fabric cover worked well to prevent loose silt material from mobilizing within the LDW. The geotextile fabric was removed and disposed of before the cap was placed. The excavation area was capped with a fabric containing activated carbon, a layer of sand, and a cover consisting of quarry spoils in the channel segment (where higher velocities from the outfall discharge were expected). The activated carbon fabric was included in the cap permanently to adsorb and contain any residual PCBs in the channel area and prevent upward migration of PCBs in this area. Continued annual monitoring and sediment sampling have verified that no recontamination has occurred within the engineered cap and have demonstrated that the remaining contaminated area is limited to a small segment of the drainage channel located just below the south storm drain outfall (PPC 2003).

Activated Carbon – Reactive Core Mat (Stryker Bay, Duluth, Minnesota)
Stryker Bay in Duluth (MN) was heavily contaminated with tar and coke (Bell and Tracy 2007). Coal tar thicknesses under the water reached as much as 13 ft in some areas. Remediation involved placing six inches of sand cap and a reactive core mat (RCM), followed by six inches of sand cap over the contaminated sediments. The activated carbon-based geotextile fabric, a reactive cap, allowed the cap thickness to be less than a traditional sand cap, and provided stability and physical isolation. According to the *First Five-Year Review Report* (USACE 2003b), the remedial action was

complete and was found to be protective of human health and the environment as intended by the 2000 Record of Decision (ROD) because soils above the direct exposure cleanup levels identified in the ROD for industrial use were removed.

Activated Clay Cap (Willamette River, Portland, Oregon)

In 2004, as part of the cleanup of the McCormick and Baxter Superfund site, the east bank and bed of the Willamette River in Portland (OR) were capped with an organoclay sediment layer to contain high concentrations of COCs, including pentachlorophenol (PCP), creosote, chromium, and arsenic (Aquatechnologies.com, Oregon Department of Environmental Quality [ODEQ] 2005). Over most of the site, the cap consists of a 2-ft-thick layer of sand. In more highly contaminated areas, a 1-ft organoclay layer was placed beneath a 5-ft-thick layer of sand. The organoclay consists of bentonite or hectorite clay modified to be hydrophobic, to have an affinity for non-soluble organics, and especially to prevent breakthrough of non-aqueous phase liquid through the cap. The design of the sediment cap incorporated different types of armoring to prevent erosion of the sand and organoclay layers. In the *Third Five-Year Review Report* (ODEQ 2011), the remedy for the sediment OU was determined to be protective of human health and the environment because the remedy required by the ROD is working as intended.

Granular Bentonite, Sand/Soil/Bentonite Slurry, and AquaBlok™ (Lower Grasse River, Massena, New York)

Pilot studies conducted in 2001 in the Lower Grasse River, Massena, (NY) evaluated capping with various materials as a cleanup alternative for remediating PCB-contaminated sediments (Quadrini et al. 2003). Materials such as a 1:1 sand/top soil mixture, granulated bentonite (clay), and AquaBlok™ were tested as single components or mixtures. Optimal results were achieved with a 1:1 sand/top soil cap applied via a clamshell attached to a barge-mounted crane. Few apparent short-term impacts were noted during the pilot project, as well as negligible water quality impacts. However, in 2003, cap monitoring data indicated significant loss of cap material, and in some cases, significant but localized scouring of underlying sediment (up to 2 ft), that translated into redistribution of the PCBs buried in the river sediments in the upper approximately 1.8 miles of the Lower Grasse River (Quadrini et al. 2003). The possible cause was an ice jam that formed on the river during the spring ice breakup. Consequently, an ice breaking demonstration project was conducted in 2007, the results of which were incorporated into the analysis of alternatives report to evaluate remedial options for the river (Alcoa 2007).

AquaBlok™/Sand (Anacostia River, Washington, D.C.)

A major demonstration of several active-addition reactive cap designs has been conducted on the Anacostia River in Washington, D.C. (EPA 2007c). The objective of this demonstration project was to provide information on the design, construction, placement, and effectiveness of these augmented caps. Various cap technologies were evaluated, including sand (as a demonstration control), AquaBlok™, coke breeze (with potential to sequester and retard the migration of organic contaminants through sorption), and apatite (which encourages precipitation and sorption of metals). The performances of these caps were evaluated in terms of physical stability, hydraulic seepage, and impacts on benthic habitat and ecology. Monitoring of the caps over an approximately three-year period using a multitude of invasive and non-invasive sampling and monitoring tools was used in assessing performance. Results indicate that the AquaBlok™ was highly stable, and likely more stable than traditional sand capping material even under very high bottom shear stresses. The AquaBlok™ material was also characteristically more impermeable, and it is potentially more effective at controlling contaminant flux, than traditional sand capping material. However, the low permeability AquaBlok™ cap showed evidence of heaving because of methane accumulation and release. AquaBlok™ also appeared to be characterized by impacts (lack of colonization) to benthos and benthic habitat similar to traditional sand capping material (EPA 2007c). Apatite results were not available for review in the EPA (2007c) report.

In another demonstration in the Anacostia River in 2004, a RCM was designed to accurately place a 1.25-cm thick sorbent (coke) layer in an engineered sediment cap (McDonough et al. 2006; Figure 7-4). Twelve 3.1-meter (m) x 31-m sections of RCM were placed in the river and overlain with a 15-cm layer of sand to secure it and provide a habitat for benthic organisms to colonize without compromising the integrity of the cap. Placement of the RCM did not cause significant sediment resuspension or impact site hydrology. The RCM was shown to be an inexpensive and effective method to accurately deliver thin layers of difficult to place, high value, sorptive media into sediment caps. It can also be used to place granular reactive media that can degrade or mineralize contaminants.

7.1.4.3 Capping and Overwater Structures

Overwater or floating structures (e.g., docks, piers, marina floats) preclude conventional means of installing a cap using a material barge and excavator or clamshell-based equipment. Various alternative methods are available and have been successfully implemented under these circumstances:

- A belt-conveyor system that can be controlled for angle and speed spray-deposits sand under piers and between pile bents (Figure 7-6).
- Small construction equipment (e.g., skid loader) that fits between pile bents can directly apply cap materials during low tide and where surface conditions are sufficiently stable and access is adequate for maneuvering. This approach was used successfully at the Wyckoff/Eagle Harbor West Operable Unit remediation site in 1997.
- A discharge pipeline can hydraulically deposit a sand-slurry underneath or through the overwater structure. The latter may require removing some of the pier decking. This approach was used successfully at the Wyckoff/Eagle Harbor West Operable Unit remediation site in 1997.

- Pier decks can be removed temporarily to improve access for mechanical placement, as employed at the Martinac Shipyard in the Thea Foss Waterway circa 2003.
- Grout-filled mats can be installed around pile bents, as employed in the Thea Foss Waterway circa 2003.

At intertidal locations where it is difficult to effectively place a sand cap by conventional means (e.g., where the slope is too steep or overhead obstructions exist), a shotcrete cap is an option. Shotcrete is typically composed of concrete or mortar and is pneumatically jettisoned from a nozzle at high velocity onto the surface to be coated at low tide. A shotcrete cap was installed during the Todd Shipyards sediment cleanup (McCarthy, Floyd|Snider 2005). The shotcrete application at Todd Shipyards effectively encapsulated existing debris (slag) mounds (Figure 7-7). Shotcrete can be applied to various material types and surface orientations, including steep embankments. However, shotcrete is not appropriate for use in habitat areas.

7.1.4.4 Modeling of Cap Recontamination

The potential for a conventional *in situ* isolation sand cap to be recontaminated over time by the movement of contaminants through the cap from underlying sediments was analyzed using a one-dimensional groundwater flux model (Lampert and Reible 2009) that also includes net sedimentation on top of the cap. The modeling approach and the results of the analysis are presented in Appendix C, Part 8 (Modeling Contaminant Transport through a Sediment Cap).

The analysis showed that PCB breakthrough above the assumed performance goals is not expected to occur.[4] This is true even where the assumed conditions are unfavorable (high groundwater flow, low sedimentation, and low organic carbon coefficient [K_{oc}]), because the sedimentation rate is always greater than the rate at which the contaminant front migrates through both the cap and the new sediment layer that is continually added over time. The analysis showed that cPAHs behave similarly to PCBs and therefore would also not exceed similar performance goals.

In the complete absence of sedimentation, the results show that capping is still feasible, but that minimum organic carbon requirements for cap materials may need to be specified to achieve a cap design life of more than 100 years. ENR is predicted to achieve assumed performance goals under average conditions, but may not be applicable in adverse conditions (high groundwater flow, no sedimentation, low K_{oc}).[5]

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4 The assumed performance goals for cap modeling are: 1) sediment concentrations not exceeding 100 µg/kg dw total PCBs in the top 10 cm within 100 years, and 2) porewater concentrations below 0.03 µg/L at the sediment/water interface within 100 years.

5 Analysis of ENR generally assumes that placed ENR sand mixes with underlying sediment. This analysis assumed that a thin ENR sand layer (15 cm) did not mix with underlying sediment. Therefore, the analysis is exploratory.

Cap or ENR material specification and applicability of ENR would be evaluated during remedial design.

For the 45-cm clamming point of compliance direct contact scenario, the results show that capping with a 3-ft sand cap is feasible, even in the absence of sedimentation. However, minimum organic carbon requirements for cap materials may need to be specified to achieve a cap design life of more than 100 years.

Specific locations within the LDW, such as Ash Grove Cement (RM 0.1E) and the Duwamish Shipyard (RM 1.35W), have historical high concentrations of metals (e.g., arsenic) in the subsurface. For this reason, remedial design should address the potential for dissolved metals (such as inorganic arsenic) to migrate through a proposed cap to surface sediment and surface water (Palermo et al. 1998). The potential for bioturbation and/or diffusion should also be considered during remedial design of caps.

Although cap modeling results presented in Appendix C (Part 8) indicate amendments may not be necessary as a component of cap design to reduce transport of hydrophobic organics (e.g., PCBs and PAHs), remedial design should identify whether the mobility and bioavailability of metals (such as arsenic) need to be reduced and incorporate any special needs into the design. Several studies (Pattanayak et al. 2000, Mohan and Pittman 2007) report the extensive research conducted on effective removal of arsenic through activated carbon adsorption mechanisms. Many other candidates appear interesting for arsenic adsorption, such as activated alumina, clay, silica sand, and organic polymers, which are known to be good adsorbents that can be regenerated *in situ*. Absorptive capacity should be considered in the design phase.

7.1.4.5 Capping Technology Summary

For developing and evaluating remedial alternatives in the FS, conventional sand cap and armored cap process options have been selected to represent the technology as a whole. Sand caps may be applied to net depositional areas, and armored caps may be applied to areas within the LDW subject to episodic erosion. Reactive caps, although not evaluated in this FS for LDW-wide application, may be appropriate and cost-effective depending on location-specific circumstances.

Section 8 of the FS identifies areas suitable for capping based on evaluating the potential for propeller scour, outfall scour, ship wakes, water depths required for vessel navigation and berthing, slopes, habitat requirements, and erosion associated with high-flow conditions in the LDW. Locations requiring armoring are also considered.

7.1.5 Monitored Natural Recovery (MNR)

Natural recovery of sediments refers to the ability of natural processes such as chemical and biological degradation as well as physical burial by incoming sediments to reduce contaminant concentrations over time (Figure 7-8). Where conditions support natural recovery and natural recovery is included in the remedial alternative, a monitoring program will be instituted as a key component of MNR to assess if, and at what rate, risks are being reduced and whether progress is being made toward achieving the cleanup objectives. The monitoring program associated with an MNR remedy generally combines physical, chemical, and possibly biological testing to track progress toward achieving the cleanup objectives. As with any risk-reduction approach that takes time to reach remediation goals, remedies that include MNR frequently rely upon institutional controls, such as seafood consumption advisories, to control human exposure during the recovery period (EPA 2005b). In the event that MNR does not achieve or progress sufficiently toward achieving performance objectives, contingency actions such as capping, ENR/*in situ* treatment, or dredging may be required. Establishing decision rules with targets and time frames for the performance of MNR is an essential component of an adaptive site management framework (Magar et al. 2009).

As discussed in Section 5, new material transported into the LDW from upstream will tend to settle and bury some of the contaminated sediments. This burial, combined with surficial mixing (both from bioturbation by benthic organisms and resuspension caused by physical processes), is the principal ongoing natural recovery process within the LDW. The majority of COCs in LDW sediments are resistant to chemical and biological degradation and dissolution. These mechanisms are not likely to make important contributions to natural recovery in the LDW. Thus, it is reasonable to expect that the primary factor in determining how quickly natural recovery will occur (assuming sources are adequately controlled) is the burial or sediment deposition rate. Recovery is expected to be more rapid in areas with intermediate to high net sedimentation rates and slow where net sedimentation rates are low or where the potential exists for either significant scour or episodic erosion. The bed composition model (BCM) (see Section 5) was developed as a tool to predict recovery as a function of both location within the LDW and of the concentrations of contaminants coming into the LDW from upstream and lateral (e.g., stormwater) sources.

7.1.5.1 Sediment Remediation Projects with an MNR Component

Examples of sediment remediation projects where MNR is a component of a combined remedy or where natural recovery trends have been observed are provided below.

Duwamish/Diagonal EAA (Seattle, Washington)

Data collected during the Duwamish/Diagonal EAA project (Anchor 2007) lend some empirical support to natural recovery potential in the LDW. This project involved a combination of removal (dredging), capping, and thin-layer sand placement. Surface sediment contaminant concentrations are being monitored on and adjacent to the actively remediated areas of the project site (Figure 7-5). Monitoring data associated with the cap and thin-layer sand placement are discussed below in Section 7.1.6. The data collected from stations peripheral to the actively remediated areas are plotted versus time in Figure 7-5 (center chart). The trends suggest that contamination from resuspension and dispersal during the dredging operation may have been responsible for total PCB concentrations remaining high and are consistent with data generated during the investigative phase of the project in the mid-1990s. Since that time, total PCB concentrations have declined by 50% or more in five of the eight perimeter locations, presumably as a result of natural recovery processes (see Appendix F). Net sedimentation rates ranging from 0.7 to 3.1 cm/yr were estimated from radioisotope core data in the Duwamish/Diagonal area, consistent with the STM model predictions (see Appendix F, Figure F-2). The average concentration of the perimeter stations (Figure 7-5) have already decreased (after 5 years) to below modeled predictions of recovery 10 years following remediation (Stern et al. 2009). However, dispersion of some of the newly placed capping material appeared to have initially influenced some immediately adjacent noncapped areas, thereby contributing to the decrease in PCB concentrations seen in the first post-capping year. Unpublished PCB data from 2010 sampling indicate that the total PCB concentration has decreased by approximately 67% from that observed in 2009 (Williston, personal communication, 2010) indicating the area is continuing to recover.

Slip 4 EAA (Seattle, Washington)

Additional empirical support for natural recovery in the LDW can be discerned from the Slip 4 surface sediment dataset, as shown in Figure 7-9, although the conditions in the slip are somewhat different than those in the LDW outside of the slip. This figure shows where surface sediment samples were collected and analyzed for total PCBs within the Slip 4 EAA. These data were divided into two groups representing conditions observed before 1999, and conditions observed in 2004 (see Figure 7-9). The two datasets were analyzed statistically and determined to be significantly different ($p<0.05$; Mann-Whitney two-sample test). The mean total PCB concentration in the 2004 dataset (830 µg/kg dw) is 24% of the mean concentration in the pre-1999 dataset (3,200 µg/kg dw). However, sampling of Slip 4 surface sediments in 2010 revealed increasing PCB concentrations within the EAA, which highlighted the need for additional source control actions (Ecology 2011a). Net sedimentation rates ranging from 1.6 to 3.2 cm/yr have been estimated from radioisotope core data in the Slip 4 area, contributing to the process of natural recovery; these estimated rates are consistent with the STM model predictions (see Appendix F, Figure F-2).

Sangamo Weston/Twelve-Mile Creek/Lake Hartwell (Pickens, South Carolina)

Lake Hartwell and its tributary Twelve-Mile Creek are heavily contaminated with PCBs, which were discharged by the Sangamo Weston Inc. facility between 1955 and 1977. MNR, in combination with institutional controls (fish consumption advisories), was selected by EPA as the main remedy for Operable Unit 2. Net sedimentation rates of 5 to 15 cm/yr, confirmed by radioisotope geochronology, and burial by progressively cleaner sediment over time is the dominant physical process for recovery. Field measurements show a gradual recovery of surface sediments from peak concentrations of approximately 40 mg/kg dw to around 1 mg/kg dw in more recent samples (Magar et al. 2003). In addition, sedimentation for the Twelve-Mile Creek arm of Lake Hartwell has been accelerated by the release of accumulated sediment from three upstream dams. Chemical transformation (i.e., PCB dechlorination) has also been observed via PCB congener analysis of sediment cores with depth and age. This natural process has been found to be slow and limited as a result of anaerobic subsurface sediment, but it has reduced the long-term risks associated with potential sediment resuspension (Magar et al. 2009).

Annual monitoring has been conducted through sediment sampling (at 21 locations within the tributary and lake), fish tissue sampling (at 6 lake locations), and bioaccumulation studies (in caged *Corbicula* clams) to track the progress toward achievement of cleanup objectives. Despite the substantial historical decrease in PCB sediment concentrations (below the 1 mg/kg dw cleanup level), fish tissue concentrations have not decreased accordingly (Magar et al. 2004, Magar et al. 2009). PCB concentrations in catfish fell below the Food and Drug Administration (FDA) tolerance level of 2 mg/kg wet weight (ww) for several years, but this trend has not been sustained since 2005. The other five fish species monitored show no clear trend of decreasing PCB concentrations. Fish consumption advisories remain in effect for Twelve-Mile Creek and Lake Hartwell, because PCB concentrations in fish continue to exceed the FDA tolerance level of 2.0 mg/kg ww.

James River (Hopewell, Virginia)

The chlorinated pesticide Kepone (chlordecone, a carcinogenic chlorinated hydrocarbon) was made and discharged between 1974 and 1975 through the municipal sewage system, surface runoff, and solid waste dumping into the James River estuary in Hopewell (Virginia). Average Kepone concentrations in the channel sediments ranged from 20 to 193 µg/kg dw.

MNR was selected as the main remedy for all areas of the site, and the dominant natural recovery processes were dispersion (in high-energy areas) and physical isolation through natural sedimentation (in low-energy areas). Radioisotope geochronology showed evidence of natural sedimentation within the estuary, ranging from less than 1 cm/yr to greater than 19 cm/yr, with an average of at least 8 cm/yr at 8 of the 21 sediment sampling locations (Magar et al. 2009).

Although Kepone tissue concentrations in James River fish reached as high as 5 mg/kg ww in 1975, the average tissue concentrations had fallen below the FDA action level of 0.3 mg/kg ww by 1986 (Luellen et al. 2006). The last exceedance of the action level in striped bass was measured in 1995, according to the Virginia Department of Environmental Quality (VA-DEQ 2011). However, Kepone continues to be detected in about 94% of fish tissue samples above reporting limits. Continued detections of Kepone are believed to be related to coastal disturbances related to severe weather (Luellen et al. 2006, Magar et al. 2009). The observed decline in fish contamination over the years is thought to be the result of the Kepone being sequestered in the tidal basin sediments of the James River and thus becoming less available to contaminate fish (Lawson 2004).

A fish consumption advisory is still in effect for Kepone, and the VA-DEQ continues to monitor Kepone levels in fish tissue and sediment to address concerns about contaminated sediment resuspension after high-energy events (Magar et al. 2009).

Bremerton Naval Complex (Puget Sound, Washington)

The cleanup of Puget Sound Bremerton Naval Shipyard Complex (PSNS), located on the Sinclair Inlet of Puget Sound at Bremerton (WA), included extensive dredging, capping, ENR, and long-term monitoring of surface sediments to assess natural recovery (EPA 2000c). The marine area of concern (Operable Unit B) in the PSNS is a subtidal section of the inlet, with water depths generally less than 15 m. Baseline total PCB concentrations in sediments within the area of concern were around 13 mg/kg organic carbon (oc) (with a maximum measured concentration of 61 mg/kg oc) (Merritt et al. 2010).

Three rounds of post-remedy monitoring (2003, 2005, and 2007) have been completed, including measures to verify the integrity of remedy components and assess progress toward cleanup goals. In addition, bathymetric surveys, sub-bottom profiling, and collection and analysis of sediment cores were performed. These activities have confirmed that dredging, capping and ENR remedy components are functioning as planned, and that ongoing sediment deposition and mixing (MNR) are occurring naturally (URS 2009).

Sampling of marine sediments throughout Operable Unit B and Sinclair Inlet were also conducted. In 2007, the geometric mean for Operable Unit B Marine sediment total PCB concentrations, estimated on an area-weighted average basis, was 4.5 mg/kg oc (URS 2009); this value exceeded the cleanup goal of 3 mg/kg oc, but it was less than the 2003 and 2005 area-weighted geometric mean values (6.7 and 6.1 mg/kg oc, respectively).

Total PCB concentrations in English sole tissue samples were also analyzed. The 2007 arithmetic mean English sole total PCB concentration was 0.033 mg/kg ww, above the remedial goal of 0.023 mg/kg ww (URS 2009) and well below the concentration of 0.085 mg/kg ww obtained in 2003.

Trend analyses for Operable Unit B Marine performed on the 2003, 2005, and 2007 sediment samples predicted a decreasing trend and indicated that the cleanup goals established in the ROD may be achieved within 10 years after remediation (<3 mg/kg oc for PCBs) and the long-term goal of <1.2 mg/kg oc for PCBs may be achieved by 2017 (EPA 2000, URS 2009, Leisle and Ginn 2009).

7.1.5.2 MNR Summary

NRC (2007) projected that MNR is likely to be a component of many large-scale sediment remediation projects with temporal goals. In the LDW, natural recovery is predicted to occur at varying rates at specific locations within the LDW, as supported by the LDW examples above, modeling, and comparison of co-located sediment samples collected over time (see Appendix F). For these reasons, MNR is retained as a remedial process option for developing the remedial alternatives in this FS. LDW-wide reductions in average concentrations of COCs such as PCBs are necessary to reduce resident fish and shellfish tissue concentrations. Hence, MNR is also evaluated as an LDW-wide "polishing step" for all of the remedial alternatives considered in this FS.

7.1.6 Enhanced Natural Recovery (ENR)

ENR refers to the application of thin layers of clean granular material, typically sand, to a sediment area targeted for remediation. Application thicknesses of approximately 6 inches are common, producing an immediate reduction in surface contaminant concentrations (Figure 7-7). Essentially, ENR reduces the time for sediment concentration reductions over what is possible by relying solely on natural sediment deposition where burial is the principal recovery mechanism (EPA 2005b). Thus, areas that are stable (not expected to erode) and are recovering naturally (albeit slowly) are candidates for ENR. Although ENR is best employed in areas not subject to scour, it may be appropriate in some cases to employ engineered aggregate mixes or engineered synthetic products to ensure stability (Palermo et al. 1998, Agrawal et al. 2007).

Unlike capping, which typically has a much greater application thickness, surface sediment contaminant concentrations in areas that undergo remediation by ENR are expected to be influenced by benthic recolonization and associated bioturbation. These processes result in the mixing of underlying contaminated sediment with the cleaner near-surface material. This is important for remedial design where a surface sediment concentration threshold is typically established below which MNR is appropriate (i.e., cannot be achieved in an acceptable time scale) and above which other active technologies (e.g., dredging or capping) should be considered.

The FS assumes that half of the ENR footprint would warrant amendment with a material such as activated carbon for *in situ* treatment. This assumption provides a basis for estimating costs and comparing the remedial alternatives; however, during remedial design, the emphasis on ENR or *in situ* treatment will depend on location-specific factors and additional testing of the implementability of these technologies. The composition of ENR/*in situ* treatment will depend on additional evaluation during remedial design; it may include carbon amendments, habitat mix, and/or scour mitigation specifications to increase stability and enhance habitat.

7.1.6.1 ENR Sediment Remediation Projects

Examples of ENR sediment remediation projects are provided below.

Ketchikan Pulp Company (Ketchikan, Alaska)

A thin-layer placement was successfully applied in 2001 over the sediments offshore of a former sulfite pulp mill (Ketchikan Pulp Company-KPC) in Ward Cove, Alaska (Merritt et al. 2009, Becker et al. 2009). The primary COCs were ammonia and 4-methylphenol. These COCs are not bioaccumulative. Diffusion of contaminants from underlying sediment was identified as the dominant mode of chemical transport responsible for toxicity to organisms in surface sediment.

The thin-layer cap of fine-grained to medium-grained sand was placed over 28 acres of native sediments to a thickness ranging from 15 to 30 cm (Merritt et al. 2009). In 2004 and 2007, the first and second monitoring events were conducted, and included evaluations of sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities. Concentrations of both COCs in the thin-layer strata were low in 2004, indicating ENR effectiveness. The clean sand placement material was not being noticeably affected by upward migration of the COCs from underlying native sediment; the concentrations of COCs remained low in 2007. For sediment toxicity, amphipod survival was about 93 to 96% in 2004 and remained high in 2007 (92 to 95%). Benthic communities had begun recolonization by 2004 and total abundance increased substantially in 2007 (Becker et al. 2009).

Duwamish/Diagonal EAA Project (Seattle, Washington)

In response to observed increases in surface sediment concentrations of total PCBs adjacent to a portion of the primary dredging and cap area at the Duwamish/Diagonal EAA, a thin layer of sand (9 inches, to ensure a minimum 6-inch coverage everywhere) was placed in February 2005 over 4 acres of sediment, providing immediate reduction in exposures, and reducing total PCB concentrations to between 1 and 32 µg/kg dw (Figure 7-5; Anchor 2006b). Prior to dredging and capping, this adjacent area had an average total PCB concentration of 46 mg/kg oc. Immediately following cap placement, that average tripled to 136 mg/kg oc. This increase in total PCB concentrations was attributed to resuspension and dispersal of contaminated sediment (i.e., dredging residuals) during the removal action. Within the ENR area, total PCB concentrations immediately following thin layer placement were well below the SQS (at a mean of 7 µg/kg dw[6]) because of the clean material placed, achieving its goal of immediately reducing PCBs to below predredge surface sediment concentrations. Subsequent years have shown a slight increase in the PCBs concentrations (Stern et al. 2009). The slight increase is likely due to resuspension of the surrounding sediments and by deposition of upstream and lateral load contributions according to the inputs to the area used in the STM. Modeling, supported by monitoring data and physical measurements of the sediment surface layer, has also shown that the thin sand layer is not significantly mixing with the underlying sediment, consistent with measured bioturbation depths (Stern et al. 2009).

A comparison of the 2008 and 2009 total PCB averages of 8 and 5 mg/kg oc, respectively, to the 2003 predredging/capping average of 46 mg/kg oc (almost a six-fold decrease) demonstrates that ENR continues to maintain exposures below the SQS.

Based on diver probing surveys conducted in April 2009, the thickness of the ENR sand layer exhibited a minor decrease from 2006 to 2009. The estimated thickness of the ENR sand layer ranged between 5 and 10 inches at 11 different sampling locations, while 1 to 8 inches of silt were observed to have accumulated on the surface of the ENR layer.

[6] Total organic carbon content in the March 2005 sampling event was too low to calculate oc-normalized data.

When silt and sand are considered together, the average thickness was 12.8 inches (Anchor QEA 2009). These results are consistent with deposition and bioturbation processes as originally anticipated in the ENR area, but also indicate the presence of a stable surface over a period of time. Post-placement bathymetric monitoring was also conducted and nearly all of the Duwamish/Diagonal cleanup area exhibited accretion over the 5-year period following completion of the ENR remedy.

7.1.6.2 ENR Technology Summary

ENR has sufficient merit and has been sufficiently demonstrated in sediment remediation projects elsewhere to carry this technology forward in developing LDW remedial alternatives. ENR may be applied to broad areas of the LDW with lower levels of contamination, net sedimentation, and where significant erosion is not a concern.

7.2 Institutional Controls

Institutional controls are non-engineered measures that may be selected as remedial or response actions either by themselves or in combination with engineered remedies, such as administrative and legal controls that minimize the potential for human exposure to contamination by limiting land or resource use (EPA 2000e). The National Contingency Plan (NCP) sets forth environmentally beneficial preferences for permanent solutions, complete elimination rather than control of risks, and treatment of principal threats to the extent practicable. Where permanent and/or complete elimination are not practicable, the NCP creates the expectation that EPA will use institutional controls to supplement engineering controls as appropriate for short- and long-term management to prevent or limit exposure to hazardous substances, pollutants, or contaminants. It states that institutional controls may not be used as a sole remedy unless active measures are determined not to be practicable, based on balancing trade-offs among alternatives (40 CFR 300.430 [a][1][iii]).

EPA recommends that where it may provide greater protection, multiple institutional controls should be used in combination, referred to as "layering" by EPA. Institutional controls may be an important part of the overall cleanup at a site, whenever contamination is anticipated to remain following active remediation at concentrations that exceed cleanup levels. Institutional controls may be applied during remedy implementation to minimize the potential for human exposure (as temporary land use or exposure limitations). These controls may also extend beyond the end of construction (or be created at that time) or even after cleanup objectives are achieved to ensure the long-term protectiveness of remedial actions that leave contaminants on site above cleanup levels (as long-term or permanent limitations, e.g., protecting a contaminant barrier like a sediment cap from being accidentally breached).

Institutional controls potentially applicable to cleanup of the LDW site are identified and discussed below. This section describes specific individual controls in sufficient detail to allow for a comparison of remedial alternatives that includes various types and degrees of reliance on institutional controls. An integrated Institutional Controls

Implementation Plan is anticipated for the LDW after the ROD is issued that meets specific location, tribal treaty rights, and community needs. These considerations are discussed further in the FS as part of the development and evaluation of remedial alternatives (Sections 8 and 9).

EPA guidance broadly lists four types of institutional controls: governmental controls, proprietary controls, enforcement tools, and informational devices. However, governmental controls such as the permitting of some (point source but not non-point source) discharges to, or dredging and filling of the LDW, as well as some enforcement controls, such as consent decrees or administrative orders under which settling parties implement remedies including institutional controls, are not discussed at any depth in this FS because they do not inform the choices among alternative remedies. These governmental controls are, for remedy selection purposes, uniform across all alternatives and options (i.e., permitting requirements cannot be changed by remedy selection in the ROD), and consent decrees will be used if responsible parties implement any or all of any remedial action EPA selects in the ROD as required by Section 122(d) of CERCLA. Therefore, the most important institutional controls, or aspects of them, for the development of remedial alternatives are emphasized below. Enforcement tools, even though they are used, for example, to establish enforceable proprietary controls pursuant to consent decrees or orders, are discussed under the category of informational devices. It should be clear at this point that many categories overlap and that the agency guidance that created them was intended to be helpful in analyses rather than necessarily invent divisible categories (e.g., proprietary controls have government enforcement mechanisms to ensure their continuation, and some informational devices can be related to or enhanced by governmental enforcement programs):

- Proprietary controls
- Informational devices
  - Monitoring and notification of waterway users
  - Seafood consumption advisories, public outreach, and education
  - Enforcement tools
  - Environmental Covenants Registry.

These types of institutional controls are outlined below.

7.2.1 Proprietary Controls

Proprietary controls are recorded rights or restrictions placed in property deeds or other documents transferring property interests that restrict or affect the use of property. Covenants are a grant or transfer of contractual rights. Easements are a grant of property rights by an owner, often for a specific purpose (e.g., access, utility, and environmental, among other types of easements). Covenants and easements are essentially legally binding arrangements that allow or restrict usage of property for one or more specific objectives (e.g., habitat protection, protection of human health, etc.). They commonly survive the transfer of properties through real estate transactions and are binding on successors in interest who have not participated in their negotiation. This distinguishes covenants and easements from ordinary contracts or transactions between or among parties. At cleanup sites, covenants and easements commonly control or prevent current and future owners from conducting or allowing activity that could result in the release or exposure of buried contamination as long as necessary. Potential activities controlled or prohibited may include in-water activities (e.g., anchoring, spudding, vessel or tug maneuvering) and construction activities (e.g., pile driving and pulling, dredging, and filling) where buried contamination may become exposed as a result of the activity, as long as it is an activity the owner may legally control. Selecting a less expensive remedy in the form of a proprietary control that limits future property uses in ways a more expensive remedy would not, involves a complex balancing of interests by EPA and Ecology. For example, a proprietary control can lower remedial costs for a former owner at the expense of the redevelopment options of a current owner, who acquired the property after it was contaminated. For this reason, among others, EPA policy and guidance stress assessing reasonably anticipated future land use as an important part of remedy selection generally, and specifically stress limiting use of institutional controls.

Traditionally, covenants or easements were only enforceable by whomever they were granted to, and their successors, depending on how they were crafted. In Washington State, MTCA gave Ecology the right to enforce covenants created under MTCA. More recently, Washington passed its Uniform Environmental Covenants Act (UECA), which allows EPA, as well as the state (in addition to the parties to an UECA covenant), to enforce environmental covenants. For this reason, UECA covenants are anticipated to be the primary proprietary control used in LDW environmental cleanup actions.

Parties with sufficient ownership interests in shorelines and aquatic land could grant UECA covenants that would help ensure that remedial measures (such as sediment caps) are not disturbed. However, UECA covenants may not be implementable or practicable for the publicly-owned, working industrial waterway portions of the LDW where the balancing of interests is especially complex, where access and use are in any case difficult to control, and where the extent of the authority of public entities with ownership or management rights to grant covenants with the full range of controls commonly included in UECA covenants, is uncertain. Another uniquely important interest to consider is the extent to which public entity granted covenants may interfere with tribal treaty-protected seafood harvesting, in particular.

7.2.2 Informational Devices

7.2.2.1 Monitoring and Notification of Waterway Users

The LDW ROD could include an enhanced notification, monitoring, and reporting program for areas of the LDW where contamination remains following cleanup activities. Under such a program, the protection of areas where contamination remains above levels needed to meet cleanup objectives, including areas where capping or CAD containment technology have been utilized, could be enhanced. Such areas could be periodically monitored (by vessels and/or surveillance technology), with vessels performing the dual role of educating potential violators of the existence of activity restrictions, and promptly reporting violations of use restrictions to EPA or Ecology, or the U.S. Coast Guard (USCG) if the area were formally designated as a Restricted Navigation Area (RNA) by formal USCG rulemaking as described in Section 7.2.2.3, Enforcement Tools. Notification to waterway users could further be provided through enhanced signage and other forms of public notice, education, and outreach. A mechanism for the review of any USACE navigation dredging plans and other Joint Aquatic Resource Permit Application (JARPA) construction permitting activity could be established. The review would identify any projects that may compromise containment remedies (cap or CAD) or potentially disturb contamination remaining after remediation, which would include a requirement to promptly notify EPA and Ecology during the permitting phase of any project that could affect cleanup remedies. This mechanism would serve as a backup to an existing Memorandum of Agreement between EPA and USACE for coordinating such permitting, especially if that agreement were to lapse or be discontinued for any reason by either agency in the future.

Additional measures could include: establishing a LDW cleanup protection hotline private citizens could call or email to report potential violations, with a requirement that reports be investigated and conveyed to EPA and Ecology (and the USCG for any RNAs) under specified protocols; and developing and implementing periodic seafood consumption surveys to identify, by population group and geographical location, which seafood species are consumed, where they are consumed, and in what quantities they are consumed. This information would be used to update the Institutional Control Implementation Plan as appropriate and improve seafood consumption advisories and associated public outreach and education. Additional monitoring of the effectiveness of these tools can be used to adapt this approach, as discussed in the next section. The effectiveness of all these measures could be re-evaluated periodically to assess which ones should be continued or be modified.

7.2.2.2 Seafood Consumption Advisories, Public Outreach, and Education

The Washington State Department of Health (WDOH) publishes seafood consumption advisories in Washington. The WDOH currently recommends no consumption of resident seafood from the LDW. Salmon are not resident in the LDW; they are anadromous species that spend most of their lives outside of estuaries like the LDW. WDOH recommendations for Duwamish salmon are the same as for Puget Sound as a whole (e.g., no more than one meal per week of Chinook salmon). The WDOH maintains a web site that includes its advisories and provides publications and other educational forums that cover healthy eating and seafood consumption. In addition, WDOH seafood consumption advisories are posted on signs at public access locations around the LDW. Following these advisories is wholly voluntary, which makes advisories, as a necessity, a last resort. Advisories would also be fundamentally inconsistent with tribal fishing rights secured under treaties of the United States if they were relied on in lieu of cleanup measures intended to provide seafood suitable for consumption. More information can be found at http://www.doh.wa.gov/ehp/oehas/fish/rma10.htm.

The Washington State Department of Fish and Wildlife (WDFW) develops and enforces seasonal restrictions on recreational fishing and seasonal and daily catch limits per individual for various seafood species. WDFW licensing and LDW enforcement activities presumably limit resident LDW seafood consumption to some unknown degree. All recreational fishers over 15 years of age must have a fishing license and comply with specific size, species, and seasonal restrictions on fishing for fish and shellfish throughout the Puget Sound region. In the LDW, all resident fish and shellfish should not be consumed according to WDOH advisories. While WDFW regulations summarize the WDOH seafood consumption advisories, which may enhance their reach and effectiveness, they do not prohibit fishing or shellfishing within the LDW. It is lawful to seasonally collect and consume certain fish and shellfish from the LDW.

Some level of seafood consumption advisories will likely be necessary into the foreseeable future to reduce human health risks from seafood consumption. This is because of the technical impracticability of achieving the seafood consumption cleanup levels under any of the remedial alternatives. Concerns associated with the use of these ICs include the burden placed on tribes exercising their treaty rights and other fishers who use the LDW. Relying on seafood consumption advisories to further reduce human health risks may require fishers to change behavior or make cultural adjustments. This burden is difficult to value precisely given the broad range of needs different fishers may have. Given the diversity of the community that can access the LDW, including tribal members, recreational users, low-income, and non-English-speaking people, additional measures to enhance the effectiveness of seafood consumption advisories and thereby enhance confidence in relying upon them, should be fully and aggressively explored.

An enhanced approach called community-based social marketing was adopted at the Palos Verdes Superfund site in California to reduce the limitations of seafood consumption advisories (EPA 2009a, 2009b). This approach, pioneered by Doug McKenzie-Mohr of St. Thomas University in Canada in 1999, as cited in EPA (2009a), can be summarized broadly as:

- Researching to establish and quantify baseline behaviors and size/demography of different populations and to identify culturally-specific barriers and benefits.
- Defining desired behaviors and understanding barriers to achieving those behaviors; definition of incentives for overcoming barriers and achieving behavior change.
- Creating effective messages/incentives and effective delivery and monitoring mechanisms.
- Implementing culturally-appropriate outreach to all target populations using brief, clear, tested messages and incentives.
- Following up on research after a time period to monitor and evaluate levels of behavior change and to modify the approach as needed.

Application of community-based social marketing concepts in the LDW, modeled after the program and experience-base developed for the Palos Verdes site, could improve the effectiveness of existing seafood consumption advisories for protecting human health.

A collaborative advisory group could be convened to develop an LDW-specific framework and technical approach. Likely participants would include EPA, Ecology, WDOH, WDFW, and other interested federal, state, and local government agencies such as the National Oceanic and Atmospheric Administration, the Seattle Department of Neighborhoods, and ethnically-specific community group leaders, as well as non-governmental organizations and settling parties. A key mandate of the advisory group would be the founding of a small, credible, and knowledgeable core team to facilitate the effort (e.g., develop and complete surveys to better understand affected populations [demographics], and potential incentives for and barriers to improving the effectiveness of seafood consumption advisories).

The overarching goal of this effort would be to develop and implement a public outreach and education program that focuses on incentives and activities that research indicates have the greatest likelihood of adoption and would make the greatest substantive difference in environmental health. Ideally, the program would be coordinated with other health-based initiatives such as the City of Seattle's urban agriculture initiative.

Implementation of the outreach and education program could be accomplished in a number of ways, stressing culturally-appropriate teams, objective and credible participants, and a systematic approach to applying, documenting, and quantifying results of the approach. The advisory group would recommend program elements based on ideas generated by the group and the affected communities, and a review of approaches demonstrated to have caused positive behavior changes at other sites. It would also recommend appropriate programmatic changes as needed based on the evolution of monitoring and survey-based information. Example elements of the outreach and education program for enhancing the effectiveness of seafood consumption advisories include:

- Establish a website to provide up-to-date information on seafood contaminant concentrations and consumption advisories.
- Increase the use of signs containing advisory information at fishing locations.
- Conduct outreach efforts at fishing locations on a regular and periodic basis.
- Ensure all recreational anglers receive seafood consumption advisory information when purchasing licenses.
- Disseminate advisory-related information at community health facilities, schools, and at community-based functions such as health fairs.
- Encourage medical and other health professionals to communicate risks to the public.

A significant difference between the Palos Verdes site and the LDW is the presence in the LDW of tribal fishing rights secured by treaties of the United States. Nothing in this section or anywhere in this FS is intended to suggest that exercise of such rights, or the underlying cultural traditions, would be precluded by seafood consumption advisories and related programs to reduce contaminated seafood consumption as part of LDW remedial action. For this reason, the seafood consumption advisories, and public outreach education programs should be developed in consultation with affected tribes to develop accommodations for such tribes to the greatest extent practicable. A significant limitation of the Palos Verdes enhancement to conventional seafood consumption advisories is that individual responses remain entirely voluntary.

7.2.2.3 Enforcement Tools

As mentioned above in the context of the potential development of monitoring and notification programs as a selected component of remedial action for the LDW, RNAs are created by the promulgation of formal rules by the USCG. RNAs represent an enforceable means of protecting containment remedies and other areas where contamination remains from anchoring and other physical interference, particularly where UECA covenants or other proprietary controls may not be achievable, such as within Commercial Waterway District #1. To the extent that RNAs may potentially interfere with seafood harvest activities, particularly tribal harvests, engineered or other alternative means of accommodating fish harvest should be devised (e.g., alternative means of allowing anchoring or tying off a net within a RNA-created no-anchor zone).

Although this option has the significant potential to regulate potential impacts associated with anchorage, barge spudding, and tugboat propeller wash, it could restrict maritime commerce or preclude commercial activities generally necessary for construction, maintenance, and operation of commercial piers, depending on where the RNA was located. Like proprietary controls generally, even for sediment areas in private ownership, RNAs require a careful and often highly complex balancing of competing interests, and may only be useful in certain locations or circumstances. Whenever the government limits or adversely affects property rights, it may be subject to takings claims by affected persons based on the Fifth Amendment to the Constitution of the United States.

7.2.2.4 Environmental Covenants Registry

Placement and maintenance of LDW areas, with containment remedies (cap or CAD) or anywhere where contamination remains above levels needed to meet cleanup objectives, on Ecology's Environmental Covenants Registry in its Integrated Site Information System) would provide information regarding applicable restrictions (RNAs and proprietary controls) to anyone who uses or consults the state registry.

7.2.3 Institutional Controls Summary

In summary, it must be emphasized that all of the institutional controls described in this section are difficult to enforce. Privately owned sediments, like publicly owned sediments, in an urban commercial waterway are generally substantially more difficult to guard or restrict uses of than upland properties. Further, it is anticipated that some people, including tribal members with treaty-protected harvest rights, will choose to fish and consume what they catch regardless of seafood consumption advisories and robust public outreach and education programs. For these reasons, institutional controls will be relied on only to the extent necessary to develop practicable remedial actions for the LDW.

7.3 Monitoring

Monitoring is an important assessment and evaluation tool for collecting data and is a requirement of remedial alternatives conducted under CERCLA and MTCA. Monitoring data are collected and used to assess the completeness of remedy implementation, remedy effectiveness, and the need for contingency actions. The sampling and testing process options common to most sediment remediation projects are as follows:

- Sediment quality (e.g., chemistry, grain size distribution)
- Sediment toxicity
- Surface water quality (e.g., conventional parameters and contaminant concentrations)
- Contaminant concentrations in porewater

- Contaminant concentrations in fish and shellfish tissue
- Physical (e.g., visual inspections, bathymetry).

Typically, these sampling and testing process options are prescribed components of project monitoring plans which, in turn, focus on different aspects of the remedial action. For example, monitoring during the construction phase has different objectives than the operation and maintenance (O&M) monitoring that follows construction. Five different monitoring concepts that form the basis for individual or combined monitoring plans, depending on project-specific circumstances, are described below.

In addition, source control monitoring (addressed under Tier 4 of the source control strategy, see Section 2.4) and evaluation within upland drainage basins will be required by Ecology in parallel with in-water monitoring for remedial actions and may include parties other than those responsible for performing the remedial action. The goal of source control monitoring is to determine the potential for recontamination in areas that have already been remediated and become subsequently recontaminated above LDW cleanup levels. Type and scope of source control monitoring is not discussed in the FS since this varies on a site by site basis.

7.3.1 Baseline Monitoring

Baseline monitoring establishes a statistical basis for comparing physical and chemical site conditions prior to, during, and after completion of a cleanup action. Baseline monitoring for the LDW will likely entail the sampling and analysis of sediment, surface water, and tissue samples in accordance with a sampling design that enables such a statistical comparison of conditions.

7.3.2 Construction Monitoring

Construction monitoring during active remediation is area-specific and short-term and is used to evaluate whether the project is being constructed in accordance with plans and specifications (i.e., performance of contractor, equipment, and environmental controls). This type of monitoring evaluates water quality in the vicinity of the construction operations to determine whether contaminant resuspension and dispersion are adequately controlled.

Further, bathymetric monitoring data establish actual dredge prisms or the placement location and thickness of cap material.

7.3.3 Post-construction Performance Monitoring

Post-construction performance monitoring at the conclusion of in-water construction evaluates post-removal sediment conditions in dredging or containment areas. Both chemical and physical data are collected to determine whether the work complies with project specifications.

7.3.4 Operation and Maintenance (O&M) Monitoring

O&M monitoring refers to data collection for the purpose of tracking the technology performance, long-term effectiveness, and stability of individual sediment cleanup areas.

In capping areas, O&M monitoring typically consists of analysis including COCs, grain size, TOC, and cap thickness using sediment or porewater matrices. A combination of tools, including bathymetry soundings, surface grab samples, sediment cores, diver surveys, peepers, staking, and/or settlement plates is used to evaluate cap performance. Some of these tools are also used for ENR and MNR performance monitoring.

7.3.5 Long-term Monitoring

Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action to assess risk reduction and progress toward achievement of cleanup objectives. Data collected under long-term monitoring yields information reflecting the combined actions of sediment remediation and source control.

7.3.6 Monitoring Summary

Monitoring is an essential element of remedial alternatives developed in this FS. Appendix K set forth key assumptions and an overall framework for monitoring using the process options and monitoring objectives described above. Appendix K also cross references these monitoring terms and concepts with those used in MTCA.

7.4 Ancillary Technologies

7.4.1 Barge Dewatering

Dewatering mechanically dredged sediment on transfer barges prior to additional sediment handling (e.g., off-loading and disposal) is an important interim management step. Dewatering produces a more consolidated sediment load and reduces the volume of water that would otherwise need to be managed elsewhere (e.g., at a transloading facility or at a landfill). Typically, the dewatering step occurs on a transfer barge within the dredge operations area by gravity settling and separation. In the past, the separated water was decanted directly back to the receiving water without further treatment. This confines the discharge to the area that is already seeing elevated turbidity as a result of dredge operations. Barge dewatering in this manner is typical of sediment remediation projects conducted in the Puget Sound region and this FS assumes it will be part of the remedial alternatives for costing purposes. As discussed below, more recent projects have included treatment.

Examples of Puget Sound region projects that used this technology are provided below. Each was implemented in compliance with project-specific water quality certifications.

Todd Shipyards (Seattle, Washington)

A patented (General Construction Company) sloping drain barge was used on this project. The technique involved ballasting one end of the barge with ecology blocks to create a sloping deck surface, which in turn, promotes gravity drainage to the down-slope end of the barge (Figure 7-10). The down-slope end of the barge is equipped with an overflow weir. The separated water was released directly back into the receiving water without further treatment.

Denny Way and East Waterway Phase 1 (Seattle, Washington)

For these two projects, dredged material was placed on flat-deck barges equipped with fabric-lined scuppers to allow gravity drainage of sediment. Sediment was retained in the barge, while the separated water was decanted directly back into the receiving water through the scuppers without further treatment.

Hylebos Waterway Sediment Remediation (Tacoma, Washington)

Dredged material was placed in hopper barges for gravity dewatering. Excess water from the hopper barge was decanted, treated to the water quality standards set for the project, and released back into the waterway. During the initial project phase, water treatment consisted of adding flocculants followed by routing the water through a series of weirs to enable suspended solids removal prior to discharging the water to the water body. During the final phase, a combination of flocculants and mixing tanks were used to treat the water prior to release to the water body.

Slip 4 Non-Time Critical Removal Action (Seattle, Washington)

For the recently completed Slip 4 project (one of the EAAs), a barge-based process was used that filtered the decant water through geotubes and several layers of geotextile fabric, and then drained the filtered water through granular activated carbon. While not a required element of the Slip 4 project, this step reduced turbidity in the return water. The project was completed in compliance with the water quality permit issued for the project.

7.4.2 Wastewater Treatment Associated with Sediment Remediation

Remedial alternatives that involve the removal and upland handling of contaminated sediment invariably generate wastewater that must be managed, treated, and discharged in a manner consistent with ARARs. Wastewater treatment technologies (e.g., for treatment of stormwater or industrial wastewater) are standard, myriad, and ubiquitous in their application to a wide variety of site-specific conditions. Treatment trains using conventional equipment are capable of treating water generated during sediment remediation projects to levels consistent with ARARs.

Section 8 assumes wastewater treatment would be required at a transloading facility to manage water generated from dewatering of sediments. Selection of an appropriate treatment train for this wastewater would require characterizing the wastewater properties and, potentially, conducting some treatability testing. The process options likely to be employed are expected to be standard and commercially available. For example, a common treatment train consists of gravity separation to remove suspended solids, media (e.g., sand) filtration, and adsorption on granular activated carbon for removal of dissolved organic compounds. Depending on dissolved metals concentrations, a chemical coagulation/flocculation process step might also be required. Discharge of treated water, similar to the soil-washing water treatment discharge (Section 7.1.2.2), would likely be directly back to the LDW after treatment, and would be governed by a CWA 401 water quality certification.

Discharge to the King County Metro sewer system could also be considered where the discharge meets flow (i.e., capacity) and chemical parameter limits. This approach would be an off-site disposal action, potentially requiring pretreatment to achieve discharge criteria and comply with all permit requirements (e.g., daily discharge volume, etc.), so as not to contribute to an overflow event (e.g., holding tanks for monitored flow).

7.4.3 Best Management Practices

Implementation of best management practices (BMPs) is widely considered essential to sediment cleanup projects (NRC 2007). BMPs are particularly important for environmental dredging to minimize release to the environment of contaminated material (sediment, water, debris) from the dredging footprint, and during barge transport, off-loading, and upland rehandling.

Environmental dredging to remove COCs also causes some residual sediment contamination (Palermo 2008). Contaminated sediments that are dislodged or suspended by the dredging operation are subsequently redeposited on the bottom either within or adjacent to the dredging footprint. The primary causes for this residual contamination are described in Section 7.1.1.2.

Resuspended residuals generally accumulate (settle) above the dredging cutline in thin layers, and are characterized by fine-grained sediment, being unconsolidated, having a high moisture content, and possibly existing as a fluid mud layer. The constituent COC concentrations in the residual layer can be approximated using the average dredge prism concentration (Hayes and Patmont 2004). The residual layer can be present within and adjacent to the dredge prism.

Potential BMPs to evaluate during design for dredging residuals and water quality management include:

- Remove debris prior to dredging.
- Minimize residuals generation by dredge control and design, such as carefully controlling depth, location, and cutting action to maximize sediment capture and minimize sloughing and bottom impacts. Optimize the fill efficiency of a dredge bucket to minimize both free-water capture and overfill fallback.

- Control speed of bucket through the water column to minimize loss of adhered sediment.
- Allow sediment-filled bucket to drain before fully emerging above the water surface.
- Contain drippage during the overwater swing of a filled bucket (e.g., by placing an empty barge or apron under the swing path during offloading).
- Wash bucket prior to lowering back into the water column.
- Use environmental or sealed bucket if practicable and if proper sediment conditions exist.
- Start dredging in upslope areas and move downslope to minimize sloughing.
- Plan multiple dredge cuts: limit initial cut depths to avoid sloughing of the cut bank; plan initial cut(s) to remove most of the contamination; and design a final "cleanup" cut into subsurface "clean" sediment to lower the average dredge prism COC concentrations.
- Use floating and/or absorbent booms to capture floating debris or oil sheens.
- Use conventional construction stormwater BMPs to control and reduce the silt burden in runoff from barges or rehandling areas.
- Develop and implement a post-dredging residuals monitoring and management plan.
- Monitor natural recovery of dredged area.
- Place a thin-layer sand cover (ENR) to address residuals.

The use of silt curtains around the dredging operations to reduce the transport of suspended solids is an engineering control that can be employed under certain circumstances. However, the effectiveness of a silt curtain is primarily determined by the hydrodynamic conditions at the site (usually relatively shallow, quiescent water, without significant tidal fluctuations are preferred), the quantity and type of suspended solids, the mooring method, and the characteristics of the barrier. Often, strong currents (greater than 2.5 ft/second) are problematic, and any application and deployment of silt curtains for high velocities would require special design and engineering features (USACE 2008a). In the Puget Sound region, silt curtains are not frequently used in areas where there are large tidal excursions, high-flow velocities, conflicts between dredging activities and navigation, or other technical limitations.

The specific array of BMPs or engineering controls implemented during cleanup will be location-specific and will be determined during design of the remedial alternative. Often, the remedial design specifications define certain BMPs along with performance requirements (such as water quality standards) to which the contractor must adhere. The contractor typically is required to provide additional details on specific BMPs in their work plans. Monitoring and adaptive management are common practices that will be used to refine and optimize BMPs throughout the duration of the project to ensure compliance with the project performance requirements. Representative BMPs have been identified as part of the FS remedial alternatives to develop cost estimates.

7.5 Summary of Representative Process Options for the FS

The shaded rows of Table 7-4 show the representative technology process options carried forward to Section 8 for potential development and evaluation of remedial alternatives. Consistent with CERCLA guidance, alternate process options may be considered during remedial design.

The suite of technologies and institutional controls is consistent with most of the sediment feasibility studies and cleanup projects conducted to date within the Puget Sound region and around the country. Further, it is consistent with recent deliberations and reports that have emerged from the sediment remediation community nationwide (NRC 2007). These reports conclude that a limited number of engineering approaches are available to address sediment cleanup and that some combination of dredging, disposal, capping, ENR, and MNR will invariably be at the core of almost every future major project.

Table 7-1 Initial Screening of Candidate Remedial Technologies

GRA	Technology Type	Process Option	Description
No action	None	Not applicable	No active remedy or monitoring.
Institutional controls	Proprietary controls and informational devices (EPA 2000)	Proprietary controls	Mechanisms in deeds or other instruments transferring property that restrict or affect the use of property
		Seafood Consumption Advisories, Education and Public Outreach	Public advisories that consumption of resident LDW fish and shellfish (and sediment contact) may present health risks.
		Monitoring and notification of waterway users	Regulatory constraints on uses such as vessel wakes, anchoring, and dredging. Physical constraints, such as fencing and signs, placed on property access points that limit human access to areas that pose a health risk.
		Enforcement Tools	Agency consent decrees or orders overseeing implementation of institutional controls and monitoring. Restrictive Navigation Areas, per Coast Guard formal rulemaking, could be an enforceable means of protecting containment remedies and other areas from anchoring and other physical interference, particularly where UECA covenants or other proprietary controls may not be achievable.
		Site Registry	Placement and maintenance of site information on the State Registry (Ecology's Hazardous Sites list and Site Register) would provide information regarding restrictions on the property.
Monitoring	Physical and chemical assessment	Baseline Monitoring	Establishes a statistical basis for comparing site conditions before, during, and after the cleanup action.
		Construction Monitoring	Short-term monitoring during remediation used to evaluate whether the project is being constructed in accordance with specifications (i.e., water quality monitoring, bathymetric surveys, discharge monitoring, inspection surveys, sediment monitoring).
		Post-construction Performance Monitoring	Post-construction performance monitoring evaluates post-removal surface and subsurface sediment conditions in dredging or containment areas to confirm compliance with project specifications.
		Long-term Operation and Maintenance Monitoring	Long-term operation and maintenance monitoring of dredging areas, containment, and/or disposal sites (i.e., CAD sites, ENR, and capping areas) required to ensure long-term effectiveness and continued stability of the structure.
		Long-term Monitoring	Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action.
Monitored natural recovery	Chemical/physical transport and degradation	Combination	Desorption, dispersion, diffusion, dilution, volatilization, resuspension, and transport.
	Biological degradation	COC metabolism	Chlorine atoms are removed from PCB molecules by bacteria; however, toxicity reduction is not directly correlated to the degree of dechlorination. PAHs may be partially or completely degraded.
	Physical-burial processes	Sedimentation	Contaminated sediments are buried (by naturally occurring sediment deposition) to deeper intervals that are less biologically available. (Resuspension and transport are minor components of MNR.)

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Enhanced natural recovery	Thin-layer placement	Placement of thin layer to augment natural recovery	Application of a thin layer of clean sand and natural resorting, sedimentation, or bioturbation to mix the contaminated and clean sediments, resulting in acceptable contaminant concentrations.
Containment	Capping	Conventional sand cap	Placement of clean sand over existing contaminated bottom to physically isolate contaminants.
		Conventional sediment / clay cap	Use of dredged fine-grained sediments or commercially obtained clay materials to achieve contaminant isolation.
		Armored cap	Coarse granular material such as: cobbles, pebbles, or larger material are incorporated into the cap to prevent erosion in high-energy environments or to prevent cap breaching by bioturbators (example: membrane gabions).
		Composite cap	Soil, media, and geotextile cap placed over contaminated material to inhibit migration of contaminated pore water and/or inhibit bioturbators.
		Spray cap	Placement of capping materials (usually concrete) by spraying concrete or mortar from a nozzle at high velocity onto a surface via pressure hoses with either a dry or wet mix process.
		Reactive cap	Incorporation of materials such as granular activated carbon or iron filings to provide chemical binding or destruction of contaminants migrating in porewater.
Removal	Dredging	Hydraulic dredging	Hydraulic dredges use a cutter head, and suction provided by an on-board pump(s) to agitate, entrain, and hydraulically transport sediment via pipeline to a land-based sediment handling facility or slurry discharge location.
		Mechanical dredging	A barge-mounted floating crane on a derrick barge maneuvers a dredging bucket. The bucket is lowered into the sediment; when the bucket is withdrawn, the jaws of the bucket are closed, retaining the dredged material.
		Mechanical dredging (excavator)	Excavator dredges use a barge-mounted excavator with fixed arm linkages (boom and stick), instead of cables, to position the clamshell bucket at the target elevation for sediment removal.
	Excavating	Dry excavation	Sediment is removed by upland-based conventional excavation (backhoe) equipment. Removal during low tides may not require sheet-pile walls or cofferdams. This removal option may include erecting sheet-pile walls or a cofferdam around the contaminated sediments to dewater.
In Situ treatment	Biological*	*In situ* slurry biodegradation*	Anaerobic, aerobic, or sequential anaerobic/aerobic degradation of organic compounds with indigenous or exogenous microorganisms. Oxygen, nutrients, and pH are controlled to enhance degradation. Requires sheet piling around entire area and slurry treatment performed using aerators and possibly mixers.
		In situ aerobic biodegradation*	Aerobic degradation of sediment *in situ* with the injection of aerobic biphenyl enrichments or other co-metabolites. Oxygen, nutrients, and pH are controlled to enhance degradation.
		In situ anaerobic biodegradation*	Anaerobic degradation *in situ* with the injection of a methanogenic culture, anaerobic mineral medium, and routine supplements of glucose to maintain methanogenic activity. Nutrients and pH are controlled to enhance degradation.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
In Situ treatment (cont)	Biological	Imbiber Beads™*	A "cover blanket" of Imbiber Beads™ placed over contaminated sediments to enhance anaerobic microbial degradation processes and allow exchange of gases between sediments and surface water. The beads are spherical plastic particles that would adsorb PCB vapors generated.
	Chemical*	Aqua MecTool™ oxidation*	A caisson (18' by 18') is driven into the sediment and a rotary blade is used to mix sediment and add oxidizing agents such as ozone, peroxide, or Fenton's reagent. A bladder is placed in the caisson to reduce TSS and the vapors may be collected at the surface and treated.
		In situ oxidation*	Oxidation of organics using oxidizing agents such as ozone, peroxide, or Fenton's reagent.
		Electro-chemical oxidation*	Proprietary technology in which an array of single steel piles is installed and low current is applied to stimulate oxidation of organics.
	Physical-extractive processes*	Sediment flushing*	Water or other aqueous solution is circulated through contaminated sediment. An injection or infiltration process introduces the solution to the contaminated area and the solution is later extracted along with dissolved contaminants. Extraction fluid must be treated and is often recycled.
		In situ slurry oxidation*	An array of injection wells is used to introduce oxidizing agents such as ozone to degrade organics.
	Phy*sical-immo*bilization	Aqua MecTool™ stabilization*	A caisson (18' by 18') is driven into the sediment and a rotary blade is used to mix sediment and add stabilizing agents. A bladder is placed in the caisson to reduce TSS and the vapors may be collected at the surface and treated.
		Vitrification*	Uses an electric current *in situ* to melt sediment or other earthen materials at extremely high temperatures (2,900-3,650 °F). Inorganic compounds are incorporated into the vitrified glass and crystalline mass and organic pollutants are destroyed by pyrolysis. *In situ* applications use graphite electrodes to heat sediment.
		Ground freezing*	An array of pipes is placed *in situ* and brine at a temperature of -20 to -40°C is circulated to freeze soil. Recommended only for short duration applications and to assist with excavation.
		Activated Carbon Amendment **	Activated carbon (powder, granules, or pellets) serves as an amendment to the bioactive surface layer of sediment. Hydrophobic organic contaminants adsorb to activated carbon particles, reducing porewater contaminant concentrations and bioavailability for uptake by organisms.
		Organoclay Amendment **	Organoclay products for use in sediment remediation consist of mineral clay, polymer additives, and an aggregate core for densification. Organoclays bind contaminants through replacement of metal ions with amines or other functional groups, physically isolate the contaminated sediment from receptors (because of low permeability), and stabilize sediment by preventing resuspension and transport of contaminants.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Ex Situ treatment	Biological*	Landfarming/ Composting*	Sediment is mixed with amendments and placed on a treatment area that typically includes leachate collection. The soil and amendments are mixed using conventional tilling equipment or other means to provide aeration. Moisture, heat, nutrients, oxygen, and pH can be controlled to enhance biodegradation. Other organic amendments such as wood chips, potato waste, or alfalfa are added to composting systems.
		Biopiles*	Excavated sediments are mixed with amendments and placed in aboveground enclosures. This is an aerated static pile composting process in which compost is formed into piles and aerated with blowers or vacuum pumps. Moisture, heat, nutrients, oxygen, and pH can be controlled to enhance biodegradation.
		Fungal biodegradation*	Fungal biodegradation refers to the degradation of a wide variety of organopollutants by using fungal lignin-degrading or wood-rotting enzyme systems (example: white rot fungus).
		Slurry-phase biological treatment*	An aqueous slurry is created by combining sediment with water and other additives. The slurry is mixed to keep solids suspended and microorganisms in contact with the contaminants. Upon completion of the process, the slurry is dewatered and the treated sediment is removed for disposal (example: sequential anaerobic/aerobic slurry-phase bioreactors).
		Enhanced biodegradation*	Addition of nutrients (oxygen, minerals, etc.) to the sediment to improve the rate of natural biodegradation. Use of heat to break carbon-halogen bonds and to volatilize light organic compounds (example: D-Plus [Sinre/DRAT]).
	Chemical*	Acid extraction*	Contaminated sediment and acid extractant are mixed in an extractor, dissolving the contaminants. The extracted solution is then placed in a separator, where the contaminants and extractant are separated for treatment and further use.
		Solvent extraction(s)*	Contaminated sediment and solvent extractant are mixed in an extractor, dissolving the contaminants. The extracted solution is then placed in a separator, where the contaminants and extractant are separated for treatment and further use (example: B.E.S.T.™ and propane extraction process).
	Chemical/ Physical	Reduction/ Oxidation*	Reduction/oxidation chemically converts hazardous contaminants to nonhazardous or less toxic compounds that are more stable, less mobile, and/or inert. The oxidizing agents most commonly used are hypochlorites, chlorine, and chlorine dioxide.
		Slurry oxidation*	The same as slurry-phase biological treatment with the exception that oxidizing agents are added to decompose organics. Oxidizing agents may include ozone, hydrogen peroxide, and Fenton's reagent.
		Dehalogenation*	Dehalogenation process in which sediment is screened, processed with a crusher and pug mill, and mixed with sodium bicarbonate (base catalyzed decomposition) or potassium polyethylene glycol. The mixture is heated to above 630 °F in a rotary reactor to decompose and volatilize contaminants. Process produces biphenyls, olefins, and sodium chloride.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Ex Situ treatment (cont)	Chemical/ Physical (cont)	Soil washing	Contaminants sorbed onto fine soil particles are separated from bulk soil in an aqueous-based system on the basis of particle size. The wash water may be augmented with a basic leaching agent, surfactant, pH adjustment, or chelating agent to help remove organics and heavy metals.
		Radiolytic dechlorination*	Sediment is placed in alkaline isopropanol solution and gamma irradiated. Products of this dechlorination process are biphenyl, acetone, and inorganic chloride. Process must be carried out under inert atmosphere.
	Physical	Particle Separation	Contaminated fractions of solids are concentrated through gravity, magnetic, or sieving separation processes.
		Solar detoxification*	Through photochemical and thermal reactions, the ultraviolet energy in sunlight destroys contaminants.
		Solidification	The mobility of constituents in a “solid” medium is reduced through addition of immobilization additives.
	Thermal	Incineration*	Temperatures greater than 1,400°F are used to volatilize and combust organic contaminants. Commercial incinerator designs are rotary kilns equipped with an afterburner, a quench, and an air pollution control system.
		High Temperature Thermal Desorption*	Temperatures in the range of 600-1,200°F are used to volatilize organic contaminants. These thermal units are typically equipped with an afterburner and baghouse for destruction of air emissions. Wastes are heated to volatilize water and organic contaminants. A carrier gas or vacuum system transports volatilized water and organics to the gas treatment system (examples: X*TRAX™, DAVES, Tacuik Process, and Holoflite™ Dryer).
		Low Temperature Thermal Desorption*	Temperatures in the range of 200-600°F are used to volatilize and combust organic contaminants. These thermal units are typically equipped with an afterburner and baghouse for treatment of air emissions.
	Thermal (cont)	Pyrolysis*	Chemical decomposition is induced in organic materials by heat in the absence of oxygen. Organic materials are transformed into gaseous components and a solid residue (coke) containing fixed carbon and ash.
		Vitrification*	Current technology uses oxy-fuels to melt soil or sediment materials at extremely high temperatures (2,900-3,650°F).
		High-pressure oxidation*	High temperature and pressure are used to break down organic compounds. Operating temperatures range from 150-600°C and pressures range from 2,000-22,300 MPa (examples: wet air oxidation and supercritical water oxidation).

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Disposal	On-site disposal	Level-bottom cap*	Relocation of contaminated sediment to discrete area and capping with a layer of clean sediments. Provides similar protection as capping, but requires substantially more sediment handling that may cause increased releases to surface water.
		Contained Aquatic Disposal (CAD)	Untreated sediment is placed within a lateral containment structure (i.e., bottom depression or subaqueous berm) and capped with clean sediment.
		Confined Disposal Facility (CDF)	Untreated sediment is placed in a nearshore CDF that is separated from the river by an earthen berm or other physical barrier and capped to prevent contact. A CDF may be designed for habitat purposes.
	Off-site Disposal	Subtitle D landfill	Off-site disposal at a licensed commercial facility that can accept nonhazardous sediment. Regional landfills can accept both dewatered and wet sediments.
		Subtitle C landfill	Off-site disposal at a licensed commercial facility that can accept hazardous dewatered sediment removed from dredging or excavation. Dewatering required to reduce water content for transportation.
		TSCA-licensed landfill*	Off-site disposal at a licensed commercial facility that can accept TSCA sediment. Dewatering required to reduce water content for transportation.
		DMMP open water non-treated (if acceptable) disposal	Treated or separated sediment is placed at the Elliott Bay DMMP disposal site. Requires that the placed sediment be at, or below, DMMP disposal criteria for priority pollutants and potentially bioaccumulative contaminants.
		Upland MTCA confined fill (commercial/industrial – beneficial use)*	Treated or untreated sediment is placed at an off-site location. Requires that sediment be at, or treated to, MTCA cleanup levels at an off-site location and meet nondegradation standards. Location may require cap or other containment devices based on analytical data.
		Upland MTCA fill (residential/clean – beneficial use)	Treated or untreated sediment is placed at an off-site location. Requires that sediment be at, or treated to, a concentration at or below MTCA cleanup levels for unrestricted land use and meet nondegradation standards.
		In-water beneficial use	Sediments treated to below DMMP guidelines may be beneficially reused for habitat creation, capping, or residual management.

Notes:

Shaded technologies and process options are retained at end of initial screening as potentially feasible at the end of the Table 7-2 series, where more detailed screening information is provided. These process options were retained at the conclusion of the detailed screening and are evaluated in Table 7-3 for applicability in the LDW with the exception of institutional controls, which do not lend themselves to comparison on the same terms as other technologies. Institutional controls are discussed only within Section 7.2 and are not included in Tables 7-2 and 7-3.

A detailed description of these process options is not included in the FS text. Details regarding these technology and process options are provided in the document *Identification of Candidate Cleanup Technologies for the Lower Duwamish Waterway* prepared by The RETEC Group Inc. (2005). These process options were eliminated in the detailed screening process shown in Table 7-2 series. The *in situ* treatment (activated carbon and organoclay amendments) have been added to this table as a result of recent advances in these technologies and project case studies now available for review.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; DMMP = Dredged Material Management Program; ENR = enhanced natural recovery; GRA = general response action; MTCA = Model Toxics Control Act; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; TSCA = Toxic Substances Control Act; TSS = total suspended solids; UECA = Uniform Environmental Covenants Act

Table 7-2a Detailed Screening of Process Options: No Action, Institutional Controls, and Monitoring

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Screening Decision	Cost
No Action	None	Not Applicable	—	Retained per NCP requirement	Technically implementable for conditions within the LDW.	—	—	Retained per NCP requirement	Low
Institutional Controls				All retained				All retained	Low
Monitoring	Physical and Chemical Assessment	Baseline Monitoring	Can be effective for evaluating changes.	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate
Monitoring	Physical and Chemical Assessment	Construction Monitoring	Can be effective for evaluating changes.	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Low
Monitoring	Physical and Chemical Assessment	Post-construction Performance Monitoring	Can be effective for evaluating changes	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Low
Monitoring	Physical and Chemical Assessment	Operation and Maintenance Monitoring	Can be effective for evaluation and maintenance of LDW following remedial actions	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate
Monitoring	Physical and Chemical Assessment	Long-term Monitoring	Can be effective for evaluating sediment, tissue and water quality over an extended period of time following remedial actions	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate to High

Note:
COC = contaminant of concern; CTM = Candidate Technologies Memo; FS = feasibility study; GRA = general response action; NCP = National Contingency Plan

Table 7-2b Detailed Screening of Process Options: Monitored Natural Recovery and Enhanced Natural Recovery

GRA	Technology Type	Process Option	Effectiveness Screening: LDW COCs	Effectiveness Screening: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Implementability: Screening Decision	Cost
Monitored Natural Recovery	Chemical Degradation	Natural Désorption, Diffusion, Dilution, Volatilisation	Potentially effective for immobilizing COCs through TOC or sulfide sorption.	Retained for further evaluation	Technically implementable for conditions within the LDW	—	—	Retained for further evaluation in the FS	Low
Monitored Natural Recovery	Biological Degradation	COC Metabolism (aerobic and anaerobic)	Effective for SVOCs and PAHs but does not result in complete destruction of PCBs or TBT in acceptable time frame. Not applicable to metals.	Retained for further evaluation	Technically implementable for conditions within the LDW	—	—	Retained for further evaluation in the FS for SVOCs only	Low
Monitored Natural Recovery	Physical/Burial Processes	Natural Sedimentation and Burial (resuspension and transport are minor components of MNR)	Potentially effective for LDW COCs via deposition and reburial. Requires demonstration of long-term deposition and burial.	Retained for further evaluation	Technically implementable for conditions within the LDW	Preliminary results at some projects show some success.	—	Retained for further evaluation in the FS	Low
Enhanced Natural Recovery	Thin-layer Placement	Thin-layer Placement	Effective for all LDW COCs. Applicable: 1) at areas where MNR processes are demonstrated, but faster recovery is required; or 2) as a residual management tool after completion of a removal action.	Retained for further evaluation	Technically implementable for conditions within the LDW.	Thin-layer placements for ENR and residuals management have been applied in multiple locations in Puget Sound and nationally.	—	Retained for further evaluation in the FS	Low to Moderate

Note:

CTM = Candidate Technologies Memorandum; COC = contaminant of concern; ENR = enhanced natural recovery; FS = feasibility study; GRA = general response action; MNR = monitored natural recovery; PAHA = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; SVOC = semivolatile organic compound; TOC = total organic carbon; TBT = tributyltin

Table 7-2c Detailed Screening of Process Options: Containment Process Options

GRA	Technology Type	Process Option	Effectiveness		Implementability				Cost
			LDW COCs	Screening Decision	Site Conditions	Available and Demonstrated	Innovative Technology	Screening Decision	
Containment	Capping	Conventional Sand Cap	Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained for consideration throughout the LDW	Applicable to LDW conditions. Easily applied *in situ;* however, scouring must be considered. Decreased water depth may limit future uses of waterway and may impact flooding, stream bank erosion, navigation, and recreation.	Conventional sand caps have been applied in multiple locations in Puget Sound and nationally.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Conventional Sediment/Clay Cap	Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact. Sediment with silt and clay is effective in limiting diffusion of contaminants. Sediment caps are generally more effective than sand caps for containment of contaminants with high solubility and low sorption	Retained for consideration throughout the LDW	Generally applicable to LDW conditions. Placement of clay caps is considered in shallow water depth areas where minimal cap thickness is required. Special engineering controls will be needed to place clay cap in the LDW.	Conventional sediment caps using river-dredged sediments have been applied in multiple locations in Puget Sound and nationally. Application of clay caps is relatively new, but demonstrated.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Armored Cap	Applicable to LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained for limited use in high-energy sections of the LDW	Applicable to areas of LDW where increased velocities from river flow, or potential scouring associated with propeller wash might be expected. Decreased water depth may limit future uses of waterway and may impact flooding, stream bank erosion, navigation, and recreation. Limited use in intertidal areas that support clamming and recreational activities.	Armored caps have been implemented at several sites in Puget Sound and nationally.	—	Retained for limited use in the FS for high-energy sections of the LDW.	Low to Moderate
		Composite Cap (geotextile, HDPE)	Effective for LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants. Can be used: 1) to limit cap thickness, 2) for low solids underlying sediments where additional floor-support is required, 3) as a bioturbation barrier, or 4) as a barrier for areas where methane generation may be an issue.	Retained for consideration throughout the LDW	Applicable to LDW site conditions. Application must consider that decreased water depth may limit future uses of waterway and impact flooding, stream bank erosion, navigation, and recreation. Limited use in intertidal areas that support clamming and recreational activities.	Application of composite capping is relatively new, but commercially demonstrated for projects with similar size and scope.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Spray Cap	Confines COCs by encapsulating with shotcrete (usually concrete) placed over underlying surface.	Retained for consideration throughout the LDW	Applicable to hard to access areas under piers and wharves. Shotcrete cap reduces the habitat value of the intertidal sediment bed.	Shotcrete was used at the Todd Shipyards effectively encapsulating existing debris (slag) mounds under dock structures from the aquatic environment.	Demonstrated effective at recent Puget Sound region remediation project.	Retained for consideration in the FS for application in hard to access areas under piers or wharf structures.	Low to Moderate
		Reactive Cap	Effective for LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained	Reactive caps may be applicable to site conditions on the LDW. Limited use in intertidal areas that support clamming and recreational activities.	Addition of materials to increase sorptive capacity of cap has been implemented in Puget Sound. Long-term effectiveness data may be available during the LDW FS.	Reactive capping is an innovative technology that is in the demonstration phase on the Anacostia River. Results of those tests are expected during the LDW FS.	Retained for consideration in the FS as an innovative technology.	Low to Moderate

Notes:

COC = contaminant of concern; FS = feasibility study; GRA = general response action; HDPE = High-density polyethylene

Table 7-2d Detailed Screening of Process Options: Treatment Process Options

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
In Situ Treatment	Biological	*In Situ* Slurry Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		In Situ Aerobic Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		In Situ Anaerobic Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		Imbiber Beads™	Potentially applicable to PCBs and SVOCs, not metals. No data on effectiveness with TBT. Not demonstrated for remediation of sediments. Removal and disposal of the blanket is not demonstrated.	Eliminated	—	—	—	—	—
	Chemical	Aqua MecTool™ Oxidation	Technology is effective for PCBs, SVOCs in soils. Process should be effective for TBT, but not metals.	Retained for further consideration	Could be applicable to conditions in LDW. Requires treating sediments in place using caisson and proprietary injectors.	Not demonstrated in pilot- or full-scale sediment projects. Technical difficulties in field trials injecting high air flows into caisson with standing water while preventing generation of TSS.	Not considered innovative or available during LDW FS.	Eliminated	—
		In Situ Oxidation	Has not been demonstrated to be effective for LDW COCs in sediments.	Eliminated	—	—	—	—	—
		Electro-chemical Oxidation	Applicability for use in water is not known. No demonstrated sediment application.	Eliminated	—	—	—	—	—
	Physical-Extractive Processes	Sediment Flushing	Bench scale effectiveness for all LDW COCs.	Retained for further consideration	Potentially applicable to LDW. Requires in-water steel piling around treatment area and extensive water quality monitoring outside piles.	No known pilot or full-scale applications.	Not considered innovative or available during LDW FS.	Eliminated	—
		In Situ Slurry Oxidation	Not demonstrated in full-scale applications effective for LDW COCs. Requires in-water steel piling around treatment area and extensive water quality monitoring outside piles.	Eliminated	—	—	—	—	—
	Physical-Immobilization	Aqua MecTool™ Stabilization	Proprietary technology that has been effective in stabilizing metals, PCBs and SVOCs in soil. No data available on TBT, but physical process likely to be effective on butyltins.	Retained for further consideration	Could be applicable to conditions in LDW. Requires treating sediments in place using caisson and proprietary injectors.	Proprietary technology that was tested in a pilot-scale application in Wisconsin with coal tar-contaminated sediments, and found to be not implementable. Previous trials with this technology created water treatment problems inside the caisson.	Not considered innovative or available during LDW FS.	Eliminated	—
		Activated Carbon Amendment	Effective at adsorbing organic contaminants in sediment applications. Pilot studies (in five pilot-scale demonstration projects in the United States and Norway) and research indicates technology has promising long-term effectiveness. Carbon-amended sediment provides a suitable habitat for benthic communities.	Retained for further consideration	Potentially applicable to LDW. Easily applied *in situ*; may require armoring in scour areas.	Demonstrated effective in recent pilot-scale remediation projects (San Francisco-CA, Lower Grasse River-NY, Canal Creek-MD, and Trondheim-Norway) in various aquatic environments (tidal mudflat, freshwater river, marine harbor, deep-water fjord, tidal creek, and marsh).	Activated carbon amendment is considered innovative and available during the LDW FS.	Retained for consideration in the FS	Low to Moderate
		Organoclay Amendment	Effective at adsorbing organic contaminants in sediment applications. Long-term effectiveness shown in pilot-scale demonstration projects in Anacostia River (Washington, DC).	Retained for further consideration	Potentially applicable to LDW. Easily applied *in situ*; may require armoring in scour areas.	Demonstrated effective at the Anacostia River in a recent pilot-scale remediation project that used AquaBlok® (proprietary clay polymer composite).	Organoclay amendment is considered innovative and available during the LDW FS.	Retained for consideration in the FS	Low to Moderate

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
In Situ Treatment	Physical-Immobilization	Vitrification	Effective at stabilizing COCs in soil applications, but requires less than 60% water content. Remaining sediment surface may not provide suitable habitat. No known sediment applications.	Eliminated	—	—	—	—	—
		Ground Freezing	Not permanently effective for LDW COCs. Long-term effectiveness in presence of standing water has not been demonstrated. Standing water likely provides a significant sink for cold temperatures and would substantially increase cost.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Landfarming/ Composting	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation.	Eliminated	—	—	—	—	—
		Biopiles	Not effective for metals, PCBs, dioxin or TBT. Used for reducing concentrations of petroleum constituents in soils. Applied to treatment of nonhalogenated VOCs and fuel hydrocarbons. Requires large upland area.	Eliminated	—	—	—	—	—
		Fungal Biodegradation	Not effective for metals, PCBs, dioxins or TBT. No known full-scale applications. High concentrations of contaminants may inhibit growth. The technology has been tested only at bench scale.	Eliminated	—	—	—	—	—
		Slurry-phase Biological Treatment	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation. Large volume of tankage required. No known full-scale applications.	Eliminated	—	—	—	—	—
		Enhanced Biodegradation	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation.	Eliminated	—	—	—	—	—
	Chemical	Acid Extraction	Suitable for sediments contaminated with metals, but not applicable to PCBs or SVOCs. No data on TBT.	Eliminated	—	—	—	—	—
		Solvent Extraction	Potentially effective for treating sediments containing PCBs, dioxins, or SVOCs. Not applicable to metals. No data on TBT. Extraction of organically-bound metals and organic contaminants creating residuals with special handling requirements. At least one commercial unit available.	Retained for further consideration	Potentially applicable to dewatered (dry) sediments on the LDW containing primarily organic contaminants such as PCBs. Extracted organic contaminants from the process will need to be treated or disposed. Requires pre-treatment that involves screening of sediments.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale.	This technology has been used to demonstrate under the EPA SITE program, but there are no data for similar implementation of this technology for large-scale PCB-impacted sediment. No current or planned projects.	Eliminated	—
		Solvent extraction: Solvent Electron Technology (SET™)	Effective for SVOCs and PCBs, but not metals. No data on TBT. Full scale system commercially available for treatment. Mobile units can be set up to meet project requirements. Nationwide TSCA treatment permit for SET™ issued by EPA for mobile PCB chemical destruction in soils.	Retained for further consideration	Potentially applicable to dewatered (dry) sediments on the LDW. This technology results in destruction of PCBs and other organic contaminants. Operates on a closed loop system and does not produce secondary hazardous waste or off-gas.	Not demonstrated in pilot- or full-scale sediment projects.	—	Eliminated	—
		Solvent Extraction; Peroxide and Ferrous Iron Treatment	Oxidation using liquid hydrogen peroxide (H_2O_2) in the presence of native or supplemental ferrous iron (Fe^{+2}) produces Fenton's Reagent which yields free hydroxyl radicals (OH^{-}). These strong, nonspecific oxidants can rapidly degrade various organic contaminants.	Retained for further consideration	Potentially applicable to LDW.	Technology is neither commercially available nor demonstrated on a project of similar size and scope.	This technology has been used for pilot studies for treating PAH-impacted sediment from Utica Harbor, but there are no data for similar implementation of this technology for PCB-impacted sediment. No current or planned projects.	Eliminated	—

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
Ex Situ Treatment (continued)	Chemical (continued)	Solvent Extraction: High Energy Electron Beam Irradiation	Full-scale system commercially available for treatment of PCBs and SVOCs, and process is limited to slurried soils, sediments, and sludges. Slurrying is a required pre-treatment for this technology. Not demonstrated to be effective in sediments. Pilot-scale testing has been performed to treat wastewaters with organic compounds. Metals are not amenable to treatment. No data on TBT.	Retained for further consideration	Potentially applicable to slurried sediments in the LDW consisting primarily of organic contaminants such as PCBs.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale.	This technology demonstrated under the EPA SITE program to treat wastewater with organic compounds, but no data for similar implementations are available for PCB-impacted sediment. No current/planned projects.	Eliminated	—
		Reduction/ Oxidation	Target contaminant group for chemical redox is inorganics. Less effective for nonhalogenated VOCs, SVOCs, fuel hydrocarbons, and pesticides. Not cost-effective for high contaminant concentrations because of large amounts of oxidizing agent required.	Eliminated	—	—	—	—	—
		Dehalogenation	PCB and dioxin-specific technology. Generates secondary waste streams of air, water, and sludge. Similar to thermal desorption, but more expensive. Solids content above 80% is preferred. Technology is not applicable to metals.	Eliminated	—	—	—	—	—
		Slurry Oxidation	Applicable to SVOCs, but not PCBs or metals. TBT treatment unknown. Large volume of tankage required. No known full-scale applications. High organic carbon content in sediment will increase volume of reagent and cost.	Eliminated	—	—	—	—	—
		Soil Washing with Air Stripping	Full-scale testing of Biogenesis™ Advanced washing process showed demonstrated effectiveness for metals, SVOCs and PCBs in sediments. Limited data suggests not effective for TBT. High recalcitrant (e.g., PCB) contaminant concentration, increased percentage of fines, and high organic content increases overall treatment costs.	Retained for further consideration	Potentially applicable to dewatered sediments on the LDW. Would require upland processing space, storage capacity for dredged sediments, wastewater treatment, and discharge. Treated residuals would still require disposal.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale. Tests to date have been on 15,000 cy.	Full-scale testing has been performed. Mobile units available for setup. Continuous flow process designed to process up to 40 cy of sediments per hour for the full-scale system.	Retained as innovative technology to consider further in the FS.	Moderate to High
		Radiolytic Dechlorination	Only bench-scale testing has been performed. Difficult and expensive to create inert atmosphere for full-scale project.	Eliminated	—	—	—	—	—
	Physical	Particle Separation	Reduces volumes of COCs by separating sand from fine-grained sediments. Some bench scale testing has suggested that at high PCB concentrations, the sand fraction retains levels that still require landfilling.	Retained for further consideration	Potentially applicable dredged sediments in the LDW.	Separation technologies available and have been used in several programs of similar size and scope.	—	Retained to consider further in the FS.	Low
		Solar Detoxification	The target contaminant group is VOCs, SVOCs, solvents, pesticides, and dyes. Not effective for PCBs, dioxins or TBT. Some heavy metals may be removed. Only effective during daytime with normal intensity of sunlight. The process has been successfully demonstrated at pilot scale.	Eliminated	—	—	—	—	—
		Solidification	Bench-scale studies have added immobilizing reagents ranging from Portland cement to lime cement, kiln dust, pozzolan, and proprietary agents with varying success. Dependent on sediment characteristics and water content. Lime is particularly effective at volatilizing PCBs in wet sediment (by a phase transfer mechanism).	Retained for further consideration	Potentially applicable to LDW.	Lime has been successfully added to dredged material at other projects. Considered for use during the dewatering operation to remove excess water and prepare material for disposal.	—	Retained to consider further in the FS.	Moderate

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
Ex situ Treatment (continued)	Thermal	Incineration	High temperatures result in generally complete decomposition of PCBs and other organic contaminants. Effective across wide range of sediment characteristics but fine grained sediment difficult to treat. Not effective for metals.	Retained for further consideration	Technically applicable to LDW site conditions. Especially effective and potentially required where COCs exceed TSCA limits (e.g., PCB >50 ppm). Only a small portion of LDW sediments are above TSCA.	Only one off-site fixed facility incinerator is permitted to burn PCBs and dioxins. Metals not amenable to incineration. No data on TBT, but should be effective. Mobile incinerators are available for movement to a fixed location in close proximity to the contaminated sediments.	—	Eliminated	—
		High-temperature Thermal Desorption (HTTD) then Destruction	Target contaminants for HTTD are SVOCs, PAHs, PCBs, TBT and pesticides, which are destroyed by the heating process. Metals not destroyed.	Retained for further consideration	Technically applicable to LDW site conditions. Especially effective and potentially required where COCs exceed TSCA limits (e.g., PCB >50 ppm).	Technology readily available as mobile units that would need to be set up at a fixed location in close proximity to the contaminated sediments. Cement-Lock® Technology demonstration projects partially destroyed organics and encapsulated metals in the product matrix. The Cement-Lock® product passes the TCLP test for priority pollutants.	Cement-Lock® Technology -Two demonstration projects started. Both experienced equipment related problems and were shut down.	Eliminated	—
		Low-temperature Thermal Desorption (LTTD)	Target contaminants for LTTD are SVOCs and PAHs. May have limited effectiveness for PCBs. Metals not destroyed. Fine-grained sediment and high moisture content will increase retention times. Widely-available commercial technology for both on-site and off-site applications. Acid scrubber will be added to treat off-gas.	Retained for further consideration	Potentially applicable to LDW.	Demonstrated effectiveness at several other sediment remediation sites. Vaporized organic contaminants that are captured and condensed need to be destroyed by another technology. The resulting water stream from the condensation process may require further treatment.	—	Eliminated	—
		Pyrolysis	High moisture content increases treatment cost. Generates air and coke waste streams. Target contaminant groups are SVOCs and pesticides. It is not effective in either destroying or physically separating inorganics from the contaminated medium	Eliminated	—	—	—	—	—
		Vitrification	Thermally treats PCBs, SVOCs, TBT, and stabilizes metals. Successful bench-scale application to treating contaminated sediments in Lower Fox River, and in Passaic River.	Retained for further consideration	Potentially applicable to LDW.	Not commercially available or applied on similar site and scale.	No known pilot or full-scale applications in sediments planned.	Eliminated	—
		High-pressure Oxidation	Predominantly for aqueous-phase contaminants. Wet air oxidation is a commercially-proven technology for municipal wastewater sludges and destruction of PCBs is poor. Supercritical water oxidation has demonstrated success for PCB destruction.	Eliminated	—	—	—	—	—

Notes:

1. Costs indicated here are relative to incineration costs.
2. Institutional controls are retained as potentially feasible and applicable to the LDW, and carried forward in the detailed screening; however, they do not lend themselves to comparison on the same terms as other technologies. Therefore, they are discussed only within Section 7.2 and are not included in Tables 7-2 and 7-3.

COC = contaminant of concern; CTM = Candidate Technologies Memorandum; cy = cubic yards; EPA = U.S. Environmental Protection Agency; FS = feasibility study; GRA = general response action; HTTD = high-temperature thermal desorption; LTTD = low-temperature thermal desorption; MNR = monitored natural recovery; NCP = National Contingency Plan; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; SET™ = Sediment Electron Technology; SVOC = semivolatile organic compound; TBT = tributyltin; TCLP = Toxicity Characteristic Leaching Procedure; TOC = total organic carbon; TSCA = Toxic Substances Control Act; TSS = total suspended solids; VOC = volatile organic compound

Table 7-2e Detailed Screening of Process Options: Removal Process Options

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Implementability: Screening Decision	Cost
Removal	Dredging	Hydraulic Dredging	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Best suited to low density, high water solids with little debris. Requires nearshore dewatering facilities and right-of-way for slurry pipeline. Water treatment and disposal required.	Hydraulic environmental dredging is available and demonstrated in similar size projects, but is less frequently used for projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
		Mechanical Dredging	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Better suited for higher density, low water solids, and more effective at handling debris. Environmental buckets suitable for softer materials with low debris; clamshell buckets suitable for harder, dense sediments.	Mechanical environmental dredging is available and demonstrated in similar size projects, and is commonly employed for projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
		Mechanical Dredging (Excavator)	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Better suited for higher density, low water solids, and more effective at handling debris. Environmental excavators are suited for all materials (soft and dense), better able to handle debris, but may be depth limited.	In-water excavators are available and demonstrated in similar size projects, including projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
	Dry Excavation	On-land or Intertidal excavator, backhoes, specialty equipment	Applicable to all LDW COCs. Effective for nearshore and/or intertidal areas where depths limit conventional dredging equipment	Retained for further consideration for intertidal or nearshore areas in the LDW	Limited in application to nearshore shallow and/or intertidal areas that can be reached from shore or by specialty equipment designed to work on soft unconsolidated sediments.	Equipment is commercially available and has been applied on projects of similar scope in Puget Sound.	—	Retained for consideration in the FS for shallow and/or intertidal areas of the LDW.	Moderate

Note:
COC = contaminant of concern; FS = feasibility study; GRA = general response action

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
No Action	None	Required by NCP	Applicable to all LDW COCs.	Applicable to all COCs. Effective where risk assessment demonstrates low to no risk to human health and environment.	COCs remain in place.	Applicable throughout LDW where COC concentrations are low.	1) Readily implemented with no construction or monitoring requirements; 2) Minimal impact on industrial and shipping uses of waterway.	1) Requires source controls to be in place.	Low
Monitoring	Physical and Chemical Assessment	Monitoring	Applicable to all LDW COCs.	Can be effective for evaluating changes during implementation phase and over the long-term	1) A lot of variability in data results, difficult to discern trends; 2) Relationships not well understood for some contaminants.	Applicable to all subtidal areas of LDW.	1) Readily implementable; 2) Minimal impact on industrial and shipping uses of waterway; 3) Good for risk communication to public.	1) Requires long-term financial commitment to ensure maintenance of engineered structures (i.e., cap, CAD) and monitoring/sampling.	Moderate
Monitored Natural Recovery	Chemical Degradation	Combination of natural desorption, diffusion, dilution, volatilization, resuspension, and transport	Effective principally to LDW organic COCs including SVOCs and PCBs. Inorganics not subject to degradation.	Effective where chemical degradation of COCs is demonstrated to occur in the short- and long-term.	1) Effective where risk assessment demonstrates low to no risk to human health and environment; 2) Physical/chemical degradation demonstrated for SVOCs, but less effective for metals, PCBs, TBT and pesticides; 3) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 4) Potentially low level of short-term effectiveness for ecological receptors because COCs remain in place, but can provide adequate long-term protection; 5) Requires implementation of long-term monitoring study and risk assessment objectives.	Applicable to all areas of the LDW.	1) Readily implemented with no construction requirements; 2) Minimal impact on current or future industrial and shipping uses of waterway; 3) May be used in conjunction with other technologies in a combined alternative.	1) Must be implemented in conjunction with a well-designed, long-term monitoring program; 2) May require future active remediation where MNR risk-expectations are not achieved.	Low
	Biological-Degradation	COC Metabolization (aerobic and anaerobic)	Effective principally to SVOCs. PCBs and TBT will degrade, but not within an acceptable time frame. Metals will not degrade.	Biodegradation is a demonstrated and proven remedial technology for volatiles and SVOCs. Effective where degradation of COCs are demonstrated to occur in the short- and long-term.	1) Biological degradation less effective for PCBs and TBT; 2) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) Less effective for ecological receptors because COCs remain in place; 4) Requires implementation of long-term monitoring study and risk assessment objectives.	Applicable in areas with low concentrations of SVOCs in well-mixed sediments.	1) Readily implemented with no construction requirements; 2) Minimal impact on current or future industrial and shipping uses of waterway; 3) May be used in conjunction with other technologies in a combined alternative; 4) Implemented in areas with biodegradable COCs.	1) Must be implemented in conjunction with a well-designed long-term monitoring program; 2) May require future active remediation where MNR risk-expectations are not achieved.	Low
	Physical Burial Processes	Sedimentation/ Burial Resuspension and Transport (minor components of MNR)	Effective for all LDW COCs where concentrations are low.	1) Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants; 2) Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact.	1) Requires implementation of long-term monitoring study and risk assessment objectives; 2) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) Less effective for ecological receptors because COCs remain in place; 4) COCs not actively removed and remain in place. 5) Facilitates PCB contamination of the marine food chain when resuspension and transport occur	Applicable where geochronological studies and hydrodynamic modeling demonstrate long-term sedimentation and burial processes are in-place.	1) Readily applied and demonstrated process; 2) Can be combined with institutional controls until long-term risk-objectives are demonstrated; 3) Minimal impact on industrial and shipping uses of waterway.	1) Requires long-term monitoring and continuing financial commitment until risk-objectives are achieved; 2) Associated institutional controls may limit future uses of waterway.	Low

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
Enhanced Natural Recovery	Enhanced Physical Burial	Thin-layer placement to augment natural sedimentation	Effective for all LDW COCs where MNR processes are demonstrated.	ENR dilutes COC concentrations while not resulting in the resuspension and transport of contaminants that occurs with dredging.	1) Requires implementation of long-term monitoring study and risk assessment objectives; 2) Short-term impacts to human and ecological health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) COCs not actively removed, but attenuated by addition of clean sediments.	Applies where data and modeling indicate placement of a thin-layer of material, combined with natural recovery processes will result in achievement of risk-based sediment objectives. Particularly useful for critical habitat areas, and/or shallow intertidal areas where active remedial methods could result in unwanted habitat loss. Potentially suitable for management of dredge residuals.	1) Puget Sound-demonstrated technology with local construction knowledge; 2) Sediment for thin-layer placement readily available.	1) Requires long-term monitoring, institutional controls and continuing financial commitment until cleanup objectives are achieved; 2) Institutional controls may limit future uses of waterway.	Low
Containment	Capping	Conventional Sand Cap	Applicable principally to PAHs, other SVOCs, metals, and PCBs; Limited applicability to VOCs.	1) Demonstrated effectiveness for isolating contaminants in the LDW; 2) Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants; 3) Capping does not result in the resuspension and transport of contaminants that occurs with dredging.	1) Sand cap may be subject to bioturbation and release of buried COCs; 2) Sand caps may be susceptible to propeller and/or high-flow scour, methane generation, and earthquakes; 3) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 4) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable to subtidal areas where sediments have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COCs into the clean cap surface.	1) Readily applied and demonstrated technology. Local construction experience; 2) Capping materials readily available from navigation dredging at the Upper Turning Basin.	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway; 4) Impacts to flooding, stream bank erosion, navigation, and recreation must be addressed in design.	Low to Moderate
		Conventional Sediment/Clay Cap	Applicable principally to COCs with potentially higher solubilities and lower sorption.	1) Sediment with high fines (silt and clay) and or TOC is effective in limiting diffusion of contaminants. Sediment caps are generally more effective than sand caps for containment of contaminants with high solubility and low sorption; 2) Natural TOC present in conventional sediments more effective at adsorbing COCs such as PCBs.	1) Clay liners in caps are potentially more susceptible to breaches caused by methane generation through the cap; 2) Caps may be susceptible to propeller and/or high-flow scour, methane generation, and earthquakes; 3) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 4) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable in sections of LDW with low erosion potential and where placement of finer-grained material can be managed. May be useful in nearshore, or intertidal applications where thinner caps with higher sorbtive capacities are required. Sediments must still have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COCs into the clean cap surface.	1) Readily applied and demonstrated technology; 2) Placement of high TOC and/or high fine sediments minimizes thickness of cap in areas with shallow water depth; 3) Materials readily available through upland sources or from navigation dredging at other systems.	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway; 4) Impacts to flooding, stream bank erosion, navigation, and recreation must be addressed in design; 5) Utilization of navigation dredged material for capping has potential logistical issues.	Low to Moderate
		Armored Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	Effective in combination with conventional caps to isolate contaminants and protect cap against physical erosion and/or bioturbation.	1) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 2) Armor rock may be less productive habitat for benthic organisms. 3) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable in conjunction with other cap configurations in areas of LDW, but can be applied where erosion potentials are higher.	(1) Readily applied and demonstrated technology; 2) Armor placement can be used to minimize thickness of cap in areas with shallow water depth; 3) Armor materials can be combined with habitat-enhancing materials (e.g., "Fish Mix").	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway.	Low to Moderate
		Composite Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	1) Provides physical isolation of COCs from the overlying water column; 2) Assists in preventing bioturbation breaches of caps and prevents direct contact between aquatic biota and contaminants; 3) Rigid HDPE layers used in small areas to assist in NAPL containment, control hydraulic gradient, and methane containment and diffusion.	1) Composite caps at other sites have resulted in catastrophic breaches as a result of methane generation under the cap; 2) Rigid HDPE layers do not have long-term demonstrated effectiveness; 3) Use of geotextiles may not be necessary for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact; 4) Geotextiles by themselves do not limit advective or diffusive flux of COCs; 5) Requires long-term monitoring, institutional controls, and financial commitment.	Composite caps with impermeable layers such as HDPE are generally applicable where control of NAPL or groundwater movement is needed in a limited area. Composite caps may also be potentially applicable in intertidal areas where physical separation between receptors and COCs are required, but where minimal change to the slope or bathymetric configuration is needed.	1) Increasingly applied technology; 2) Placement of geotextile or rigid HDPE can be used to minimize thickness of cap in areas with shallow water depth.	1) Requires specialty equipment for placement, sinking, and securing to the sediment floor; 2) Tidal ranges in the LDW can affect ability to place materials; 3) Requires long-term monitoring and financial commitment.	Low to Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: COCs	Effectiveness: Advantages	Effectiveness: Disadvantages	Implementability: Site Conditions	Implementability: Advantages	Implementability: Disadvantages	Cost[1]
Containment	Capping	Spray Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	Good for application under hard to access areas such as piers and wharves. Provides good physical barrier between contaminants and overlying surfaces.	1) Creates a hard surface. If habitat surface values are required, habitat-suitable material would need to be placed on top of the shotcrete. 2) Must be applied in the dry with time to set. Areas of application are limited to high intertidal areas. 3) Requires long-term monitoring and maintenance, institutional controls, and a potential requirement for replacement habitat.	Labor intensive process to implement in difficult working conditions under docks and piers.	Good for application under hard to access areas such as piers and wharves.	1) Potentially dangerous work because of obstructions, slippage, and presence of contaminants next to workers applying the shotcrete. 2) Requires specialty equipment to place the shotcrete. 3) Tidal ranges can affect placement location. 4) Not applicable in habitat areas.	High
		Reactive Caps	Potentially applicable to all LDW COCs as described for conventional sand and/or conventional sediment caps.	Similar to advantages described for other caps. Provides an additional level of contaminant-sorbing materials to caps.	Long-term effectiveness not demonstrated. Retained as innovative technology. Requires long-term monitoring, institutional controls, and financial commitment. Probably not acceptable in beach areas.	Applicable in conjunction with other cap configurations in areas of LDW.	Adds an additional level of environmental protection with contaminant sorbing materials. May allow for construction of thinner caps.	1) Requires specialty equipment for placement, sinking, and securing to the sediment floor; 2) Tidal ranges in the LDW can affect ability to place materials; 3) Requires long-term monitoring and financial commitment; 4) Long-term implementability not demonstrated. Retained as innovative technology.	Low to Moderate
Removal	Dredging	Hydraulic Dredging	Applicable to all LDW COCs at higher concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	1) Effective removal with lower resuspension and recontamination/residual rate relative to mechanical dredging; 2) Can be readily incorporated into treatment trains such as chemical and/or physical separation.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of low solid sediments, generally less than 20 ft. of water depth and low levels of debris.	(1) Various hydraulic dredges readily available on the West Coast and at least one dredging contractor has equipment on the LDW; (2) More effective lateral and vertical cut control may be achieved, relative to mechanical dredges; (3) High utility when used in conjunction with CDFs; (4) Local experience of use for the Sitcum and Blair Waterway projects.	1) Hydraulic dredges limited in heavy-debris environments; 2) Environmental hydraulic dredges are depth limited, and difficult to size to accommodate steady solids flow under varying tidal regimes; 3) Requires separation of solids from water, resulting in large volumes of water that may require treatment prior to discharge back to LDW; 4) Treatment facilities must be located near-waterway with enough land space to accommodate retention basins, mechanical dewatering equipment, sand and carbon filtration, and transfer of dewatered material to trucks or trains for transfer to regional landfill.	Moderate to High
		Mechanical Dredging	Applicable to all LDW COCs at concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	Effective for removal in areas with high debris and sediments with high sand or heavy clay content that require digging buckets.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of high percentage solids sediments, including areas with heavy debris, sand, and clay. Mechanical dredging is not depth restricted, and not affected by tidal exchange.	1) Various mechanical dredges, including environmental buckets and clamshells readily available on the LDW and in Puget Sound; 2) Recent construction experience in LDW and Puget Sound with skilled operators; 3) Environmental buckets useful in softer, unconsolidated materials with low debris; 4) Digging buckets (e.g., clamshells) useful in harder clays or compacted sediments, or where debris is high; 5) Existing infrastructure for barge transport, off-loading, and transfer to railcars for transport to regional landfills; 6) Depth and tidal limitations within the LDW do not restrict use of mechanical buckets.	1) Not all river segments may be accessible to a barge-operated mechanical dredge; 2) Can result in potentially higher resuspension and residual rates than hydraulic dredges; 3) Lower vertical and horizontal operational control relative to hydraulic dredges.	Low to Moderate
		Mechanical Dredging (Excavator)	Applicable to all LDW COCs at concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	Effective for removal in areas with high debris and sediments with high sand or heavy clay content that require digging buckets.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of high percentage solids sediments, including areas with heavy debris, sand, and clay. Mechanical dredging is not depth restricted, and not affected by tidal exchange.	1) Equipment is available to the Puget Sound region but to lesser extent than standard clamshell dredges; 2) Recent construction experience in LDW and Puget Sound with skilled operators; 3) Offer high level of vertical and horizontal control during dredging.	1) Not all river segments may be accessible to a barge-operated mechanical dredge; 2) Can result in potentially higher resuspension and residual rates than hydraulic dredges; 3) Lower vertical and horizontal operational control relative to hydraulic dredges.	Low to Moderate
	Excavating	Dry Excavating	Applicable to all LDW COCs. Effective for nearshore and/or intertidal areas where depths limit conventional dredging equipment	1) Contaminated sediments removed; 2) Residuals can be minimized or eliminated by dry excavation.	Effective only in relatively small and narrow shoreline areas of limited intertidal bands. Requires either only working during low tides, or using cofferdams or sheet pile walls to create a contained, dry area.	Limited in application to nearshore shallow and/or intertidal areas that can be reached from shore or by specialty equipment designed to work on soft, unconsolidated sediments.	Equipment and construction experience in Puget Sound.	1) Construction costs may involve contingencies to address potential spills and leaks	Low to Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
In Situ Treatment	Physical-Immobilization	Activated Carbon Amendment	Applicable to certain LDW COCs at concentrations that pose unacceptable risks to human health and the environment	1) Contaminants adsorb to activated carbon particles; 2) porewater concentrations (sediment-to-water fluxes), contaminant concentrations, and bioavailability for uptake by organisms are reduced; and 3) promising pilot-scale results.	May require armoring in areas susceptible to propeller and/or high-flow scour. Requires long-term monitoring, institutional controls, and financial commitment. Retained as innovative technology. Long-term effectiveness not demonstrated at full scale.	Easily implementable, and applicable to most areas of the LDW. Sand could be mixed with the activated carbon as a form of modified ENR.	1) Recently demonstrated implementable technology; 2) activated carbon for placement readily available, and 3) commercial products have been developed to improve the deployment of the activated carbon, by using a weighting particle (sand, gravel, etc.) coated with an inert binder and activated carbon.	1) Can require specialized equipment depending on application method; 2) requires long-term monitoring.	Low to Moderate
		Organoclay Amendment	Applicable to certain LDW COCs at concentrations that pose unacceptable risks to human health and the environment	1) Chemically binds metal ions, replacing them with amines or other functional groups; 2) physically isolates the contaminated sediment from receptors (because of low permeability of clay); 3) stabilizes sediment preventing resuspension and transport of contaminants, and 4) promising pilot-scale results.	May require armoring in areas susceptible to propeller and/or high-flow scour. Requires long-term monitoring, institutional controls, and financial commitment. Retained as innovative technology. Long-term effectiveness not demonstrated at full scale	Easily implementable, and applicable to most areas of the LDW.	1) Recently demonstrated implementable technology; 2) organoclays for placement commercially available.	1) Can require specialized equipment depending on application method; 2) requires long-term monitoring.	Low to Moderate
Ex Situ Treatment	Chemical/Physical	Soil Washing	Applicable to all LDW COCs. Principal application would be for high volumes of organic-contaminated sediments.	1) Full-scale testing demonstrated ability to take high concentrations of COCs and treat to equivalent of MTCA soil standards; 2) Potential beneficial reuse for residuals.	1) Tests to date have treated hazardous waste-level materials. No data on treatment of lower concentrations of contaminants; 2) Effective treatment when starting with high sands materials—lower effectiveness when treating low solids and high fine-grained sediments; 3) Solid-waste classification in Washington state unclear, which may require disposal of treated materials at a Subtitle D landfill.	Applicable to potential dredge areas containing organic and coarse-grained sediment.	1) Readily implementable, resulting in reduced contaminated sediment volume; 2) System could be coupled with hydraulic dredging for continuous treatment train; 3) Mobile units are available 4) Continuous flow process designed to process up to 40 cy of sediments per hour for the full-scale system; 5) May be available for potential beneficial reuse.	1) Waste streams include hydraulic-dredge decant water, reagents used in soil washing, and the treated residuals; 2) Water will require filtration and treatment prior to discharge; 3) Treated residuals may require off-site disposal; 4) Volume/long-term supply of sediments to be treated and local market for beneficial use products affect the economics of implementing this technology.	Moderate
	Physical	Particle Separation	Only applicable to adsorptive COCs that would adhere to the fine-grained soil. Offers greatest utility and cost saving benefits where concentrations of COCs would otherwise require incineration or Subtitle C disposal.	1) Demonstrated effectiveness for reduction in volume of highly contaminated sediments with a high percentage of sand-content; 2) Used to increase effectiveness of dewatering dredged material.	1) Not effective for contaminants with high concentrations and high organic content; 2) Previous work at other sites with PCB-contaminated sediments has shown that PCBs are retained on sand particles (as emulsion), requiring Subtitle D disposal.	Applicable to potential dredge areas containing higher sand content.	1) Readily implementable, resulting in reduced contaminated sediment volume; 2) Can be combined with soil washing to improve contaminant separation and/or destruction; 3) Mobile units are available; 4) Separated sand may be available for potential beneficial reuse, capping, or disposal at DMMP Elliott Bay site.	Will require disposal of separated waste stream at a Subtitle D landfill. Fines could also require Subtitle C disposal or incineration.	Moderate
		Solidification	Applicable to all LDW COCs. Principal application would be for high volumes of PCB-contaminated sediments that exceed hazardous waste criteria and would otherwise require incineration or Subtitle C disposal.	1) Lime has been successfully added to dredged material at other projects; 2) Effective during the dewatering operation to remove excess water and prepare material for disposal.	High contaminant concentration and high water content results in higher project costs.	Applicable to all dredge areas of LDW.	1) Readily implementable; 2) Reagent materials readily available.	1) Immobilizing reagents, ranging from Portland cement to lime cement, kiln dust, pozzolan, and proprietary agents, have been applied with varying success. Dependent on sediment characteristics and water content; 2) Contaminants remain in place. Stabilized product requires disposal in regulated landfill.	Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
Disposal	On-Site	Contained Aquatic Disposal (CAD)	Applicable to all LDW COCs below hazardous waste designations.	1) Demonstrated local experience and effectiveness in the LDW and Puget Sound; 2) Effective containment of metals, organics, and PCBs; 3) Can be designed to include habitat enhancement for salmonids.	1) CADs must be engineered to withstand bioturbation, advective flux, and release of buried COPCs, propeller and/or high-flow scour, and earthquakes; 2) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 3) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable to subtidal areas where sediments have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COPCs into the clean cap surface.	1) Technically readily implemented within the LDW with contaminated sediments contained on-site; 2) Local construction experience; 3) Excavated clean pit-materials can be used for beneficial uses and/or to cap CAD; (4) Does not interfere with current industrial uses of LDW.	1) Volume-limited on LDW as a large area would be required to accommodate dredged sediments; 2) Requires long-term commitment to monitoring with the potential for additional actions if CAD fails; 3) Requires permanent institutional controls (e.g., deed restrictions, dredging moratorium) that may affect future development and uses of the LDW; 4) Requires concurrence with land owner.	Low to Moderate
		Confined Disposal Facility (CDF)	Applicable to all LDW COCs below hazardous waste designations.	1) Demonstrated local experience and effectiveness in Puget Sound; 2) Effective containment of metals, SVOCs and PCBs; 3) A subtidal CDF could be designed to include habitat enhancement for salmonids.	1) CDFs must be engineered to withstand advective flux and release of buried COCs, propeller, and/or high-flow scour, and earthquakes; 2) Filling of nearshore lands would result in unavoidable loss of aquatic lands that will require mitigation.	Requires large suitable near-shore or upland containment site. Former slips or similar in-water areas would be best suited to construct a CDF.	1) Puget Sound-demonstrated technology with local construction knowledge; 2) Cap sediments or soils readily available; 3) Could contain large volumes of contaminated sediments, depending upon site availability; 4) Beneficial upland industrial and/or residential reuse of filled site.	1) Site-limited on LDW. Few potential locations without other current uses; 2) Requires long-term commitment to monitoring with the potential for additional actions if CDF fails; 3) Requires permanent institutional controls (e.g., deed restrictions, dredging moratorium) that may affect future development and uses of the LDW; 4) Requires concurrence with land owner.	Moderate to High
	Off-Site	Subtitle D Landfill	Applicable to all LDW COCs below hazardous waste designations.	Subtitle D landfills highly effective for long term, permanent containment of contaminated materials.	COCs contained, but not permanently destroyed.	Applicable throughout LDW for both dewatered and wet sediments.	1) Several licensed landfills in Washington exist that can receive dredged materials in Puget Sound; 2) Transfer facilities for moving sediments from LDW to the landfills exist on-site; 3) Transport infrastructure in-place on the LDW; 4) Options exist for moving wet sediments - eliminating need for on-site dewatering facilities.	1) Transfer and barge offload facilities may not be present at the time the project is completed so a separate offload facility may need to be constructed; 2) Landfills in Eastern Washington and Eastern Oregon requires train transport with potential for spillage.	Moderate
	Off-Site	Subtitle C Landfill	Applicable to all LDW COCs exceeding hazardous waste designations.	Subtitle C landfills are federally-regulated facilities and are highly effective for long-term, permanent containment of highly contaminated materials.	1) COCs contained, but not permanently destroyed; 2) Requires dewatering of dredged sediments.	Applicable throughout LDW for dewatered sediments	Option for disposal of listed, hazardous wastes	Transport of hazardous materials to facility expensive.	High
		DMMP Open Water Disposal	Applicable to all LDW COCs in sediments that are separated or treated to below the DMMP disposal standards.	DMMP is a well-established and effective program with a long-term track record of monitoring to verify environmental protectiveness.	None	Applicable throughout LDW	The DMMP disposal site is located in nearby Elliott Bay	Sediments that require remediation are not likely to meet the open water disposal criteria.	Low
		MTCA Reuse (upland or in-water beneficial reuse)	Applicable to all LDW COCs in sediments that are either below, or treated-to below the reuse standards for uplands and in-water.	Beneficial reuse of sediments	Some residual COCs may remain after treatment	Applicable throughout LDW	Potential use of sediments that meet the MTCA Level A soil requirements as upland fill, or other beneficial upland uses including daily landfill cover. Potential beneficial reuse as in-water ENR, capping material, and habitat enhancement. May be implementable for high volumes of materials with low concentrations of COCs, or for treated sediments.	No specific beneficial upland reuse has been identified. As such, requires the additional costs for transport of material and/or tipping fee to send to landfill.	None

Notes:

1. Cost assessment is based on the relative cost of a process option in comparison to other process options within a given technology type.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; CTM = Candidate Technologies Memorandum; cy = cubic yards; DMMP = Dredged Material Management Program; ENR = enhanced natural recovery; FS = Feasibility Study; GRA = general response action; HDPE = High-density polyethylene; HTTD = high-temperature thermal desorption; LTTD = low-temperature thermal desorption; MNR = Monitored Natural Recovery; MTCA = Model Toxic Control Act; NCP = National Contingency Plan; PCB = polychlorinated biphenyls; SVOC = semivolatile organic compound; TBT = tributyltin; TOC = total organic carbon; TSCA = Toxic Substances Control Act; TSS = total suspended solids; UECA = Uniform Environmental Covenants Act

Table 7-4 Remedial Technologies and Process Options Retained for Potential Use in Developing Remedial Alternatives

General Response Action	Technology Type	Process Option	Comments Related to Technology Assumptions for the FS
No Action	None	Not Applicable	Per NCP requirements
Institutional Controls	Proprietary controls and Informational Devices (EPA 2000)	Proprietary Controls	Access to much of the LDW shoreline from the uplands is already restricted by general security measures put in place by private and public property owners. The LDW is a public waterway and public access to nearshore areas is generally not prohibited.
		Seafood Consumption Advisories, Education, and Public Outreach	Public advisories regarding fish and shellfish consumption are currently posted for the entire LDW. Public advisories regarding sediment contact risks are not currently posted. Advisories are a likely element of all remedial alternatives and will remain in place until monitoring data confirms that the advisories can be modified or removed entirely.
		Monitoring and Notification of Waterway Users	As needed, these will be tailored to specific remediation activities and site constraints.
		Enforcement Tools	CERCLA or MTCA consent decrees for settling potentially responsible or liable parties, or unilateral orders for non-settling parties, issued by EPA or Ecology are anticipated.
		Site Registry	Provides information on applicable restrictions associated with Restricted Navigation Areas and other proprietary controls.
Monitoring	None	Baseline Monitoring	Establishes a statistical basis for comparing conditions before and after the cleanup action.
		Construction Monitoring	Short-term monitoring during remediation used to evaluate whether the project is being implemented in accordance with specifications (i.e., water quality monitoring, bathymetric surveys)
		Post-Construction Performance Monitoring	Post-construction performance monitoring evaluates post-removal surface and subsurface sediment conditions in dredging or containment areas to confirm compliance with project specifications.
		Operation and Maintenance Monitoring	Operation and maintenance monitoring of dredging areas, containment, and/or disposal sites (i.e., CAD sites, ENR and capping areas) required to ensure long-term effectiveness and continued stability of the structure.
		Long-term Monitoring	Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action.
Monitored Natural Recovery (MNR)	Natural Physical, Biological, and Chemical Recovery	Multiple potential mechanisms: burial (sedimentation), immobilization, desorption, dispersion, diffusion, dilution, volatilization, resuspension, biological degradation.	Surface sediment chemistry is monitored over time to track recovery by multiple physical, chemical, and biological mechanisms that operate naturally in the estuarine environment of the LDW. Burial by the comparatively cleaner sediments coming into the LDW from the Green River is the principal mechanism for recovery in the LDW. Natural recovery is operative in the waterway as supported by analysis of the empirical data and predicted by the STM. Areas potentially suitable for MNR must be depositional, not subject to significant physical disturbances from high river flows, vessel propeller scour, anchor drag, and routine dredging. Future construction activity in MNR zones is not precluded; however, the applicant/owner must be prepared to appropriately handle any contaminants that may be encountered as part of the project.
Enhanced Natural Recovery (ENR)	Thin-layer Placement	Placement of a thin layer of granular media (e.g., sand) to augment natural recovery	ENR differs from MNR with respect to the modification of initial conditions (i.e., placing clean material onto the contaminated sediment surface). In other respects, siting, monitoring, and future use restrictions and considerations are the same. Placement also can serve as a means of managing contaminated dredging sediment residuals, called thin-layer sand placement. The composition of ENR/*in situ* treatment may include carbon amendments and/or habitat mix.
Containment	Capping	Conventional Sand Cap	Conventional capping is restricted to net deposition areas that are not subject to appreciable sustained or episodic erosion. Cap thickness must be sufficient to prevent reintroduction of buried contaminants into biologically active zone (upper 10 cm).
		Conventional Sediment/Clay Cap	Cap thickness must be sufficient to prevent reintroduction of buried contaminants into biologically active zone (upper 10 cm).
		Armored Cap	If capping is considered in erosion areas, armoring will likely be required to maintain the cap integrity.
		Spray Cap (Technology not addressed by CTM)	Shotcreting is potential approach for confining, isolating contaminants under dock or overwater structures. The shotcrete application at Todd Shipyards effectively encapsulated existing debris (slag) mounds from the aquatic environment.
		Composite Cap	Application would be location- and contaminant-specific where space or pollutant constraints indicate conventional sand capping is not adequate.
		Reactive Cap	Application would be location- and contaminant-specific where space or pollutant constraints indicate conventional sand capping is not adequate.
Removal	Dredging	Hydraulic Dredging (including diver-assisted dredging)	Hydraulic dredging has several constraints that limit its project-wide application: the cost and logistics of managing large volumes of water including large land area adjacent the dredging area; potential for water quality impacts; debris leads to operational difficulties and dredging inaccuracies; interruption of waterway use caused by placement of the hydraulic discharge pipeline in the LDW. Application of hydraulic dredging in the LDW may be appropriate on a small scale (e.g., diver-assisted dredging in under dock/pier areas) or on location-specific basis.
		Mechanical Dredging	Demonstrated effective in the Puget Sound region and nationwide sediment remediation projects. Readily available and least-cost dredging option in the Puget Sound region.
		Mechanical Dredging (Excavator)	Excavator dredges offer a high level of control in the placement of the dredge bucket because it uses fixed linkages instead of cables. This yields a higher degree of accuracy resulting in less volume of dredged sediment and reduced water quality impacts as compared to a conventional derrick barge. Often used for debris removal and/or shallow in-water dredging operations.
	Excavating	Dry Excavating	Generally applicable to nearshore areas above elevation -2.0 ft MLLW or 25-ft reach from top of bank.

Table 7-4 Remedial Technologies and Process Options Retained for Potential Use in Developing Remedial Alternatives (continued)

General Response Action	Technology Type	Process Option	Comments Related to Technology Assumptions for the FS
In Situ Treatment	Physical/Immobilization	Activated Carbon Amendment	Demonstrated effective in nationwide sediment remediation projects at pilot-scale level. Readily available and low-cost *in situ* treatment technology. Sand could be mixed with the activated carbon as a form of modified ENR (see above).
		Organoclay Amendment	Demonstrated effective in nationwide sediment remediation projects at pilot-scale level. Readily available and low-cost *in situ* treatment technology. May require armoring in LDW areas susceptible to propeller and/or high-flow scour.
Ex Situ Treatment	Chemical/ Physical	Soil Washing	Mechanically dredged sediment is screened to remove oversize debris and is then processed through a series of unit operations resulting in the following products or waste streams: wastewater, sludge (fines fractions), and sand/gravel. Wastewater requires treatment, the sludge is typically disposed (upland landfill), and the sand/gravel component may be reused for in-water applications if it tests suitable for beneficial use pursuant to the Washington State Sediment Management Standards (i.e., less than SQS criteria). Soil washing/particle separation is potentially effective and implementable in the LDW where the percentage of sand in the sediment exceeds ~ 30% by weight. It is anticipated that most of the COCs will concentrate on the remaining sludge (fines fraction), which will then need disposal. This concentrating process, if too great, could cause the sludge to be designated as hazardous waste.
	Physical	Separation	Presented as unit costs in FS.
		Solidification	If future designs require further water reduction methods and to remove free water prior to landfilling.
Disposal	On-site Disposal	Contained Aquatic Disposal (CAD)	The overall space (volume) capacity for CAD is limited. However, adequate capacity may be available to contain substantial portion of the contaminated dredged sediment for those alternatives requiring the least amount of dredging. However, for most alternatives, CAD will not be adequate for project-wide application, but could serve to contain a portion of the contaminated sediment. Substantial implementability logistics issues need to be addressed with CAD. Also, constraints with long-term institutional controls (e.g., conflict if located within established dredging areas) and multiple agency approvals to authorize the site are a concern.
		Confined Disposal Facility (CDF)	Not applicable to LDW site-wide application because of limited locations (and capacity) without other current uses. May be applicable for smaller-scale location-specific application.
	Off-site Disposal	Subtitle D Landfill	Applies specifically to sediment that is characterized as non-hazardous in accordance with federal or state regulations. Regional landfills that can accept nonhazardous sediment are Allied Waste Inc. (Roosevelt, Washington) and Waste Management (Arlington, Oregon).
		Subtitle C Landfill	Applies specifically to sediment that is characterized as hazardous or dangerous in accordance with federal or state regulations. This condition is not expected to occur on a large scale and more likely will be limited to localized hot spot removal areas, if triggered at all.
		Dredged Material Management Program (DMMP) Open water Disposal	This is a potentially viable disposal option where the average concentration of COCs in the entire dredged material management unit is determined to be less than the DMMP disposal requirements.
		Beneficial Use (In-Water and Upland)	Sediment that tests suitable for beneficial use pursuant to the Washington State Sediment Management Standards (i.e., less than SQS criteria) may be beneficially reused for habitat creation, capping, or residual management. In case of treatment (e.g., soil washing), the sediment may qualify for beneficial reuse.

Notes:

Representative site-wide process options included in the development of the remedial alternatives and cost estimates for this FS. Other process options may have location-specific applicability; but not site-wide applicability.

1. These technologies and process options were screened and retained in Tables 7-2a through 7-2e, and summarized in Table 7-3 with the exception of institutional controls, which do not lend themselves to comparison on the same terms as other technologies. Institutional controls are discussed only within Section 7.2 and are not included in Table 7-3.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; CTM = Candidate Technologies Memo; DMMP = Dredged Material Management Program; Ecology = Washington State Department of Ecology; ENR = enhanced natural recovery; EPA = U.S. Environmental Protection Agency; MNR = monitored natural recovery; MTCA = Model Toxics Control Act; MLLW = mean lower low water; NCP = National Contingency Plan; SQS = Sediment Quality Standards; STM = Sediment Transport Model

Table 7-5 Sediment Dredging and Handling Methods Used on Representative Projects in the Puget Sound Region

Sediment Remediation Project	Completed	Dredge Method	Disposal Method	Predicted Volume of Dredged Sediment (cubic yards)	Actual Volume of Dredged Sediment (cubic yards)
Wyckoff/Eagle Harbor West Operable Unit	1997	Mechanical	CDF	1,300 to 9,200	6,000
Norfolk Sediment Remediation	1999	Mechanical	Subtitle D landfill and Subtitle C landfill	4,050	5,190
Cascade Pole Site	2001	Mechanical	CDF	n/a	40,000
Puget Sound Naval Shipyard	2001	Mechanical	CAD	300,000	n/a
Weyerhaeuser	2002	Mechanical	Landfill	n/a	n/a
Hylebos Waterway – Area 5106	2003	Hydraulic	CDF	20,000	n/a
East Waterway	2004	Mechanical	Subtitle D landfill	n/a	n/a
Lockheed Shipyard	2004	Mechanical	Subtitle D landfill	46,625	70,000
Todd Shipyard	2004	Mechanical	Subtitle D landfill	116,415	220,000
Duwamish/Diagonal	2004	Mechanical	Subtitle D landfill	42,500	66,000
Middle Waterway	2004	Mechanical	CDF	75,000	109,000
Hylebos Waterway – Segments 3-5	2004	Mechanical	CDF	n/a	>100,000
Pacific Sound Resources	2004	Mechanical	Subtitle D landfill	3,500	10,000
Head of Hylebos Waterway	2005	Mechanical	Subtitle D landfill	217,000	419,000
Thea Foss – Wheeler Osgood Waterways	2005	Hydraulic/Mechanical	CDF	620,000[a]	422,535
Denny Way	2007	Mechanical	Subtitle D landfill	13,730	14,400

Notes:

a. Volume from combined projects from Commencement Bay Nearshore/Tideflats Explanation of Significant Differences (EPA 2000e)

CAD = contained aquatic disposal; CDF = confined disposal facility; n/a = not available

Figure 7-1 Pilot-scale Demonstrations of Activated Carbon Amendment Delivery into Sediment

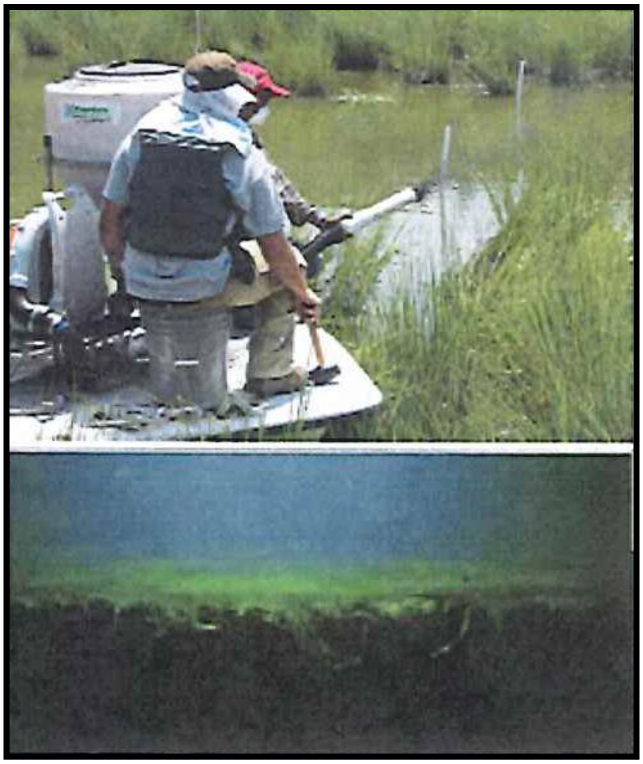
A) *Application of activated carbon in a tidal mudflat at Hunter's Point Naval Shipyard, San Francisco Bay, CA using two application devices (2004 and 2006). The Aquamog (top) using a floating platform approached the site from water and used a rototiller arm while the slurry injection system (bottom) was land based and applied a carbon slurry directly into sediment.*

Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech*., 45, 1163–1168, 2011.

B) *Application of activated carbon under 15 feet of water at Lower Grasse River, NY (2006). The site was enclosed with a silt curtain and application was performed using a barge mounted crane. Placement and mixing of the activated carbon was achieved using two devices: 1) a 7-by 12-foot rototiller-type mixing unit (top); and 2) a 7-by-10-foot tine sled device (bottom).*

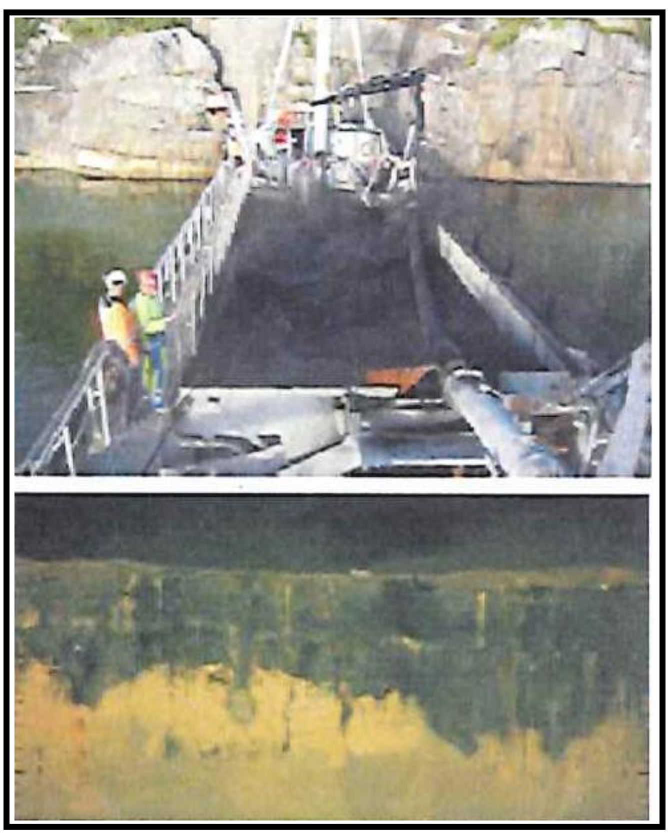
Source: 2006 Activated Carbon Pilot Study Project (thegrasseriver.com).

Figure 7-1 Pilot-scale Demonstrations of Activated Carbon Amendment Delivery into Sediment (continued)



D) *Application of activated carbon in a pelletized form (SediMite™) using an air blown dispersal device (top) over a vegetated wetland impacted with PCBs near the James River, VA (2009). Picture below illustrates bioturbation induced breakdown and mixing of pelletized carbon with a fluorescent tag in a laboratory aquarium (bottom).*

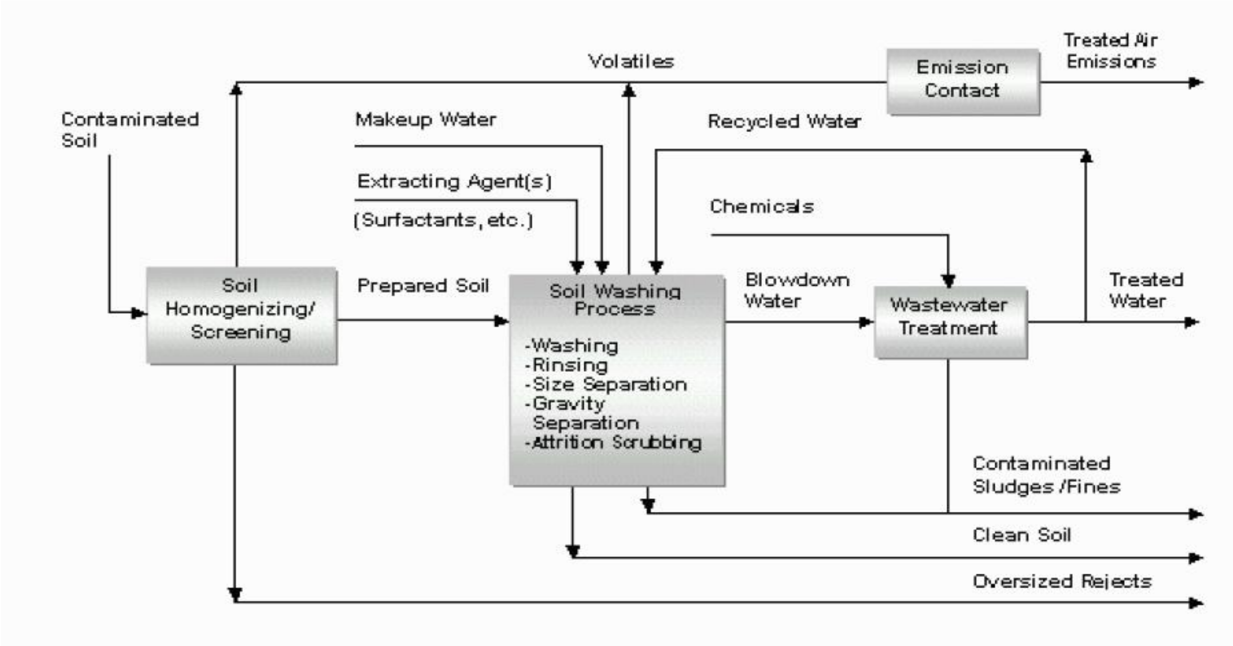
Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech.*, 45, 1163–1168, 2011.



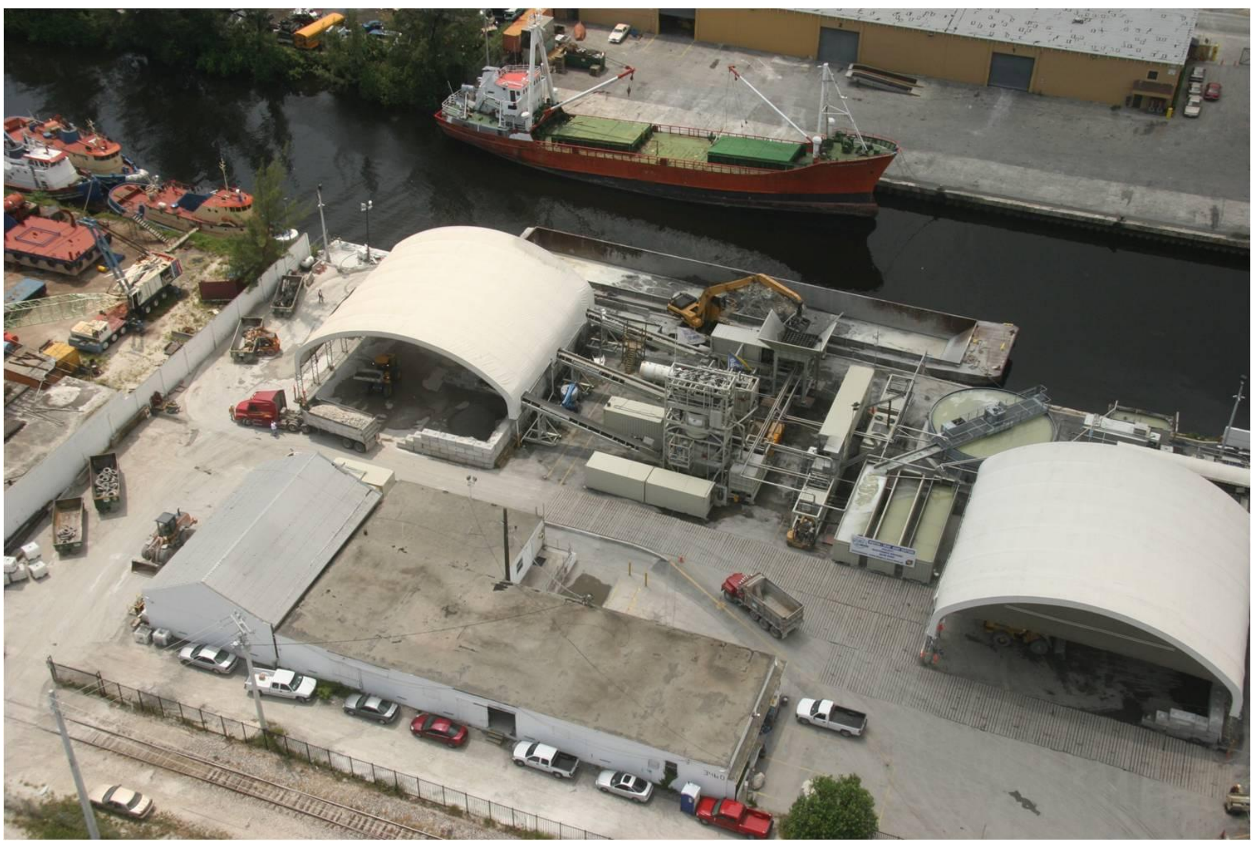
E) *Application of activated-carbon-clay mixture at 100- and 300-ft depth, Grenlandsfjords, Norway (2009), led by NGI and NIVA. A hopper dredger was used to pick up clean clay from an adjacent site. After activated-carbon-clay mixing, the trim pipe was deployed in reverse to place an activated-carbon-clay mixture on the sea floor. Sediment-profile imaging and sediment coring (bottom figure) showed that placement of an even active cap was successful.*

Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech.*, 45, 1163–1168, 2011.

Figure 7-2 Soil Washing



Process Diagram *http://www.frtr.gov/matrix2/section4/4-19.html*

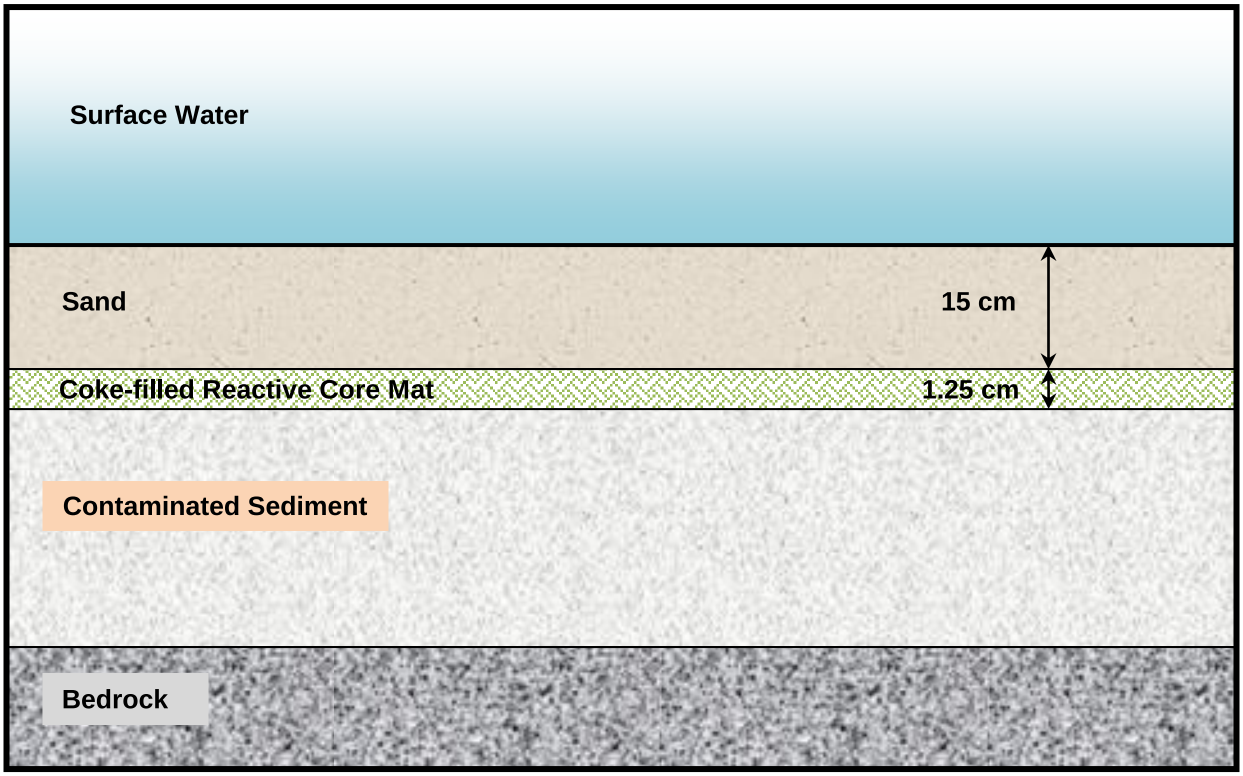


Soil Washing. Miami River Soil /Sediment Separation Plant.
Source: Boskalis-Dolman 2006

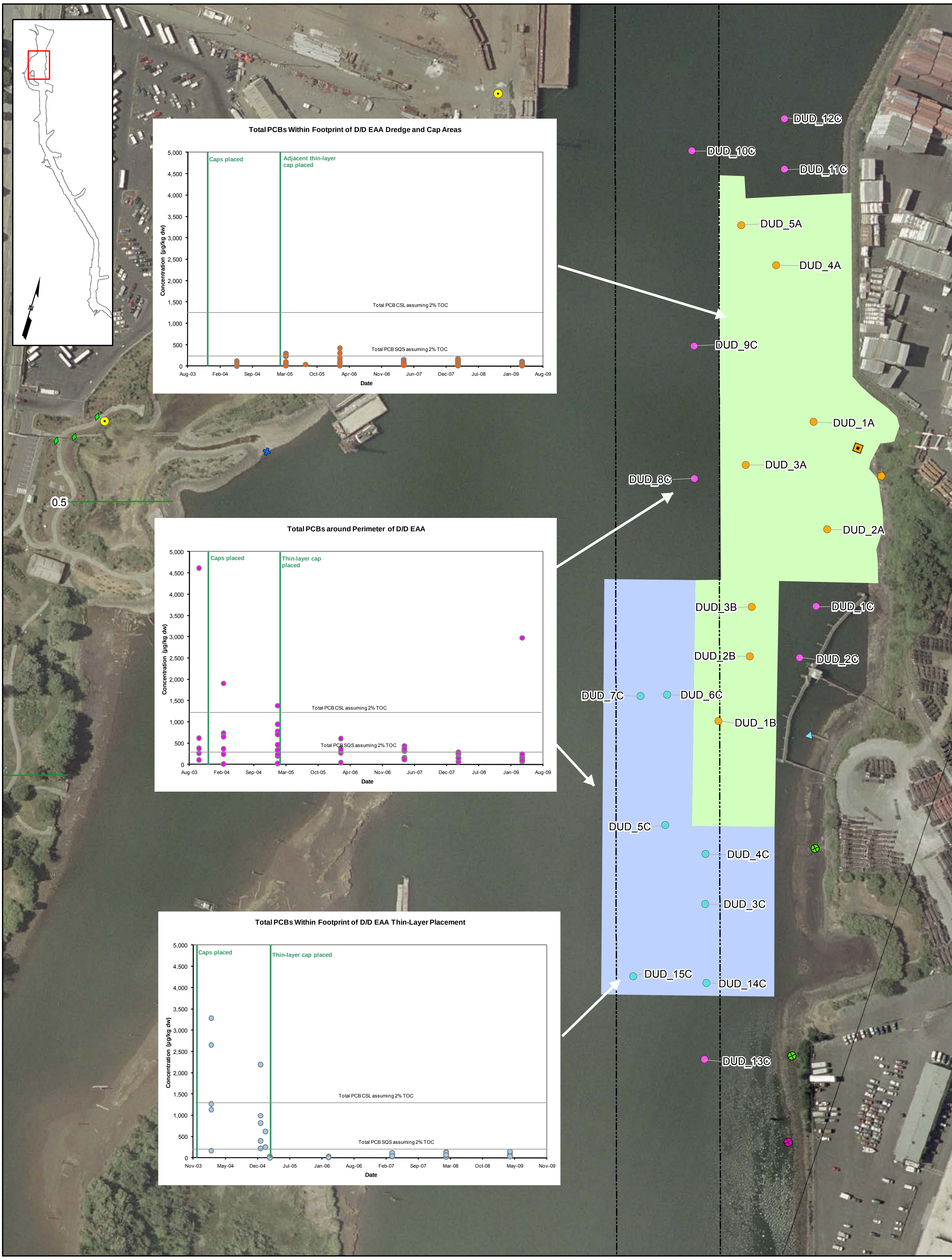
Figure 7-3 Mechanical Placement of Cap at Ward Cove, Alaska

Source: Candidate Technologies Memorandum, Retec 2005.

Figure 7-4 Schematic of Reactive Cap from Anacostia River



Note:
This reactive core mat (RCM) was designed to accurately place a 1.25-cm thick sorbent (coke) layer in an engineered sediment cap in twelve 3.1-m × 31-m sections. The RCM was overlain with a 15-cm layer of sand to secure it. It was placed in the Anacostia River (Washington D.C.) during the Anacostia River Active Capping demonstration project in April of 2004 (McDonough et al. 2006).



Notes:
1. USGS 2002 photograph provided by Windward Environmental.
2. Data provided by Windward Environmental in Access database accompanying Final Remedial Investigation (Windward 2010).
3. CSO= combined sewer overflow; EOF= emergency overflow.
4. Outfalls shown were identified during a City of Seattle low-tide survey in 2003 (Herrera 2004). Some locations were initially identified using drainage maps from Ecology's National Pollutant Discharge Elimination System (NPDES) permit files and other relevant agency databases. These locations were later surveyed in the field. Review of agency files and interviews with agency and LDWG personnel provided additional outfall-specific information. Some locations were field-verified by LDWG members; some additional outfall locations were identified during these subsequent verifications.

Legend

Dredge and Cap (2003-2004)
Thin-layer Sand Placement (2005)

Outfall Location

- Private storm drain
- Permitted private storm drain
- Public storm drain
- CSO/storm drain
- EOF
- Stream, channel, or swale
- Abandoned
- Pipe of unresolved origin and/or use

Sampling Location

- Dredging/Cap
- Thin-layer Sand Placement
- Perimeter

River Mile Marker
Navigation Channel

0 120 240 480 Feet

AECOM
Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.39	Surface Sediment Total PCB Trends at Duwamish/Diagonal EAA
DATE: 10/31/12 DWRN:MVI/sea Revision: 0	FIGURE 7-5

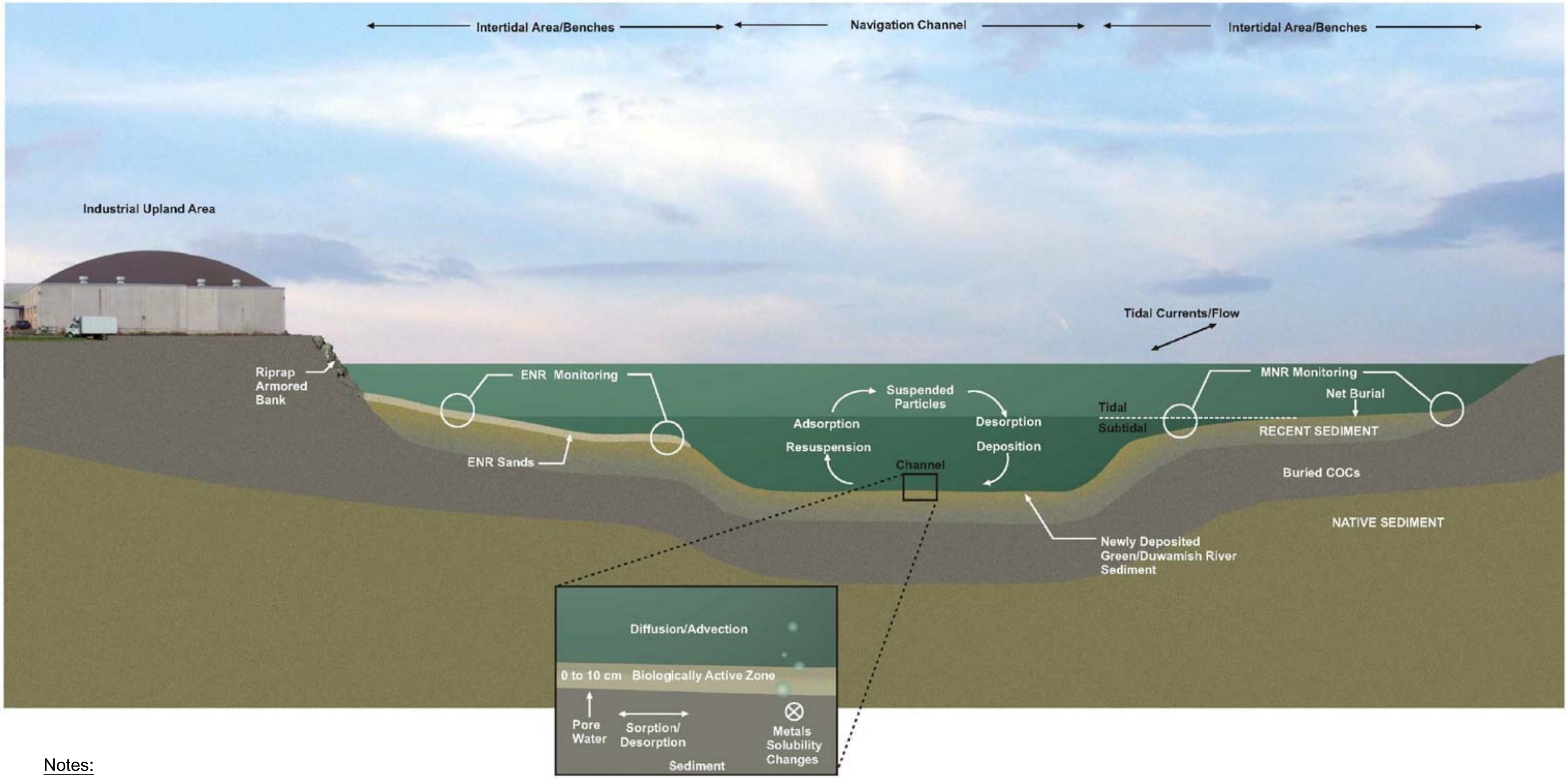
L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 7\Figure7-5SurfsedDD.mxd

Figure 7-6 Placement of Under-pier Capping Sand between Bents by Sand Throwing Barge

Source: Interim Construction Inspection Report, Todd Shipyards (McCarthy and Floyd|Snider 2005)

Figure 7-7 Finished Shotcrete Surface on Debris Mound

Source: Interim Construction Inspection Report, Todd Shipyards (McCarthy and Floyd|Snider 2005)



Notes:

COCs = Contaminants of Concern
ENR = Enhanced Natural Recovery
MNR = Monitored Natural Recovery

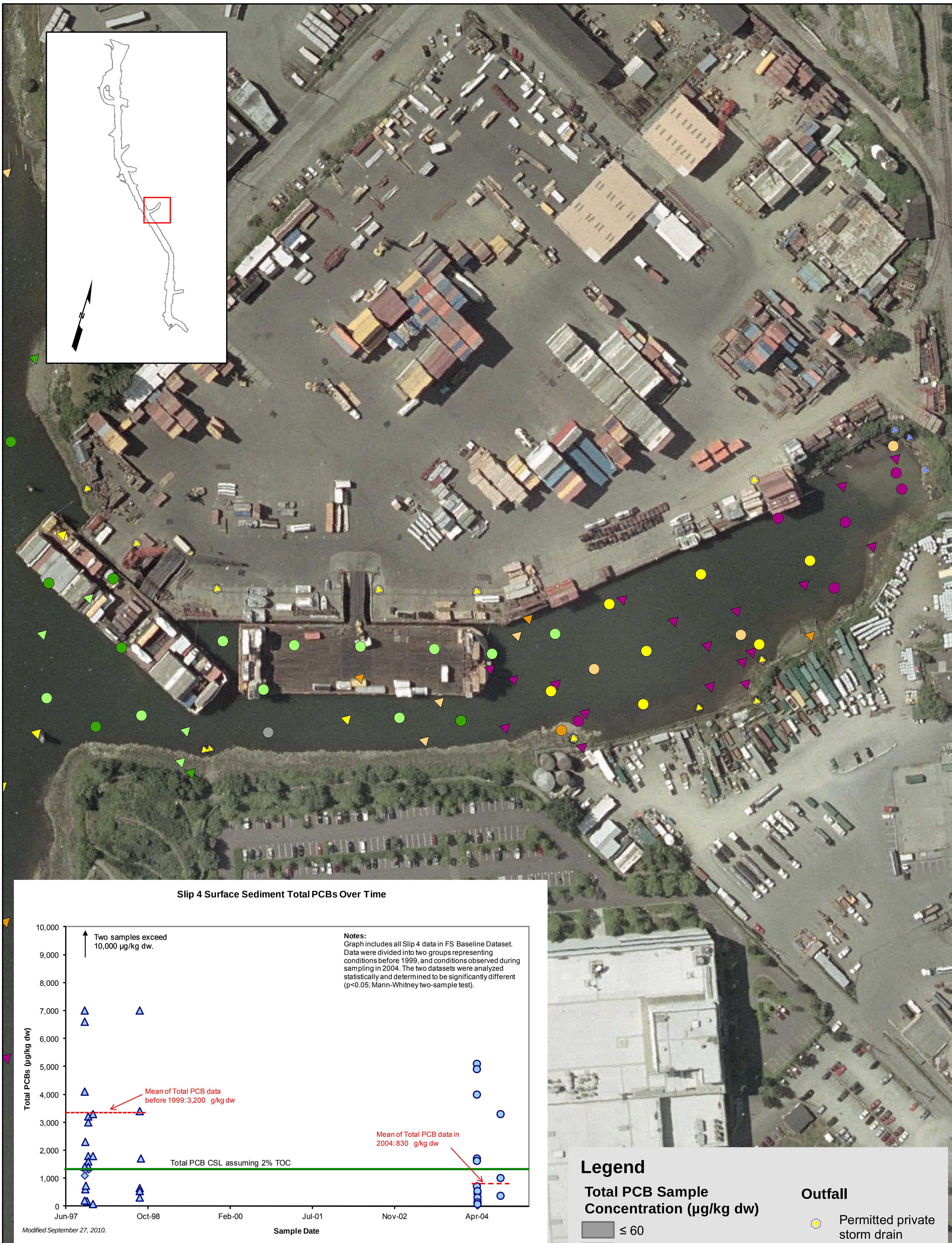
AECOM Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

LOWER DUWAMISH WATERWAY
FINAL FEASIBILITY STUDY
60150279-14.39

CONCEPTUAL DIAGRAM OF MONITORED NATURAL RECOVERY AND ENHANCED NATURAL RECOVERY

DATE: 10/31/12 | DRWN: E.M./SEA

FIGURE 7-8



Notes:
1. USGS 2002 photograph provided by Windward Environmental.
2. Outfalls shown were identified during a City of Seattle low-tide survey in 2003 (Herrera 2004). Some locations were initially identified using drainage maps from Ecology's National Pollutant Discharge Elimination System (NPDES) permit files and other relevant agency databases. These locations were later surveyed in the field. Review of agency files and interviews with agency and LDWG personnel provided additional outfall-specific information. Some locations were field-verified by LDWG members; some additional outfall locations were identified during these subsequent verifications.
3. SQS value of 240 µg/kg dw based on conversion of 12 mg/kg oc to a dry weight value using 2% TOC.

Feet
0 50 100 200

AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.39			Surface Sediment Total PCB Trends in Slip 4
DATE: 10/31/12	DWRN:MVI/sea	Revision: 1	FIGURE 7-9

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 7\Figure7-9Slip4EAA.mxd

Figure 7-10 Sloping Drain Barge (Hylebos Waterway, Tacoma, WA)

Source: "Barge Dewatering on Contaminated Sediment Projects" WEDA Pacific Chapter 2007 Annual Meeting Honolulu, Hawaii. Presented by Integral Consulting Inc., October 26, 2007

4 Remedial Action Objectives and Preliminary Remediation Goals

This section of the feasibility study (FS) identifies narrative remedial action objectives (RAOs) and numerical preliminary remediation goals (PRGs) for cleanup of the Lower Duwamish Waterway (LDW). RAOs for the LDW describe what a proposed cleanup remedy is expected to accomplish to protect human health and the environment (EPA 1999b) PRGs are the contaminant endpoint concentrations or risk levels associated with each RAO that are believed to be sufficient to protect human health and the environment based on available site information (EPA 1997b).

The step of identifying narrative RAOs provides a transition between the findings of the human health and ecological risk assessments and development of remedial alternatives in the FS. The RAOs pertain to the specific exposure pathways and receptors evaluated in the risk assessments and for which unacceptable risks were identified.

RAOs are developed herein for cleanup of contaminated sediment in the LDW Superfund site. Surface water within the site is also a medium of concern. However, no active remedial measures are anticipated for the water column. Improvements in surface water quality are expected following sediment cleanup and implementation of upland source control measures. Further, water quality monitoring will be part of long-term monitoring for the site.

PRGs are intended to protect human health and the environment and to comply with applicable or relevant and appropriate requirements (ARARs) for specific contaminants (EPA 1991b). For the LDW, PRGs are numerical concentrations or ranges of concentrations in sediment that protect a particular receptor from exposure to a hazardous substance by a specific pathway. The PRGs are expressed as sediment concentrations for the identified risk drivers because the alternatives in this FS address cleanup of contaminated sediments. PRGs are not developed in this FS for surface water because actions to directly address water quality are not included among the FS alternatives. Instead, surface water quality will be discussed as water quality ARARs, which are equivalent to PRGs. The RAOs, ARARs, and PRGs presented here may be modified and will be finalized by the U.S. Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology) in the Record of Decision (ROD).

4.1 Development of Remedial Action Objectives

The RAOs are narrative statements of the medium-specific or area-specific goals for protecting human health and the environment. RAOs describe in general terms what the sediment cleanup will accomplish for the LDW. RAOs help focus the development and evaluation of remedial alternatives and form the basis for establishing PRGs.

EPA's *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988) specifies that RAOs are to be developed based on the results of the human health risk assessment (HHRA) and ecological risk assessment (ERA). Other EPA guidance (EPA 1991a, 1999a) states that RAOs should specify:

- The exposure pathways, the receptors, and the contaminants of concern (COCs)
- An acceptable concentration or range of concentrations for each exposure pathway.

Section 2 summarized the remedial investigation (RI), including the chemical and physical conceptual site model. Section 3 summarized the results of the risk assessments, which identified receptors, exposure pathways, risk drivers, and, where calculable, risk-based threshold concentrations (RBTCs). The RAOs presented here were crafted based on the RI and findings from the baseline ERA and HHRA (Windward 2010, 2007a, 2007b).

4.1.1 Remedial Action Objectives for the Lower Duwamish Waterway

The results of the baseline HHRA and ERA indicate that remedial action is warranted to reduce unacceptable human health and ecological risks posed by COCs in LDW sediments. Unacceptable risks were estimated for certain human health exposure scenarios (through seafood consumption and direct contact exposure pathways) and for certain ecological risks (for benthic organisms and for other ecological receptors).

For human health, EPA defines a generally acceptable risk range for excess cancer risks as between one in ten thousand (1×10^{-4}) and one in one million (1×10^{-6}) (i.e., the "target risk range") and for non-cancer risks a hazard index (HI)[1] of 1 or less is considered acceptable (EPA 1991a). Excess cancer risks greater than 10^{-4} or HIs greater than 1 generally warrant a response action (EPA 1997b).

To establish cleanup levels and remedial action levels (RALs), the Washington State Model Toxics Control Act (MTCA) specifies that individual excess cancer risks for identified COCs should be 1×10^{-6} or less, and total excess cancer risks (all carcinogens combined) should not exceed one in one hundred thousand (1×10^{-5}). Cleanup levels should be adjusted downward to take into account exposure to multiple hazardous substances if the total excess cancer risk exceeds 1×10^{-5}. MTCA also specifies that risks resulting from exposure to multiple hazardous substances may be apportioned among hazardous substances in any combination as long as: 1) the total excess cancer risk (all carcinogens combined) does not exceed 1×10^{-5}; and 2) the health threats resulting from exposure to two or more non-carcinogenic hazardous substances with similar types of toxic response does not exceed an HI of 1 (WAC 173-340-708).

1 HIs are calculated as the sum of hazard quotients with similar non-cancer toxic endpoints.

Based on guidance provided by EPA under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and other requirements provided in MTCA/Sediment Management Standards (SMS), four RAOs have been identified for the cleanup of LDW sediments. These RAOs are identified below, and a discussion of each RAO follows.

RAO 1: Reduce human health risks associated with the consumption of resident LDW fish and shellfish by reducing sediment and surface water concentrations of COCs to protective levels.

Lifetime excess cancer risks from human consumption of resident LDW seafood are estimated to be greater than 1×10^{-6} for some individual carcinogens, and greater than 1×10^{-4} for carcinogens cumulatively under reasonable maximum exposure (RME) seafood consumption scenarios. In addition, the estimated non-cancer risks exceed an HI of one (see Tables 3-4a and 3-4b of Section 3). These estimated risks warrant response actions to reduce exposure.

Total polychlorinated biphenyls (PCBs), arsenic, and carcinogenic polycyclic aromatic hydrocarbons (cPAHs) are the primary risk drivers that contribute to the estimated risks based on consumption of resident seafood. As discussed in Section 3, although risks associated with consumption of dioxins/furans in resident seafood were not quantitatively assessed in the baseline HHRA, those risks were assumed to be unacceptable; thus, dioxins/furans are also considered risk drivers with respect to the consumption of resident seafood.

Achieving RAO 1 requires that site-wide average[2] concentrations of COCs in sediment be reduced, which in turn is expected to reduce tissue COC concentrations in fish and shellfish exposed to these sediments. Exposure of fish and shellfish to COCs in sediment occurs within the biologically active zone. As reported in the RI (Windward 2010), this zone is estimated to be the upper 10 cm of sediment. Deeper, undisturbed sediments contribute negligibly to the risks addressed by this RAO if contaminants in these deeper sediments do not migrate into the biologically active zone. However, deeper sediments that contain contaminants at concentrations above action levels and that are potentially subject to disturbance (e.g., erosion, propeller scour, earthquakes) or otherwise may migrate into the biologically active zone through advection or other mechanisms may warrant response actions to satisfy this RAO.

With regard to seafood consumption, bioaccumulative COCs enter the food web from both sediment and water. For example, the food web model used to predict tissue PCB concentrations (refer to Appendix D of the RI; Windward 2010) assumes that the

2 The FS uses average concentrations to evaluate the effectiveness of alternatives in attaining RAOs. In practice, compliance with clean-up levels will be based on the 95% upper confidence limit on the mean (UCL95).

exposure of fish and shellfish to PCBs occurs through their exposure to both sediments and surface water.

Substantial reductions in the concentrations of such COCs in sediment achieved through remediation should also reduce the concentrations of those COCs in surface water, thereby contributing to reducing their concentrations in fish and shellfish tissue and ultimately reducing human health risks, as stated in RAO 1. The relationships between sediment, surface water, and tissue concentrations are complex, and will be assessed through long-term monitoring of the remedial actions.

RAO 2: Reduce human health risks associated with exposure to COCs through direct contact with sediments and incidental sediment ingestion by reducing sediment concentrations of COCs to protective levels.

Lifetime excess cancer risks from human direct contact and incidental sediment ingestion RME scenarios (netfishing, tribal clamming, and beach play) are estimated to be within EPA's 10^{-4} to 10^{-6} target risk range (Tables 3-6a and 3-6b of Section 3) for the individual risk drivers. Some individual excess cancer risks exceed 1×10^{-6}, and total risks from all risk drivers exceed 1×10^{-5}, both of which are MTCA thresholds. Therefore, the risks associated with these exposure pathways warrant response actions to reduce exposure. No HIs were greater than 1 for any of the direct contact or incidental ingestion sediment RME scenarios, with the exception of one individual beach (Beach 4). Total PCBs, arsenic, cPAHs, and dioxins/furans are the primary risk drivers that contribute to the estimated excess cancer risks, and total PCBs are also a risk driver for noncancer risks based on direct contact.

Achieving RAO 2 requires that average concentrations of COCs be reduced at locations and depths within the sediment where people have the potential to be exposed. For netfishing activities, exposure is over the entire LDW and to surface sediments (0 to 10 cm). Direct contact risks in the beach play and clamming areas are assumed to result from exposure to the upper 45 cm depth interval, which accounts for potential exposures to children and clammers, who may dig holes deeper than 10 cm. Deeper sediments in other areas do not contribute appreciably to these risks unless they could be exposed by future disturbances (e.g., erosion, propeller scour, earthquakes). Achieving and maintaining this RAO may include response actions to address deeper sediments containing concentrations of the risk drivers above action levels if such disturbances of the overlying sediments over time may potentially expose these sediments.

RAO 3: Reduce risks to benthic invertebrates by reducing sediment concentrations of COCs to comply with the Washington State SMS.

The SMS provide both chemical and biological effects-based criteria. The numerical SMS chemical criteria are available for 47 contaminants or groups of contaminants (i.e., sediment quality standards [SQS] and cleanup screening levels [CSL]). These numerical chemical criteria are based on apparent effects thresholds (AETs) developed for four different benthic endpoints by the Puget Sound Estuary Program (PSEP) (Barrick et al. 1988). An AET is the highest "no effect" sediment concentration of a specific contaminant above which a significant adverse biological effect always occurred among the several hundred samples used in its derivation. In general, the lowest of the four AETs for each contaminant was identified as the SQS; the second lowest AET was identified as the CSL. According to the SMS (WAC 173-204), locations with all contaminant concentrations less than or equal to the SQS are defined as having no acute or chronic adverse effects on biological resources, locations with any contaminant concentrations between the SQS and the CSL are defined as having minor adverse effects, and locations with any contaminant concentration greater than the CSL are defined as having more pronounced adverse effects (refer to Section 5 of the RI, Windward 2010).

The baseline ERA (Windward 2007a) reported that 41 contaminants were detected in surface sediment at one or more locations within the LDW at concentrations exceeding their respective SQS (see Table 3-1, Section 3 of this FS). Thus, the ERA determined that these 41 contaminants are COCs because they pose a risk to the benthic invertebrate community. These 41 COCs are designated as risk drivers for this pathway.

Benthic organisms reside primarily in the biologically active zone (uppermost 10 cm) of intertidal and subtidal sediments of the LDW (Section 2 of the RI, Windward 2010). Deeper sediments in areas subject to disturbance (e.g., erosion, propeller scour, earthquakes) that contain COCs at concentrations above the SQS may warrant response actions to satisfy RAO 3.

RAO 4: Reduce risks to crabs, fish, birds, and mammals from exposure to COCs by reducing concentrations of COCs in sediment and surface water to protective levels.

The ERA (Windward 2007a) determined that exposure to seven contaminants, identified as COCs, exceeded toxicity benchmarks for fish, birds, or mammals. In consultation with EPA and Ecology, total PCBs were designated as the risk driver associated with seafood consumption based on estimated risks to river otters. Thus, achievement of RAO 4 is based on addressing PCB risk to river otters (see Section 3.1.3 for discussion of other ecological COCs).

River otters are indirectly exposed to PCBs in sediment primarily through the consumption of prey. Therefore, achieving this RAO requires that site-wide average concentrations of PCBs in sediment be reduced, with the expectation that sediment cleanup will reduce PCB concentrations in fish and shellfish, and that concentrations of the remaining six COCs identified for this exposure pathway will also be reduced to acceptable levels for other receptors (Windward 2010).

The potential for exposure of prey to COCs occurs primarily within the biologically active zone (upper 10 cm of sediment). Deeper sediments, if left undisturbed, contribute negligibly to the risks addressed by this RAO. Deeper sediments in areas subject to disturbance (e.g., erosion, propeller scour, earthquakes) that contain COCs at concentrations above action levels may warrant response actions to satisfy RAO 4.

Remediation will reduce COC concentrations in the LDW sediments; this in turn should also reduce those same COC concentrations in surface water, thereby contributing to a reduction of their concentrations in the tissue of fish and shellfish (including prey species). The relationships between sediment, surface water, and tissue concentrations are complex, and will be assessed through long-term monitoring following completion of the remedial actions.

4.1.2 Role of Source Control

Controlling sources of contamination to the LDW to the maximum extent practicable is an explicit MTCA expectation when natural attenuation is part of the remedial action (WAC 173-340-370). Active sediment remediation of COCs that have accumulated in sediments over time will address a major portion of the risks addressed in each RAO; however, without continued source control to keep reducing COC inputs to the LDW, sediments will likely recontaminate and water quality may continue to be impaired. Source control must include continued involvement by the Source Control Work Group (SCWG) to protect the long-term investments in the LDW cleanup.

Contaminated media from within the LDW drainage basin can affect sediments through several pathways, which can be organized into seven general types based on the origin of contamination, pathways to sediments, and the types of source control available:

- Direct discharge into the LDW (e.g., CSOs, storm drains)
- Surface water runoff or sheet flow
- Spills and/or leaks to the ground, surface water, or directly into the LDW
- Groundwater migration/discharge
- Bank erosion/leaching
- Atmospheric deposition
- Transport of resuspended contaminated sediments.

Understanding how each of these potential sources and pathways may impact a given sediment area is a complex undertaking and beyond the scope of this FS. Whether additional localized source control actions, beyond what has already been done, are needed before in-water work can begin will be considered in remedial design. This will require a recontamination/source control assessment study that varies in scope and magnitude depending on the specific project area.

Currently, source identification and implementation of effective control efforts in the LDW watershed are supported by a cooperative interagency program with the goal of identifying sources of potential contamination and recontamination in coordination with sediment cleanups and promoting their control. Ecology, as the lead entity for implementing source controls in the LDW, formed the LDW SCWG in 2002, which conducts several source control activities within the LDW area. The SCWG is composed primarily of public agencies responsible for source control, including EPA, Seattle Public Utilities, King County, and the Port of Seattle. The LDW source control strategy (Ecology 2004) also identifies various regulatory programs at EPA and Ecology that are called upon as needed for source control as well as several *ad hoc* members of the SCWG, including the City of Tukwila, Puget Sound Clean Air Agency, and Washington State Departments of Transportation (WSDOT) and Health (WDOH). All LDW SCWG members are public agencies with various source control responsibilities; the group's collective purpose is to share information, identify issues and data gaps, develop action plans for source control tasks, coordinate implementation of various source control measures, and share progress reports on these activities. Individually, these agencies are able to use their regulatory authority to promote source control in the LDW via source tracing sampling, stormwater and combined sewer overflow (CSO) programs, permits, hazardous waste management and pollution prevention programs, inspection and maintenance programs, water quality compliance and spill response programs, and environmental and pathway assessments.

Ecology's *Lower Duwamish Waterway Source Control Strategy* (Ecology 2004) is consistent with sediment source control protocols described in EPA guidance (2002b) and the SMS (Ecology 1995). The strategy describes the process and timing for implementing source control and the roles of various regulatory agencies responsible for conducting source control (e.g., SCWG) and enforcement. The strategy also provides for tracking and documenting source control progress in the LDW.

The focus of the LDW source control strategy is to identify and manage sources of COCs to waterway sediments in coordination with sediment cleanups and to prevent post-cleanup recontamination to levels exceeding cleanup goals established in the ROD to the extent practicable (Ecology 2004). Specific goals for the source control program are:

- Minimize the potential for contaminants in sediments to exceed the SMS criteria (as stated in WAC 173-204) and the LDW sediment cleanup levels (to be established in the ROD).
- Achieve adequate source control that will allow sediment cleanups to begin.
- Increase opportunities for natural recovery of sediments.

- Support long-term suitability and success of current and future habitat restoration opportunities.

Source control started in 2002 and is an ongoing, iterative process that continually produces new information. During remedial design, the work accomplished by Ecology and other public entities will serve as a foundation for any additional source control investigations and actions necessary before implementing various components of the sediment cleanup.

4.2 Applicable or Relevant and Appropriate Requirements (ARARs)

CERCLA Section 121(d) requires remedial actions to achieve (or formally waive) ARARs, which are defined as any legally applicable or relevant and appropriate standard, requirement, criterion, or limitation under any federal environmental law, or promulgated under any state environmental or facility siting law that is more stringent than the federal law. Similarly, MTCA requires that all cleanup actions comply with all legally applicable or relevant and appropriate requirements in applicable state and federal laws, as set forth in WAC 173-340-710. Given these substantive similarities in language between CERCLA and MTCA on the role of legal requirements, the FS uses the term ARARs to identify requirements that will satisfy or comply with both statutes. This subsection identifies ARARs for cleanup of the LDW. Section 9 of this document evaluates whether the remedial alternatives developed for cleanup of the LDW comply with these ARARs.

The National Contingency Plan (40 CFR 300.5) defines applicable requirements as the more stringent among those cleanup standards, standards of control, and other substantive requirements, criteria, or limitations promulgated under federal environmental or state environmental or facility siting laws that specifically address a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstances found at a CERCLA site. A requirement may not be applicable, but nevertheless may be relevant and appropriate. Relevant and appropriate requirements address problems or situations sufficiently similar to those encountered at CERCLA and MTCA sites that their use is well-suited to the particular site. Relevant and appropriate requirements have the same effect as applicable requirements. They are not treated differently in any way.

Washington State has promulgated environmental laws and regulations to implement or co-implement several major federal laws through federally approved programs, for example, the Clean Water Act, Clean Air Act, and RCRA. The ARAR is the more stringent of either a federal requirement or a state requirement. Because this FS is being conducted under a joint CERCLA and MTCA order, applicable or relevant and appropriate provisions of MTCA and the SMS are considered to be ARARs for CERCLA, as well as governing requirements under MTCA. MTCA is a particularly important CERCLA ARAR. As will be seen, its background standards for final sediment cleanups are more stringent, and its allowable excess cancer risk standards are considerably more stringent. CERCLA permits risk-based cleanup standards within a range of 10^{-4} to 10^{-6} excess cancer risks. EPA policy and guidance recommends trying to achieve the more stringent 10^{-6} standard but accepts lesser standards within the range based on many factors. MTCA requires risk-based cleanup standards to be set at one in one million (1×10^{-6}) excess cancer risk levels for all individual carcinogens (such as PCBs) at a site, and a total excess cancer risk of one in one-hundred thousand (1×10^{-5}) for all carcinogens cumulatively at a site. Procedural requirements under state laws (e.g., MTCA disproportionate cost analysis methodology) are not CERCLA ARARs, but are required to comply with MTCA.

Table 4-1 lists and summarizes ARARs for the LDW site. Some ARARs prescribe minimum numerical requirements or standards for cleanup of specific media such as sediment, surface water, fish tissue, and groundwater. Other ARARs place requirements or limitations on actions that may be undertaken as part of a remedy. Table 4-2 lists other requirements or laws that are not considered ARARs by EPA and Ecology, generally because their primary purpose is not environmental protection (or state facility siting), but rather, for example, historical preservation of archaeological artifacts, endangered species, or workplace protection. Consideration of or compliance with requirements under these laws is anticipated for implementing most of the alternatives in this FS. While all federal, state, and local laws have to be complied with (except the need to acquire federal, state, or local permits for onsite cleanup work), it is helpful in considering remedial alternatives to list other laws or requirements alongside ARARs that will be implemented.

Some ARARs contain numerical values or methods for developing such values. These ARARs establish minimally acceptable amounts or concentrations of hazardous substances that may remain in or be discharged to the environment, or minimum standards of effectiveness and performance expectations for the remedial alternatives. RBTCs based on risks to human health or the environment may dictate setting more stringent standards for remedial action performance, but they cannot be used to relax the minimum legally prescribed standards in ARARs. The rest of this subsection focuses on ARARs containing specific minimum numerical standards.

There are no federal ARARs providing numerical standards for hazardous substances, pollutants, or contaminants in sediment. However, Washington State has promulgated numerical standards in the SMS for the protection of benthic invertebrates, and these regulations are cross-referenced in MTCA. Under CERCLA, the SMS criteria are considered ARARs and are promulgated standards for the LDW under MTCA. However, although the SMS contain narrative standards to protect human health and other biological resources, no SMS or other state numerical sediment criteria have been established to protect human health, including human consumers of seafood, or for other biological resources such as birds, fish, or mammals. Cleanup levels or standards for protection of these receptors are derived from RBTCs developed during the risk assessments performed during the LDW RI (Windward 2010).

Surface water (i.e., the water column) is also a medium of concern in the LDW. Therefore, federal water quality criteria (WQC) developed to protect ecological receptors and human consumers of fish and shellfish are relevant and appropriate requirements or minimum levels or standards for remedial action pursuant to CERCLA Section 121 (d)(2)(A)(ii) and RCW 70.105D.030(2)(e). Under CERCLA and MTCA, state water quality standards (WQS) approved by EPA are generally applicable requirements under the Clean Water Act (CWA). National recommended federal WQC established pursuant to Section 304(a)(1) of the CWA are compiled and presented on the EPA website at http://www.epa.gov/waterscience/criteria/wqctable/. Although these criteria are advisory for CWA purposes (to assist states in developing their standards), the last sentence of CERCLA Section 121(d)(2)(A)(ii) makes them minimum cleanup levels or standards, where relevant and appropriate under the circumstances, for CERCLA site remedial actions.

Consequently, the more stringent of the federal WQC and the state WQS are the cleanup levels or standards for the site. Washington State WQS for the protection of aquatic life are found at WAC 173-201A-240. The numerical criteria for aquatic life meet the federal requirements of Section 303(c)(2)(B) of the CWA and are at least as stringent as the federal WQC. Table 4-3 presents state and federal marine and freshwater values that have been developed for aquatic life and human health WQC. Specific considerations for compliance with federal and state aquatic life WQC and human health WQC are discussed in Section 4.2.2 of the RI (Windward 2010).

4.3 Process for Development of Preliminary Remediation Goals

PRGs are the COC endpoint concentrations initially identified for each RAO that are believed to be sufficient to protect human health and the environment based on available site information (EPA 1997b). The PRGs are used in the FS to guide the geographic definition of areas of potential concern (AOPCs) and the evaluation of proposed sediment remedial alternatives. PRGs are not final CERCLA/MTCA cleanup levels and standards. EPA and Ecology will select CERCLA/MTCA cleanup levels and standards in the ROD.

PRGs are developed in this subsection for each risk-driver COC, and are expressed as sediment concentrations that are intended to achieve the corresponding RAO. PRGs are based on considering the following factors:

- ARARs, including MTCA risk requirements, and SMS criteria
- RBTCs based on the human health and ecological risk assessments

- Background concentrations if protective RBTCs are below background concentrations
- Analytical practical quantitation limits (PQLs) if protective RBTCs are below concentrations that can be quantified by chemical analysis.

This section presents the numerical criteria in these categories to enable a comprehensive analysis and identification of PRGs. The pertinent information is then compiled and numerical PRGs are identified for each risk driver and each RAO.

4.3.1 Role of ARARs

Certain PRGs in this FS are set based on MTCA's more stringent (than CERCLA) excess cancer risk standards and its requirement that final cleanups achieve natural background levels when RBTCs are below background. The SMS (WAC 173-204) also contain numerical sediment contaminant concentration criteria pertinent for protecting the marine benthic invertebrate community (and hence the SMS criteria apply to PRGs for RAO 3).[3]

The SMS chemical and biological criteria are applied on a point basis to the biologically active zone of the sediments (i.e., upper 10 cm). Under the SMS, sediment cleanup standards may be established on a site-specific basis within an allowable range of contamination. The SQS, also called the sediment cleanup objective, and the CSL, also called the minimum cleanup level (MCUL), define this range. WAC 173-204-570(4) specifies that the site-specific cleanup standards shall be as close as practicable to the cleanup objective (the SQS) but in no case shall exceed the minimum cleanup level (the CSL). For this reason, in developing PRGs and analyzing alternatives, the SQS is used in this FS.[4] This WAC subsection also states that the cleanup standards shall be defined in consideration of the net environmental effects, cost, and engineering feasibility of different cleanup alternatives. The following WAC subsection (WAC 173-204-570(5)) emphasizes that all cleanup standards must ensure protection of human health (for which there are no SMS numerical criteria) and the environment (which encompasses receptors beyond the benthic invertebrate community). The SMS also require that contaminant concentrations (and toxicity) meet the cleanup standards within a reasonable time frame, as defined by a number of factors in WAC 173-204-580(3)(a).

As described in Section 4.2, surface water quality criteria are ARARs for the site because the water column is part of the site. The water column is affected by the sediment contaminant concentrations, as well as other factors, including ongoing releases, inflowing water from the Green/Duwamish River system, direct discharges to the LDW, and aerial deposition. However, the water column cannot practicably be directly

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[3] The SMS are ARARs under CERCLA and promulgated numerical standards under MTCA.

[4] Co-located sediment toxicity test results that "pass," (i.e., indicate no toxicity) override exceedances of the SMS numerical criteria only for determining compliance with RAO 3.

remediated. Thus, while surface water is included as a medium of concern to be addressed by RAOs 1 and 4, surface water quality ARARs have not been identified as numerical PRGs at the site. However, because the WQC are CERCLA ARARs, the quality of LDW surface water will have to meet the more stringent of the federal and state aquatic life and human health WQC (Table 4-3) or be waived at or before completion of CERCLA remedial action.

Significant water quality improvements are anticipated as a result of sediment remediation and source control. Water quality monitoring will be part of the selected remedy to help measure the efficacy of sediment remediation and source control, and to assess compliance with ARARs. The remedial alternatives developed and evaluated in this FS may not comply with all surface water quality standards, or with natural background sediment standards required under MTCA in lieu of protective human seafood consumption RBTCs, in which case surface water quality and MTCA ARAR waivers could be issued by EPA at or before the completion of the remedial action. Potential ARAR waivers are listed in Section 121(d)(4) of CERCLA. The most common waiver is for technical impracticability, the standards for which are explained in detail in comprehensive EPA guidance designed to ensure a rigorous evaluation, and that only genuine demonstrated technical impracticability will qualify.

4.3.2 Role of RBTCs

The RI developed site-specific sediment RBTCs (summarized in Section 3.3 of this document) for each of the risk-driver COCs. RBTCs for human health were calculated based on risks associated with the direct sediment contact RME scenarios and seafood consumption RME scenarios. RBTCs for wildlife receptors were calculated based on prey consumption by river otters. For the benthic invertebrate community, RBTCs were set at the SQS and CSL.

Total PCBs, cPAHs, arsenic, and dioxins/furans are the risk drivers for the human seafood consumption pathway. Sediment RBTCs for total PCBs were calculated for the 1×10^{-4} excess cancer risk level and are applied as site-wide average concentrations.[5] As discussed in Section 3.3, sediment RBTCs based on the seafood consumption pathway were not calculated for arsenic and cPAHs, because correlations between sediment contaminant concentrations and receptor tissue concentrations could not be established. Sediment RBTCs were also not calculated for dioxins/furans. Fish and shellfish tissue data were not collected for this risk driver during the RI because it was determined that sediment concentrations would exceed RBTCs, which would be more stringent than

5 For the excess cancer risk levels of 1 in 1,000,000 (1×10^{-6}) and 1 in 100,000 (1×10^{-5}) and for the non-cancer HQ of 1, even at a total PCB concentration of 0 µg/kg dw in sediment, the food web model predicted total PCB concentrations in tissue that would result in a risk estimate greater than the risk levels for the RME seafood consumption scenarios because of the contribution of total PCBs from water alone, even at concentrations similar to those in upstream water (i.e., 0.3 ng/L). Therefore, sediment RBTCs for these risk levels were represented as "< 1" (see Table 3-9).

natural background, resulting in natural background concentrations in sediment being the PRG for dioxins/furans.

Total PCBs, cPAHs, arsenic, and dioxins/furans are also the human health risk drivers for the direct sediment contact pathway. Sediment RBTCs for these hazardous substances were presented in Table 3-10 for each of the three direct sediment contact RME scenarios (i.e., netfishing, tribal clamming, and beach play). These sediment RBTCs are average concentrations applied to the spatial area over which exposure would reasonably be expected.

A total PCB sediment RBTC was calculated to protect wildlife. It protects river otters as the most sensitive representative wildlife species from the ERA, based on their consumption of prey species (Windward 2007a). The RBTC is applied as a site-wide average concentration.

4.3.3 Role of Background Concentrations

Both CERCLA and MTCA consider background hazardous substance concentrations when formulating PRGs and cleanup levels. Both recognize that setting numerical cleanup goals at levels below background is impractical (because of the potential for recontamination to the background concentration). MTCA (WAC 173-340-200) defines natural background as the concentrations of hazardous substances that are consistently present in an environment that have not been influenced by localized human activities. Thus, under MTCA, a natural background concentration can be defined for man-made compounds even though they may not occur naturally (e.g., PCBs deposited by atmospheric deposition into an alpine lake). According to CERCLA guidance, natural background refers to substances that are naturally present in the environment in forms that have not been influenced by human activity (e.g., naturally occurring metals).

MTCA cleanup levels cannot be set at concentrations below natural background (WAC 173-340-705(6)). Similarly, CERCLA guidance states that natural background concentrations establish a limit below which a lower cleanup level cannot be achieved (EPA 2005b).

Both cleanup programs also recognize that natural and man-made hazardous substance concentrations can occur at a site in excess of natural background concentrations, not as a result of local site-related releases but caused by human activities in areas remote from the site and natural processes that transport the contaminants to the site (e.g., atmospheric uptake, transport, and deposition). CERCLA defines “anthropogenic background” as natural and human-made substances present in the environment as a result of human activities, but not related to a specific release from the CERCLA site undergoing investigation and cleanup (EPA 2002c). MTCA defines the term “area background” as media-specific concentrations that are consistently present in the environment in the vicinity of a site that are attributable to human activities unrelated to specific releases from the site. CERCLA generally does not require cleanup to concentrations below anthropogenic background concentrations. In states that have a more stringent state standard, CERCLA cleanups must try to meet state ARARs, or EPA must waive the ARAR at or before completion of the remedial action. MTCA defines natural background as the cleanup standard required for final remedies when natural background concentrations are higher than the calculated risk-based cleanup levels (i.e., RBTCs). Thus, a CERCLA remedy in Washington State that cannot achieve natural background concentrations is not final unless this MTCA requirement is achieved or waived, or residual risks are otherwise sufficiently controlled. Under MTCA, because a waiver is not available, a remedy that cannot achieve natural background concentrations remains “interim” by default (see WAC 173-340-430) unless it is technically impossible to implement a more permanent cleanup action for all or a portion of the site (see WAC 173-340-360(2)(e)(iii)), and residual risks can be sufficiently controlled with institutional controls.

As a result, PRGs have been set at natural background concentrations for hazardous substances that have risk-based concentrations below natural background concentrations. EPA and Ecology recognize that natural background concentrations are unlikely to be achieved at the site and that long-term sediment contaminant concentrations following active sediment remediation will be governed primarily by concentrations in incoming sediment from the Green/Duwamish River system and new or continuing releases from other sources subject to further source control actions (see Section 5). Long-term monitoring will be used to determine what the technically practicable lower limits are for site concentrations, as well as where source control should continue to be focused. When these lower limits are reached, as demonstrated by monitoring data, a CERCLA technical impracticability (TI) waiver of the MTCA ARAR, in conjunction with institutional controls, could be used to provide administrative closure of the LDW cleanup. The TI waiver would address the gap between the technically practicable limit and natural background concentrations. Under MTCA, sufficient institutional controls that address remaining human health risks may similarly allow a final cleanup determination, where it is technically impossible to implement a more permanent cleanup action for all or a portion of the site (see WAC 173-340-360(2)(e)(iii)).

4.3.4 Natural Background in Sediment

This section presents estimates of natural background concentrations for total PCBs, arsenic, cPAHs, and dioxins/furans in sediment.[6] To characterize natural background, marine sediment data were compiled from areas within Puget Sound that have not been influenced by localized human activities. These data represent non-urban, non-localized concentrations that exist as a result of natural processes and/or the large-scale distribution of these hazardous substances from anthropogenic sources.

6 EPA and Ecology will set natural background concentrations and remediation goals in the ROD.

The Dredged Material Management Program (DMMP) (comprised of the U.S. Army Corps of Engineers [USACE], EPA, Ecology, and the Washington State Department of Natural Resources [DNR]) collected sediment data throughout Puget Sound in the summer of 2008 and documented the results in a study called *Final Report: Puget Sound Sediment PCB and Dioxin 2008 Survey, OSV BOLD SURVEY REPORT* (EPA OSV *Bold* Survey; EPA 2008b). EPA and Ecology have determined that the 95% upper confidence limit on the mean (UCL95) of the data from the EPA OSV *Bold* Survey will be used in this FS for natural background concentrations. Data were collected from 70 sampling locations throughout Puget Sound, as well as from the area around the San Juan Islands and the Strait of Juan de Fuca. Locations for each target sampling station are displayed in Figure 4-1. A subset of these sample locations (N = 20) were located within four reference areas (Carr Inlet, Samish Bay, Holmes Harbor, and Dabob Bay) established by Ecology. In each of these reference areas, five target sediment sampling locations were located based on a stratified random sampling design. The remaining 50 sample locations were spread throughout Puget Sound and the straits of Georgia and Juan de Fuca and were intended to represent areas outside the influence of urban bays and known point sources. At five stations, a duplicate sample (or field split) was collected for quality assurance purposes. Samples were analyzed for the full suite of DMMP contaminants, including semi-volatile organic compounds, PAHs, PCB Aroclors and PCB congeners, organochlorine pesticides, and trace metals, as well as for sediment conventionals (e.g., total organic carbon [TOC], grain size, percent solids). Summary statistics (see Table 4-4) were then calculated for the EPA OSV *Bold* Survey data for each of the four human health risk drivers using the statistical software ProUCL version 4.00.04. Statistical analyses of these sediment data did not adjust for the spatial bias resulting from repeated sampling of four reference areas, or other spatial aspects of how the sample locations were distributed.

4.3.4.1 Natural Background for Arsenic in Sediment

Arsenic was detected in all of the samples from the EPA OSV *Bold* Survey (Table 4-4). Concentrations ranged from 1.1 to 21 milligrams per kilogram dry weight (mg/kg dw), with a mean concentration of 6.5 mg/kg dw, and an UCL95 of 7.3 mg/kg dw. Using the UCL95 statistic, the background concentration for arsenic is rounded to 7 mg/kg dw.

4.3.4.2 Natural Background for Total PCBs in Sediment

Total PCBs as Aroclors were below reporting limits in the majority of sediment samples from the EPA OSV *Bold* Survey (Table 4-4). The PCB congener method, with its lower reporting limits, produced a detection frequency of 100%, based on quantifying at least one PCB congener in each sample. Total PCBs in each sample were calculated by summing the concentrations of all detected PCB congeners, consistent with the protocol in the SMS for reporting total PCBs by summing the concentrations of all detected PCB Aroclors. Using the congener results, total PCB concentrations ranged from 0.01 to 10.6 micrograms per kilogram (µg/kg) dw, with a mean of 1.2 µg/kg dw and an UCL95 of 1.5 µg/kg dw. Using the UCL95 statistic, the background concentration for total PCBs is rounded to 2 µg/kg dw.

4.3.4.3 Natural Background for cPAHs in Sediment

The detection frequency for cPAHs in the EPA OSV *Bold* Survey was 87%, based on quantifying at least one cPAH compound in each sample (Table 4-4). Total cPAHs in each sample were calculated by summing the concentrations of all detected cPAH compounds multiplied by their respective benzo(a)pyrene potency equivalency factors (PEFs), along with half the reporting limits of any undetected cPAH compounds multiplied by their respective PEFs. Concentrations ranged from 1.3 to 57.7 µg toxic equivalent (TEQ)/kg dw, with a mean concentration of 7.1 µg TEQ/kg dw and an UCL95 of 8.9 µg TEQ/kg dw.[7] Using the UCL95 statistic, the background concentration for cPAHs is rounded to 9 µg TEQ/kg dw.

4.3.4.4 Natural Background for Dioxins/Furans in Sediment

The detection frequency for dioxins/furans in the EPA OSV *Bold* Survey was 100%, based on quantifying at least one congener in each sample (Table 4-4). The total TEQ of dioxins/furans (relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) in each sample was calculated by summing the concentrations of certain detected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective toxic equivalency factors (TEFs), along with half the reporting limits of undetected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective TEFs. Concentrations ranged from 0.2 to 11.6 ng TEQ/kg dw, with a mean of 1.4 ng TEQ/kg dw (Table 4-4) and an UCL95 of 1.6 ng TEQ/kg dw.[8] Using the UCL95 statistic, the background concentration for dioxins/furans is rounded to 2 ng TEQ/kg dw.

4.3.5 Role of Practical Quantitation Limits

Both CERCLA and MTCA allow consideration of PQLs when formulating PRGs to address circumstances in which a concentration determined to be protective cannot be reliably detected using state-of-the-art analytical instruments and methods. For example, if an RBTC is below the concentration at which a contaminant can be reliably quantified, then the PRG for that contaminant may default to the analytical PQL.

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7 The uncertainty associated with handling the undetected cPAH data is negligible. To determine how nondetects affected the overall statistics, a sensitivity analysis was run. For this analysis, the concentrations of the undetected cPAH compounds were set to zero. The concentrations of the individual detected cPAH compounds were multiplied by their respective PEFs and the products were summed. The results indicate a mean of 6.9 µg TEQ/kg dw and an UCL95 of 8.0 µg TEQ/kg dw.

8 The uncertainty associated with handling the undetected dioxin/furan data is negligible. To determine how nondetects affected the overall statistics, a sensitivity analysis was run. For this analysis, the concentrations of the undetected dioxin/furan congeners were set to zero. The concentrations of the individual detected dioxin/furan congeners were multiplied by their respective TEFs and the products were summed. The results indicate a mean of 1.2 ng TEQ/kg dw and an UCL95 of 1.5 ng TEQ/kg dw.

MTCA defines the PQL as:

> *…the lowest concentration that can be reliably measured within specified limits of precision, accuracy, representativeness, completeness, and comparability during routine laboratory operating conditions, using department approved methods* (WAC 173-340-200).

In simpler terms, the PQL is the minimum concentration for an analyte that can be reported with a high degree of certainty.

Tables 4-5 and 4-6 list the risk-driver specific PQLs developed for the RI sediment sampling programs and documented in the associated quality assurance project plans. These PQLs represent the lowest values that can be reliably quantified when the sample matrix (in this case, sediment) is free of interfering compounds that can reduce sensitivity and raise reporting limits. Also, these tables present the range of actual sample PQLs reported by the laboratories for the data in the RI database. These results reflect the range of what the laboratories were able to achieve given the composition of and matrix complexity associated with LDW sediment samples.

Analytical quantitation limits are generally not expected to exceed RBTCs, SQS, or natural background concentrations for samples of low matrix complexity. However, empirical evidence from the RI suggests that, on a case-by-case basis, matrix interferences have the potential to preclude quantification to concentrations below the PRGs (and ultimately the cleanup levels and standards) established for cleanup of LDW sediments.

4.4 Preliminary Remediation Goals

PRGs for sediment are derived from a comparison of ARARs, RBTCs, background concentrations, and PQLs. For each RAO and risk driver, the PRG is the higher value between the natural background concentration and the lowest RBTC.[9] PQLs were also considered and were not found to influence selection of the PRGs (i.e., all PRGs are above PQLs). The RAOs and PRGs are used in Section 6 of the FS to identify AOPCs and were considered in selecting the RALs. Section 9 compares estimated concentrations of risk drivers to PRGs as one measure of the effectiveness of the remedial alternatives.

Tables 4-7 and 4-8 summarize the analysis and selection of sediment PRGs for the risk-driver COCs. Table 4-7 focuses on the four human health risk drivers and the wildlife risk driver, and is subdivided to address the various spatial applications of the PRGs for each RAO. Table 4-8 contains the PRG analysis for the remaining SMS risk drivers (i.e.,

[9] SQS and CSL values for the 41 SMS risk-driver COCs are the RBTCs for protection of benthic organisms. Sediment RBTCs were calculated (see Section 3) for protection of ecological receptors (river otters) and humans.

the risk-driver COCs for RAO 3). PRGs were not developed for the other COCs identified in the RI. The potential for risk reduction for the other COCs following remedial action is evaluated in Section 9.

The PRGs identified in Tables 4-7 and 4-8 are derived from RBTCs, natural background, or SQS values. The PRGs are applied on either a point basis or an average basis over a given exposure area depending on the COC, exposure pathway, and receptor of concern. PRGs for RAOs 1, 2, and 4 are applied on a site-wide average basis that requires a sediment spatially-weighted average concentration (SWAC) over the applicable exposure area to be below the PRG. These SWACs have been calculated to evaluate and compare remedial alternatives; ultimate compliance for remedial actions will be based on the UCL95.

For RAO 1, the numerical PRG for total PCBs is natural background because the sediment RBTCs[10] are below natural background for the RME seafood consumption scenarios. RBTCs were not derived for dioxins/furans (see Section 3.2.4), but were presumed also to be below natural background levels for the RME seafood consumption scenarios. Therefore, natural background is the PRG for dioxins/furans for RAO 1. Arsenic and cPAH PRGs were not identified for the human health seafood consumption pathway (RAO 1). Excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. Based on data collected during the RI, there is no credible relationship between cPAH or arsenic concentrations in sediment and concentrations in clam tissue (Section 8 of the RI, Windward 2010). However, the development and evaluation of remedial alternatives in the latter sections of the FS discuss the need for future investigations of the sediment/clam tissue relationships for arsenic and cPAHs. Further, meeting the PRGs defined in Tables 4-7 and 4-8 should lead to reductions in sediment concentrations of arsenic and cPAHs (see discussion of RALs in Section 6). PRGs based on natural background are unlikely to be achieved by any of the remedial alternatives developed in this FS. This is partly because of COC concentrations in inflowing sediment from the Green/Duwamish River system, as predicted in the bed composition model used in this FS (see Section 5). In addition, the urban setting of the LDW will make it difficult to achieve natural background for PCBs and dioxins/furans. However, in accordance with MTCA, natural background concentrations were used in this FS for setting background-based PRGs.

For RAO 2, PRGs are based on the sediment RBTCs (1×10^{-6} or natural background, whichever is higher) developed for three exposure scenarios: netfishing, tribal clamming, and beach play. PRGs are applied on a spatially-weighted average basis over

[10] Sediment RBTCs were calculated only for the 1×10^{-4} risk threshold. The contribution of PCBs in water alone (even at concentrations similar to those in upstream water) was high enough to result in seafood consumption risks for Adult and Child Tribal RME and Asian and Pacific Islander RME scenarios exceeding the 1×10^{-6} and 1×10^{-5} excess cancer risk thresholds even in the absence of any contribution from sediment (Table 3-9).

a given exposure area (e.g., site-wide for netfishing). Except for arsenic, the PRGs for the RAO 2 risk drivers are based on their RBTCs. The arsenic PRG for RAO 2 is based on natural background, which may be difficult to achieve by any of the remedial alternatives developed in this FS, for the same reasons explained above for total PCBs and dioxins/furans for RAO 1.

For RAO 3, the SMS numerical criteria apply on a point basis (Table 4-6). As noted in Section 4.3.1, WAC 173-204-570(4) specifies that the site-specific cleanup standards shall be as close as practicable to the cleanup objective (the SQS) but in no case shall exceed the minimum cleanup level (the CSL). For this reason, the PRGs for RAO 3 in this FS are set to the SQS. However, where co-located toxicity test data are available, sediment toxicity results override the numerical criteria for RAO 3. (However, toxicity test results do not override PRGs for RAOs 1, 2, and 4 because toxicity test results are only relevant for an assessment of effects on benthic fauna, not on other ecological or human receptors.)

For RAO 4, the PRG for seafood consumption by ecological receptors is set to the sediment RBTC for river otter (hazard quotient less than 1).

Table 4-1 ARARs for the Lower Duwamish Waterway

Topic	Standard or Requirement	Regulatory Citation: Federal	Regulatory Citation: State	Comment
Sediment Quality	Sediment quality standards; cleanup screening levels		Sediment Management Standards (WAC 173-204)	The SMS are MTCA rules and an ARAR under CERCLA. Numerical standards for the protection of benthic marine invertebrates.
Fish Tissue Quality	Concentrations of contaminants in fish tissues	Food and Drug Administration Maximum Concentrations of Contaminants in Fish Tissue (49 CFR 10372-10442)		The Washington State Department of Health assesses the need for fish consumption advisories.
Surface Water Quality	Surface Water Quality Standards	Ambient Water Quality Criteria established under Section 304(a) of the Clean Water Act (33 USC 1251 et seq) http://www.epa.gov/ost/criteria/wqctable/	Surface Water Quality Standards (RCW 90-48; WAC 173-201A)	State surface water quality standards apply where the State has adopted, and EPA has approved, Water Quality Standards that are more stringent than Federal recommended Water Quality Criteria established under Section 304(a) of the Clean Water Act. Both chronic and acute standards, and marine and freshwater are used as appropriate.
Land Disposal of Waste	Disposal of materials containing PCBs	Toxic Substances Control Act (15 USC 2605; 40 CFR Part 761)		
	Hazardous waste	Resource Conservation and Recovery Act Land Disposal Restrictions (42 USC 7401-7642; 40 CFR 268)	Dangerous Waste Regulations Land Disposal Restrictions (RCW 70.105; WAC 173-303, 140-141)	
Waste Treatment Storage and Disposal	Disposal limitations	Resource Conservation and Recovery Act (42 USC 7401-7642;40 CFR 264 and 265)	Dangerous Waste Regulations (RCW 70.105; WAC 173-303)	
Noise	Maximum noise levels		Noise Control Act of 1974 (RCW 80.107; WAC 173-60)	
Groundwater	Groundwater quality	Safe Drinking Water Act MCLs and non-zero MCLGs (40 CFR 141)	RCW 43.20A.165 and WAC 173-290-310	For on-site potable water, if any.

Table 4-1 ARARs for the Lower Duwamish Waterway (continued)

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Dredge/Fill and Other In-water Construction Work	Discharge of dredged/fill material into navigable waters or wetlands	Clean Water Act (33 USC 401 et seq.; 33 USC 141; 33 USC 1251-1316; 40 CFR 230, 231, 404; 33 CFR 320-330) Rivers and Harbors Act (33 USC 401 et seq.)	Hydraulic Code Rules (RCW 75.20; WAC 220-110)	For in-water dredging, filling, or other construction.
	Open-water disposal of dredged sediments	Marine Protection, Research and Sanctuaries Act (33 USC 1401-1445; 40 CFR 227)	DMMP (RCW 79.90; WAC 332-30-166)	
Solid Waste Disposal	Requirements for solid waste handling management and disposal	Solid Waste Disposal Act (42 USC 215103259-6901-6991; 40 CFR 257-258)	Solid Waste Handling Standards (RCW 70.95; WAC 173-350)	
Discharge to Surface Water	Point source standards for new discharges to surface water	National Pollutant Discharge Elimination System (40 CFR 122, 125)	Discharge Permit Program (RCW 90.48; WAC 173-216, 222)	
Shoreline	Construction and development		Shoreline Management Act (RCW 90.58; WAC 173-16); King County and City of Seattle Shoreline Master Plans (KCC Title 25; SMC 23.60); City of Tukwila Shoreline Master Program (TMC 18.44)	For construction within 200 feet of the shoreline.
Floodplain Protection	Avoid adverse impacts, minimize potential harm	Executive Order 11988, Protection of Floodplains (40 CFR 6, Appendix A); FEMA National Flood Insurance Program Regulations (44 CFR 60.3Ld)(3)).		For in-water construction activities, including any dredge or fill operations. Includes local ordinances: KCC Title 9 and SMC 25.09.
Critical (or Sensitive) Area ARAR	Evaluate and mitigate impacts		Growth Management Act (RCW 36.70a); King County Critical Area Ordinance (KCC Title 21A.24); City of Seattle (SMC 25.09); City of Tukwila Sensitive Area Ordinance (TMC 18.45)	

Table 4-1 ARARs for the Lower Duwamish Waterway (continued)

Topic	Standard or Requirement	Regulatory Citation: Federal	Regulatory Citation: State	Comment
Habitat for Fish, Plants, or Birds ARAR	Evaluate and mitigate habitat impacts	Clean Water Act (Section 404 (b)(1)); U.S. Fish and Wildlife Mitigation Policy (44 CFR 7644); U.S. Fish and Wildlife Coordination Act (16 USC 661 et seq.); Migratory Bird Treaty Act (16 USC 703-712)		
Pretreatment Standards	National Pretreatment Standards		40 CFR Part 403; Metro District Wastewater Discharge Ordinance (KCC) to be considered (as is local requirement)	
Environmental Impact Review	State Environmental Policy Act		State Environmental Policy Act RCW 43.21C; WAC 197-11-790)	Applicable to MTCA cleanups. Because the LDW is under a joint EPA/Ecology Order, Ecology has determined that CERCLA requirements are the functional equivalent of NEPA and SEPA

Notes:

ARAR = applicable or relevant and appropriate requirement; CERCLA = Comprehensive, Environmental Response, Compensation, and Liability Act; CFR = Code of Federal Regulations; DMMP = Dredged Material Management Program; EPA = U.S. Environmental Protection Agency; FEMA = Federal Emergency Management Act; KCC = King County Code; MCL = maximum contaminant level; MCLG = maximum contaminant level goal; MTCA = Model Toxics Control Act; NEPA = National Environmental Policy Act; PCB = polychlorinated biphenyl; RCW = Revised Code of Washington; SEPA = State Environmental Policy Act; SMC = Seattle Municipal Code; SMS = Sediment Management Standards; TMC = Tukwila Municipal Code; USC = United States Code; WAC = Washington Administrative Code

Table 4-2 Other Legal Requirements for the Lower Duwamish Waterway

Topic	Standard or Requirement	Regulatory Citation		Comment
		Federal	State	
Native American Graves and Sacred Sites	Evaluate and mitigate impacts to cultural resources	Native American Graves Protection and Repatriation Act (25 USC. 3001 et seq.; 43 CFR Pt. 10) and American Indian Religious Freedom Act (42 USC 1996 et seq.)		
Critical Habitat for Endangered Species	Conserve endangered or threatened species, consult with species listing agencies	Endangered Species Act of 1973 (16 USC 1531 et seq.; 50 CFR 200, 402); Magnuson-Stevens Fishery Conservation and Management Act (16 USC 1801-1884)	Endangered, threatened, and sensitive wildlife species classification (WAC 232-12-297)	Consult and obtain Biological Opinions.
Historic Sites or Structures	Requirement to avoid, minimize, or mitigate impacts to historic sites or structures	National Historic Preservation Act (16 USC 470f; 36 CFR Parts 60, 63, and 800)		Considered if implementation of the selected remedy involves removal of historic sites or structures.
Occupational Health and Safety	Requirements to provide for worker health and safety	Occupational Safety and Health Act (29 USC; 29 CFR)	Washington Industrial Safety and Health Act (RCW 49.17; WAC 296)	

Notes:

CFR = Code of Federal Regulations; RCW = Revised Code of Washington; USC = United States Code; WAC = Washington Administrative Code

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
Metals and Trace Elements									
Antimony	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	640
Arsenic	360	190	**69**	**36**	**340**	**150**	69	36	0.14[g,h]
Cadmium	3.7	1.0	42	9.3	**2.0**	**0.25**	**40**	**8.8**	n/a
Chromium (hexavalent)	**15**	**10**	**1,100**	**50**	16	11	1,100	50	n/a
Chromium (trivalent)	**550**	180	n/a	n/a	570	**74**	n/a	n/a	n/a
Copper	**17**	**11**	**4.8**	**3.1**	nc[j]	nc[j]	4.8	3.1	n/a
Lead	**65**	**2.5**	**210**	**8.1**	65	2.5	210	8.1	n/a
Mercury	2.1	**0.012**	**1.8**	**0.025**	**1.4**	0.77	1.8	0.94	0.15[i]
Nickel	1,400	160	**74**	**8.2**	**470**	**52**	74	8.2	4,600
Selenium	**20**	**5**	**290**	**71**	n/a	5	290	71	4,200
Silver	3.4	n/a	**1.9**	n/a	**3.2**	n/a	1.9	n/a	n/a
Thallium	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.47
Zinc	**110**	**100**	**90**	**81**	120	120	90	81	26,000
PAHs									
2-Chloronaphthalene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,600
Acenaphthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	990
Anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	40,000
Benzo(a)anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Benzo(a)pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Benzo(b)fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
***PAHs** (continued)*									
Benzo(k)fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Chrysene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Dibenzo(a,h)anthracene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Fluoranthene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	140
Fluorene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5,300
Indeno(1,2,3-cd)pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.018[g]
Pyrene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4,000
Phthalates									
BEHP	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.2[g]
Butyl benzyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,900
Diethyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	44,000
Dimethyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,100,000
Di-n-butyl phthalate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4,500
SVOCs									
1,2,4-Trichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	70
1,2-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,300
1,2-Diphenylhydrazine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.2[g]
1,3-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	960
1,4-Dichlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	190
2,4,6-Trichlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.4[g]
2,4-Dichlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	290

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
SVOCs (continued)									
2,4-Dimethylphenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	850
2,4-Dinitrophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5,300
2,4-Dinitrotoluene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.4[g]
2-Chlorophenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	150
3,3'-Dichlorobenzidine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.028[g]
4,6-Dinitro-o-cresol (2-methyl-4,6-dinitrophenol)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	280
Benzidine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.0002
Bis(2-chloroethyl)ether	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.53[g]
Bis(2-chloroisopropyl)ether	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	65,000
Hexachlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00029[g]
Hexachlorobutadiene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	18[g]
Hexachlorocyclopentadiene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,100
Hexachloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.3[g]
Isophorone	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	960[g] (**600**[i])
Nitrobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	690
n-Nitrosodimethylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3[g]
n-Nitroso-di-n-propylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.51[g]
n-Nitrosodiphenylamine	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	6[g]
Pentachlorophenol	20[k]	**13**[k]	**13**	**7.9**	**19**[k]	15[k]	13	7.9	3[g]
Phenol	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	860,000

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
PCBs									
PCBs	**2**	**0.014**	**10**	**0.03**	n/a	0.014	n/a	0.03	0.000064[g]
Pesticides									
4,4'-DDD	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00031[g]
4,4'-DDE	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.00022[g]
4,4'-DDT	**1.1**	**0.001**	**0.13**	**0.001**	1.1	0.001	0.13	0.001	0.00022[g]
Aldrin	n/a	n/a	n/a	n/a	**3.0**	n/a	**1.3**	n/a	0.000050[g]
Dieldrin	n/a	n/a	n/a	n/a	**0.24**	**0.056**	**0.71**	**0.0019**	0.000054[g]
Aldrin/dieldrin (sum)[l]	**2.5**	**0.0019**	**0.71**	**0.0019**	n/a	n/a	n/a	n/a	n/a
alpha-BHC	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.0049[g]
beta-BHC	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.017[g]
gamma-BHC (Lindane)	2.0	**0.08**	**0.16**	n/a	**0.95**	n/a	0.16	n/a	1.8
alpha-Endosulfan	**0.22[m]**	**0.056[m]**	**0.034[m]**	**0.0087[m]**	0.22	0.056	0.034	0.0087	89 (**2**[i])
beta-Endosulfan	**0.22[m]**	**0.056[m]**	**0.034[m]**	**0.0087[m]**	0.22	0.056	0.034	0.0087	89 (**2**[i])
Endosulfan sulfate	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	89 (**2**[i])
Endrin	0.18	**0.0023**	**0.037**	**0.0023**	**0.086**	0.036	0.037	0.0023	0.06
Endrin aldehyde	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.3
Heptachlor	**0.52**	**0.0038**	**0.053**	**0.0036**	0.52	0.0038	0.053	0.0036	0.000079[g]
Heptachlor epoxide	n/a	n/a	n/a	n/a	**0.52**	**0.0038**	**0.053**	**0.0036**	0.000039[g]
Toxaphene	**0.73**	**0.0002**	**0.21**	**0.0002**	0.73	0.0002	0.21	0.0002	0.00028[g]
Chlordane	**2.4**	**0.0043**	**0.09**	**0.004**	2.4	0.0043	0.09	0.004	0.00081[g]

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
VOCs									
1,1,2,2-Tetrachloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	4[g]
1,1,2-Trichloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	16[g]
1,1-Dichloroethene (1,1-dichloroethylene)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	7,100
1,2-Dichloroethane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	37
1,2-Dichloropropane	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	15[g]
Acrolein	n/a	n/a	n/a	n/a	**3**	**3**	n/a	n/a	9
Acrylonitrile	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.25[g]
Benzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	51[g]
Bromodichloromethane (dichlorobromomethane)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	17[g]
Bromoform	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	140[g]
Bromomethane (methyl bromide)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,500
Carbon tetrachloride	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1.6[g]
Chlorobenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1,600
Chloroform	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	470
Dibromochloromethane (chlorobromomethane)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	13[g]
Dichloromethane (methylene chloride)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	590[g]
Ethylbenzene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2,100
Tetrachloroethene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3.3[g]
Toluene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	15,000
trans-1,2-Dichloroethene	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	10,000
Trichloroethene (trichloroethylene)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	30g

Table 4-3 State and Federal Aquatic Life and Human Health Water Quality Criteria (continued)

Contaminant	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
VOCs (continued)									
Vinyl chloride	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	2.4[g]
Dioxins and Furans									
2,3,7,8 TCDD	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	5.1E-09[g]

Notes:

1. Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. Existing site-specific data or site-specific data that may be collected can be used to adjust values rather than using a default hardness value of 100 mg/L. Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same). The lower of the human health criteria (when multiple criteria are available) is also bolded.

a. Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

b. Standards are from the national recommended EPA AWQC (except where noted). National recommended EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

c. Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds.

d. Human health WQC are based on dissolved concentrations for all contaminants.

e. Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time.

f. Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

g. Human health WQC are based on 1 x 10^{-6} excess cancer risk for carcinogenic contaminants.

h. Criterion represents the inorganic fraction of arsenic.

i. Standards are from 40 CFR 131.36 (NTR), as referenced in WAC 173-201A-240. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

j. Criteria based on the biotic ligand model. The acute and chronic biotic ligand model-based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

k. The freshwater aquatic life WQC for pentachlorophenol is pH-dependent; a pH of 7.8 was assumed, which is the default assumption.

l. Aldrin is metabolically converted to dieldrin. Therefore, the sum of aldrin and dieldrin concentrations is compared with the dieldrin criteria.

m. Standards are for endosulfan.

AWQC = ambient water quality criteria; BEHP = Bis(2-ethylhexyl) phthalate; BHC = benzene hexachloride; DDD – dichlorodiphenyldichloroethane; DDE = dichlorodiphenyldichloroethylene; DDT = dichlorodiphenyltrichloroethane; DOC = dissolved organic carbon; MCL= maximum contaminant level; µg/L = microgram per liter; n/a = not available; nc = not calculated; NTR = National Toxics Rule; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; SVOC = semivolatile organic compound; TCDD = tetrachlorodibenzo-*p*-dioxin; VOC = volatile organic compound; WAC = Washington Administrative Code; WQC = water quality criteria

Table 4-4 Summary of Arsenic, Total PCB, cPAH, and Dioxin/Furan Datasets for Natural Background

Human Health Risk-Driver COC	Detection Frequency	Concentration							UCL Type
		Minimum	Maximum	Mean	Median	90th Percentile[a]	UCL95	UCL95 (rounded value)[b]	
Arsenic (mg/kg dw)	70/70	1.1	21	6.5	5.9	11.0	7.3	7	Approximate Gamma UCL95
Total PCBs as Aroclors (μg/kg dw)	6/70	2.1	31	11	4.4	8.0	6.5	7	KM (Percentile Bootstrap) UCL95
Total PCBs as Congeners (μg/kg dw)	70/70	0.01	10.6	1.2	0.6	2.7	1.5	2	Approximate Gamma UCL95
cPAHs (μg TEQ/kg dw)	61/70	1.3	57.7	7.1	4.5	14.7	8.9	9	KM (BCA) UCL95
Dioxins/Furans (ng TEQ/kg dw)	70/70	0.2	11.6	1.4	1.0	2.2	1.6	2	H-UCL95

Notes:

1. Dataset collected throughout Puget Sound by EPA in 2008 and referred to as the EPA OSV *Bold* Survey.
2. Summary statistics and UCL were calculated using ProUCL 4.00.04 statistical software.
3. Total PCBs were calculated by summing the concentrations of detected PCB Aroclors or detected PCB congeners. In cases where no PCB Aroclors were detected, the highest reporting limit for an individual PCB Aroclor was used as the value of total PCBs. Total cPAHs were calculated by summing the concentrations of all detected cPAH compounds multiplied by their respective potency equivalency factors (PEFs), along with half the reporting limits of any undetected cPAH compounds multiplied by their respective PEFs.
4. The total toxic equivalent (TEQ) of dioxins/furans (relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) was calculated by summing the concentrations of detected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective toxic equivalency factors (TEFs), along with half the reporting limits of undetected polychlorinated dibenzo-*p*-dioxin or furan congeners multiplied by their respective TEFs.

a. Using MTCAStat software, instead of EPA's ProUCL, risk drivers may be slightly higher.
b. Rounded values of UCL95s are used as natural background in this FS.

BCA = bias-corrected accelerated; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; FS = feasibility study; H-UCL = UCL based on Land's H-statistic; kg = kilogram; KM = Kaplan Meier method for calculating a UCL; µg = micrograms; mg = milligram; ng = nanogram; PCB = polychlorinated biphenyl; PEF = potency equivalency factor; TEF = toxic equivalency factor; TEQ = toxic equivalent; UCL95 =95% upper confidence limit on the mean

Table 4-5 Practical Quantitation Limits, Natural Background, and Risk-Based Threshold Concentrations for the Human Health and Ecological Risk-Driver COCs

Human Health& Ecological Risk-Driver COC	Practical Quantitation Limits			Natural Background[b]	Risk-Based Threshold Concentrations				
	EPA Method	RI QAPP RLs[a]	Range of RLs from undetected values		Spatial Scale of Exposure[c]	RAO 1: Human Seafood Consumption	RAO 2: Human Direct Contact	RAO 3: Benthic Organisms	RAO 4: Ecological (River Otter)
Total PCBs (µg/kg dw)	8082	4[d]	0.56 – 50[e]	2	Site-wide	nc (7 - 185)[f]	1,300	n/a	(128 - 159)[g]
					Tribal Clamming	n/a	500	n/a	n/a
					Beach Play	n/a	1,700	n/a	n/a
					Point	n/a	n/a	12/65[h]	n/a
Arsenic (mg/kg dw)	6010B	5	3.1 – 31	7	Site-wide	n/c[i]	3.7	n/a	n/a
					Tribal Clamming	n/a	1.3	n/a	n/a
					Beach Play	n/a	2.8	n/a	n/a
					Point	n/a	n/a	57/93[h]	n/a
cPAH (µg TEQ/kg dw)	8270D	6.3 – 20 µg/kg[j]	9.0 – 130 µg/kg[j]	9	Site-wide	n/c[i]	380	n/a	n/a
					Tribal Clamming	n/a	150	n/a	n/a
					Beach Play	n/a	90	n/a	n/a
					Point	n/a	n/a	n/a[k]	n/a
Dioxins/Furans (ng TEQ/kg dw)	1613B	1 – 10 ng/kg[l]	0.12 – 7.7 ng/kg[l]	2	Site-wide	nc (bg)	37	n/a	n/a
					Tribal Clamming	n/a	13	n/a	n/a
					Beach Play	n/a	28	n/a	n/a
					Point	n/a	n/a	n/a	n/a

Notes:

a. Reporting limits from Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006) in dry weight units on untransformed data.

b. UCL95 values are calculated from the EPA OSV *Bold* Survey dataset using ProUCL.

Table 4-5 Practical Quantitation Limits, Natural Background, and Risk-Based Threshold Concentrations for the Human Health and Ecological Risk-Driver COCs (continued)

c. The spatial scale of site-wide exposure is RAO-specific: (seafood consumption for RAO 1 and RAO 4; netfishing for RAO 2).

d. PCB RLs (as Aroclors) reported in Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006). RLs for individual PCB congeners are much lower (0.5 to 1 ng/kg).

e. Range of RLs for undetected values were queried from the RI database and represent RLs for undetected total PCBs. For samples in which none of the individual Aroclors are detected, the total PCB concentration value is represented as the highest RL of an individual Aroclor, and assigned a U-qualifier, indicating no detected concentrations. Individual undetected Aroclors were not reported because they are not included in the calculation of total PCBs when other Aroclors are detected in the sample.

f. RBTC <1 µg/kg dw at risk levels of 10^{-5} and 10^{-6}, and RBTC range of 7 to 185 µg/kg dw for the three RME seafood consumption scenarios at the 10^{-4} risk level.

g. Values represent best-fit estimates for two different dietary scenarios as reported in the RI (Windward 2010).

h. Total PCB concentration units are mg/kg oc and the two values are SQS/CSL. Arsenic concentration units are mg/kg dw and the two values are SQS/CSL.

i. Arsenic and cPAH PRGs are undefined for the human health seafood consumption pathway (RAO 1). Seafood consumption excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. There is no credible relationship, based on site data, relating cPAH or arsenic concentrations in sediment to concentrations in clam tissue (Section 8 of the RI, Windward 2010). Section 8 of the FS discusses the need for future investigations of the sediment/tissue relationships for arsenic and cPAHs.

j. cPAH TEQ RLs are based on those for the individual PAH compounds used in the TEQ calculation. All individual PAH compounds used in the cPAH calculation have an RL of 20 except for dibenzo[a,h]anthracene, which has an RL of 6.3. RLs reported for undetected values are based on calculated cPAHs and can be found in Table A-1, of Round 3 Surface Sediment QAPP Addendum (Windward 2006).

k. Low- and high-molecular weight PAHs are addressed by the SMS. Criteria are set for both groupings and for individual PAH compounds.

l. Dioxin/furan TEQ RLs are based on those for the individual congeners used in the TEQ calculation. RLs for undetected values are in Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006).

bg = natural background; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; CSL = cleanup screening level; dw = dry weight; EPA = U.S. Environmental Protection Agency; LDW = Lower Duwamish Waterway; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; nc = no value calculated; nc (bg) = not calculated, RBTC value expected to be below background; ng/kg = nanograms per kilogram; oc = organic carbon; PCB = polychlorinated biphenyl; QAPP = quality assurance project plan; RAO = remedial action objective; RBTC = risk-based threshold concentration; RI = remedial investigation; RL = reporting limit; RME = reasonable maximum exposure; SQS = sediment quality standard; TEQ = toxic equivalent; UCL95 = 95% upper confidence limit on the mean

Table 4-6 Practical Quantitation Limits and Risk-Based Threshold Concentrations for Benthic Risk-Driver COCs

Benthic Risk-Driver COC	Practical Quantitation Limits			Risk-Based Threshold Concentrations RAO 3: Sediment Management Standards[c]		
	EPA Method	RI QAPP RLs[a]	Range of RLs from Undetected Values[b]	Spatial Scale of Exposure	Sediment Quality Standard (SQS)	Cleanup Screening Level (CSL)
SMS Metals		(mg/kg dw)			(mg/kg dw)	
Arsenic	6010B	5	3.1 – 31	Point	57	93
Cadmium	6010B	0.2	0.4 – 2.5		5.1	6.7
Chromium	6010B	0.5	0.25 – 1		260	270
Copper	6010B	0.2	0.5 – 1		390	390
Lead	6010B	2	1.25 – 8		450	530
Mercury	7471A	0.05	0.02 – 0.1		0.41	0.59
Silver	6010B	0.3	0.046 – 5		6.1	6.1
Zinc	6010B	2	0.5 – 2		410	960
Dry Weight Basis SMS Organic Compounds		(µg/kg dw)			(µg/kg dw)	
4-methylphenol	8270D	6.7	8.6 – 2,000	Point	670	670
2,4-dimethylphenol	8270D	6.7	6.0 – 2,000		29	29
Benzoic acid	8270-SIM	20	13 – 3,000		650	650
Benzyl alcohol	8270-SIM	2	9.2 – 690		57	73
Pentachlorophenol	8270-SIM	10	7.6 – 4,900		360	690
Phenol	8270D	20	7.3 – 790		420	1,200
oc-normalized SMS Organic Compounds[d]		(µg/kg dw)			(mg/kg oc)	
Total PCBs	8082	4	0.56 – 50	Point	12	65
Acenaphthene	8270D	20	1.8 – 2,000		16	57
Anthracene	8270D	20	13 – 2,000		220	1,200
Benzo(a)pyrene	8270D	20	6.4 – 350		99	210
Benz(a)anthracene	8270D	20	6.4 – 200		110	270
Total benzofluoranthenes	8270D	20	n/a		230	450
Benzo(g,h,i)perylene	8270D	20	13 – 2,000		31	78
Chrysene	8270D	20	18 – 170		110	460
Dibenz(a,h)anthracene	8270D	6.3	1.0 – 2,000		12	33
Indeno(1,2,3-cd)pyrene	8270D	20	6.4 – 1,600		34	88
Fluoranthene	8270D	20	19 – 340		160	1,200
Fluorene	8270D	20	1.8 – 2,000		23	79
Naphthalene	8270D	20	1.0 – 2,000		99	170
Phenanthrene	8270D	20	18 – 200		100	480
Pyrene	8270D	20	18 – 170		1,000	1,400
HPAH	8270D	n/a	n/a		960	5,300
LPAH	8270D	n/a	n/a		370	780
Bis(2-ethylhexyl)phthalate	8270D	20	15 – 1,500		47	78
Butyl benzyl phthalate	8270-SIM	2	1.8 – 2,000		4.9	64
Dimethyl phthalate	8270D	20	1.8 – 2,000		53	53
1,2-dichlorobenzene	8270-SIM	2	0.4 – 2,000		2.3	2.3
1,4-dichlorobenzene	8270-SIM	2	0.2 – 2,000		3.1	9
1,2,4-trichlorobenzene	8270-SIM	2	0.4 – 2,000		0.81	1.8
2-methylnaphthalene	8270D	20	1.0 – 2,000		38	64
Dibenzofuran	8270D	20	1.7 – 2,000		15	58
Hexachlorobenzene	8081A	1.0	0.11 – 2,000		0.38	2.3
n-Nitrosodiphenylamine	8270-SIM	10	1.8 – 2,000		11	11

Notes:
1. All QAPP-based RLs are below the SQS except for n-nitrosodiphenylamine.
2. Background concentrations were not calculated for the COCs listed in this table because benthic RBTCs are not below natural background.

a. Reporting limits from Table A-1, Round 3 Surface Sediment QAPP Addendum (Windward 2006) in dry weight units. Low level reporting limits for contaminants analyzed by EPA method 8279-SIM from Analytical Resources, Incorporated (www.arilabs.com).
b. Range of RLs reported in Remedial Investigation dataset in instances where constituent(s) were not detected. All RLs shown in dry weight units.
c. Under the SMS, sediment cleanup standards are established on a site-specific basis within an allowable range of contamination. The SQS and CSL define this range. However, the final cleanup level will be set in consideration of the net environmental effects, cost, and engineering feasibility of different cleanup alternatives (WAC 173-204-570(4)).
d. The tabulated SMS values are oc-normalized and are screened against the RLs using the underlying apparent effects threshold concentrations, which are dry weight-based.

COC = contaminant of concern; CSL = cleanup screening level; dw = dry weight; EPA = U.S. Environmental Protection Agency; HPAH = high-molecular-weight polycyclic aromatic hydrocarbon; LDW = Lower Duwamish Waterway; LPAH = low-molecular-weight polycyclic aromatic hydrocarbon; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; oc = organic carbon; PQL = practical quantitation limit; QAPP = quality assurance project plan; RAO = remedial action objective; RL = reporting limit; SIM = selected ion monitoring; SMS = Sediment Management Standards; SQS = sediment quality standard

Table 4-7 Preliminary Remediation Goals for Total PCBs, Arsenic, cPAHs, and Dioxins/Furans for Human Health and Ecological Risk-Driver COCs

Risk-Driver COC	Preliminary Remediation Goals					
	RAO 1: Human Seafood Consumption	RAO 2: Human Direct Contact	RAO 4: Ecological (River Otter)	Basis	Statistical Metric for Application	Spatial Scale of PRG Application
Total PCBs (µg/kg dw)	2	1,300	128-159	bg (RAO 1) RBTC (RAO 2) RBTC (RAO 4)	SWAC	Site-wide
	n/a	500	n/a	RBTC	SWAC	Clamming Areas
	n/a	1,700	n/a	RBTC	SWAC	Individual Beaches
Arsenic (mg/kg dw)	n/a	7[a]	n/a	bg	SWAC	Site-wide
	n/a	7	n/a	bg	SWAC	Clamming Areas
	n/a	7	n/a	bg	SWAC	Individual Beaches
cPAH (µg TEQ/kg dw)	n/a	380[a]	n/a	RBTC	SWAC	Site-wide
	n/a	150	n/a	RBTC	SWAC	Clamming Areas
	n/a	90	n/a	RBTC	SWAC	Individual Beaches
Dioxins/Furans (ng TEQ/kg dw)	2[b]	37	n/a	bg (RAO 1) RBTC (RAO 2)	SWAC	Site-wide
	n/a	13	n/a	RBTC	SWAC	Clamming Areas
	n/a	28	n/a	RBTC	SWAC	Individual Beaches

Notes:

1. The PRGs for RAO 3 are shown separately in Table 4-8. The PRGs were developed for the 41 COCs that have been identified as benthic risk drivers for RAO 3.

a. Arsenic and cPAH PRGs are undefined for the human health seafood consumption pathway (RAO 1). Seafood consumption excess cancer risks for these two risk drivers were largely attributable to the consumption of clams. There is no credible relationship, based on site data, relating cPAH or arsenic concentrations in sediment to concentrations in clam tissue (Section 8 of the RI, Windward 2010). Section 8 of the FS discusses the need for future investigations of the sediment/tissue relationships for arsenic and cPAHs.

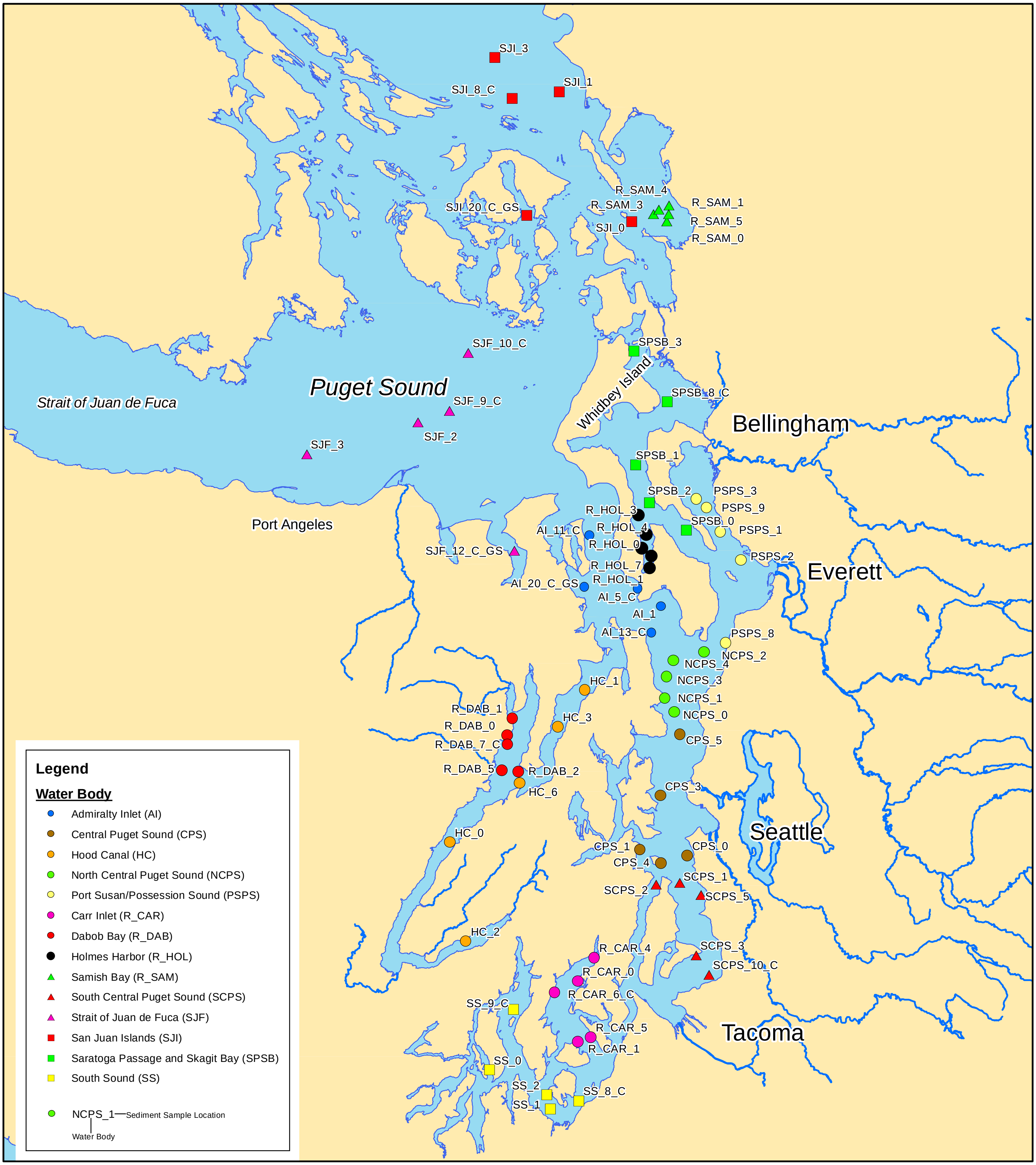
b. Although risks associated with consumption of dioxins/furans in resident seafood were not quantitatively assessed in the baseline HHRA, those risks were assumed to be unacceptable, and the associated sediment concentration was assumed to be below natural background concentrations.

bg = natural background; COC = contaminant of concern; cPAH = carcinogenic polycyclic aromatic hydrocarbon; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; ng/kg = nanograms per kilogram; oc = organic carbon; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAO = remedial action objective; RBTC = risk-based threshold concentration; SMS = Sediment Management Standards; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent

Table 4-8 Preliminary Remediation Goals for Benthic Risk-Driver COCs

Benthic Risk-Driver COC	Preliminary Remediation Goals for RAO 3			
	Value	Basis	Statistical Metric	Spatial Scale of PRG Application
SMS metals	**(mg/kg dw)**			
Arsenic	57	SQS	Point Concentration or Toxicity Test Override	Point
Cadmium	5.1	SQS		
Chromium	260	SQS		
Copper	390	SQS		
Lead	450	SQS		
Mercury	0.41	SQS		
Silver	6.1	SQS		
Zinc	410	SQS		
Dry Weight Basis SMS Organic Compounds	**(µg/kg dw)**			
4-methylphenol	670	SQS	Point Concentration or Toxicity Test Override	Point
2,4-dimethylphenol	29	SQS		
Benzoic acid	650	SQS		
Benzyl alcohol	57	SQS		
Pentachlorophenol	360	SQS		
Phenol	420	SQS		
oc-normalized SMS Organic Compounds[d]	**(mg/kg oc)**			
Total PCBs	12	SQS	Point Concentration or Toxicity Test Override	Point
Acenaphthene	16	SQS		
Anthracene	220	SQS		
Benzo(a)pyrene	99	SQS		
Benz(a)anthracene	110	SQS		
Total benzofluoranthenes	230	SQS		
Benzo(g,h,i)perylene	31	SQS		
Chrysene	110	SQS		
Dibenz(a,h)anthracene	12	SQS		
Indeno(1,2,3-cd)pyrene	34	SQS		
Fluoranthene	160	SQS		
Fluorene	23	SQS		
Naphthalene	99	SQS		
Phenanthrene	100	SQS		
Pyrene	1,000	SQS		
HPAH	960	SQS		
LPAH	370	SQS		
Bis(2-ethylhexyl)phthalate	47	SQS		
Butyl benzyl phthalate	4.9	SQS		
Dimethyl phthalate	53	SQS		
1,2-dichlorobenzene	2.3	SQS		
1,4-dichlorobenzene	3.1	SQS		
1,2,4-trichlorobenzene	0.81	SQS		
2-methylnaphthalene	38	SQS		
Dibenzofuran	15	SQS		
Hexachlorobenzene	0.38	SQS		
n-Nitrosodiphenylamine	11	SQS		

COC = contaminant of concern; µg/kg = micrograms per kilogram; mg/kg = milligrams per kilogram; n/a = not applicable; oc = organic carbon; PRG = preliminary remediation goal; SMS = Sediment Management Standards; SQS = sediment quality standard



Notes:
1. Surface sediment samples collected in 2008 by EPA aboard the Ocean Survey Vessel (OSV) *Bold*.

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Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Final Feasibility Study 60150279-14.36	**2008 Puget Sound *OSV Bold* Survey Locations**
DATE: 10/31/12 · DWRN:mvi/SEA · Revision: 0	**FIGURE 4-1**

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5 Evaluation of Sediment Movement and Recovery Potential

This section presents a summary of the sediment transport and related contaminant transport modeling, as well as empirical data, and develops an understanding of potential natural recovery based on the models and data. The overall modeling approaches are presented in this section. The sediment transport model (STM) is presented in the STM Report (QEA 2008). The sediment-related contaminant transport modeling is presented in Appendix C, Part 1. The data evaluations supporting the natural recovery analysis are presented in Appendix F.

One of the U.S. Environmental Protection Agency (EPA) guiding principles for managing sediments is to develop a conceptual site model (CSM) that considers sediment stability and evaluates the assumptions and uncertainties associated with site data and models (EPA 2005b). Model results are used to inform the CSM. A well-developed and calibrated model can assist in adaptively managing a site and adjusting or refining site predictions to the actual response of a system after various remedial actions and source control measures have either been completed or are under way. Sediment experts and site managers all recognize the unique challenges and difficulties in understanding the natural forces and man-made events that affect sediment movement, stability, and recovery potential, and that some uncertainty will always be present. Consistent with EPA's guiding principles, in this feasibility study (FS), the STM, the bed composition model (BCM), and the potential for sediments to be exposed at the surface are used to predict responses after applying the different remedial actions.

The hydrodynamic and sediment transport CSM for the Lower Duwamish Waterway (LDW), as described in Section 2, is largely influenced by the reduction and control of inflows through diversion of the rivers that historically flowed into the Green River and ongoing water management practices at the Howard Hanson Dam. Peak inflows have been greatly reduced, and the LDW has been widened and deepened to permit navigation. The increased cross-section acts as a natural sediment trap for incoming coarse-grained sediment. The STM simulates natural transport and bed evolution processes in this highly modified riverine/estuarine system. In addition, some effects of ships transiting the navigation channel and berthing areas under routine operating procedures are implicitly included in the STM/BCM by calibration to measured sedimentation rates. The goals of the LDW-wide modeling efforts for this FS are:

- Illustrate how contaminant concentrations vary spatially in the LDW via sediment movement, scour, and deposition processes and empirical trends.
- Predict contaminant fate and recovery potential for risk drivers over periods of time (e.g., 10 years) via the primary mechanisms of burial and source control.

- Demonstrate that model predictions and empirical measurements are comparable. Both the modeling results and empirical data have some measure of uncertainty; therefore, multiple lines of evidence are evaluated collectively to examine and reduce these uncertainties and to refine the CSM (EPA 2005b).
- Consider how navigation activities may disrupt natural recovery processes and affect BCM recovery predictions.

The four modeling process steps to address these goals are described below.

First, the STM results are used to look at general trends in an analysis of net sedimentation rates and to review agreement with the CSM with respect to the depositional environment in the absence of deep scour events. This is accomplished by comparing the estimated net sedimentation trends to empirical data. Empirical data include subsurface cores used to determine historical trends in net sedimentation rates and surface sediment locations that have been resampled over time. The STM is used to evaluate sediment movement as it relates to potential remedial areas and alternatives. This step includes an evaluation of net sedimentation rates, sediment transport into early action areas (EAAs), and other specific model runs to better understand sediment dynamics in the system (Section 5.1).

Second, the BCM, which takes output directly from the physical STM and applies contaminant concentrations to modeled sediment particles, was developed to predict future contaminant concentrations in surface sediments, and therefore recovery potential. The BCM is based on STM output, and BCM predictions assume that contaminant concentrations will be influenced only by sedimentation and resuspension due to natural processes. The BCM and associated empirical evidence are used in the FS to provide a predictive tool for evaluating whether contaminant concentrations in the surface layer/biologically active zone will decrease through natural recovery processes. The STM, BCM, and empirical evidence are used to evaluate whether the sediment bed is stable (i.e., not subject to significant scour, erosion, and transport) and whether the sedimentation rate is sufficient for burial of contaminated sediments to occur in the absence of navigation-caused disturbances. If these conditions are met in a given location, then monitored natural recovery (MNR) or enhanced natural recovery (ENR) may be appropriate response actions for evaluation in one or more remedial alternatives. Conversely, if natural processes are not effectively reducing concentrations of contaminants of concern (COCs) in surface sediments, then capping or dredging may be more appropriate choices (Section 5.2).

STM/BCM predictions of net sedimentation over much of the LDW are consistent with the CSM when ongoing navigation activities are assumed to constitute a minor influence on surface sediment contaminant concentrations (e.g., propeller wash does not expose, resuspend, and mix deeper subsurface contamination with surface sediment).

Third, smaller scale areas are analyzed to evaluate local recovery potential and assess whether empirical data and predictive models agree. MNR is a potential remedy that relies on ongoing, naturally-occurring processes (such as sediment deposition, mixing, and burial) to reduce COC concentrations in surface sediment. Several lines of evidence (e.g., isotope cores, sediment transport analysis, contaminant trends analysis, evaluation of erosion potential) are combined to assess whether contaminated subsurface sediments are stable, if they are effectively isolated, and whether surface sediment contaminant concentrations are predicted to decrease over time. The STM and BCM do not incorporate disturbances to bed sediments from propeller wash; therefore, bathymetric imaging data were used to identify these areas. These lines of evidence are used in the FS both when configuring remedial alternatives and when evaluating the long-term effectiveness of remedial alternatives (Sections 5.3 and 5.4). Local recovery potential under routine navigation procedures is discussed in Section 5.3.2.7.

Fourth, this FS considers the potential influence of contaminated subsurface sediments that may be exposed at the surface. Some effects of ships transiting the navigation channel and berthing areas under routine operating procedures are included in the STM/BCM by calibration to measured sedimentation rates. However, additional navigation and construction-related activities, as well as natural events, may result in sediment bed disturbance causing increased surface sediment contaminant concentrations that are not addressed by the STM/BCM. The STM and BCM were designed to consider only external and surface sediment sources of contamination to the LDW system. They were not set up to model deeper disturbance events, so this FS conducted a separate sensitivity analysis of deep sediment disturbance to consider the potential effects of such disturbance events on STM/BCM-predicted spatially-weighted average concentrations (SWACs; see Section 5.2.3).

This section of the FS focuses on details related to the six modeling goals:

- Providing an overview of the physical CSM and the STM relative to recovery.
- Discussing briefly the multiple lines of empirical evidence (i.e., sediment core trends, surface sediment sample trends at resampled stations, and physical features) that validate the STM and identify trends not accounted for by the predictive model.
- Developing a predictive recovery model (i.e., the BCM) and inputs to the BCM.
- Developing methods to either account for or assess the potential for scour to affect sedimentation and recovery in two ways: shallow mixing from routine

vessel operating procedures through resuspension and mixing; and episodic deep disturbances that result in subsurface contaminated sediments being exposed at the surface layer (thereby affecting the SWAC).

- Performing additional STM scenario runs to help answer FS-specific questions related to sediment movement and MNR and ENR recovery potential.
- Defining uncertainties of the STM model, including a brief overview of how it affects uncertainties in the fate and transport processes for risk drivers.

Potential application of MNR and ENR and general response actions are described in Section 7, Identification and Screening of Remedial Technologies. Additional STM runs are described in Appendix C. Empirical trends for individual areas of potential concern (AOPCs) are presented in Section 6.

5.1 Sediment Transport Modeling

Modeling of particle movement in and out of the LDW and sediment transport within the LDW was undertaken during the remedial investigation (RI) to better understand the CSM and support various FS elements.[1] The site-wide STM, which simulates the natural sediment resuspension and sedimentation processes active to varying degrees within the LDW (with the caveats noted above), has shown that the LDW is net depositional on a site-wide scale and is divided into Reaches 1, 2, and 3 based on hydrodynamic characteristics and geomorphology (see Section 2 for more details regarding the CSM). Model development and calibration are detailed in the *Final Sediment Transport Analysis Report* (STAR; Windward and QEA 2008) and the *Final Sediment Transport Modeling Report* (QEA 2008). This section reviews the resulting general trends in a site-wide analysis (Section 5.1.1) and evaluates the STM's ability, when combined with the BCM, to predict contaminant trends. This is accomplished by comparing the predicted trends to empirical data (Section 5.4).

5.1.1 Composition and Sources of Sediment Loads

The STM estimated the movement of sediment from three sources over time into and through the LDW:

- Sediment from the upstream Green/Duwamish River system

[1] The STM tracks particle movement, but it does not model contaminant transport processes or mechanical transport processes such as the effect of vessel traffic or waves on net sedimentation rates. The effect of vessel traffic was analyzed separately for moving and maneuvering tugs. The analysis of moving tugs is presented in the *Final Sediment Transport Analysis Report* (STAR; Windward and QEA 2008) and the effect of maneuvering tugs is summarized in Section 5.3.1 and in Appendix C, Part 7 of this FS.

- Sediment from lateral sources (i.e., storm drains, streams, and combined sewer overflows [CSOs]) that discharge to the LDW
- Surface sediment existing in the LDW bed at the onset of the model period.

The STM modeled both the transport of total suspended solids (TSS) and bed load. The transport of TSS is the movement of suspended particles in the water column. Bed load transport is the movement of sand and gravel in a thin layer (about 1 millimeter [mm] to 1 centimeter [cm] in thickness) located along the surface of the sediment bed. The Green/Duwamish River is the predominant source of sediment to the LDW. Figures 5-1a and 5-1b show that surface sediment (0 to 10 cm) in over 90% of the LDW model area will be comprised of over 50% upstream solids at the end of the 10-year model simulation and over 75% upstream solids at the end of the 30-year simulation. The STM quantified sediment loading from this upstream source using a flow-rating curve for the Green/Duwamish River based on discharge data gathered from 1960 to 1980 and from 1996 to 1998. The grain size characteristics of the in-flow material from both periods were also evaluated to determine the contribution from suspended material in contrast to bed load. Of the total upstream solids load, approximately 24% is bed load and 76% is suspended load in both the 10-year and 30-year simulation periods. Nearly all of the bed load and suspended load in the sand-size range settles in the LDW. Of the clay and silt suspended load, approximately 10% of the clay-size particles and 76% of the silt-size particles are predicted to settle in the LDW. All of the bed load entering the LDW from upstream is deposited within the Upper Turning Basin and the upstream portions of the navigation channel, which are periodically dredged by the U.S. Army Corps of Engineers (USACE). Approximately 50% of the total solids load entering the LDW from upstream is deposited in the LDW, with approximately 80% of this deposition occurring in the vicinity of the Upper Turning Basin in Reach 3 (QEA 2008, see Appendix B of the STM Report).

Sediment loads from lateral sources were derived from analyses conducted by the City of Seattle and King County (Nairn 2007; Seattle Public Utilities 2008). Storm drains, CSOs, and streams discharge into the LDW at over 200 locations. These were initially aggregated in the STM report into 21 discrete discharges at 16 locations to simplify modeling. In the STM, the total annual sediment load from the lateral sources was estimated to be 1,257 metric tons per year (MT/year); of this, 76% was attributed to storm drains, 3% to CSOs, and 21% to streams.

The distribution and magnitude of sediment loads from lateral sources were updated after the STM report (QEA 2008) was completed. These updated sediment loads are presented in Appendix C, Part 4, Scenario 2. The updated loads provide a more accurate distribution of the loads, reflecting better distribution of inputs and more actual outfall locations. Figure 5-2 illustrates the spatial distribution of the percentages of sediment from lateral sources at the end of the 10-year model simulation, using the updated lateral loads distribution. Updated lateral loads were used in all subsequent modeling in this FS. The areas with the greatest predicted lateral sediment contribution (i.e., the sediment bed after 10 years includes more than 10% lateral contribution) are limited to the following areas in the LDW: at the heads of Slips 4 and 6, Hamm Creek at river mile (RM) 4.3W, RM 1.8W, near Glacier at RM 1.5W, RM 1.2E, RM 0.3W, and in the Duwamish/Diagonal EAA at RM 0.5.

A third component of sediment load is the movement of surface sediment from one model grid cell to another. Bed sediment can be resuspended during a high-flow event, after which it either resettles nearby or is transported downstream. The STM tracks the movement of these particles throughout the LDW, from grid cell to grid cell. The ability of the STM to track the movement of particles within the LDW was used to evaluate the transport of sediment between Reaches 1, 2, and 3, as summarized in Figure 5-3 and in Appendix C, Part 4, Scenario 4.

The highest percentage of original bed sediments remaining in the surface layer after 10 years occurs in the grid cells east of Kellogg Island at RM 0.9 and at the Terminal 117 EAA (RM 3.0 to RM 3.5). The areas that have the highest percentage of original bed sediment remaining at the end of the 30-year simulation are consistently the highest throughout the simulation and are not the result of a short-term scour event. A higher percentage of original bed sediment indicates that much of the surface layer is not being replaced by upstream or lateral sediment (i.e., the bed surface sediments are not receiving much deposition and could be interpreted as having a more constant composition over time).

5.1.2 Solids Balance In and Out of the LDW

Figure 5-3 shows the mass of sediment moving through and within the three reaches of the LDW over 10-year and 30-year modeling periods. Year-to-year variation in sediment load occurs because of variability in river flow, with total sediment load increasing during years with relatively high flows. Over the 10-year period, more than 99% of the incoming sediment load (1,850,850 MT) originates from the Green/Duwamish River (upstream); less than 1% (12,580 MT, or an annual average of 85,000 MT/yr) enters the LDW from lateral sources. Over a 30-year period, a cumulative total of 6.2 million MT enters the LDW (for an annual average of approximately 207,000 MT/yr). The magnitude of the sediment mass movement increases, but the percent contribution from upstream and lateral sources is essentially the same as for the 10-year period. About 50% of the incoming solids (approximately 100,000 MT annually) deposit within the LDW and are not exported farther downstream into the East and West Waterways and Elliott Bay. Approximately 51% of the sediment that settles in the LDW is removed by periodic maintenance dredging, mostly in the Upper Turning Basin.[2] Thus, approximately 25% of the incoming sediment load remains in the LDW basin after dredging.

Bed load (heavier, larger particles that skip and travel along the sediment bed[3]) comprises 24% of the total incoming sediment load, on average, at the upstream boundary of the area modeled by the STM, with the remaining 76% entering the LDW as sediment suspended in the water column (QEA 2008). According to the STM, most of the bed load deposits above RM 4.0; the suspended sediment primarily deposits farther downstream or is transported through the system. The proportion of bed load to total load is inversely dependent on flow rate, decreasing from 30% to about 17% to 18% as the flow rate increases (24% on average). The estimated average annual bed load transported during the 30-year model period was 50,000 MT/year, with a range of 10,000 MT/year (1978) to 132,000 MT/year (1975) for low-flow and high-flow years, respectively (QEA 2008). This solids mass balance supports the CSM conclusion that the LDW is net depositional over long time periods and that lateral sources are important, but their effect is localized to the receiving sediments in the vicinity of these sources. The CSM and dredge records both indicate that the majority of the Upper Turning Basin dredged material is from upstream (Green/Duwamish River).

5.1.3 Scour Potential from High-flow Events and Vessel Traffic

Figure 5-4 shows potential scour areas derived from two processes: high-flow events and scour from vessel traffic. Areas of erosion from both high flows and vessel scour were considered during delineation of AOPCs (see Section 6).

Few areas in the LDW that show significant high-flow erosion potential (10 cm scour depth or more) also have subsurface contamination. These areas are identified in Appendix C (Part 4, Scenario 5) and are evaluated in Section 6 for the delineation of the AOPCs. Alternatively, most areas with significant subsurface contamination (greater than sediment quality standards [SQS]) do not show erosion potential beyond a few centimeters in depth during high-flow events. An analysis of how erosion and deposition impact surface COC concentrations over time is discussed in Section 5.2.

The STM models sedimentation and resuspension in the absence of deep sediment disturbance and exposure of contaminated subsurface sediment. No available transport model has the capacity to include anthropogenically induced resuspension and transport with confidence. Development and validation of the STM is most reliable in regions where naturally occurring sedimentation dominates transport and in areas with relatively little anthropogenic activity. The effects of such anthropogenic activity on the

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2 Dredging averages 38,000 MT/yr within the navigation channel and 13,000 MT/yr in the berthing areas. The average total dredged is 51,000 MT/yr.

3 The mean percent of fines in surface sediment of Reach 3 is 34%. The mean percent of fines in surface sediment of Reaches 1 and 2 is 69%. Bed load is mostly sand and gravel-sized particles. See Appendix C, Part 3, Tables 5a and 5b for more information.

STM/BCM are separately evaluated in Section 5.3.1.2 by modifying long-term BCM SWAC estimates to include episodic disturbance of surface and subsurface sediments.

The 100-year high-flow event produces a maximum erosion depth of less than 1 foot (less than 30 cm) in limited areas (see Figure 2-9). Most of these areas do not show COC concentrations at this depth that are greater than the SQS and that are not already expressed as SQS exceedances at the surface. Subsurface COC concentrations in areas with scour greater than 10 cm are analyzed in Appendix C (Part 4, Scenario 5) and discussed in Section 5.3.2.5.

Although this FS focuses on single high-flow events, the 30-year hydrograph record used for the STM analysis included numerous high-flow events of more than 10,000 cfs. In some years, two high-flow events occurred in the same year. Therefore, the STM inherently accounts for multiple scour events in the same year (Appendices D and F in QEA 2008).

5.2 Bed Composition Model (BCM)

Output from the STM was coupled with contaminant concentrations in sediments from various sources to enable prediction of future surface sediment contaminant concentrations under various remedial action scenarios. This analysis is termed the BCM. This section of the FS describes the BCM, its applications, and its limitations.

Output from the STM is directly applied to the BCM. A basic and conservative assumption is that all contaminants are strongly bound to sediment particles. The BCM is conservative with respect to sediment concentrations because it only accounts for contaminant movement associated with particles (i.e., transport, resuspension, burial) and assumes no loss of contaminant mass via other physical, chemical, or biological degradation processes (e.g., desorption, diffusion, volatilization, biotransformation, dechlorination, etc.). Other degradation processes explored at other sites are documented at the end of this section to provide some context for understanding these processes. The BCM does not account for contaminant transfer from sediments to the water column. However, polychlorinated biphenyl (PCB) flux from sediments to the water column and to biota was estimated in the food web model (RI Appendix D; Windward 2010).

The BCM is used later in the FS as one line of evidence to evaluate recovery potential of LDW sediments (Section 6), to identify and screen remedial technologies (Section 7), and to develop and evaluate remedial alternatives (Sections 8 and 9). The sensitivity of the BCM is also investigated by looking at how changes in input parameters affect the output (Section 9). Sediment disturbance resulting from episodic emergency and high-power ship maneuvering and maintenance/construction is not included in the BCM. The potential influence of these disturbances on the sediment bed is discussed in Section 5.3.1.

5.2.1 The BCM Calculation

The BCM is a spreadsheet-based tool that predicts COC concentrations at individual model grid-cell locations[4] in the surface sediment layer (0 to 10 cm) by using a simple mass balance formula (RETEC 2007c, Appendix C):

$$C_{(time)} = C_{bed}{*}f_{bed\ (time)} + C_{lateral}{*}f_{lateral\ (time)} + C_{upstream}{*}f_{upstream\ (time)} \qquad \textbf{Equation 5-1}$$

Where:

- f_{bed}, $f_{lateral}$, and $f_{upstream}$ are, respectively, the fractions of surface sediment sourced from existing bed sediment, from lateral source sediment, and from upstream Green/Duwamish River sediment in each grid cell at a specific point in time. These surface sediment fractions change over time and are direct outputs of the surface sediment layer of the STM. The sum of these fractions in each grid cell is 1.
- C_{bed}, $C_{lateral}$, $C_{upstream}$ are the concentrations of a COC associated with each sediment source. These concentrations are derived from existing bed contaminant concentrations, lateral source samples (i.e., stormwater and CSO discharges), and upstream (Green/Duwamish River) lines of evidence.

An example of how the BCM computation uses the STM output is shown in Figure 5-5. Additional mechanics of the BCM are provided in Appendix C.

As noted in Equation 5-1, the sediment composition fractions (f) vary with time because the STM output varies with time[5] and ongoing sediment transport changes the bed composition of each fraction. The concentration terms for the lateral source and upstream sediments ($C_{lateral}$ and $C_{upstream}$) are assumed to be constant over time for modeling purposes, representing current best estimates of the long-term average inputs over time.[6] The derivation of these values is discussed in greater detail in Section 5.2.3. The BCM assigns the same COC concentration (input value) to the lateral source and upstream sediments regardless of the variability observed over time or spatially (such as among different outfalls for the lateral sources). The bed concentration (C_{bed}) is the

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4 STM grid cells are taken directly from the STM setup, as described in the STM report (QEA 2008), and overlain with inverse distance weighting 10-ft by 10-ft chemistry grid cells in the BCM. Consequently, the BCM calculates results for 100-ft^2 areas.

5 STM output in 5-year increments is used in the BCM runs. The STM runs continuously for the entire 30-year simulation period at time steps on the order of minutes. The FS presents results in 5- or 10-year increments following the start of remedy construction. For remedial scenarios that take longer than 30 years to implement, the simulation starts over at the beginning of the 30-year hydrograph used for the STM.

6 However, high and low "sensitivity" concentrations were also used as input values to bracket the range of uncertainty in the input values and demonstrate the effects from anticipated reductions in contaminant concentrations over time.

best estimate of the COC concentration in the surface sediment bed at a given location at the start of the model period, defined by the FS surface sediment dataset. The BCM is implemented in a geographic information system (GIS) framework and MS Excel platform (described in Appendix B of RETEC 2007b).

The BCM (Equation 5-1) can be used to estimate COC concentrations in surface sediment at each grid cell location in the LDW as a function of time under various remedial alternatives. Where active remediation is assumed within an alternative, the grid cells contained within the actively remediated footprint receive a post-remedy bed sediment replacement value for C_{bed}. The new value is an estimate of the COC concentration that exists in the surface sediment at the completion of the remediation (see Section 5.2.3.4).

5.2.2 BCM Assumptions

The predictive accuracy of the BCM hinges on two important findings from the STM:

- Over time, the surface sediment that erodes, moves, and redeposits within the LDW originates primarily from the Green/Duwamish River, as shown in Figure 5-3. STM results indicate that movement of bedded sediment from within the LDW is a very minor component of overall sediment transport in the LDW. The effect of bedded sediment was further analyzed by a simulation that tracked the movement of bedded sediment. This analysis is presented in Appendix C, Part 4, Scenario 4 and Part 5, Scenario 6.

- The magnitude of high-flow bed scour is sufficiently minor such that subsurface sediments with COC concentrations that exceed the SQS are generally not exposed, eroded, or redistributed within the LDW. Even after a high-flow event, the bed height increases from deposition (see Appendix E, Figures E-19 through E-23 in QEA 2008). From the sediment mass balance analysis, the new sediment that accumulates is largely from the Green/Duwamish River. Given the limited movement of bed sediment during high-flow events, bed COC contaminant concentrations at the reach- or site-wide scale would not be predicted to change significantly during a high-flow event (Appendix C, Part 5).

Although the assumption of assigning the contaminant concentrations to resuspended bed sediment is not inherently mass conservative, it will not significantly impact model predictions, because: 1) in the LDW, the mass of bed sediment resuspended is much less than the mass of sediment from upstream; and 2) COC concentrations in resuspended sediments become similar to those in upstream solids over time and as the cleanup proceeds. Consequently, redistribution of existing sediments with COC concentrations that exceed the SQS is not a significant process, and future bed sediment chemistry can be reasonably estimated as a mass balance between present bed sediment and incoming sediment loads from the Green/Duwamish River and lateral sources.

These key findings are supported in three ways: 1) by the CSM (Section 2.3), 2) by a comparison of empirical trends to model estimates of net sedimentation and recovery rates (Section 5.4), and 3) by additional STM special scenario runs (Section 5.3.2) used to help refine the CSM for the FS.

In addition, the BCM assumes that:

- All COCs are permanently bound to sediment particles; degradation or phase transfer processes such as solubilization are assumed not to reduce COC concentrations over time. This assumption is generally consistent with the known properties of the COCs, and is inherently conservative because some degree of degradation or phase transfer likely occurs. The assumption could result in higher predicted concentrations in surface sediment with time.

- COC concentrations from drainage basins were derived from all storm drain and other solids sample data, but samples were collected from only a portion of the LDW drainage basin conveyances. These data are assumed to be representative of all lateral COC inputs. COC contributions from eroding bank material and groundwater were not included in the lateral source estimates.

- COC concentrations from drainage basins that have not been sampled are assumed to be similar to or lower than those in drainage basins sampled for source control evaluation. This is consistent with the sampling strategy of the Source Control Work Group (SCWG), which has focused first on areas with the most significant sediment contamination and associated outfalls identified as being the most likely sources of contaminated sediments to the LDW. The COC concentrations derived from the empirical data are then applied to all lateral sources in the model.

- The biologically active zone for most of the LDW is approximately 10 cm, and therefore the top 10-cm model layer represents exposure concentrations for benthic organisms. This depth is consistent with results from the sediment profile imaging (SPI) analyses conducted in the LDW for the Washington State Department of Ecology (Ecology 2007b) and King County (King County 2007a), as described in the RI (Windward 2010).[7] The 95% upper confidence limits (UCL95) on the mean of maximum sediment feeding void depths for benthic organisms (a conservative measure of the biologically active zone) used in the Ecology dataset was 11 cm with a mean

[7] The assumption of 10 cm can be reasonably applied as the biologically active zone in the LDW based on several factors: representativeness of entire benthic community, relationship with void depths, and central tendency of void depths (Windward 2010).

of 10 cm. The King County dataset was even shallower (9 cm with a mean of 8 cm). The 10-cm depth is used as the STM and BCM assumption for the active mixed layer.

5.2.3 Input Values to the BCM for Risk Drivers

Concentrations of risk drivers associated with the three sources or types of solids (i.e., upstream, lateral, and bed sediments) were estimated as inputs to the BCM. Samples from media representative of these three sources were analyzed for several COCs over a period of years, and the resulting concentrations were selected for use in the BCM based on summary statistics from compiled datasets. Some best professional judgment was incorporated into these datasets with assumptions about current and potential future conditions, including future source control efforts, the amount of solids entering the LDW, and potential biases of particular datasets. In selecting the BCM lateral input parameters, the median, the mean, and the 90th percentile of the datasets were used as the low, mid-range, and high values, respectively. High values were removed from the dataset, as described in Section 5.2.3.2, because it was assumed that they would be addressed by ongoing source control actions. For the BCM upstream input parameters, mean values of the most representative of several upstream datasets were selected for the low and mid-range input values, and the UCL95 was used for the high input value. High, medium, and low post-remedy bed sediment replacement values were derived assuming varying degrees of mixing of clean sediments in the remediated footprint with contaminated sediments remaining in the rest of the LDW, as described in Section 5.2.3.4. Selected values and ranges for the BCM input values for total PCBs, arsenic, carcinogenic polycyclic aromatic hydrocarbons (cPAHs), and dioxins/furans are provided in Tables 5-1a through 5-1c. The ranges of concentrations reported from various data sources are provided in Tables 5-2a through 5-2d.

5.2.3.1 Contaminant Concentrations Associated with Upstream Solids

Contaminant concentrations associated with Green/Duwamish River solids were compiled from various data sources, which are described in Appendix C, Part 3. These data provide multiple lines of evidence that characterize the contaminant concentrations associated with sediments entering the LDW from the Green/Duwamish River system. Data from the various studies were used to develop a range of input values for each risk driver (Table 5-1a).

The data sources evaluated included:

- Upstream whole-water samples collected by King County
- Upstream centrifuged suspended solids samples collected by Ecology
- Upstream surface sediment samples (containing fines greater than 30%) collected by Ecology between RM 5.0 and 7.0

- Upstream surface sediment samples from RM 5.0 to 7.0 included in the RI dataset
- Core data collected by the USACE to characterize sediment prior to dredging in the navigation channel from RM 4.3 to 4.75, which is assumed to represent the Green/Duwamish River combined bed load and suspended material that settles in the upper reach of the LDW.

The upstream King County whole-water concentrations were normalized to the value of the concurrently collected TSS, so that the concentration units were comparable with the sediment concentration units (i.e., both on a dry weight basis).[8]

A subset of the Ecology upstream surface sediment data was developed by excluding samples that contained less than 30% fines. This approach accommodates the systematic differences in grain size distributions between upstream (e.g., mid-channel) data and average conditions in the LDW. Both the full dataset and the subset with fines greater than 30% were used as lines of evidence to develop the range of BCM upstream input parameters.

Upstream surface sediment samples from RM 5.0 to 7.0, included in the RI dataset, were evaluated, but were not used in selecting BCM input values. The rationale for this approach is explained in Appendix C, Part 3. Instead, the more recent upstream surface sediment data collected by Ecology were used. The upstream surface sediment data had lower total PCB and cPAH concentrations than other upstream lines of evidence. This may reflect the coarser (i.e., sandier) material encountered during sampling that is characteristic of bed load[9] being transported down the Green/Duwamish River—very little of which is transported beyond the Upper Turning Basin. The surface sediments upstream of the LDW are generally coarser than those in the LDW because there is little net sedimentation upstream of the Upper Turning Basin as a result of higher stream velocities above RM 4.75.

The subsurface sediment cores collected by the USACE to characterize sediment prior to dredging in the navigation channel from RM 4.3 to 4.75 represent the Green/Duwamish River bed load and suspended material that settles in the upper reach of the LDW.[10] The Upper Turning Basin is a natural sink for incoming sediment loads from upstream, and

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8 Normalizing to TSS likely produces a high estimate of the COC concentration on sediment particles because some of the COC mass is likely dissolved or on colloidal particles that do not settle in the LDW.

9 Bed load is heavier, sandier material that travels along the bed surface; it is not suspended in the water column and thus, typically travels shorter distances than do suspended solids.

10 The RI summarized USACE cores in the Upper Turning Basin from RM 4.0 to 4.75. The FS screened this dataset to exclude the potential influence of sources (e.g., Hamm Creek) in the downstream portion between RM 4.0 and 4.3. The FS dataset also includes more recent data collected by USACE above RM 4.3.

because the navigation channel is dredged every 2 to 4 years from RM 4.0 to 4.75, this area is a good indicator of suspended solids settling in the upper reach of the LDW.

The upstream solids values selected for use in the BCM were based on these four datasets as values representing the best estimate concentrations of the risk-driver COCs entering and settling in the LDW. Each dataset contains information that represents, to a degree, the COC concentrations in sediment particles that enter and deposit within the LDW. As discussed below, these datasets are considered reasonable lines of evidence for developing incoming concentrations to the LDW from upstream, although each type of data collection tends to bias the results toward lower or higher values (e.g., low percent fines versus high percent fines; single collection events instead of seasonal collection events; potential influence of sources). In general, the value representing a mid-range of the various lines of evidence was considered for the input value, and then values representing upper and lower bounds were selected for the high and low sensitivity input values, respectively. One goal of including a range in the input values is to account for uncertainty in all the datasets representing upstream inputs and show how these data ranges affect the predictions of natural recovery for the remedial alternatives.

For total PCBs and cPAHs, the means of the LDW RM 4.3-4.75 USACE core data were selected as the upstream input values (35 microgram per kilogram dry weight [µg/kg dw] and 70 µg toxic equivalent [TEQ]/kg dw, respectively). To address sensitivity around the mid-range value for both total PCBs and cPAHs, the low upstream input values were the means of the Ecology upstream surface sediment samples containing fines greater than 30%. The high upstream input values were the UCL95s of the TSS-normalized King County whole-water datasets.

For arsenic, the selected upstream input value was the mean (9 milligrams per kilogram dry weight [mg/kg dw]) of the Ecology upstream samples containing fines greater than 30%. The mean of the LDW RM 4.3 to 4.75 USACE core data (7 mg/kg dw) was selected as the low sensitivity value. The high sensitivity value (10 mg/kg dw) was the UCL95 of the Ecology upstream sediment samples containing fines greater than 30%. King County surface water TSS-normalized data and Ecology centrifuged solids data were not used in the selection of BCM upstream values for arsenic because the UCL95 for both of these datasets would have resulted in much higher modeled surface sediment concentrations than in the LDW baseline dataset. It is likely that these two datasets, especially the surface water dataset, contain finer particulates with higher arsenic concentrations than those that deposit in the LDW. These finer particles tend not to settle in the LDW (approximately 50% of the Green/Duwamish River solids [bed load and suspended solids combined] do not settle in the LDW).

For dioxins/furans, the Ecology upstream sediment samples (containing fines greater than 30%) and the Ecology upstream centrifuged solids were the only datasets used for selecting the BCM input values; there were neither core data from RM 4.3 to 4.75 nor whole-water dioxin/furan data among the other datasets. Because of the smaller datasets and the desire to evaluate a range of input values, a slightly different approach was used to select dioxin/furan BCM input values. The midpoint between the means of the two datasets is the mid-range value (4 ng TEQ/kg dw); the low sensitivity value is the mean of the Ecology upstream sediment samples containing fines greater than 30% (2 ng TEQ/kg dw); and the high sensitivity value is the midpoint between the mean and UCL95 of the Ecology upstream centrifuged solids dataset (8 ng TEQ/kg dw).

Dry weight concentrations for COCs based on upstream surface sediment samples may be biased low and may underrepresent the concentrations associated with the fraction of solids entering the LDW that have finer grain size and higher organic carbon concentrations. Silt- and clay-sized suspended solids represent 67% of the sediment entering the LDW. As a result of the settling of most sand-sized particles in Reach 3, silt- and clay-sized particles make up only about 35% of the sediment that settles in Reach 3, but more than 90% of the sediment that settles in Reaches 1 and 2. Case study literature and LDW data exist that support the relationship between COC concentrations, organic carbon content, and particle size. The relationship between particle size and organic carbon content and the various methods to account for these relationships and their potential effect on model results is explored in Section 5.3.3.

5.2.3.2 Contaminant Concentrations Associated with Lateral Source Sediments

Contaminant concentrations associated with storm drains and CSOs were evaluated to estimate concentrations associated with lateral source sediments.[11] The storm drain solids and CSO data were collected as part of ongoing source control programs for the LDW. All available storm drain data were compiled by Seattle Public Utilities (SPU) for source samples collected in areas draining to the LDW through June 2009 by SPU, the Boeing Company, and King County. These data included storm drain solids collected from on-site and right-of-way catch basins, in-line grab samples, and in-line sediment traps. The storm drain solids data were used to generate a range of lateral input concentrations for total PCBs, arsenic, and cPAHs for use in the BCM. Storm drain solids and sediment data collected near large stormwater outfalls draining urban areas in the greater Seattle area were used to establish BCM lateral input values for dioxins/furans. The King County CSO whole-water data were also considered and found to support the ranges of BCM lateral input values estimated from the storm drain solids dataset. Consequently, the same COC concentration values were used for both storm drains and CSOs and were also assumed for the stream inputs.

The lateral input values selected for use in the BCM are estimates, based on the assumption that contaminant concentrations in storm drain solids will decrease as a

11 Lateral source sediments include inputs from storm drains, CSOs, and streams discharging to the LDW.

result of source control efforts in the LDW drainage basin. The following assumptions were made for the BCM input values:

- The mid-range, or best-estimate, input value is a pragmatic assessment of what might be achieved in the future with anticipated levels of source control. This value is based on mean/median concentrations observed in the lateral dataset after excluding the highest concentrations in the dataset to represent control of high and medium priority sources.
- The high sensitivity value is a conservative representation of near future conditions assuming only modest success in management of high priority sources already identified by the SCWG.
- The low sensitivity value is an estimate of the best that might be achievable in 30 to 40 years with increased coverage and continued aggressive source control.

The assumed level of source control was based on best professional judgment of the SCWG and what is currently known about the distributions and current source(s) of each COC within the LDW drainage basin. The BCM input values reflect potential levels of source control that could occur over time. To simulate potential lateral inputs after implementing varying degrees of source control, the source tracing datasets were screened to remove all values above various concentrations already targeted for source control. Summary statistics were then generated for each level of assumed source control (high, medium, low). Table 5-1b presents the best-estimate BCM input values for lateral sources. The summary statistics for the four human health risk drivers (total PCBs, arsenic, cPAHs, and dioxins/furans) are provided in Tables 5-2a through 5-2d.

A general summary of the lateral input values selected for the BCM is presented below. The lateral sources memo (King County and SPU 2010) found in Appendix C, Part 3 describes the selection of the lateral input values in more detail. It should be noted that the high lateral input value is not intended to represent what sources could potentially exist throughout the drainage basins tributary to the LDW. This high value is used only to determine sensitivity of the model and the implications of inadequate source control at individual discharge locations; it is not an estimate of actual source loads or a target value for source control work. Similarly, the low sensitivity value should not be construed as a prediction of source control efficiency or as a determination of source control effectiveness or completeness. The actual effectiveness of source control can only be assessed after the fact because "complete" source control is the aggregate of many different actions applied to any given media, pathway, or source of COCs.

Total PCBs

Prior to generating summary statistics for total PCBs and to avoid skewing the summary statistics, the data were flow-weighted, including data from these targeted and known source areas: Rainier Commons, North Boeing Field/Georgetown Steam Plant, Terminal 117, and Boeing Plant 2/Jorgensen Forge. Flow-weighting takes into account the relative contribution of a specific contaminant by adjusting its concentration based on the land area and estimated annual runoff volume relative to the total contributing area in the LDW drainage basin. To reflect potential levels of source control that could occur over time, a range of screening concentrations was used to select the BCM lateral values for total PCBs. The mid-range BCM input value (300 μg/kg dw) is represented by the mean of data after excluding concentrations greater than 5,000 μg/kg dw.

Screening values of 2,000 and 10,000 μg/kg dw total PCBs were used to define the low and high BCM sensitivity values, respectively. If all samples with a total PCB concentration above a screening value of 2,000 μg/kg dw are removed from the dataset, the median of the remaining data is 100 μg/kg dw. This value was selected as the low BCM sensitivity value (100 μg/kg). When all samples with total PCB concentrations above a screening value of 10,000 μg/kg dw are removed from the dataset, the 90th percentile value of the remaining data is 1,000 μg/kg dw, which was selected as the high BCM sensitivity value.

cPAHs

Unlike total PCBs, cPAHs are expected to be difficult to control due to urbanization and major transportation routes in the LDW basin, and a multitude of current sources. Consequently, a more cautious approach was taken with the source tracing dataset by excluding cPAH concentrations above a single source control level of 25,000 μg TEQ/kg dw. Data for cPAHs were not flow-weighted because cPAH concentrations in the storm drain solids samples do not show a distinct geographic distribution, and higher concentrations of cPAHs are found throughout the LDW drainage basins, typically in drainage structures (catch basins and oil/water separators) at facilities engaged in transportation-related activities (e.g., bus and airport operations), maintenance facilities, service stations, foundries, and fast food facilities. The mean (1,400 μg TEQ/kg dw) of the data, excluding all samples with cPAH concentrations greater than 25,000 μg TEQ/kg dw, was selected as the BCM input value. The median (500 μg TEQ/kg dw) was selected as the low sensitivity value. The 90th percentile (3,400 μg TEQ/kg dw) was selected as the high BCM sensitivity value.

Arsenic

For arsenic, two different screening values (the SQS and cleanup screening level [CSL]) were used to reflect different potential levels of source control. The mid-range BCM input value of 13 mg/kg dw was selected based on the mean of the dataset, excluding all samples with arsenic concentrations above a screening value of 93 mg/kg dw (the CSL). The 90th percentile of the same dataset is 30 mg/kg dw, and this value was selected to represent the high BCM sensitivity value. If all samples with arsenic concentrations above a screening value of 57 mg/kg dw (the SQS) are removed from the dataset, the median of the remaining data is 9 mg/kg dw. This value was selected as the low BCM sensitivity value.

Dioxins/Furans

Available storm drain solids data for dioxins/furans were also used along with surface sediment sample data collected for the LDW RI in the vicinity of storm drains throughout the Greater Seattle metropolitan area to establish BCM lateral input values. By combining these two datasets (because the storm drain solids dataset was small compared to the other risk-driver datasets) and excluding one outlier, BCM lateral values were selected for dioxins/furans. The mean of 20 ng TEQ/kg dw was selected as the BCM input value; the median of 10 ng TEQ/kg dw as the low BCM sensitivity value; and the UCL95 of 40 ng TEQ/kg dw as the high BCM sensitivity value. In addition, the UCL95 rather than the 90th percentile was used to establish the high BCM sensitivity value, because it resulted in a more reasonable upper end estimate for the sensitivity analysis.

King County CSO Whole-Water Samples

In addition to the storm drain solids dataset, whole-water samples collected from CSOs by King County for analyses of PCBs, arsenic, and cPAHs were also considered when developing BCM lateral values. For both total PCBs and cPAHs, whole-water concentrations were divided by their sample-specific TSS concentrations to calculate TSS-normalized concentrations. This gives a conservative estimate that is likely biased high because it is assumed that all of the PCBs and cPAHs are on the particulate fraction and none are in the dissolved or colloidal phases. For arsenic, paired total and dissolved concentrations were used to estimate the portions of the total arsenic concentrations associated with the particulate fraction. These were then divided by the sample-specific TSS concentrations to calculate a TSS-normalized concentration for arsenic. Whole-water samples collected from CSOs in the LDW had not been analyzed for dioxins/furans at the time this document was prepared. Summary statistics for CSO data are provided in the lateral source memo (King County and SPU 2010) found in Appendix C, Part 3.

5.2.3.3 Contaminant Concentrations of Existing Bed Sediments

Existing bed sediment contaminant concentrations were developed by spatially interpolating surface sediment data from the FS baseline dataset for total PCBs, arsenic, and cPAHs. An inverse distance weighting (IDW) algorithm was used to interpolate the data. The IDW methodology is documented in Appendix A.

Existing bed sediment concentrations for dioxins/furans were developed by applying Thiessen polygons to the dioxin/furan surface sediment data from the FS baseline dataset. For Washington State Sediment Management Standards (SMS) contaminants, SQS and CSL exceedances at surface sediment stations were also spatially applied using

Thiessen polygons. In this case, dry weight or organic carbon (oc)-normalized concentrations were compared to SQS/CSL or apparent effects threshold criteria, as appropriate for each contaminant. Thiessen polygons were designated as a pass, SQS exceedance, or CSL exceedance. Sediment toxicity results trumped SMS chemistry results. For example, a Thiessen polygon with a contaminant CSL exceedance, but a toxicity pass, was coded as a pass.

Collectively, these risk drivers comprise the FS baseline dataset used to map "existing conditions" in the LDW. The FS baseline dataset spans about 18 years (1991 to 2009) of data collection efforts. It is likely that current concentrations of some COCs at stations sampled many years ago may now be lower than what is reflected in the FS baseline dataset (see Appendix F).

5.2.3.4 Post-Remedy Bed Sediment Replacement Values

In areas that would be actively remediated under different cleanup alternatives, the existing bed sediment concentration (C_{bed}) is replaced with a value representing near-term (0 to 2 years) conditions following the cleanup. The post-remedy surface sediment conditions are influenced by multiple factors. This subsection describes the assumptions used to model the post-cleanup concentrations.

Experience at other sediment remediation sites has shown that contaminant concentrations in the sediment bed shortly after the completion of dredging or capping cannot be assumed to be zero and are often above background (NRC 2007, EPA 2005b, Anchor 2003). This occurs because: 1) some degree of residual surface contamination always exists from the resettling of contaminated sediments suspended during remedial activities; 2) material used for capping of subsurface sediment exposed after dredging contains low concentrations of these COCs; and 3) existing adjacent sediments can become resuspended and then deposited in remediated areas.

Post-remedy bed sediment replacement values within a remediated area reflect an assumed combination of clean backfill material (e.g., from capping or ENR, and using or not using post-dredge residuals management) and the average concentration of surrounding unremediated sediments. To derive a replacement value based on this assumption, estimates of both values are required. The UCL95 values for the 2008 EPA Puget Sound Ocean Survey Vessel (OSV) *Bold* survey (EPA OSV *Bold* survey) data were used to estimate the contaminant concentrations in clean backfill. These data correspond to natural background estimates for Puget Sound.[12]

However, once clean material is placed, other sediments start settling on the backfill. These sediments are some combination of upstream and lateral inputs, resuspended bed sediments, and dredge residuals. For the purposes of this FS, the average concentration of bed sediments that will not be actively remediated was assumed to be

[12] Data were also collected from the Strait of Juan de Fuca and Strait of Georgia.

representative of this mixture of inputs onto the clean backfill. The average concentration of unremediated sediments was derived using the SWACs outside of remediated areas. The average concentrations remaining outside of AOPC 1 and outside AOPC 2 for Alternative 6 (see Section 6 for AOPCs and Section 8 for alternative footprints) were used in this analysis. The post-remedy bed sediment replacement value was applied to the actively remediated footprint. Clean material was assumed not to be deposited outside of the active footprint.[13]

To calculate a range of post-remedy bed sediment replacement values, the following ratios of clean material to the post-remedy SWAC were assumed: 50:50 for the mid-range BCM input value, 75:25 for the low sensitivity value, and 25:75 for the high sensitivity value.

Post-remedy bed sediment replacement values for total PCBs, arsenic, cPAHs, and dioxins/furans are presented in Table 5-1c. The degree of residual contamination is dependent on several factors, including the type of remedial activity, specific design elements, construction methods, best management practices, engineering controls, and contingency measures (discussed further in Section 7.1). Therefore, post-remedy bed sediment replacement values for use as input parameters to the BCM were developed as a range using the proportioning values described above and best professional judgment. The same post-remedy bed sediment replacement value is applied to areas that are to be dredged, capped, undergo ENR, or have a thin-layer placement of sand inside the dredge footprint for residuals management.

5.2.4 Inputs and Application of the BCM for Other SMS Contaminants

The BCM can also be used to estimate future SQS and CSL exceedances for SMS contaminants. In the BCM, a particular SMS contaminant is selected for each point, and the BCM assigns that point into one of three categories in the future: below the SQS, SQS exceedance (but below the CSL), or CSL exceedance. The BCM equation (Equation 5-1) can be used to estimate future concentrations for any contaminant having available upstream and lateral input values. For the FS, these calculations were conducted on a subset of the SMS contaminants, termed “representative” contaminants. This subset was chosen from the full list of SMS contaminants because: 1) not every SMS contaminant has lateral and upstream data available; 2) several SMS contaminants had very low detection frequencies; and 3) indicator SMS contaminants within a specific class (e.g., PAHs) may well represent the behavior of that class. The representative SMS contaminants were identified by querying the database and counting the number of samples that exceeded the SQS for each contaminant. Those with the most frequently

13 The post-remedy bed sediment replacement value was not applied outside of the active remedial footprint because a thin layer of sand will be applied to manage dredge residuals where needed. It was assumed that such application would, on average, return any sediments affected by residuals outside of the dredge footprint to preconstruction concentrations.

detected exceedances were selected to represent a group/class (Table 5-3). They include bis(2-ethylhexyl)phthalate (BEHP) (phthalate group); chrysene, fluoranthene, and phenantherene (PAH group); and mercury and zinc (metal group). Arsenic and total PCBs were also included to assess the spatial distribution of these risk drivers in a manner consistent with the other SMS contaminants. Detected SQS/CSL exceedances for total PCBs were assessed using sample-by-sample oc-normalizations to ensure that detected exceedances were not missed in the interpolated IDW maps based on dry weight (see Table 5-2a).

After the initial representative SMS contaminant list was established, locations were identified that exceeded the SQS for other SMS contaminants, and additional SMS contaminants were added to the list so that at least one representative SMS contaminant was identified for each location. As a result, butylbenzyl-phthalate, phenol, acenaphthalene, and indeno(1,2,3-cd)pyrene were added. Table 5-3 lists these SMS contaminants and the upstream and lateral values established for each.

For each location that had a detected SQS exceedance in the FS baseline dataset, the maximum exceedance ratio above the SQS and the SMS contaminant responsible for that exceedance were determined. Typically, the SMS contaminant responsible for the highest exceedance was one of the representative SMS contaminants, and was usually total PCBs.[14] If the SMS contaminant with the maximum exceedance ratio was not in the representative SMS contaminant list, a representative SMS contaminant of the same chemical class that also exceeded the SQS at that location was used in the BCM. The future BEHP concentrations were also predicted by the BCM for each location because this SMS contaminant is a concern due to lateral sources.

5.2.4.1 Input Values for Representative SMS Contaminants

Lateral input values were determined by querying the City of Seattle's lateral source database (SPU 2010). Upstream input values were derived from the USACE Dredged Analysis Information System (DAIS) core database using data through 2009 (USACE 2009b, 2009c). For the City of Seattle data, all storm drain solids data were queried for each COC. The log-normal mean of the dataset was then calculated and used as the lateral inflow value for that contaminant (Table 5-3) after outliers were removed. The USACE core data from the Upper Turning Basin, RM 4.3 to 4.75, were used to represent the incoming sediment from upriver because that is the only upstream dataset analyzed for all SMS contaminants over a sufficient period of time. The data were screened to include only those collected after 1990 (prior data were excluded). The median of the dataset for each contaminant was then calculated and used as the upstream value for that contaminant. Table 5-3 lists the lateral and upstream inflow values used for each representative contaminant. No post-remedy bed sediment replacement values were used for these points. If a point was located in an actively remediated area, it was

14 Several locations were sampled only for PCBs.

considered to be remediated below the SQS and removed from further bed composition modeling at that location.

5.2.4.2 BCM Equation Using Lateral and Upstream Input Parameters

For those locations where the detected concentration of any SMS contaminant exceeded the SQS at the start of the modeling period (and was not a toxicity pass), the BCM equation was run using Equation 5-1. The upstream and lateral input values discussed in Section 5.2.4.1 were employed for the contaminant selected to represent that location. Equation 5-1 was also used to estimate exceedances at the end of 10 years for BEHP, a contaminant that chronically exceeds the SQS and is generally associated with non-point source lateral discharges.

Because the lateral and upstream input parameters are on a dry weight basis, the BCM Equation 5-1 was run for the representative SMS contaminants using dry weight concentrations. For each SMS contaminant modeled at a location and having oc-normalized SMS criteria, the dry weight concentrations predicted for each time period modeled were compared to the baseline dry weight concentration. This process yielded a percent reduction that was then applied to the baseline oc-normalized concentration. If the resulting value exceeded the SQS, then the station was considered to be an SQS exceedance at the end of the modeling period.

5.2.5 BCM Output and Model Sensitivity

The output of the BCM is predicted contaminant concentrations for each grid cell[15] at specified time intervals (i.e., 5, 10, 15, 20, 25, 30, 35, 40, and 45 years). Summary statistics, such as site-wide and area-specific SWACs can be calculated for the distributions of surface sediment concentrations and used in assessing remedy effectiveness. Area-specific statistics can be calculated to assess beach play and potential clamming area-focused remedies.

Sensitivity runs of the BCM are used to evaluate the effect of varying contaminant concentrations associated with upstream and lateral source sediments and post-remedy bed sediment replacement values (in remediated areas) on bed sediment concentrations over time. The sensitivity of the BCM was investigated by looking at how changes in input parameters affect the output (Appendix C, Part 5).

When evaluating model uncertainty, it is important to understand that the contaminant concentration in a specific area is not as straightforward as selecting a specific cell and assuming that the concentration in that cell is accurately represented by the BCM value. For developing the initial contaminant concentration, the BCM uses a 10 ft × 10 ft cell size to capture the spatial scale of surface sediment contaminant concentrations used in IDW interpolations (see Appendix A). The BCM grid is used for computing SWACs in

[15] The BCM analysis uses grid cell sizes of 10-ft by 10-ft, the same as those used for the IDW interpolation of surface sediment concentrations.

this FS. However, it should not be construed that the 10 × 10-foot grid is appropriate for design purposes and the grid should not be used beyond this FS. Remedial design should be based on data and analysis specific to a design area.

Existing surface sediment contaminant data are more sparsely located in some areas and the initial contaminant concentration for a grid cell of interest may be represented using a data point that was collected anywhere from a few feet up to more than one hundred feet from the location of the grid cell. Nevertheless, when averaged over larger areas, model results are still relevant. However, the BCM model resolution on finer scales is limited not only by resolution of initial condition data but also by STM grid cell resolution[16] and other factors (such as representation of lateral load distribution). For example, specific "hot spots" may cover only a small part of an STM grid cell that extends from the bank to fairly deep water. The model-predicted current velocity and sedimentation rate are assumed to be spatially constant over this STM grid cell. The actual current velocity, and therefore sedimentation rate, may vary substantially over this STM grid cell, especially for cells that are near-channel or near-shore. The current velocity and sedimentation rate may be representative of the average for the area covered by the STM grid cell, but may not accurately represent these parameters within some subdomain of the STM grid cell. It will always be important to investigate and understand model input and processes (such as the scale of predicted sedimentation rates from the STM) when evaluating the appropriate size of areas where BCM-predicted contaminant concentrations are valid.

5.3 Additional Analyses Related to Natural Recovery Potential

The STM and the BCM presented above address most of the processes that affect natural recovery. However, this FS assesses several processes not explicitly addressed in the RI (Windward 2010) and the Final STM report (QEA 2008). These include:

- The effect of tugs on sediments in berthing areas (disturbance activity)
- Additional model scenario runs using the calibrated STM to answer several specific FS questions
- Influence of grain size and organic carbon on sediment contaminant concentrations.

The following sections discuss these other processes that may affect natural recovery.

5.3.1 Incorporating Effects of Disturbance Activity

The STM and BCM predict changes to the sediment bed for long time periods from natural processes and estimated contaminant loadings. However, STM and BCM

[16] STM grid cells range in size from range from 0.1 to 4 acres, with the median area of a grid cell being 0.5 acre (e.g., a 100 ft-by-200 ft area is roughly 0.5 acres).

predictions do not incorporate long-term changes to the sediment bed that could be caused by deep disturbance of sediments (i.e., up to 2 ft), such as:

- Emergency and high-power (i.e., outside of routine operating procedures) tug or ship maneuvering, ship grounding, small boat activities in shallow water, and construction and maintenance-related activities in the LDW may cause deep scour (Section 5.3.1.3), which mixes subsurface sediments with surface sediments, resulting in higher contaminant concentrations at the surface.

- Seismic events (earthquakes) could result in liquefaction-induced ground movements that could damage in-water and upland infrastructures and could result in deep disturbance of subsurface contamination, resulting in higher contaminant concentrations at the surface.[17]

Such disturbances would likely be isolated and infrequent, but the cumulative effects could be of concern over the long term. Several approaches were utilized to increase our understanding of how BCM-predicted SWAC values are influenced by both natural and anthropogenic processes. This section discusses two topics:

- Influence on bed erosion of vessels maneuvering in the navigation channel and in areas deep enough to accommodate vessel drafts based on propeller shear stress modeling

- Areas where episodic, high-energy disturbance activity can expose more highly contaminated underlying sediments.

5.3.1.1 Propeller-Scour Model of Maneuvering Vessels

Propeller scour from tugs transiting the navigation channel under routine operating procedures in the LDW was evaluated in the STAR (Windward and QEA 2008). The analysis showed that the maximum scour from tugs transiting the navigation channel is less than 1 cm within the navigation channel and approximately 1 to 2 cm on the benches adjacent to the navigation channel. The higher potential scour on the benches is due to tugs traveling on the edge of the navigation channel adjacent to shallower depths on the benches.

Assuming that sediments resuspended by propellers redeposit near the resuspension site, then anthropogenic scour in the navigation channel and benches acts only as a mixing process in the surface layer, augmenting the mixing induced by bioturbation (which is typically greatest within the top 10 cm of sediment). The STM assumes a 0- to 10-cm mixed layer of sediment at the surface; hence, the effects of propeller scour

[17] Although earthquakes can also result in admixture of subsurface and surface sediments, this potential disturbance is not explicitly discussed in this section, because the range of effects is not readily modeled with the information currently available. However, see Section 8.1.3 for more information.

associated with vessels moving in the navigation channel are consistent with the STM assumptions for tugs operating in the navigation channel.

However, the propeller scour analysis presented in the STAR is not applicable to tugs or vessels maneuvering in areas shallower than the navigation channel or when emergency and high-power operations are needed. Tugs may occasionally need to use more power while maneuvering barges in and out of berths, and tugs may be stationary for longer periods of time (while still operating their propellers).

A modeling approach developed by the USACE was applied to the LDW for maneuvering vessels. This model was developed with an analysis of currents and shear stresses induced by towboats and barges on the Mississippi River (Maynord 2000). The methods and model were used for computing bottom currents and shear stresses caused by moving barges and propeller scour in the LDW. A detailed discussion of the Maynord model is presented in Appendix C, Part 7. Briefly, the model maps the velocity and the associated shear stress induced by the propellers that reaches the river bottom. The shear stress time series and the sediment characteristics at the river bottom determine the amount of scour that will occur over a period of time. The velocity is related to the amount of power applied by the tug. However, tugs may operate at higher power for short periods of time. The applied power under different operating conditions and durations was determined from interviews with tug operators. The analysis followed a similar approach as in the STAR (Windward and QEA 2008), using the same two tugs for model input parameters. The larger tug, *Sea Valiant*, operates downstream of the First Avenue South bridge (RM 2.1), while the smaller tug (*J.T. Quigg*) is able to operate in shallower water upstream of the bridge.

No precise methods are available to relate propeller-induced shear stress to sediment erosion. However, rough estimates of the scour magnitude can be developed. Based on the analysis,[18] localized deep (more than 10 cm) vessel scour may occur for tugs operating in shallow water and at higher power, as described by tug operators working under emergency conditions (see Appendix C, Part 7). Vessel scour depth is strongly affected by the distance between the propeller and the sediment bed, with substantially less scour in deeper water. Other factors influencing propeller scour are propeller angle, thrust, blade configuration, and duration of the high-power event under stationary conditions. For most berthing areas and operational conditions (in deeper water operations under normal power conditions), the depth of scour is estimated to be 10 cm or less, which would not necessarily disturb and expose subsurface contaminant concentrations (see Appendix C, Part 7). However, as described in Section 5.3.1.2, infrequent events can scour more than 10 cm. Results of this scour analysis, combined with empirical evidence of scour, have been incorporated into the FS in two ways: the

[18] This analysis was limited to the vertical depth of the Sedflume core data collected during the RI (about 30 cm).

development of recovery categories (Section 6) and in the technology assignments for individual remedial alternatives (Section 8). The following section discusses other components of scour.

5.3.1.2 Episodic Deep Disturbances Leading to Exposure of Subsurface Contamination

Potential influences on SWAC from routine vessel operations are described above. However, less frequent and episodic events in an active navigation area such as the LDW may induce disturbance of subsurface sediments, exposing subsurface contamination. In this FS, this process is called deep disturbance. Deep disturbances may involve ships operating with excessive propeller power, ship groundings, emergency maneuverings, or seismic events. Maintenance operations such as dock construction/maintenance and vessel maintenance may also cause deep disturbance.

The STM/BCM models were not set up to model deeper disturbance events, so this FS conducted a separate sensitivity analysis of deep sediment disturbance to consider the potential effects of such disturbance events on STM/BCM-predicted SWACs. This disturbance analysis introduces an additional, local source of contamination: the subsurface sediment bed. Natural processes (apart from earthquakes) and routine ship operations in the LDW will not typically mix the surface 0- to 10-cm layer with deeper subsurface sediments except in areas that were identified on the basis of known ship activity and from precision bathymetry, which suggested deeper erosion (Section 5.3.2.7). However, some lines of empirical evidence (geochronology cores and sediment concentration profiles) suggest that in some areas subsurface sediments may have been disturbed as a result of anthropogenic activity. There is evidence, based on contaminant profiles in some cores and geochronological data, that deep disturbance events may have hindered recovery at localized areas. The frequency and magnitude of these events is unknown. Influence of such events on BCM SWAC projections was analyzed in Appendix M, Part 5, and results are compared in Section 10. Changes in the long-term SWAC, based on potential exposure of contamination remaining in the subsurface sediment after dredging or capping, are estimated for each alternative as a function of the long-term SWAC, the size of the area disturbed, and the average contaminant concentration remaining in the subsurface after remediation. Because the total area of deep disturbance is unknown, results are presented as change in SWAC as a function of acreage that has experienced deep disturbance. Because the frequency of such events is also unknown, this FS assumes that disturbed areas would have to be exposed continuously to produce a measurable difference in the long-term model-predicted SWAC of 25%. This 25% threshold is considered the minimum change needed to detect a difference between two SWAC values (see Section 9.1.2.1). Results for the deep disturbance analysis (provided in Section 10) range from 11 to 43 acres (2% to 10% of the total LDW acreage).

5.3.2 Additional Special Scenario STM Runs

Six additional scenarios were run using the STM to further understand the movement of sediment particles within the LDW and the potential effects on the natural recovery analysis. The additional runs assessed:

1) Potential for recontamination of EAAs
2) Effect of more detailed distribution of discharges from lateral sources on the bed composition
3) Movement via tidal currents of resuspended sediment from reaches downstream of the Upper Turning Basin upstream into the Upper Turning Basin
4) Movement and deposition of sediment between Reaches 1, 2, and 3
5) Fate of sediment scoured from depths greater than 10 cm
6) Tracking of existing bed sediment movement
7) Natural recovery hindered in selected berthing areas.

A description of each of these scenarios and a summary of the results are presented in Table 5-4. A detailed accounting of scenarios 1 through 6 is presented in Appendix C, Parts 4 and 5. The findings of this work are generally consistent with the CSM (see Section 2) and support key assumptions and analyses inherent in the BCM and the assignment of remedial technologies (Section 8). The primary findings of the special scenario STM runs are discussed below.

5.3.2.1 Scenario 1: Potential Recontamination of EAAs

The purpose of this scenario was to assess the potential for remediated EAAs to be recontaminated over time by areas located outside of the EAA footprints that would be allowed to recover naturally. This may affect decisions concerning the timing and sequencing of remedial activities at specific EAAs.

The results of this analysis indicate it is unlikely that remediated areas will be recontaminated by unremediated areas unless the areas are adjacent to each other. Material resuspended from unremediated areas during high-flow events is estimated to account for less than 5% of the material that settles in remediated EAA footprints over a 10-yr period[19] (see Figure 5-6). The BCM analysis on this scenario indicates that recontamination of EAAs above the SQS (the SQS was used as a point of comparison for this analysis because other potential remedial action levels [RALs] vary by alternative)

[19] Only a few grid cells have been identified as having non-EAA source material in the range of 5 to 20% and most of these are in Reach 2. The average across the LDW is generally less than 5%.

is more likely to occur near outfalls as a result of lateral source inputs than to scour and settling of bed sediment from outside EAAs.

5.3.2.2 Scenario 2: Distributed Discharges from Lateral Sources

This scenario examined certain simplifying assumptions that were used in the STM for lateral discharge locations (for storm drains and streams), and refined those assumptions to better account for actual lateral discharge distribution. In the original STM (QEA 2008), all Duwamish watershed discharges were aggregated into 16 discharge points along the LDW. The discharge points consolidated total area runoff from storm drains to the major outfalls and did not include the more widely distributed smaller outfalls located along the shoreline. CSOs that discharge to the LDW were also included, but these were modeled at their actual locations.

In this distributed discharges modeling scenario, finer drainage basin delineations were used to more accurately reflect actual drainage subbasins and outfalls (pipe locations) of storm drains, resulting in 13 major storm drains, 9 CSOs, and 11 waterfront areas that discharge to the LDW through numerous small outfalls. The revised load estimates and drainage basins for storm drains, creeks, and City CSOs (SPU 2008) were presented and are summarized in Appendix C. Because the distributed load simulation more accurately represents the distribution of lateral loads along the shoreline, it was carried forward as the FS base case loading condition. The lateral loads used in the FS base case are shown in Figure 5-7.

5.3.2.3 Scenario 3: Movement of LDW Bed Sediment into the Upper Turning Basin

This scenario examined the degree to which bed sediments from elsewhere in the LDW may become resuspended, transported upstream, and deposit in the Upper Turning Basin (above RM 4.0). The Upper Turning Basin sediment composition and chemistry is only minimally affected (less than 0.01%) by sediment moving upstream with tidal currents (Figure 5-8). Figure 5-8 shows the geographic distribution of sediment settling in Reach 3 but originating from downstream of RM 4.0 (from Reaches 1 and 2). Only the area between RM 4.0 and 4.1, Slip 6, and a few other isolated grid cells in Reach 3 are estimated to have more than 0.01% sediment contribution from bed sediment downstream of RM 4.0, and even these areas are less than 0.05%. This estimate is in agreement with the 10-year sediment mass balance, which indicates that about 240 MT moves from Reaches 1 and 2 and is expected to deposit in Reach 3 (see Scenario 4). This is extremely small compared to the estimated total sedimentation in Reach 3 of 2.3 million MT over 30 years; 99.99% of this sedimentation is from upstream sediments. Based on this analysis and the contribution of sediments from lateral sources (see Section 5.3.2.2), the sediment in the Upper Turning Basin and the navigation channel above RM 4.1 should not be adversely affected by sediments transported from other portions of the LDW. The BCM analysis for this scenario shows that the predicted COC concentrations in the Upper Turning Basin are for the most part very low and negligibly affected by the amount of sediment deposited from downstream. This analysis also supports the use of Upper Turning Basin sediments in the navigation channel (RM 4.3 to RM 4.75) as representing the COC concentrations in sediments originating from the Green/Duwamish River.

5.3.2.4 Scenario 4: Movement of Bed Sediments between Reaches

This scenario examined the degree to which bed sediments in one reach of the river may be resuspended and transported to another reach. These results may be important in assessing recontamination potential between reaches and in assessing if locations would be important for sequencing the remedial alternatives. Sediment exchange (either upstream or downstream) is strongest between Reach 1 and Reach 2, while Reach 3 primarily contributes sediment to downstream reaches with very little sediment transported from downstream reaches back to Reach 3 (Figure 5-9). In addition, much of the bed sediment that is resuspended in a reach resettles in that same reach.

Reach 3 receives a large amount of sediment from the Green/Duwamish River as a combination of suspended load and bed load, the latter consisting mostly of sand. This reach is regularly dredged by the USACE, particularly in the Upper Turning Basin. Maintenance dredging, applied by the USACE on the cycles that we have seen in the past, should not change current natural recovery processes because it primarily removes sand that is not readily transported downstream and therefore is not a significant component of net sedimentation and natural recovery in Reaches 1 and 2.

5.3.2.5 Scenario 5: Sediment Scoured from Greater than 10 cm Depth

This analysis was used to evaluate whether scour and transport of deeper sediments may influence the waterway-wide SWAC during an extreme high-flow event. Scour during a 100-year high-flow event was analyzed in the STM report as a 30-day simulation (QEA 2008). Scour in excess of a 10-cm depth (up to about 22 cm) occurs in portions of the LDW from RM 2.9 to RM 3.9 and in isolated areas between RM 4.2 and RM 4.7. Most of these areas are in the navigation channel.

Sediment scoured from below 10 cm during a 100-year high-flow event was modeled over a 10-year period. In Figure 5-10a, the STM estimates that approximately 200,000 MT of sediment settles in the LDW during a 100-year high-flow event and of that amount, approximately 70,000 MT is eroded from the bed. However, as shown in Figure 5-10b, only about 6,600 MT of the sediment that settles is eroded from below 10 cm, which is only about 3% of the deposition during the 100-year high-flow event. Consequently, sediment eroded from below 10 cm during high-flow events, and mostly from Reach 2, makes a negligible contribution to sediment transport and deposition in the LDW during those high-flow events. In Reach 2, about 45% of eroded material is estimated to redeposit in the same reach (3,800 MT deposited out of 8,700 MT eroded) while deposition of upstream sediment and eroded shallow sediments from other areas of the LDW is estimated to be approximately ten times this amount. Consequently, erosion and redeposition of sediment scoured below 10 cm makes a negligible contribution to the potential for redistribution of subsurface sediment between reaches during high-flow events. In addition, very few sediment cores in these potential scour areas had SQS exceedances and those with exceedances were located in or adjacent to EAAs (see Appendix C, Part 4).

The areas estimated to have greater than 10 cm of scour total about 22 acres (Figure 5-11: and see Appendix C, Part 4, Scenario 5). Subsurface bed sediments (below the 10-cm depth) are generally more contaminated than surface sediments (0- to 10-cm depth). However, core data indicate that only a few areas have contaminant concentrations above the SQS or CSL in areas prone to natural erosion. The total area with surface exceedances above the SQS in areas with more than 10 cm of scour during high-flow events is 5.4 acres; of that, 1.5 acres are in the EAAs. In summary, empirical and modeling data indicate that the majority of subsurface sediment eroded will not have significantly higher contaminant concentrations.

5.3.2.6 Scenario 6: Movement of Existing Bed Sediment

This scenario was conducted to track the movement of sediment within the LDW. In the BCM, the initial COC concentration in the bed sediment at a given point is assumed to be unchanged through time. This means that the changes in COC concentrations at any given location are attributable only to the net sedimentation of upstream and lateral source sediments and mixing with bed sediments at that location. In actuality, bed sediments from other areas of the LDW are resuspended and settle throughout the waterway. The movement of resuspended bed sediment (distal sediment) and its effect on COC concentrations was evaluated by separately tracking the deposition of resuspended bed sediment and original bed sediment over time. This allows the COC concentration to change as a result of deposition of bed sediment as well as deposition of upstream and lateral source sediments. The STM analysis results are presented in Appendix C, Part 5 (LDW STM Bed-tracking Scenario Simulation).

The STM output was used in a BCM analysis with four contaminant inputs, one each for upstream, lateral, bed, and distal sediments. To account for the effect redeposition of sediment would have in a reach (the distal fraction), the total PCB concentration on resuspended sediment was based on a weighted average of the mass of sediment resuspended from each of the three reaches multiplied by the reach-wide SWAC for the reach where the sediment originated. For example, the PCB concentration associated with distal sediment from Reach 3 uses the SWAC from Reach 3 as the input value. This is an approximation that does not strictly conserve contaminant mass. However, it provides a check on the standard BCM analysis and shows the importance of resuspension and redeposition of bed sediment relative to other processes in the LDW on future SWACs. This analysis was conducted with the assumption that remediation of the EAAs had been completed.

This analysis indicates that accounting for bed sediment movement produces no substantial change to the total PCB SWAC at the end of 10 years, both on a site-wide and reach-wide basis (Table 5-5). The calculated total PCB SWAC, when this effect is considered, is unchanged in Reaches 1 and 3, and 6% lower in Reach 2. Site-wide, the decrease in predicted SWAC is approximately 1%. The changes are small because throughout the LDW, resuspended bed sediment that resettles in the LDW is a small component of the sediment mass balance. The resuspended bed sediment that settles in the LDW is only 5%, 12%, and 9% of the total mass of sediment depositing in Reaches 1, 2, and 3, respectively (see Appendix C, Part 5). In Reach 2, which has the highest fraction of bed sediment that resettles, most of the sediment that resettles originates in Reach 3, where total PCB concentrations are generally lower than in the other reaches. Overall, this simulation shows that redistribution of existing bed sediment by high-flow events has a minor effect on recovery predictions. The largest change is in Reach 2; however, the approach used in the BCM base case analysis likely underestimates natural recovery in Reach 2 compared to a model that actually tracks the movement of individual sediment particles.

5.3.2.7 Scenario 7: Natural Recovery Hindered in Selected Scour and Berthing Areas

In localized areas where high levels of routine ship activity occur and depths are sufficiently shallow to permit disturbance of the sediment bottom, natural recovery may still be occurring, but over longer periods.[20] Propeller scour from ordinary ship maneuvering activities temporarily resuspends surficial bed sediment, after which a portion of that material resettles in the same footprint, with the coarser material more likely to resettle and fines more likely to be transported away, depending on tides and currents. A constant source of incoming material from upstream also amends the bed sediment so that any exposed contaminant concentrations are reduced over time. Regular maintenance dredging in the navigation channel and active berthing areas indicates that net sedimentation is occurring and that sediment removal is required to maintain acceptable water depths for navigation. Empirical trends, where data are available, show that burial and sediment recovery are occurring in most of these areas (see Appendix F). Berthing areas were considered on a case-by-case basis during development of technology assumptions.

Some empirical data indicate that recovery may be hindered by normal navigation activities. These activities only rarely induce deep disturbance but, by continual resuspension of the unconsolidated surface sediment layer could reduce accumulation of layers of cleaner upstream sediments. To examine effects of such navigation activities on BCM predictions, a scenario was developed that assumes that natural recovery does not occur in areas considered prone to regular anthropogenic resuspension and transport of sediments (i.e., the berthing areas). At many of these locations, the STM

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[20] This is for normal or routine operating conditions. See Section 5.3.1.2 for evaluation of extreme, episodic conditions.

indicates sedimentation during the recovery period. In this sensitivity analysis, the initial bed contaminant concentrations in potential scour areas and berthing areas are held constant for all BCM analyses throughout the 10-year period modeled in the BCM (i.e., no sedimentation and recovery). This assumption is the best available approach to bound uncertainty pertaining to effects of vessel scour on surface concentrations predicted by the BCM.

Areas held constant in this analysis were selected to include areas of potential scour from routine navigational activities: 1) berthing areas with net sedimentation rates less than 0.5 cm/yr (see Figure 2-11), and 2) vessel scour areas identified using sun-illuminated maps (Figure 5-4). This method has several limiting assumptions. Specifically, sun-illuminated maps are a snapshot in time of bed locations that have been disturbed by ship activity. The areas identified using this method may change in the future. Therefore, the selected areas for propeller scour are not intended as a robust indicator of all areas that may be influenced by propeller scour.

A BCM sensitivity was conducted over the 10 years following construction in order to compare the site-wide and reach-wide total PCB SWACs for the base case to a case with constant bed sediment total PCB concentration in potential scour areas.

Alternative 3: 10-Year Model Condition[a]	Total PCB SWAC (µg/kg dw)			
	Site-Wide	Reach 1	Reach 2	Reach 3
Base Case (includes modeled recovery in vessel scour areas)	62	68	61	39
Holding Cells Constant in vessel scour areas and berthing areas with net sedimentation rates <0.5 cm/yr	69	75	72	42

a. Exploratory test case condition at 10 years following remedy completion of Alternative 3 using mid-range BCM values, FS baseline data, and model assumptions used in the Draft Final FS.

This bounding exercise indicates that estimates of total PCB SWAC are not very sensitive to scour effects from normal operation of transiting vessel traffic. Vessel traffic can have some influence on SWACs (by hindering natural sedimentation and recovery), but this effect is less than a 25%[21] difference (considered the minimum detectable difference between SWAC estimates). For this scenario, the SWAC is about 10% higher for site-wide and reach-wide total PCB SWACs, except in Reach 2, which is 18% higher. However, scour and the resuspension of freshly deposited material may result in greater increases in localized areas and will need to be factored into remedial design in potential areas where natural recovery is hindered by vessel scour (see Section 6).

5.3.3 Influence of Grain Size and Organic Carbon on Sediment Chemistry

Hydrophobic compounds, such as PCBs, more readily adsorb to the organic substances attached to sediment particles than they do to the inorganic surface of sediment

[21] A threshold of 25% is considered the minimum change needed to detect a difference between two SWAC values (see Section 9.1.2.1).

particles. As a result, the amount of organic carbon influences the potential adsorption of PCBs (and other hydrophobic COCs) to the particles. In addition, higher contaminant concentrations are generally associated with finer-grained sediment (clay/silt). This may be particularly important in the LDW as the grain-size distribution becomes finer from upstream to downstream (Figure 5-12), and the risk drivers are positively correlated with total organic carbon (TOC) and percent fines in the LDW (see Appendix C, Part 3b, Table 8).

Contaminant concentrations in the BCM were assigned equally to all grain sizes. In this evaluation, the sensitivity of the BCM is tested to determine the influence that size fractionation of COCs has on SWAC results. Total PCBs were assigned to the four STM particle size classes (Classes 1A [less than 10 microns], 1B [10 to 62 microns], 2 [62 to 250 microns], and 3 [250 to 2,000 microns]) in varying concentrations based on particle size (for additional details of this analysis, see Appendix C, Part 9). Three different partitioning approaches were used for assigning total PCB concentrations to the different particle size fractions (Table 5-6a). The results of the three analyses are shown in Table 5-6b.

Overall, this sensitivity analysis demonstrated that different approaches to assigning total PCB concentrations by size fraction did not substantially change the results for the BCM analysis unless the assumptions produced an increase in mass loading of total PCBs. For example, Approaches 2 and 3 demonstrated that the SWAC would decrease (14%) or remain approximately the same for cases where mass loading of the COC was not changed. This is because higher PCB concentrations are being assigned to Class 1A particles compared to the other size classes, but 90 percent of the Class 1A material passes through the LDW without settling. Approach 1 resulted in an increase in the site-wide SWAC by approximately 42% because the approach also increased the PCB loading from upstream and lateral sources by approximately 100%.

Preferential partitioning of contaminants to finer size fractions is well documented in the literature and can affect the distribution and bioavailability of contaminants. To account for this preferential partitioning, dry weight values are often normalized to the amount of organic carbon present in a sample (i.e., oc-normalization; Michelsen 1992). Many of the SMS contaminants have oc-normalized criteria.

5.4 Empirical Trends and STM/BCM Reliability

The reliability of the STM to estimate net sedimentation rates, and of the BCM to predict changes in contaminant concentrations, is supported by empirical trends (i.e., net sedimentation rates from time markers in cores and changes in contaminant concentrations over time). Consistency between empirically-derived net sedimentation rates and the STM and between the BCM and empirical trends in COC concentrations in surface sediments lends credibility to the STM/BCM prediction of natural recovery in the future. Contaminant trends in surface sediments were evaluated both by changes in risk-driver concentrations by depth in cores and by changes in their concentrations over time at resampled surface sediment locations. Appendix F presents these empirical data and the methods by which these data were evaluated. This section summarizes the findings presented in Appendix F.

Net sedimentation rates calculated from time markers (Pb210, Cs137, and contaminant peak dating) in cores that supported net sedimentation are in general agreement with rates estimated by the STM. Seven out of the 62 cores (11%) in the LDW provided no data on recovery rates, had low concentrations such that trends could not be determined, or indicated disruption to recovery. Chemical trends in most cores and at most resampled surface sediment stations show reductions in risk-driver concentrations over time. Both of these findings demonstrate that recovery is occurring in much of the LDW (as discussed and presented below for total PCBs, cPAHs, and other SMS contaminants). In areas either where these lines of evidence are not similar to one another or to the STM outputs, or where recovery is not predicted by the BCM, more attention is given to ascertain the reasons for these differences (see Appendix F). In some small-scale areas, these lines of evidence suggest that recovery is not occurring, and these areas are incorporated into assignment of recovery categories (see Section 6).

5.4.1 Net Sedimentation Rates

Net sedimentation rates were estimated from 74 cores for which time markers could be identified (Table F-3; Figure 5-13). These markers provide evidence of new material being deposited in the LDW, showing that burial, the dominant recovery mechanism, is occurring. The time markers were used to calibrate the net sedimentation rates estimated by the STM. STM calibration is discussed in Appendix F of the STAR report (Windward and QEA 2008). This analysis is also discussed in Appendix F of this FS. In the RI (Windward 2010), the depth of the peak total PCB concentration in each core was used to support the sedimentation rates estimated from the STM, and this analysis is discussed below in Section 5.4.1.1. Some cores indicated either no recovery or reduced recovery. The causes for these discrepancies are unclear. In some cases, the cores may not have been deep enough to show the time markers, concentrations were too low to detect trends, surface concentrations were too high from ongoing sources, or the area may have been previously dredged or otherwise disturbed. Deep disturbance may remove freshly deposited cleaner sediments or mix surface and subsurface sediments, resulting in exposure of higher contaminant concentrations at the surface.

System-wide statistical analysis suggests that the STM tends to underpredict sedimentation when compared to empirical data, and thus underpredict natural recovery potential. However, many of these sedimentation-rate underpredictions occur in Reach 3, which has very high sedimentation rates; thus, it does not influence model recovery predictions because both model and empirical data indicate rapid recovery. In Reaches 1 and 2, with less overall sedimentation compared to Reach 3, net sedimentation is sometimes underpredicted and sometimes overpredicted by the model. Several cores in these reaches did not have time markers preserved in the core profile from which to estimate sedimentation and recovery. Reaches 1 and 2 generally have lower empirically-derived net sedimentation rates compared to model predictions, as well as several cores that did not exhibit discernible recovery, and therefore the STM may somewhat over-predict recovery in these reaches. The base-case best-estimate STM predictions should be confirmed in localized areas during remedial design where MNR is being considered.

5.4.1.1 Vertical PCB Concentration Trends Compared to Net Sedimentation Rates

The PCB "peak" analysis presented in the RI (Windward 2010) combined information on depth patterns in PCB sediment chemistry (from sediment cores) with net sedimentation and erosion estimates from the STM to determine whether vertical patterns of total PCB concentrations are consistent with the STM's estimated net sedimentation rates and the CSM (Figure 5-14). Much of the sediment contamination in the LDW, and particularly PCB contamination, is believed to have originated from historical sources in the LDW.[22] In undisturbed depositional areas with no ongoing or recent sources, PCB concentrations should be higher in deeper core intervals than in shallower intervals. In areas with little or no deposition, localized disturbances, or ongoing or recent secondary sources (e.g., erosion of contaminated upland soil), this pattern may be altered, with higher PCB concentrations in the shallowest core intervals or relatively even distribution among core intervals.

Assuming that an area is depositional and has not been disturbed, the depth of the maximum total PCB concentration within a core should be a function of both the time since peak PCB use and release and the estimated rate of net sedimentation (from the STM). As a result, the expected depth of peak (or maximum) total PCB concentration was estimated for each core using Equation 5-2.

$$D = (T_c - T_m) \times S \qquad \textbf{Equation 5-2}$$

Where:

- D = expected depth of peak total PCB concentration (cm)
- T_c = year of core collection
- T_m = assumed year of maximum concentration in surface sediment, corresponding to the assumed peak in PCB use and releases to the LDW

22 Peak PCB use was recorded in Puget Sound sediment cores between 1960 and 1970 (Van Metre and Mahler 2005; Battelle 1997); the commercial production of PCBs was banned in 1978, and they were subsequently phased out. Although PCBs historically used in paints, caulking, and other products continue to be released into the LDW, it is believed these ongoing sources represent a smaller contribution to the LDW than historical releases.

- S = net sedimentation rate (cm/yr) estimated from the STM for the grid cell containing the core (or the closest grid cell for cores outside the STM domain).

General uncertainties associated with estimating the depth of the peak total PCB concentration include uncertainties in the net sedimentation rate estimated by the STM and uncertainty in the estimate of the year of the peak release of PCBs. In addition, uncertainty is associated with identifying the exact depth of the peak total PCB concentration within a core because of compositing within each core section. Uncertainty is particularly high at locations where the core intervals analyzed were 3 feet (ft) or greater and is lowest at locations where the core was sectioned into 0.5-ft intervals. Location-specific uncertainties include the possibility of sediment disturbance near berthing areas or local structures, and the potential for localized PCB releases to continue after the peak use/release date. To address the uncertainty in the year of maximum historical PCB releases to the LDW, a range of estimated depths of the peak total PCB concentration was calculated for each core (i.e., estimated depths within each core were calculated by assuming maximum PCB releases in 1960, 1965, and 1974).[23] These depth estimates were then compared to the depth of the peak total PCB concentration in each core. If the observed depth of the peak total PCB concentration was at or deeper than the estimated depth, the core was considered to be consistent with the CSM, and with the STM's estimated net sedimentation rates.

Of the 366 cores available in the RI dataset, 157 cores were used in the analysis and 209 cores were not used because the type of information needed for the analysis was not available for those cores. Cores were excluded if at least one of the following conditions were met:

- Only one core interval was analyzed for total PCBs
- No core interval was analyzed within the depth range of the expected peak
- PCBs were not detected in any core interval
- The sediment was disrupted by dredging prior to sampling.

Of the 157 cores included in the analysis, 110 cores (70%) had peak total PCB concentrations at depths equal to or greater than the estimated depths, consistent with the STM's estimated net sedimentation rates. Forty-seven cores (30%) had maximum total PCB concentrations that were shallower than the estimated depth range based on net sedimentation rates from the STM, or the concentrations were too diffuse to detect a significant peak at depth. For recovery estimates, the LDW model and field data are divided into three reaches. Reach 3 (the upper LDW) includes high rates of

23 The analysis used both nationwide trends for PCB peak release (1960 and 1965; Van Metre and Mahler 2004; Battelle 1997), and the year of a PCB spill in Slip 1 (1974; Blazevich et al. 1977).

sedimentation and most maintenance dredging occurs in this reach. None of the cores in Reach 3 had maximum PCB concentrations at depths that were less than model predictions. Reach 2 includes both areas of high sedimentation and areas where no sedimentation was evident (net scour). Of the cores in this reach, 35% had maximum PCB concentrations at depths that were less than model predictions and 2% showed no discernible trend. Reach 1, which is near the mouth of the LDW, has lower sedimentation rates compared to Reach 3. Of the cores in this reach, 25% had maximum PCB concentrations at depths that were less than model predictions and 5% showed no recovery.

5.4.2 Chemical Trends at Resampled Surface Sediment Locations

Generally, chemical trends in resampled surface sediment locations show that recovery is occurring over much of the LDW, which supports the BCM findings of decreasing contaminant concentrations over time. Resampled surface sediment locations are surface sediment samples collected at different times from the same station (within 10 ft of one another). The contaminant concentrations in the LDW surface sediments have heterogeneous, but restricting the distance between older and newer locations to 10 ft reduces the uncertainty introduced by comparing samples from different locations. Appendix F describes the details, statistical results, and limitations associated with this type of comparison (analytical accuracy, etc.).

In the FS dataset, the data from 70 resampled stations (67 locations with 3 outliers excluded, and excluding those collected at the Norfolk Area and Duwamish/Diagonal EAAs) were grouped into two populations: older/original data and newer (FS baseline) data (see Table 5-7). The statistical difference between total PCB concentrations in these two groups was evaluated to provide evidence of general LDW-wide trends using simple data distributions. The comparisons of total PCB concentrations between the older and newer data show a 62% decrease in the mean value. As shown in Table 5-7, the 25th and 90th percentiles of these datasets also decreased by 31% and 64%, respectively, revealing that, in general, the empirical data support the STM findings that the LDW is recovering (at least for PCBs). Table 5-7 also summarizes these trends for arsenic, cPAHs, and BEHP. These data demonstrate that, on average, total PCBs, cPAHs, and BEHP concentrations are decreasing over time (more than or equal to a 50% reduction in concentration) while arsenic is in equilibrium (see Appendix F) and relatively close to urban background levels (see Appendix J).[24] For total PCBs and cPAHs, the mean for the older dataset is more than 20 times higher than their mid-range BCM upstream input values (Table 5-1a). For arsenic, the mean of the older dataset is only 4 times higher than the mid-range BCM upstream input value. This means that new sediment from upstream will have a greater effect on reducing concentrations of total PCBs and cPAHs over time than on reducing concentrations of

[24] The arsenic data have a narrower range of concentrations in the LDW than the other risk drivers, and are more similar to background conditions.

arsenic. Station-by-station results are presented in Appendix F for total PCBs, arsenic, cPAHs, BEHP, and SMS contaminants with detected exceedances in either the newer or older data.

5.5 Uncertainties Related to Predictive Modeling

The goal of an uncertainty analysis is to both qualitatively and quantitatively define the degree of confidence in site characterization data, both conceptual and predictive site models, and predictions of the results of remedial actions to the degree possible.[25] Bounding the certainty of estimates, especially in modeling, is a developing science. In accordance with an EPA guidance document (EPA 2005b), the potential areas of uncertainty to be identified and addressed in an FS include the CSM, data uncertainty, temporal uncertainty, spatial variability, and quantitative uncertainty. Several elements of uncertainty related to the predictive models (STM and BCM) are described below.

5.5.1 Net Sedimentation Uncertainty

Extensive sensitivity analyses were conducted on the STM and are described in detail in the STM report (QEA 2008). Sensitivity analyses were conducted on both high-flow event simulations and long-term, net sedimentation simulations. The net sedimentation sensitivity analysis showed that the model was most sensitive to the upstream sediment load and the settling speed of the fine-grain sediment classes, which make up the majority of the incoming sediment load. In this FS, because two, site-wide, independent datasets were not available for net sedimentation, uncertainty and sensitivity analyses both utilize the same input parameters. An appropriate measure for uncertainty in model predictions and application in this FS is the spatial scale analysis (QEA 2008; see Figure 2-13 from the STM Report). This analysis examined the accuracy of the model with respect to estimating net sedimentation rates from the large scale (LDW-wide) to the small scale (location-specific areas). This analysis found that the capability of the model was not affected by spatial scale (minimal bias), and that, on average, the model is able to estimate net sedimentation rates to within ±0.5 cm/yr on a typical net sedimentation rate of 1 cm/yr.

The incoming sediment load and depth of scour are affected by high-flow events. The STM used Green/Duwamish River flows from 1960 to 1989 as input flows. The maximum flow rate and upstream sediment loading for these years are shown on Figures 5-15a and 5-15b. The figures indicate that the upstream sediment load was

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[25] Sensitivity analysis differs from uncertainty analysis in terms of goals and inputs. A sensitivity analysis looks at how the model responds to a range of input values, which may be extreme or not realistic, but are designed to stress the model and produce changes from the calibrated model results. Uncertainty analysis addresses the model's resolution, that is, its ability to replicate natural processes in light of unaccounted processes. Uncertainty analyses should be based on realistic and statistically defensible methods for developing a reasonable set of input parameters and conditions, which are then used to demonstrate a range in model results in order to inform decision-makers of potential model errors.

below average for the first 10 years of the simulation. Consequently, the STM and BCM may be conservatively predicting net sedimentation through the first 10-year modeling period.

The flow period represented in the STM (1960 to 1989) and shown on Figures 5-15a and 5-15b is representative of current conditions. Annual precipitation since 1989 and up to the present has not changed significantly. Global warming is also not expected to change average annual precipitation significantly (Mote and Salathe 2009). By the late 1990s, when the U.S. Geological Survey (USGS) sediment loading study was conducted, the Green/Duwamish River basin was already under control by the Howard Hanson Dam and heavily developed with agricultural, urban, and suburban land uses. For these reasons, Green/Duwamish River flows and sediment loads are not expected to change substantially in the future as long as the river flow continues to be dam controlled in a manner generally consistent with historical water management practices.

5.5.2 STM Uncertainty – Lower and Upper Bound Simulations

The effects of uncertainty in STM inputs on model estimates were analyzed and quantified in the STM report (QEA 2008; see Section 2.8 and Appendix D.6 of the STM). The results of the input parameter sensitivity analysis were used to generate reasonable lower- and upper-bound limits on the base-case results, which are based on the calibration parameter set. The upper- and lower-bound cases were a result of changing the upstream sediment loading and settling speed of Class 1A and 1B solids. The base-case upstream loading rates were developed from two USGS studies to provide a good estimate of the magnitude of Green/Duwamish River input to the LDW, and the Class 1A and Class 1B settling rates were selected during the STM calibration process because they were reasonable and because they best match the empirically-derived LDW net sedimentation rates. Therefore, the values for these two model input parameters in the STM base case were reliably defined by site-specific data and model calibration.

The base-case simulations provide the best estimates of net sedimentation rate, but the reasonable lower- and upper-bound simulations provide an acceptable range of net sedimentation rates resulting from uncertainty in model inputs, with the "true" value of net sedimentation rate being within this range. As noted in Section 5.4.1, field sedimentation data are sparse and variable by reach and location, and the STM predictions will need to be confirmed for areas where MNR is proposed during remedial design. The highest empirically-derived net sedimentation rates occur in Reach 3 and were higher than model predictions; therefore, the STM may under-predict recovery there. Reaches 1 and 2 generally have lower empirically-derived net sedimentation rates compared to model predictions, as well as several cores that did not exhibit discernible recovery, and therefore the STM may somewhat over-predict recovery in these reaches.

To demonstrate the effect of model parameters on long-term changes in bed composition, the upper- and lower-bound results have been analyzed and used to estimate uncertainty in the predicted half-time of bed-source content[26] in surface-layer (0 to 10 cm) sediment for the long-term, multi-year (e.g., 21-year calibration period) simulations. Half-time values of bed-source content in surface-layer sediment were estimated using relationships between net sedimentation rates and half-time values developed from model results presented in the STM report (QEA 2008). The approximate relationship between half-time of bed-source content and net sedimentation rate can be used to estimate the spatial distributions of half-time and recovery potential if the starting concentrations are known.

Generally, the half-time of bed-source content in surface-layer sediment tends to decrease as the net sedimentation rate increases, see Section F.2 and Figure F-37 of the STM report (QEA 2008). In general, most areas have a half-time of less than 10 years based on net sedimentation rates of 1.0 cm/yr or more. This analysis indicated a general trend of decreasing half-life of bed-source content with an increasing net sedimentation rate. Spatial distributions of the net sedimentation rate for the lower- and upper-bound simulations are shown in figures in Appendix C, Part 6. The best-fit model prediction from the bounding exercise is about 5 to 10 years (±5 years if the net sedimentation rate is more than 1 cm/yr and longer with lower net sedimentation rates). Because the bounding exercise does not represent the calibrated dataset, this characterization of uncertainty is more appropriate for those regions farther from the locations where the model was calibrated. Areas near calibrated locations have significantly lower levels of uncertainty. This level of uncertainty is acceptable for the FS. The uncertainty in the reasonable lower- and upper-bound STM runs and its effect on PCB concentrations are discussed in Section 5.5.4.

5.5.3 Uncertainty around the BCM Contaminant Input Values

For the BCM, uncertainty exists in the assumptions about contaminant concentrations in lateral and upstream sources (from both non-point and point sources). This uncertainty will exist well into the future based on the variable nature of these sources, but is managed by expressing BCM inputs as a range of concentrations (low, high, and best-estimate values). These input values are based on actual data collected over the past 20 years. BCM uncertainty is managed by bracketing the best-estimate BCM value with lower- and upper-bound BCM input values representing the mean, UCL95, or percentiles of the existing data. For the lateral inputs, the low and high estimates are meant to capture a range of uncertainty associated with potential future source control measures.

[26] The half-time is defined as the time needed for 50% of material in the initial surface layer (0 to 10 cm) of the sediment bed to be replaced with depositing sediments.

These input values were estimated from summary statistics for various datasets (surface water, surface sediment, in-line sediments, catch basin solids, etc.). Each dataset has some degree of sample uncertainty associated with it, relating to aspects such as the matrix from which the sample was collected, the location from which the sample was collected, the differences in TOC and grain size among the datasets, the time (season, river flow, portion of storm event [e.g., first flush]) of sample collection, ongoing source control efforts, and other aspects that can affect contaminant concentrations in a sample. The high end of the range (high lateral, high Green/ Duwamish River, and high post-remedy bed sediment replacement values) is intended to capture variability in the source concentrations, worst-case recontamination potential, and regular, seasonal high flows from urbanized areas. The low end of the range (low lateral values, low Green/ Duwamish River, and low post-remedy bed sediment replacement values) represents a non-conservative set of assumptions that is considered likely to underestimate future contaminant concentrations. The probability that site conditions will produce a high-high-high contaminant concentration (lateral, Green/Duwamish, bed) is likely very small.[27] A similar low probability of occurrence exists for the low-low-low end of the range.

Another source of uncertainty related to lateral inputs is the fact that lateral contributions to the LDW can come from many different sources, including storm drains, CSOs, surface water runoff, and atmospheric deposition anywhere along the LDW and in its drainage basin. These sources were aggregated into 11 waterfront areas and 16 discharge points to the LDW for the purposes of sediment transport modeling. Of these, only the CSOs have measured discharge flows; runoff flows are estimated for other discharges. Some localized discharge points may not be adequately characterized by the 11 general waterfront areas. In addition, CSO control plans will result in reduced flows in the future for many CSOs.

Similar uncertainty exists for the post-remedy bed sediment replacement values used as input in the BCM. These values represent the bed sediment contaminant concentrations in the near-term (0 to 2 years) following completion of active remediation, including influence from multiple recontamination mechanisms. Evidence from other sediment sites shows that post-construction COC concentrations become higher than detection limits and natural background after this initial time frame. Limitations in the

27 The likelihood of occurrence for the high-high-high contaminant concentration (lateral, Green/Duwamish, bed) is the product of the likelihood of each occurring independently. The likelihood of the upper bound representing the contaminant concentration for either upper, lateral, or bed source material is small. Therefore, the likelihood of all three upper bounds occurring is much smaller. It should be noted that a contaminant concentration value for any of these three variables that is higher than the medium, but less than the upper bound is not small. One can expect that the probability of occurrence of any combination is highest for medium-medium-medium and decreases moving toward either upper-upper-upper or lower-lower-lower combinations. The shape of this distribution is unknown.

dredging/capping equipment leave behind dredging residuals that resettle within the remedial footprint. Residual COC concentrations are typically proportional to the average COC concentration of the dredged material, and typically higher than the COC concentration in surrounding sediments (see Section 9 for a discussion on dredging residuals for each alternative). Post-construction surface sediments in the LDW may come into equilibrium with the sediments surrounding the remediated area. The equilibrium concentration of COCs in the sediment bed may be higher than the COC concentration in upstream sediments because of increased urbanization as one moves downstream toward downtown Seattle (more cars, vessel traffic, non-point sources, air emissions, accidental spills, and storm drain runoff). To address this uncertainty, the best-estimate for the post-remedy bed sediment replacement value is bracketed by low and high BCM input values that are a combination of clean backfill material (based on natural background concentrations) and the surrounding unremediated sediments, assuming various proportioning percentages, as described in Section 5.2.3.4. In addition, the effect of the post-remedy bed sediment replacement values on predicted total PCB concentrations for selected alternatives is presented in Appendix M.

By using many lines of evidence and a range of input values derived from these data, some quantitative analysis of the uncertainty is provided, and confidence in the model representing long-term conditions over time is increased. However, it is also uncertain how these input concentrations may change over time. In summary, these BCM input values are considered adequate for the purposes of assembling remedial alternatives (Section 8) and evaluating the short- and long-term effectiveness of the alternatives (Section 9) in the FS.

5.5.4 Combined STM and BCM Uncertainty

Both the STM and BCM have uncertainty associated with model input values, process descriptions, and discretization. Uncertainty in STM predictions that results from uncertainty in the input parameters was extensively examined in the STM report (QEA 2008). The uncertainty analysis in the STM report was used to develop reasonable and maximum upper and lower bounding simulations. The reasonable upper- and lower-bound simulations provide a realistic range of net sedimentation rates for the LDW and were used to examine the effect of STM uncertainty on BCM results. The maximum simulations were considered unrealistic and not carried forward in the BCM uncertainty analysis. The results from these bounding simulations are discussed in Section 5.5.2 and in Appendix C, Part 6. Uncertainty in the BCM chemistry input values is discussed in Section 5.5.3.

The STM base-case composition results were taken at the end of the 10-year model run for reasonable upper and lower bounding simulations as input to the BCM to compute the total PCB SWAC for each simulation following remediation of the EAAs. This analysis is presented in Appendix C, Part 6. The STM bounding simulations are presented in Section 2.8 of the STM report (QEA 2008). Reasonable upper and lower bounds are defined as net sedimentation rates that varied by ± 1 cm/yr from the STM base case. This provides a greater than 95 percent confidence interval around the data.[28] The reasonable lower to upper STM simulations produced a range in total PCB SWACs from 65 to 101 µg/kg dw or about -16% and +31% from the base case prediction, respectively (see Appendix C, Part 6, Table 5). However, the STM base case (with lower to upper BCM input values) produced a range in total PCB SWACs from 49 to 122 µg/kg or about -36% and +58% from the base case prediction, respectively. The analysis showed the total PCB SWAC is more sensitive to the range of BCM chemistry input values than it is to the range of net sedimentation rates from the reasonable upper and lower bounding STM simulations. Although the SWAC range based on BCM bounding is greater than the range based on STM bounding, both are still sufficiently large that they must be accounted for in future assessments. The range of total PCB SWAC values attributable to STM and BCM uncertainty is illustrated in Appendix C, Part 6, Figure 11.

5.5.5 BCM Input Values for Other SMS Contaminants

A total of 41 COCs with SMS criteria were identified for the protection of benthic invertebrates. It was not practical to run the BCM 41 times to evaluate recovery potential for every SMS contaminant. Therefore, a smaller subset of representative contaminants was selected because:

- Many co-occur with other SMS contaminants (e.g., PAHs)
- Groups of contaminants have similar modes of toxicity (e.g., phthalates)
- Lateral source data have not been collected, or at least compiled, for every contaminant
- Many of these COCs do not have widespread SQS exceedances in the LDW.

Application of the BCM using representative SMS contaminants is based on the fact that the representative contaminants account for the majority of the SQS exceedances and the assumption that all SMS contaminants within a group will behave/recover in a similar manner. Uncertainty exists with this simplifying assumption. In reality, each SMS contaminant may have a different starting concentration, recovery and/or recontamination potential, sediment-water partitioning dynamics, bioavailability based on organic carbon content, and lateral and upstream sources. Estimated exceedances of the SQS and CSL at the end of the 10-year modeling period may be biased high or low relative to the representative SMS contaminant predictions. This uncertainty will be managed during remedial design and by refinement of the CSM for remedial areas.

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[28] The confidence interval for the reasonable upper and lower bounds was not specifically defined in the STM analysis. However, the 95 percent confidence interval was defined as a net sedimentation rate of ± 0.5 cm/yr.

5.5.6 Age and Spatial Extent of Contaminant Data

Over the past 18 years, numerous investigations have been conducted to determine the nature and extent of sediment contamination associated with past and present contaminant releases at various locations within the LDW. These investigations have included in-water investigations involving surface and subsurface sediment sampling, toxicity testing, shoreline habitat inventories, seep surveys, and porewater sampling. These data have been aggregated into the FS baseline dataset. There is uncertainty associated with these data related to detection limits that exceed the screening criteria, especially in older data; contaminant compositing with depth; and interpolation between sampling points. An additional large source of uncertainty is the age of the data. Many of the surface sediment data comprising the FS baseline dataset are over 10 years old and do not represent current conditions. Active remedial technologies are being assigned to particular areas based on surface sediment exceedances that may have improved (or worsened) over the past few years. Because the CSM and empirical data have shown that the LDW is recovering (in many areas), there is likely a high bias introduced into the assembly of alternatives. Remedial alternatives are being assembled on fairly conservative assumptions that no recovery has occurred between when the data were collected and now. This source of uncertainty is being managed in two ways: 1) the modeling is conservative and does not account for 10 years or more of potential recovery from when the sample was collected; and 2) areas with older data, but which are predicted to recover, will be subject to verification monitoring (see Sections 6 and 8) to confirm current contaminant concentrations and degree of recovery. Other sources of data uncertainty such as vertical and horizontal extent of contamination, elevated detection limits, and SMS compliance may also be refined during remedial design.

5.5.7 Chemical Degradation and Transport Processes

Many of the LDW risk drivers (total PCBs, cPAHs, BEHP, arsenic, dioxins/furans, and other SMS contaminants) have similar fate and transport properties in that they are strongly bound to sediment particles and do not readily degrade. Compounds that readily degrade or desorb from sediments are not persistent in sediments because the concentration declines naturally over time. Persistent contaminants cause long-term sediment contamination. The following discussion focuses on PCBs because a large body of research exists for this COC at many sites across the country. However, for most of the COCs, degradation and desorption processes decrease the concentrations in sediment over time. By not accounting for these processes, the analysis is conservative with respect to sediment contamination and natural recovery because it will overestimate both long-term sediment concentrations and the time required for natural recovery to occur.

PCBs, in particular, are stable compounds that do not degrade easily. Under certain conditions, they may be broken down by chemical, thermal, and biological processes (Erickson 1986). In the environment, photolysis (breakdown by light) is the only significant chemical degradation process, but it is not likely a significant means of PCB losses from sediments because of low PCB solubility and limited penetration of sunlight into the solid media (the sediment bed) (Hutzinger et al. 1974). Microbial processes are the main route of environmental degradation of PCBs in sediments. Reductions in the sediment concentrations of PCBs can happen via desorption from sediments into the overlying water column and volatilization. The breakdown of PCBs is generally discussed below, and implied for many other risk drivers; it is assumed to be occurring in the LDW, although these processes have not been modeled in the FS. Changes in PCB concentrations in the sediment bed can be translated into predicted concentrations of PCBs in fish and shellfish tissue via the PCB food web model (FWM) developed for the LDW (Appendix D of the RI, Windward 2010; see Section 9.3.5.2 of this FS for a discussion of uncertainties associated with FWM estimates). Section 3 evaluated whether varying water concentrations account for the effects of desorption and how other inputs into the water column would affect tissue concentrations (see Figure 3-2).

The King County model was used to predict contaminant concentrations in the water column; it employed containment flux from the sediment bed to estimate desorption of PCBs into the water column.

The effects of varying PCB concentrations in the water column and the site-wide sediment SWACs on predicted residual risks from seafood consumption are discussed in Section 9; results are presented in Appendix M.

5.5.7.1 Microbial Degradation

The viability of biodegradation as a natural method of sediment recovery for sediment-bound PCBs has been documented in several studies (RETEC 2002; Appendix F).

PCBs can undergo microbial degradation in natural environments under both aerobic (i.e., in the presence of oxygen) and anaerobic (i.e., in the absence of oxygen) conditions. PCBs are a class of 209 individual contaminants (PCB congeners), in which 1 to 10 chlorine atoms are attached to a biphenyl molecule. Most Aroclors (commercially produced groups of PCBs) contain 60 to 90 different PCB congeners, with varying numbers and positions of the chlorine atoms on the biphenyl rings.

Microbes degrade PCBs by breaking the carbon-to-carbon bond of PCBs, or by substituting the chlorine atoms with hydrogen atoms in the PCB molecule under aerobic and anaerobic conditions, respectively (McLaughlin 1994). The latter method results in the transformation of PCB congeners into other less chlorinated PCB congeners in a process called dechlorination (Abramowicz 1990). Aerobic degradation, on the other hand, results in a net PCB loss from a given PCB inventory. In river sediments, aerobic conditions are typically found in the top few centimeters of the sediment bed, while anaerobic conditions are found at greater depths below the sediment surface.

Aerobic Degradation

Even though laboratory studies have documented the existence of naturally occurring aerobic bacteria capable of degrading a large spectrum of PCB congeners, there is little direct evidence indicating that the aerobic degradation process is effective at reducing the PCB mass under field conditions. In laboratory studies of the Hudson River, PCB losses were highest in the less chlorinated congeners (43 to 47% reduction) and lowest in the more chlorinated congeners (17 to 5% reduction) (Harkness et al. 1993 and 1994). The in-field studies yielded similar results (less than 50% reduction). A study of PCB patterns in Green Bay sediments suggests that aerobic degradation is not a significant transformation mechanism for those sediments (McLaughlin 1994).

Anaerobic Dechlorination

Reduction through dechlorination (under anaerobic conditions) is generally viewed as a viable means of biodegradation for numerous compounds, including PCBs at higher concentrations. This process can alter the toxicity of these compounds and make them more readily degradable. The extent to which PCBs can degrade depends on several factors (Bedard and Quensen 1995), including the nature of the active microbial population, the type of chlorine substitution, the chlorine configuration, the initial PCB concentration, and the substrate conditions (temperature, redox conditions, ionic strength, amount of carbon, and presence of other oily contaminants, etc.). For example, no anaerobic dechlorination of PCBs was observed in the downstream deposits of the Fox River where the maximum PCB concentration was approximately 30 mg/kg dw (limited effectiveness at lower concentrations). Dechlorination activity was limited to sediment PCB concentrations of 30 mg/kg dw or greater (McLaughlin 1994). The overall PCB loss due to microbial degradation in several Fox River sediment deposits was estimated to be less than 10% with respect to the original inventory of PCBs deposited in the river.

A similar threshold for degradation of 50 mg/kg dw was observed in Sheboygan River sediments (David 1990). For Grasse River sediments (Minkley et al. 1999a, 1999b), some dechlorination activity was suggested at total PCB concentrations below 7 to 10 mg/kg dw, but the statistical evidence of dechlorination was less strong than at higher concentrations. Attempts in a laboratory study to further dechlorinate Fox River sediments met with limited success and similar results, up to 10% dechlorination on a total chlorine basis (Hollifield et al. 1995).

In the Fox River, physical loss through desorption from sediments (into the water column) exceeded any biodegradation in the sediment. It was estimated that 33% of the original PCB mass originally deposited in the Lower Fox River was lost due to desorption.

5.5.7.2 Volatilization and Desorption

Volatilization and desorption remove contaminants from sediment particles without changing the chemical make-up of the contaminant. In desorption, the contaminant is removed from the sediment and becomes dissolved in water. Volatilization is the process of a contaminant going into the gaseous state and being released to the atmosphere.

Both of these processes are relatively weak for the COCs in the LDW. For instance, all of the inorganic compounds (with the exception of mercury) and low molecular weight PCBs generally do not undergo volatilization. For PCBs, volatilization into the air can be important in shallow arable soils, but less so for subsurface soils (Meijer et al. 2003). Limited volatilization of some organics could occur from exposed intertidal sediments at low tides, but this transport mechanism would be further limited by the high water content of the sediments. COCs may diffuse from sediment into porewater and then into the water column and/or atmosphere, but these transport pathways occur at very slow rates. Because subtidal sediments are covered with water and are not in contact with the atmosphere, a very limited amount of volatilization occurs from dissolved PCBs in the water column, rather than directly from sediments. Consequently, volatilization is not considered a major process in the dynamics of PCBs or other COCs in LDW sediment.

Desorption is related to how strongly a contaminant binds to sediment or to organic carbon in sediment. All of the COCs in the LDW strongly bind to sediment. If the COCs did not bind strongly to sediments, they would have desorbed, become dissolved in surface water, and have been discharged downstream, effectively removing them from LDW sediments. Empirical evidence demonstrates the persistence of these contaminants with depth in the LDW. Many of the organic compounds, such as PCBs, PAHs, and dioxins/ furans, are referred to as hydrophobic compounds. That is, the compounds preferentially partition to solids rather than become dissolved in water.

By not including volatilization and desorption in the natural recovery analysis, estimated future contaminant concentrations in sediment are conservative because these processes should slightly accelerate the predicted natural recovery in surface sediments.

5.5.8 High-Flow Scour Potential

As discussed in Sections 5.3.1 and 5.3.2.5, the maximum scour depth during a 100-year high-flow event is estimated by the STM to be about 22 cm for the base case, and the upper bound of estimated scour is 36 cm, based on upper-bound erosion sensitivity simulations. Areas with subsurface sediment contamination located in potential scour areas, whether from high-flow events or propeller scour, are explored in Section 6 and are included in the AOPC footprints. Scour areas defined in Section 2.3.1.1 and illustrated in Figures 2-9 and 2-10 were used to assign recovery categories in Section 6.

Section 5.3.2.5 illustrates that potential exposure and transport of subsurface sediments during high-flow events is small compared to the incoming sediment loads. To explore the net effect of propeller scour events, Appendix F illustrates that empirical chemical trends from many of the resampled surface sediment stations and sediment cores have decreasing contaminant trends (or trends in equilibrium) in scour areas. The FS assumes that scour potential (less than 10 cm) in areas with subsurface exceedances of SMS criteria is of concern even if empirical evidence indicates that some recovery and scour areas with adequate net sedimentation rates and water depth may eventually recover. Uncertainty related to scour potential with subsurface exceedances is inherently accounted for in Section 6. Areas with subsurface exceedances in potential scour areas are included in the AOPC footprints for the FS, and these areas are given equal consideration as surface exceedances in the assembly of alternatives and assignment of remedial technologies to those areas (Section 8). Active remediation is assigned to scour areas (within the depth of scour potential, typically RAL exceedances in the upper 2 ft) in the absence of empirical trends showing recovery.

A sensitivity analysis was conducted to evaluate uncertainty in STM predictions that may have resulted from uncertainty in model input parameters, including those that control erosion rates. Uncertainty in the extent of areas estimated to have erosion was less than ±50% within the area from RM 0.0 to 4.3, relative to the base-case simulation. Uncertainty in predicted sediment mass eroded ranged from about -50 to +75% within the area from RM 0.0 to 4.3 as well as in the east bench and navigation channel, and ranged from -40 to +130% in the west bench. The analysis showed that the predicted depth of scour, area of scour, and mass of sediment scoured are not very sensitive to erosion rate parameters used in the model.

5.5.9 Anthropogenic and Natural Deep Disturbance Uncertainty

Section 5.3.1.3 introduces the potential for both anthropogenic and natural disturbance of subsurface sediments in the LDW that may result in contaminant exposure. These subsurface sediments are an additional potential source of contaminant mass to LDW surface sediments, similar to upstream and lateral loadings. The RI did not extensively characterize subsurface contaminant concentrations. In addition, deep disturbance is inferred in some geochronologic and chemical records. However, these data are sparse relative to the size of the study area and the frequency, cause, and magnitude of deep disturbances cannot be estimated with confidence. The data can, however, provide general, first-order estimates of bounds on reasonable minimum and maximum acreages of continuous disturbance (0 to 45 acres). These acreage bounds are used to bound the possible effects on the predicted total PCB SWAC. This analysis is provided in Appendix M, Part 5, and results are discussed in Sections 9 and 10.

The approach used for this analysis is based on some assumptions that will overestimate the predicted SWAC with time. Specifically: 1) the same area is assumed to be repeatedly disturbed (e.g., perhaps a tug regularly has trouble maneuvering a barge into a particular spot); 2) there is no mixing of ongoing sedimentation with deeper sediment during a deep disturbance event; and 3) the subsurface concentrations never change. These conditions were not factored into the analysis and would mitigate some of the increases in SWAC predicted in the analysis. In addition to change in the SWAC, ongoing deep disturbances could result in longer recovery times being required to achieve the cleanup objectives.

5.5.10 Bathymetric Changes and Dredging of Upper Turning Basin Sediments

A hydrodynamic model was used to generate flow velocities, which were then used in the STM. The hydrodynamic model was not revised for changes in bathymetry due to scour or net sedimentation. However, the STM does track the changes in bed elevation over time as sediment is scoured or deposited. Analysis of specific model cells in the navigation channel and on the benches shows that the change in bed elevation in the first 10 years of the simulation is on the order of 10 cm (4 inches). This change in bathymetry would not be expected to affect the hydrodynamic model because the water depth is much greater than the change in bed elevation.

In Reach 3, the Upper Turning Basin has much more net sedimentation than Reaches 1 and 2. However, the Upper Turning Basin is regularly dredged. By ignoring the changes in bathymetry due to deposition in the Upper Turning Basin, the model essentially assumes that the Upper Turning Basin is continually dredged. If the hydrodynamic model and STM were modified for bathymetric changes between dredging events, the Upper Turning Basin would become shallower and more sediment would move downstream, resulting in higher net sedimentation rates downstream of the Upper Turning Basin. However, the hydrodynamic model does not consider the hypothetical possibility of a cessation of dredging at the Upper Turning Basin, and therefore retains the present mass inputs and grain-size distribution into the future.

5.6 Modeling Summary and Conclusions

In summary, predictive modeling is a useful tool for the FS to evaluate the value or effectiveness of remedial alternatives and the recovery potential of the system. Some alternatives will include MNR and others will not (see Section 8). The STM and BCM support decision-making regardless of which remedial alternative is selected. However, it is understood that both tools have a large degree of uncertainty (see discussion following bullets). For the purposes of the FS, a bounded margin of uncertainty is acceptable, but this FS assumes that this uncertainty can be further managed during remedial design and future monitoring. The modeling presented in this section concluded that:

- The LDW is net depositional over time and its physical characteristics and natural processes are reasonably well understood through fine-scale hydrodynamic and sediment transport modeling. The STM output has been supported by several lines of evidence, including chemistry profiles in sediment. Areas where the STM output doesn't match empirical data are generally found in locations with features and activities that the STM didn't incorporate (e.g., bridges and pilings, high-powered ship maneuvering, and other berthing activities).Three key outputs from the STM are used in the FS: net sedimentation rates, areas subject to scour from high-flow events, and bed composition. The third output provides the framework for predictive contaminant modeling in the BCM.

- Sediment is continually depositing within the LDW. Almost all new sediment (99%) that enters the LDW originates in the Green/Duwamish River system. The STM estimates that, on average, over 185,000 MT of sediment per year enters the LDW, with approximately 100,000 MT depositing in the LDW. Approximately 90% of the total bed area in the LDW receives 10 cm of new sediment within 10 years or less. This sediment is mixed with the existing bed sediment through various processes, including bioturbation and propeller wash. On average, the annual volume dredged over the past 15 years is approximately 51% of the deposited sediment load. An annual average of approximately 38,000 MT has been dredged within the authorized navigation channel and 13,000 MT within the berthing areas, for a total annual dredge volume of about 51,000 MT.

- Overall, the maximum net erosion depth during a 100-year high-flow event is approximately 22 cm, with most areas experiencing less than 10 cm of scour, while 82% of the LDW experiences net deposition rather than net erosion over the 30-year model period.

- The effects of propeller-induced bed scour are incorporated into the present structure of the LDW sediment bed because ship movement has been occurring for at least the past 40 years. Propeller-induced bed scour from transiting ships and typical berthing activities is viewed as an impulsive erosion-deposition process that tends to behave like an ongoing mixing process for surficial bed sediment. Transiting ships in the navigation channel are not a major source of sediment transport or erosion in the LDW, except where slightly greater erosion depths (net erosion) are possible in shallower areas adjacent to the navigation channel. However, the analysis of scour prepared for this FS does not consider some possible irregular events. These events, outside of normal operating procedures, may include emergency and high-power maneuvering of tug boats under unexpected conditions, high-powered navigation activities, ships running aground, seismic events, and disturbance resulting from riverine structure maintenance construction/repair. Such events are likely infrequent relative to ships transiting the LDW, but could result in deep disturbances that affect long-term SWACs and hinder natural recovery. These events can disturb subsurface sediments and mix subsurface contamination with the surface layer. A series of post-STM/BCM analyses were performed to address the potential importance of both routine navigation activity and episodic, high-powered navigation and maintenance construction/repair events on long-term SWACs (Appendix M, Part 5). These analyses indicate that long-term recovery and SWACs could be influenced by navigation and riverine activities in the LDW, with the magnitude of the impact dependent upon the frequency and extent of the disturbance event.

- The BCM estimates changes in risk driver contaminant concentrations over time. Output from the BCM includes contaminant concentrations (point concentrations and area-based SWACs) at 5-year increments for 45 years.
- Empirical data show that, on average, LDW surface sediment contaminant concentrations are decreasing over time, consistent with BCM predictions of surface sediment concentrations approaching equilibrium over time. Appendix F shows specific locations where the empirical data demonstrate recovery. However, recovery can be locally hindered by vertical mixing of surface and subsurface sediments disturbed by anthropogenic and natural activities.
- Contaminant input values used in the BCM (lateral source, Green/ Duwamish River upstream, and post-remedy bed sediment replacement) were derived from actual input data (catch basin solids, sediment trap samples, upstream surface sediment and surface water data, USACE sediment cores) from the Upper Turning Basin. A range of values (high-medium-low) are used to address uncertainty and potential temporal variability in the range of contaminant inputs associated with each source type.
- Both the BCM predictions and empirical contaminant trends show that natural recovery is occurring in some areas of the LDW (see Appendix F). According to the BCM, MNR is a viable technology for many (but not all) areas of the LDW with moderate levels of contamination (below the CSL), net sedimentation rates of more than 1 cm/yr, and minimal scour potential (see Section 6).
- The BCM uses the FS baseline dataset (where the data are already more than 10 years old in some areas) and assumes no recovery or age-consideration for the older data in existing bed sediments; therefore, the initial bed contaminant concentrations at the start of construction may be lower than estimated in the BCM.

- The STM and BCM are not contaminant fate and transport models, and the numerous assumptions made throughout model development were designed to provide reasonable estimates with respect to predicted sediment concentrations based on available data for model development. Many assumptions used to develop model input and process descriptions are conservative. For example, the models assume no chemical transformation or degradation over time. Mass is not conserved in the BCM; however, additional analyses presented in Appendix C were used to investigate the significance of this on predictions of natural recovery. Changes in tissue and surface water COC concentrations are predicted as sediment concentrations change (i.e., through burial, scour, and resuspension processes). Changes in seafood consumption risks are evaluated for each remedial alternative in Section 9 via the PCB FWM developed as part of the RI (Windward 2010).
- The BCM may underestimate potential COC concentrations in localized areas near active discharges due to variation in loading estimates among the outfalls. These localized areas should be evaluated for adequate source control during remedial design.

Uncertainty in both the STM and BCM is recognized in sedimentation rates, erosion depths, scour areas, and contaminant inputs over time. Varying levels of confidence can be attached to these model predictions depending on: 1) the COC (i.e., arsenic has a higher level of certainty compared to PAHs, which may have increasing concentration trends from urbanization) and 2) the location in the LDW (areas with estimated net sedimentation greater than a few centimeters have a higher expectation that natural recovery will occur because the estimated net sedimentation is much greater than model error). By using many lines of evidence and a range of input values derived from these data, the uncertainty can be bounded. Overall, the uncertainty in BCM contaminant concentration input parameters has a slightly greater effect on predictions of natural recovery than does the uncertainty in sedimentation rates. Therefore, the ranges of STM and BCM input parameters are useful tools to bracket uncertainties in the evaluation of FS alternatives. Regardless, monitoring will be needed to confirm that recovery is occurring wherever MNR is proposed.

Finally, the BCM analysis is considered adequate for estimating future COC concentrations in LDW sediments (combined with the analysis of deep disturbances and exposure of subsurface contamination in Appendix M, Part 5), assigning a range of suitable remedial technologies (Section 8), and evaluating short-term and long-term effectiveness of remedial alternatives (Section 9). Model uncertainties and limitations do not negate the use of the model as a predictive tool in this FS, but must be accounted for when considering the predicted outcomes of the remedial alternatives, as discussed in Sections 9 and 10. Sections 9 and 10 also include additional detailed analysis of the effects of deep disturbance induced by anthropogenic and natural activities on long-term SWACs. Spatial areas where model predictions agree or do not agree with empirical trends and physical site conditions are accounted for in the FS in the designation of recovery categories (Section 6).

Table 5-1a Bed Composition Model Upstream Input Parameters for Human Health Risk Drivers

Rationale

Range of concentrations considered representative of current and potential future conditions for solids entering and settling in the LDW from upstream. Four different datasets used to establish range of parameter values for upstream sources because of potential biases inherent to each.

Contaminant	BCM Parameters			Basis for BCM Upstream Input and Sensitivity Values[a]
	Input	Low	High	
Total PCBs (µg /kg dw)	35	5	80	**Input:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data (36 rounded to 35 µg /kg dw). **Low:** The mean of Ecology upstream sediment samples containing fines >30%. **High:** UCL95 of TSS-normalized King County (whole-water) (82 rounded to 80 µg /kg dw).
Arsenic (mg/kg dw)	9	7	10	**Input:** Mean of Ecology upstream sediment samples containing fines >30%. **Low:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data. **High:** UCL95 of Ecology upstream sediment samples with fines >30%.
cPAHs (µg TEQ/kg dw)	70	40	270	**Input:** Mean of LDW RM 4.3 to 4.75 DMMP (2001 – 2009) core data (73 rounded to 70 µg TEQ/kg dw). **Low:** Mean of Ecology upstream sediment samples containing fines >30% (37 rounded to 40 µg TEQ/kg dw). **High:** UCL95 of TSS-normalized King County (whole-water) (269 rounded to 270 µg TEQ/kg dw).
Dioxins and Furans (ng TEQ/kg dw)	4	2	8	**Input:** Midpoint between mean of Ecology upstream centrifuged solids and mean of Ecology upstream sediment samples containing fines >30% **Low:** Mean of Ecology upstream sediment samples containing fines >30%. **High:** Midpoint between mean and UCL95 of Ecology centrifuged solids data.

Notes:

a. Upstream BCM parameter values were revised using updated datasets and statistics reflective of current conditions (i.e., material entering the LDW from the Green/Duwamish River). The four primary datasets used for BCM parameterization are as follows (see Tables 5-2a through 5-2d for statistical summaries of supporting datasets):

- Ecology's 2008 upstream bed sediment chemistry data: This dataset was screened to exclude samples with ≤30% fines in consideration of the systematic differences in grain size distributions between upstream (e.g., mid-channel) data and average conditions in the LDW.
- TSS-normalized King County data: King County surface water data were normalized to solid fractions by dividing by the TSS in the individual sample.
- Ecology 2008 centrifuged suspended solids data: The Ecology samples are representative of sediments suspended mid-channel in the Green/Duwamish River that enter the LDW.
- Upper-reach USACE DMMP core data (RM 4.3 to 4.75): This dataset is representative of Green/Duwamish River suspended material that settles in the upper section of the LDW.

BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; DMMP = Dredged Material Management Program; dw – dry weight; fines = sum of silt and clay grain size fractions; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg – milligrams; ng = nanograms; PCBs = polychlorinated biphenyls; RM = river mile; TEQ = toxic equivalent; TSS = total suspended solids; UCL95 = 95% upper confidence limit on the mean; USACE = U.S. Army Corps of Engineers

Table 5-1b Bed Composition Model Lateral Input Parameters for Human Health Risk Drivers

Rationale

1. High – Conservative representation of current conditions assuming modest level of source control (e.g., management of high priority sources).
2. Input (Mid-range) – Pragmatic assessment of what might be achieved in the next decade with anticipated levels of source control.
3. Low – Best that might be achievable in 30 to 40 years with increased coverage and continued aggressive source control.

Contaminant	BCM Parameters			Basis for BCM Lateral Input and Sensitivity Values
	Input	Low	High	
Total PCBs[a] (µg /kg dw)	300	100	1,000	Used a range of screening concentrations to reflect potential levels of source control that could occur over time. **Input:** Mean of flow-weighted dataset excluding values >5,000 µg/kg dw (315 rounded to 300 µg /kg dw). **High:** 90th percentile of flow-weighted source tracing dataset excluding values >10,000 µg/kg dw (1,009 rounded to 1,000 µg /kg dw). **Low:** Median of flow-weighted source tracing dataset excluding values >2,000 µg/kg dw (102 rounded to 100 µg/kg dw).[a]
Arsenic[a] (mg/kg dw)	13	9	30	Screened the source-tracing dataset to exclude concentrations above assumed SMS-based source control levels (93 and 57 mg/kg dw) **Input:** Mean excluding values >93 mg/kg (the CSL). **High:** 90th percentile excluding values >93 mg/kg (the CSL). **Low:** Median of all samples, excluding values >57 mg/kg (the SQS).[a]
cPAHs[a] (µg TEQ/kg dw)	1,400	500	3,400	Screened the source-tracing dataset to exclude concentrations above an assumed source control level. cPAHs are expected to be difficult to control due to the petroleum-based economy, intensity of urbanization in the LDW, and myriad ongoing sources. **Input:** Mean of source-tracing dataset excluding values >25,000 µg TEQ/kg dw (1,370 rounded to 1,400 µg TEQ/kg dw). **High:** 90th percentile of source-tracing dataset excluding values >25,000 µg TEQ/kg dw (3,366 rounded to 3,400 µg TEQ/kg dw). **Low:** Median of source tracing dataset excluding values >25,000 µg TEQ/kg dw (490 rounded to 500 µg TEQ/kg dw).[a]
Dioxins and Furans[b] (ng TEQ/kg dw)	20	10	40	Based on combined Greater Seattle metropolitan sediment and SPU catch basin solids datasets.[b] **Input:** Mean (22 rounded to 20 ng TEQ/kg dw) **High:** UCL95 (41 rounded to 40 ng TEQ/kg dw). **Low:** Median (15 rounded to 10 ng TEQ/kg dw).

Notes:

a. Used Lower Duwamish Waterway source tracing dataset (compiled by SPU) through June 2009 as the primary basis for establishing lateral BCM parameter values for arsenic, total PCBs, and cPAHs. The dataset was screened to remove concentrations using various source control practicability assumptions (best professional judgment by the Source Control Work Group). Total PCB data were flow-weighted before generating statistics because PCBs exhibit a distinct geographic distribution with hot spots identified at Terminal 117, North Boeing Field/Georgetown Steam Plant, Rainier Commons, and Boeing Plant 2/Jorgensen Forge. These four areas have been extensively sampled and make up a significant portion of the overall source tracing dataset. Therefore, the PCB source-tracing data were flow-weighted to avoid skewing the summary statistics used in the BCM. Arsenic and cPAH data were not flow-weighted prior to the statistical analysis because these contaminants lack a pronounced geographic dependency that would warrant flow-weighting. See Tables 5-2a through 5-2d for statistical summaries of supporting datasets.

b. Parameter estimation for dioxins and furans was based on the Greater Seattle metropolitan area receiving sediment dataset collected as part of the RI (Windward 2010) and sediment and SPU catch basin solids datasets (City of Seattle 2010; data collected through 2009). The summary statistics used to estimate parameter values correspond to the combined datasets, as supported by statistical analysis, and include the removal of outliers. See Tables 5-2a through 5-2d for statistical summaries of supporting datasets.

BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; CSL = cleanup screening level; dw = dry weight; kg = kilograms; LDW = Lower Duwamish Waterway; µg = micrograms; mg – milligrams; ng = nanograms; PCBs = polychlorinated biphenyls; RI = remedial investigation; SPU = Seattle Public Utilities; TEQ = toxic equivalent; SQS = sediment quality standard; UCL95 = 95% upper confidence limit on the mean

Table 5-1c Bed Composition Model Post-Remedy Bed Sediment Replacement Values for Human Health Risk Drivers

Rationale

Range of concentrations considered representative of current and potential near-term (0-3 years) post-remedy surface sediment conditions influenced by multiple recontamination mechanisms. Values expected to vary spatially.[a]

Contaminant	SWAC Outside of AOPC 1[b]	Clean Fill Material[c]	Input and Sensitivity Values					
			Proportioned Values Using SWAC Outside of AOPC 1[d]			Proportioned Values Using SWAC Outside of AOPC 2[d,e]		
			Input	Low Input	High Input	Input	Low Input	High Input
Total PCBs (µg /kg dw)	120	2	60	30	90	20	10	40
Arsenic (mg/kg dw)	12	7	10	9	11	9	8	10
cPAHs (µg TEQ/kg dw)	270	9	140	70	200	100	50	140
Dioxins and Furans (ng TEQ/kg dw)	7	2	4	2[f]	6	n/a	n/a	n/a

Notes:

a. Actively remediated areas within the AOPC 1 footprint receive the higher input values. Actively remediated areas within AOPC 2 footprint would receive lower input values. See Section 6 for a definition of AOPCs.

b. The SWAC outside of AOPC 1 is assumed representative of concentrations adjacent to remediated areas for arsenic, total PCBs, and cPAH. The representative dioxins and furans concentration outside of AOPC 1 is based on the arithmetic mean of the point values located outside of AOPC 1. See Section 6 for definition of AOPC 1.

c. The contaminant composition of clean fill material is based on the UCL95 of 2008 EPA OSV *Bold* Survey data. Use of qualified maintenance dredged materials (e.g. from the Upper Turning Basin) for capping would, in practice, lead to higher range of post-remedy bed-sediment replacement values than calculated in this table.

d. Range of representative post-remedy bed sediment replacement values assumes combinations of clean backfill material (e.g., whether capping, ENR, or post-dredge residuals management) and surrounding representative bed sediment concentrations. Assumed proportioning percentages are as follows:

BCM Parameter	Post-Remedy Bed Sediment Replacement Value Proportioning Assumptions	
	% of Clean Import Material	% of SWAC Outside of AOPC 1
Input	50	50
Low	75	25
High	25	75

Shading indicates input value used in the BCM

e. As discussed in Section 6, a larger footprint referred to as AOPC 2 was developed. The remedial alternative that evaluates this footprint will use lower input values after all high to moderate PCB concentration areas have been remediated.

f. In this case, the "low" value of 2 is used to maintain a reasonable range of concentrations. The adjustment is considered reasonable because of the small dataset available for calculating the concentration outside of AOPC 1.

AOPC = area of potential concern; BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; OSV = ocean survey vessel; ENR = enhanced natural recovery; kg = kilograms; µg = micrograms; mg – milligrams; n/a = not available; ng = nanograms; PCBs = polychlorinated biphenyls; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; UCL95 = 95% upper confidence limit on the mean.

Table 5-2a BCM Parameter Line of Evidence Information for Total PCBs (µg/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*22*	50	21	107	82	Normalized to TSS; data from 2005 to 2008, provided by King County.
	Ecology Centrifuged Solids		*7*	14	8	54	36	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
	King County and Ecology Data Combined		*29*	42	11	120	127	Calculation of all upstream surface water data by AECOM; unpublished.
Upstream Surface Sediment	LDW RI Data		*37*	23	19	40	21	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	*30*	5	2	13	8	Data from 2008, downloaded from EIM database, stats calculated by AECOM, screened to exclude samples ≤ 30% fines; outlier excluded: 770 µg/kg dw; unpublished.
		All	*73*	3	3	6	3	Data from 2008, downloaded from EIM database; stats calculated by AECOM and outlier excluded: 770 µg/kg dw.
	LDW RI and Ecology Data Combined		*110*	8	3	23	13	Calculation of all upstream surface sediment data by AECOM and outlier excluded: 770 µg/kg dw; unpublished.
USACE Upper Turning Basin Cores	RM. 4.5 – 4.75 (1991-2009)		*10*	23	22	38	23	Calculation of DAIS core data by AECOM; unpublished.
	RM. 4.3 – 4.75 (1991-2009)		*20*	36	33	56	42	Calculation of DAIS core data by AECOM; unpublished.
Lateral Inflow								
City of Seattle Storm Drain Data	Minus samples >2,000 µg/kg dw		*625*	223	102	534	—	Flow-weighted average of storm drain solids data screened to exclude samples >2,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
	Minus samples >5,000 µg/kg dw		*692*	315	125	718	—	Flow-weighted average of storm drain solids data screened to exclude samples >5,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
	Minus samples >10,000 µg/kg dw		*755*	508	146	1,009	—	Flow-weighted average of storm drain solids data screened to exclude samples >10,000 µg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
King County CSO Water Quality Data			*28*	638	580	920	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010. Estimates biased high because method assumes all PCBs in whole-water sample in particulate phase.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data	0 – 2 years after dredging		*18*	120	120	—	150	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data	Thick Cap		—	Mean = 45 (yr 0.5), 84 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports (King County 2006; 2009).
	ENR		—	Mean = 6 (yr 0), 23 (yr 1), 62 (yr 2)				
Puget Sound Survey (*OSV BOLD*)			*70*	1	1	3	2	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 120				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 47				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2b BCM Parameter Line of Evidence Information for Arsenic (mg/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*100*	37	29	73	47	Normalized to TSS; data from 2001 to 2006. All detected arsenic concentrations associated with TSS were calculated as the difference between whole-water (i.e., unfiltered) and filtered sample data.
	Ecology Centrifuged Solids		*7*	17	14	24	22	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Upstream Surface Sediment	LDW RI Data		*24*	7	5	11	8	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	*31*	9	9	11	10	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 30% fines; unpublished.
		All	*74*	7	6	10	7	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
	LDW RI and Ecology Data Combined		*98*	7	6	10	7	Calculation of all upstream surface sediment data by AECOM; unpublished.
USACE Upper Turning Basin Cores	RM. 4.5 – 4.75 (1991-2009)		*8*	5	5	7	7	Calculation of DAIS core data by AECOM; unpublished. 1990 data excluded.
	RM. 4.3 – 4.75 (1991-2009)		*18*	7	6	12	8	Calculation of DAIS core data by AECOM; unpublished. 1990 data excluded.
Lateral Inflow								
City of Seattle Storm Drain Data		Minus samples >57 mg/kg dw	*553*	12	9	29	—	Storm drain solids data screened to exclude samples >57 mg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
		Minus samples >93 mg/kg dw	*563*	13	10	30	—	Storm drain solids data screened to exclude samples >93 mg/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer, 2010.
King County CSO Water Quality Data			*21*	9	11	13	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data		0 – 2 years after dredging	*8*	11	12	—	14	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data		Thick Cap	—	Mean = 3 (yr 0.5), 10 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports (King County 2006; 2009).
		ENR	—	Mean = 2 (yr 0), 4 (yr 1), 8 (yr 2)				
EPA OSV *Bold* Survey			*70*	7	6	11	7	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 12				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 10				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2c BCM Parameter Line of Evidence Information for cPAHs (µg TEQ/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	King County Whole Water		*18*	151	74	354	269	Normalized to TSS; data from 2008, provided by King County.
Green River Water Quality	Ecology Centrifuged Solids		*7*	138	53	400	432	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Green River Water Quality	King County & Ecology Data Combined		*25*	135	58	330	266	Calculation of all upstream surface water data by AECOM; unpublished.
Upstream Surface Sediment	LDW RI Data		*16*	55	18	135	100	Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
Upstream Surface Sediment	Ecology	Fines >30%	*31*	37	16	77	72	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤30% fines. Note: Outlier of 230 µg TEQ/kg dw was not excluded from any statistical calculations.
Upstream Surface Sediment	Ecology	Fines >50%	*18*	50	44	91	75	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 50% fines. Note: Outlier of 230 µg TEQ/kg dw was not excluded from any statistical calculations.
Upstream Surface Sediment	Ecology	All	*74*	18	9	57	43	Data from 2008, downloaded from EIM database, stats calculated by AECOM. Note: Outlier of 230 µg TEQ/kg dw was included in statistical calculations.
Upstream Surface Sediment	LDW RI and Ecology Data Combined		*90*	25	10	73	55	Calculation of all upstream surface sediment data by AECOM; unpublished.
USACE Upper Turning Basin Cores	RM. 4.5 – 4.75 (1991-2009)		*9*	37	41	63	52	Calculation of DAIS core data by AECOM; outlier excluded: 1051.5 µg TEQ/kg dw; unpublished.
USACE Upper Turning Basin Cores	RM. 4.3 – 4.75 (1991-2009)		*19*	73	57	180	134	Calculation of DAIS core data by AECOM; outlier excluded: 1051.5 µg TEQ/kg dw; unpublished.
Lateral Inflow								
City of Seattle Storm Drain Data (minus samples >25,000 µg TEQ/kg dw)			*533*	1,370	490	3,366	—	Storm drain solids data screened to exclude samples >25,000 µg TEQ/kg dw; data collected through June 2009. SPU data provided by B. Schmoyer (2010).
King County CSO Water Quality Data			*26*	1,051	714	2,728	—	TSS-normalized values of CSO water data provided by D. Williston, King County, 2010. Estimates biased high because method assumes all cPAHs in whole-water samples in particulate phase.
Post-Remedy Bed Sediment Replacement Value								
Post-Maintenance Dredge Surface Data	0 – 2 years after dredging		*8*	180	170	—	250	Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Duwamish/Diagonal Post-Capping Data	Thick Cap		—	Mean = 63 (yr 0.5), 159 (yr 3)				Calculation of D/D post-capping data by AECOM; data available in King County monitoring reports(King County 2006; 2009).
Duwamish/Diagonal Post-Capping Data	ENR		—	Mean = 11 (yr 0), 43 (yr 1), 89 (yr 2)				
Puget Sound Survey (OSV BOLD)			*70*	7	4	15	9	Calculation of Puget Sound Survey stats by AECOM.
Outside AOPC 1 Footprint			*n/a*	IDW interpolated SWAC = 270				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*n/a*	IDW interpolated SWAC = 190				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Notes: See Table 5-2d for notes.

Table 5-2d BCM Parameter Line of Evidence Information for Dioxins/Furans (ng TEQ/kg dw)

Study/Source			*No. of Samples*	Mean	Median	90th Percentile	UCL95	Comments
Green/Duwamish River Inflow								
Green River Water Quality	Ecology Centrifuged Solids		*6*	6	3	13	10	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
Upstream Surface Sediment	LDW RI Data		*4*	Range of Values (Median): 1.1 - 2.6 (1.7)				Data from 1994 to 2005 between RM 5 and 7 included in the RI baseline dataset.
	Ecology	Fines >30%	*31*	2	2	3	2	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 30% fines; unpublished.
		Fines >50%	*18*	2	2	3	3	Data from 2008, downloaded from EIM database, stats calculated by AECOM and screened to exclude samples ≤ 50% fines; unpublished.
		All	*74*	1	0.3	3	2	Data from 2008, downloaded from EIM database, stats calculated by AECOM.
USACE Upper Turning Basin Cores RM 4.3 – 4. 75 (1991-2009)			*2*	2 and 2.8 ng TEQ/kg dw				Calculation of DAIS core data by AECOM; unpublished.
Lateral Inflow								
Greater Seattle Sediment and SPU Catch Basin Solids			*23*	22	15	48	41	Calculation of stats based on combined Greater Seattle sediment and SPU catch basin solids datasets by AECOM; outlier excluded: 187 ng TEQ/kg dw; unpublished.
Post-Remedy Bed Sediment Replacement Value								
Puget Sound Survey (OSV BOLD)			*70*	1	1	2	2	Calculation of Puget Sound Survey stats by AECOM.
Post- Maintenance Dredge Area Surface Data			*3*	Mean = 8.3 ng TEQ/kg dw				Calculation of post-maintenance dredge surface data by AECOM; unpublished.
Outside AOPC 1 Footprint			*18*	Mean = 7 ng TEQ/kg dw				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.
Outside AOPC 2 Footprint			*11*	Mean = 5 ng TEQ/kg dw				Calculation of IDW interpolated SWAC by AECOM; unpublished. See Section 6 for AOPCs.

Value(s) used for central tendency BCM input value. (mid point between mean Ecology Centrifuged solids and mean upstream fines >30% used for Green/Duwamish River)

Value(s) used as basis for low-sensitivity BCM value.

Value(s) used as basis for high-sensitivity BCM value.(mid-point between mean and UCL95 Ecology Centrifuged Solids used for Green/Duwamish River)

Notes:
1. Statistics for these datasets were calculated using ProUCL 4.0, except that statistics for the City of Seattle Storm Drain Solids, King County CSO Water Quality, and Post-Remedy Bed Sediment Replacement Values datasets were calculated with Excel.
2. TEQs were calculated using one-half RL for undetected individual dioxin/furan congeners or PAH compounds.

'—' = not calculated; n/a = not available

AOPC = area of potential concern; BCM = bed composition model; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; CSO = combined sewer overflow; DAIS = Dredged Analysis Information System; D/D = Duwamish/Diagonal; dw = dry weight; EIM = Ecology Information Management Database; ENR = enhanced natural recovery; fines = sum of silt and clay grain size fractions; IDW = inverse distance weighting; kg = kilogram; LDW = Lower Duwamish Waterway; µg = microgram; mg = milligram; ng = nanograms; OSV = ocean survey vessel; PCBs = polychlorinated biphenyls; RI = remedial investigation; RL = reporting limit; RM = river mile; SPU = Seattle Public Utilities; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; TSS = total suspended solids; USACE = U.S. Army Corps of Engineers; UCL95 = 95 percent upper confidence limit on the mean

Table 5-3 BCM Input Values for Representative SMS Contaminants[a]

Contaminant	Upstream Inflow (n = 22 to 23)		Lateral Inflow (n = 531 to 579)	
	BCM Input Value (µg/kg dw)[b]	Basis	BCM Input Value (µg/kg dw)[b]	Basis
BEHP	120	Median of USACE Dredged Material Characterization Core Data (RM 4.3 to 4.75; USACE 2009a, 2009b)	15,475	Log-normal mean of City of Seattle source-tracing data through July 2009 with outliers removed[c] (SPU 2010)
Chrysene	49		1,807	
Fluoranthene	190		3,989	
Phenanthrene	53		2,010	
Mercury (mg/kg dw)	0.1		0.14	
Zinc (mg/kg dw)	64		626	
Acenaphthalene	8		209	
Butylbenzyl-phthalate	11		972	
Indeno(1,2,3-cd)pyrene	31		675	
Phenol	10		237	

Notes:

a. FS dataset used to generate summary statistics.

b. Units are in µg/kg dw, unless otherwise noted. Input values are not flow-weighted.

c. Values that were at least two times the next highest value were removed from the analysis as outliers.

BCM = bed composition model; BEHP = bis(2-ethylhexyl)phthalate; dw = dry weight; kg = kilogram; µg = micrograms; mg = milligrams; n= number of; RM = river mile; SMS = Sediment Management Standards; SPU = Seattle Public Utilities; USACE = U.S. Army Corps of Engineers

Table 5-4 Results of Additional STM Special Scenario Runs

Purpose	Description	Results
1: Potential Recontamination of EAAs	An additional bed sediment class is added to differentiate sediment within EAAs from sediment outside of EAAs. This addition results in 16 sediment variables (four size classes for each of four sediment types): EAA bed sediments, non-EAA bed sediments, lateral source sediments, and upstream Green/Duwamish River source sediments). Model is run for 10-year period to predict how unremediated areas may contribute to recontamination of remediated area, assuming EAAs have been remediated.	• Contribution from non-EAA areas to remediated EAAs is less than 5% of the surface sediments at most EAAs after 10 years.
2: Distributed Discharges from Lateral Sources	The STM input is modified to have the discharges from lateral sources distributed to more closely describe actual drainage distribution among shoreline outfalls. The updates primarily affect private nearshore drainage basins. The model is run for both 10-year and 30-year periods to compare what was reported in the STM report (QEA 2008)(the lateral load distributed via 21 outfalls) with the redistributed lateral loads used in the FS.	• Lateral source sediments are more widely distributed, often at lower percent composition, along the nearshore STM grid cells. • Lateral source sediments are more widely distributed throughout the LDW, but most of the changes only result in some areas increasing from <1.0% lateral load content to 1.0 - 2.0%. • The greatest changes were observed around Hamm Creek and between RM 2 and 3. • Updated load distribution used in all subsequent analyses; it was used in all STM base-case model runs.
3: Movement of LDW Bed Sediment into the Upper Turning Basin	10-year model run that tracks bed sediment from four sources: Upper Turning Basin, navigation channel from RM 4.0 to 4.3, bench areas upstream of RM 4.0, and all sediment downstream of RM 4.0. The model run predicts whether downstream LDW sediments resuspend and settle upstream in the Upper Turning Basin area.	• Contribution of downstream sediment to the Upper Turning Basin area is negligible (<0.01%). • Only 240 MT of sediment is transported upstream to Reach 3 from downstream areas over 10 years compared to over 800,000 MT that settles in Reach 3 from upstream. • Supports use of USACE sediment cores collected from RM 4.3 to 4.75 in navigation channel as one line of evidence of upstream solids (i.e., negligible input from downstream sediments).
4: Movement of Bed Sediments between Reaches	Evaluation of the mass balance of sediment originating from each reach that moves between reaches and out of the LDW. This scenario is conducted for the 30-year model period.	• Much of the sediment resuspended in a reach that resettles in the LDW settles within the same reach. • There is more of an exchange of sediments between Reach 1 and 2, than from Reach 1 and 2 to Reach 3. • Reach 3 sediments are widely distributed throughout the LDW, while very little sediment from Reach 1 or 2 resettles in Reach 3.

Table 5-4 Results of Additional STM Special Scenario Runs (continued)

Purpose	Description	Results
5: Sediment Scoured from Greater than 10-cm Depth	Areas that are estimated to scour greater than 10-cm depth are assigned a new variable to represent a new sediment class. The 100-year high-flow simulation is used to predict where these >10 cm scoured sediments resettle.	• Sediment eroded from below 10 cm makes up a very small fraction of the total sediment mass moving over a 100-year high-flow event. • Sediment eroded from below 10 cm is greatest in Reach 2 and lowest in Reach 1. • Most of the scour >10 cm occurs in localized navigation channel above about RM 2.9.
6: Movement of Existing Bed Sediment (bed-tracking)	An additional bed sediment class is added to differentiate bed sediment that was resuspended and redeposited into another model cell from original bed sediment over a 10-year period. This scenario tracks the movement of bed sediments with the LDW and its effect on bed composition and SWACs.	• Resuspended bed sediment makes up less than 30% of the total original + resuspended bed fraction, and typically less than 5 to 10%. • The BCM construct is considered appropriate for use in the FS.
7: Holding Cells Constant in Selected Scour and Berthing Areas (no natural recovery)	The analysis was a 10-year model run that assumed no natural recovery in areas with high-flow scour, evidence of propeller scour, and berthing areas with less than 0.5 cm/yr of sedimentation. These areas were essentially "held constant" at their FS baseline total PCB concentrations. The analysis was conducted over 10 years following construction of Alternative 3C and then compared to the site-wide and reach-wide best-estimate total PCB SWAC model predictions.	• Total PCB SWACs increased about 10% compared to best-estimate model predictions and up to 18% in Reach 2.

Note:

BCM = Bed Composition Model; cm = centimeter; EAA = early action area; FS = feasibility study; LDW = Lower Duwamish Waterway; MT = metric ton; PCBs = polychlorinated biphenyls; RM = river mile; STM = sediment transport model; SWAC = spatially-weighted average concentration; USACE = U.S. Army Corps of Engineers; yr = year

Table 5-5 Comparison of Year 10 Total PCB SWACs between the Bed Tracking Scenario and STM Base Case

Scenario	Total PCB SWACs (µg/kg dw)			
	Site-wide	Reach 1	Reach 2	Reach 3
Post-Alternative 1				
Year 0	180	190	220	56
Year 10 STM Base Case	73	84	70	40
Year 10 modified STM Bed Tracking with resuspended bed variable	72	84	66	40
Distal Sediment Concentration Input Values to the Analysis				
Distal Bed (µg/kg dw) – reach-wide post-Alternative 1 SWAC	n/a	176	117	57

Notes:

1. For a detailed discussion of the analyses supporting this table, see Part 5 of Appendix C.

dw = dry weight; kg = kilogram; µg =micrograms; n/a = not applicable; PCB = polychlorinated biphenyl; STM = sediment transport model; SWAC = spatially-weighted average concentration

Table 5-6a Total PCB Input Concentrations for the Particle Size Fractionation Analysis

Solids Source and Class	Percentage of Solids by Mass	Total PCB Concentration (µg/kg dw)			
		FS mid-range BCM Input Value	Approach 1	Approach 2	Approach 3
Green/Duwamish (Upstream) Solids					
Class 1A	70	35	80	42	38
Class 1B	18	35	80	21	38
Class 2	12	35	5	13	11
Class 3	0	35	5	3	2
Aggregate concentration on upstream solids		35	71	35	35
Lateral Source Solids					
Class 1A	55	300	1,000	422	374
Class 1B	18	300	1,000	211	374
Class 2	23	300	100	127	112
Class 3	4	300	100	25	22
Aggregate concentration on lateral solids		300	757	300	300

Notes:

1. For Green/Duwamish solids Classes 1A, 1B, and 2 are suspended load and Class 3 is bed load. However, there is very little bed load that reaches the LDW beyond river mile 4.5.
2. The Draft Final FS mid-range BCM input values are shown for reference when comparing input values for the three approaches.
3. Approach 1 essentially increases PCB mass from upstream and lateral sources by approximately 100 percent over the mid-range BCM input values, while Approaches 2 and 3 maintain the same PCB mass as in the mid-range BCM case.

Table 5-6b Effect of Particle Size Fractionation on Total PCB SWACs

LDW Reach	Total PCB SWAC (µg/kg dw) Resulting from Use of:			
	FS Mid-range BCM Input Value	Approach 1	Approach 2	Approach 3
1	84	120	78	85
2	67	100	60	66
3	40	51	23	28
Site-Wide	73	104	65	71

Notes:

BCM = bed composition model; dw = dry weight; FS = feasibility study; kg = kilogram; LDW = Lower Duwamish Waterway; µg = micrograms; PCB = polychlorinated biphenyl; SWAC = spatially-weighted average concentration

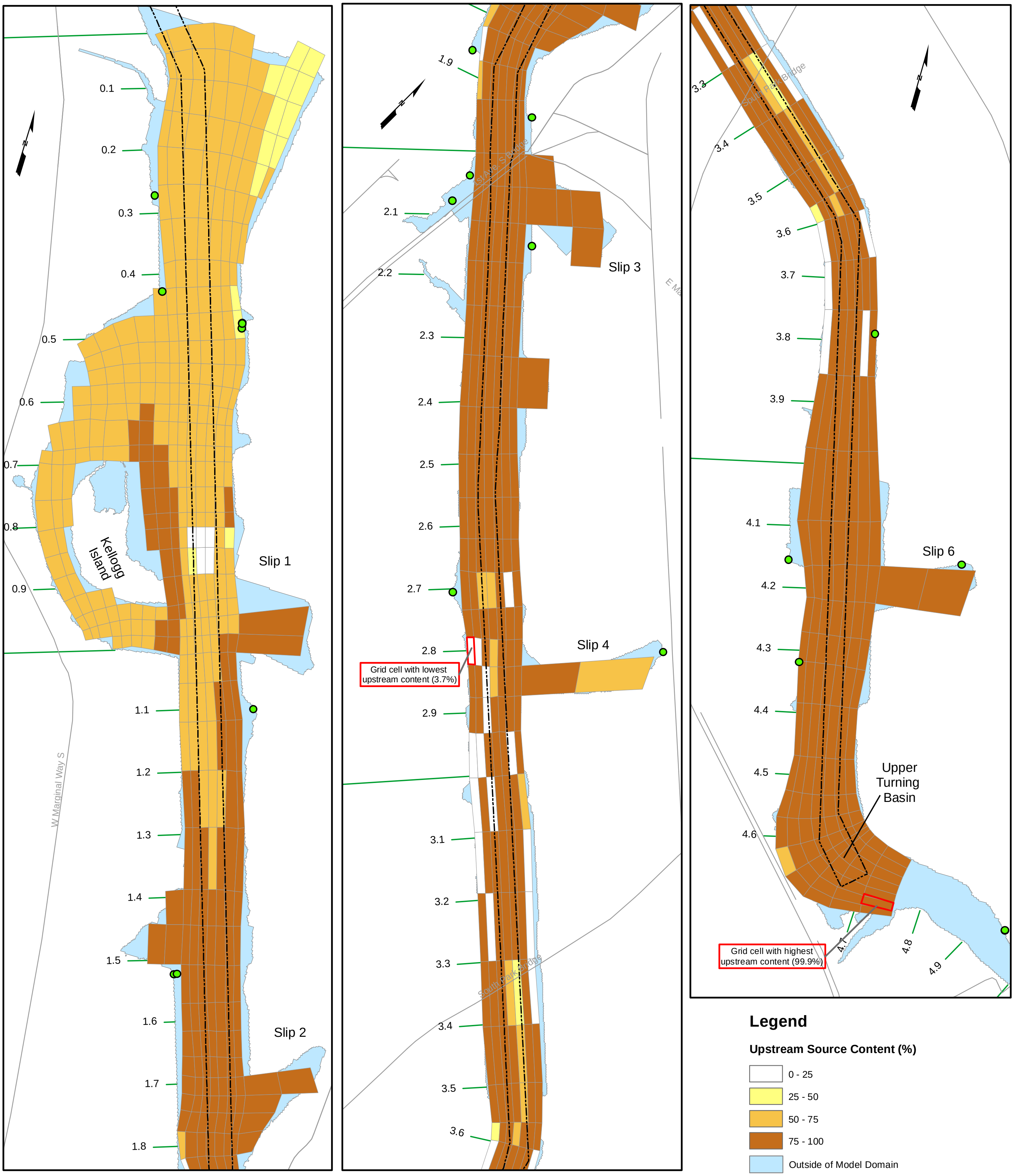
Table 5-7 Changes in Contaminant Concentrations at Resampled Surface Sediment Stations

Contaminant and Metric	Original/Older Data (1991–2006)	Newer (FS Baseline) Data (1998–2008)	Percent Decrease between Older and Newer Concentrations (%)
Total PCBs (µg/kg dw); N = 67			
25th Percentile	107	74	31
Mean	939	354	62
90th Percentile	2,141	776	64
Arsenic (mg/kg dw); N = 56			
25th Percentile	10	11	Minimal change; in equilibrium
Mean	40	35	
90th Percentile	41	40	
cPAHs (µg TEQ/kg dw); N = 53			
25th Percentile	200	145	28
Mean	1,534	437	72
90th Percentile	2,070	803	61
BEHP (µg/kg dw); N = 53			
25th Percentile	230	92	60
Mean	827	310	63
90th Percentile	1,570	606	61

Notes:

1. Newer data are co-located with older data (i.e., within 10 ft). Older data are not included in the FS baseline dataset.
2. Statistics calculated using ProUCL v.4.00.04.
3. Undetected data were set to the reporting limit.
4. Three PCB locations omitted to generate the n=67 dataset: LDW-SS110/SD-323-S; LDW-SS111/DR186; and SD-320-S/SD-DUW92. These are located within the Boeing Plant 2/Jorgensen Forge EAA.
5. Results on a station-by-station basis are provided in Appendix F.

BEHP = bis(2-ethylhexyl)phthalate; cPAHs = carcinogenic polycyclic aromatic hydrocarbons; dw = dry weight; EAA = early action area;
FS = feasibility study; kg = kilogram; LDW = Lower Duwamish Waterway; µg = micrograms; mg = milligrams; N = number of;
PCB = polychlorinated biphenyl; TEQ = toxic equivalent



Legend

Upstream Source Content (%)

- 0 - 25
- 25 - 50
- 50 - 75
- 75 - 100
- Outside of Model Domain
- Discharge Location Modeled in STM
- Road
- Navigation Channel
- River Mile Marker

Feet
0 200 400 800

Notes:
1. 10-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).
2. A grid cell with 75% sediment from upstream sources has 25% of the surface sediment from lateral and original bed sediment, totaling 100% bed composition (as solids).

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AECOM

Lower Duwamish Waterway Group
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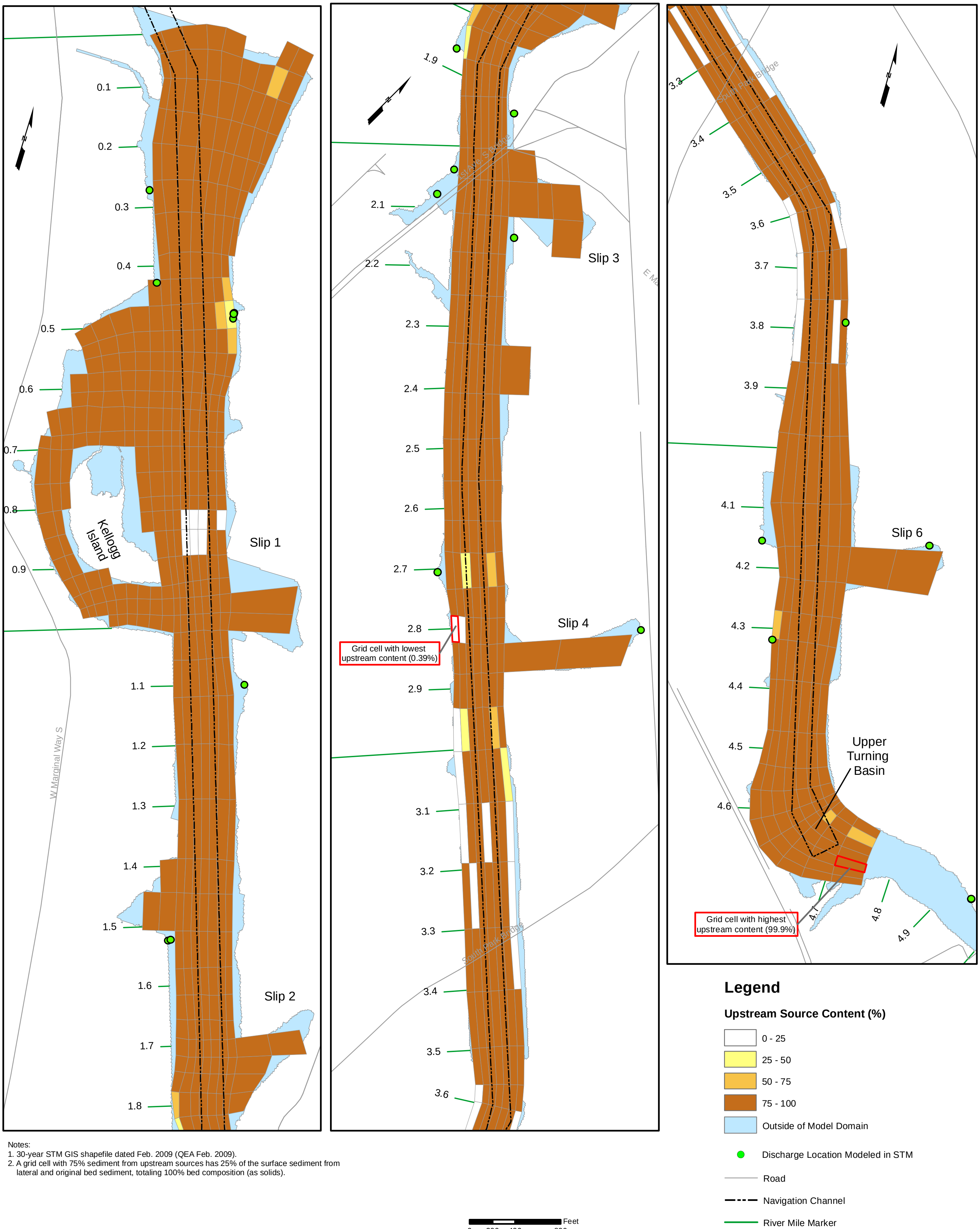
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			**Percentage of Bed Sediment from Upstream Sources after 10 Years**
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	**FIGURE 5-1a**



Notes:
1. 30-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).
2. A grid cell with 75% sediment from upstream sources has 25% of the surface sediment from lateral and original bed sediment, totaling 100% bed composition (as solids).

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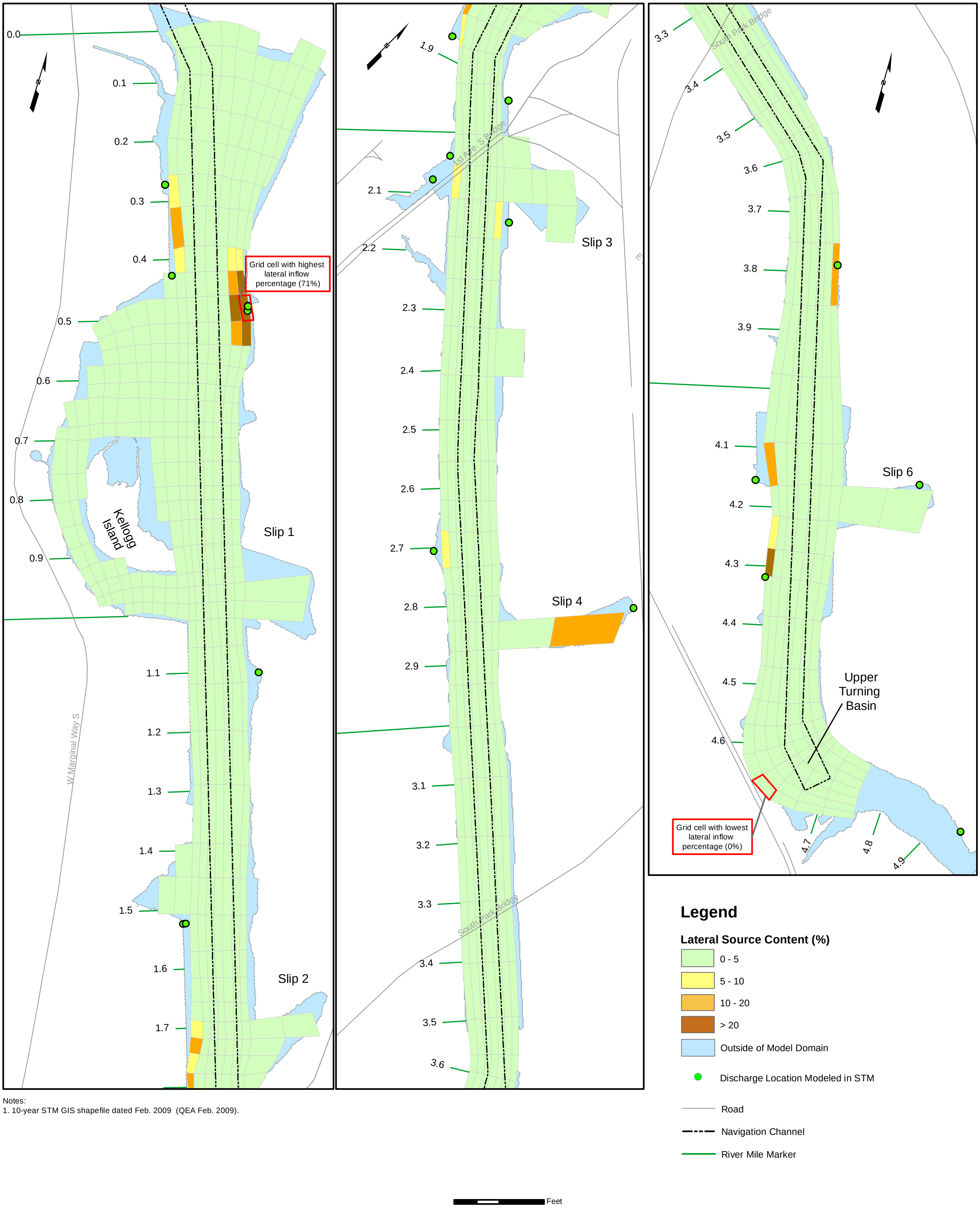
Lower Duwamish Waterway Final Feasibility Study 60150279-14.8			Percentage of Bed Sediment from Upstream Sources after 30 Years
DATE: 10/31/12	DWRN:MVI/sea	Revision: 1	FIGURE 5-1b



Notes:
1. 10-year STM GIS shapefile dated Feb. 2009 (QEA Feb. 2009).

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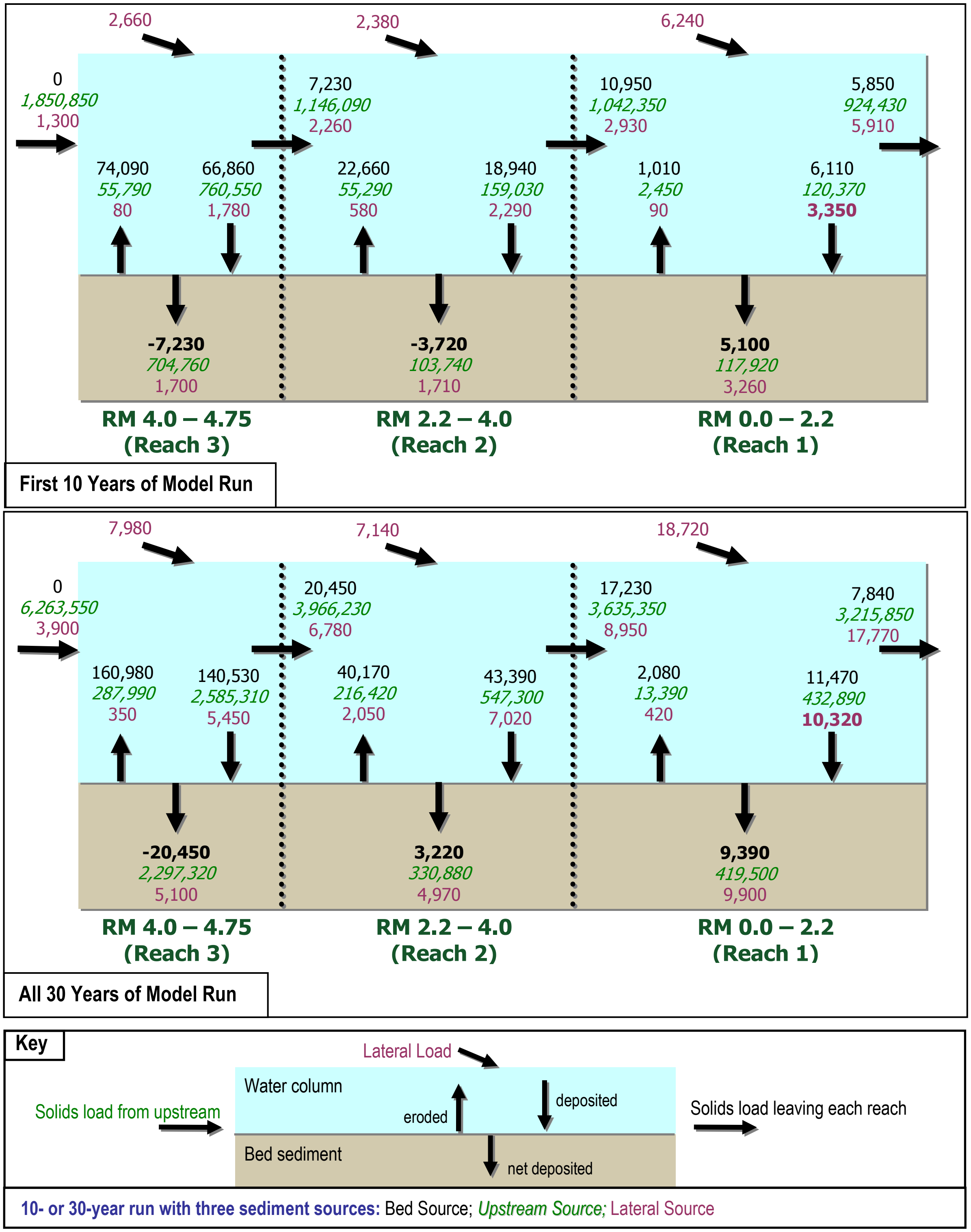
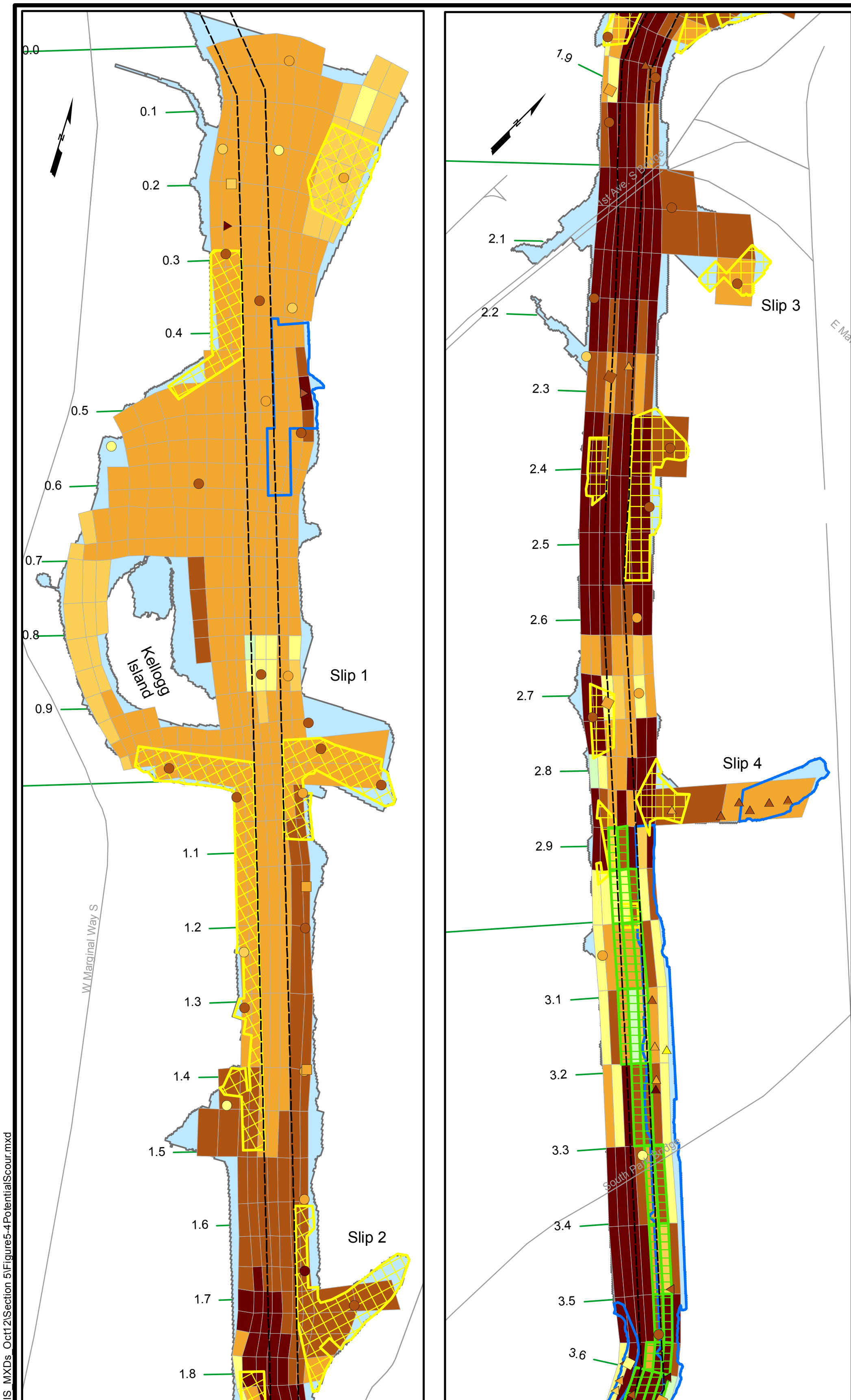
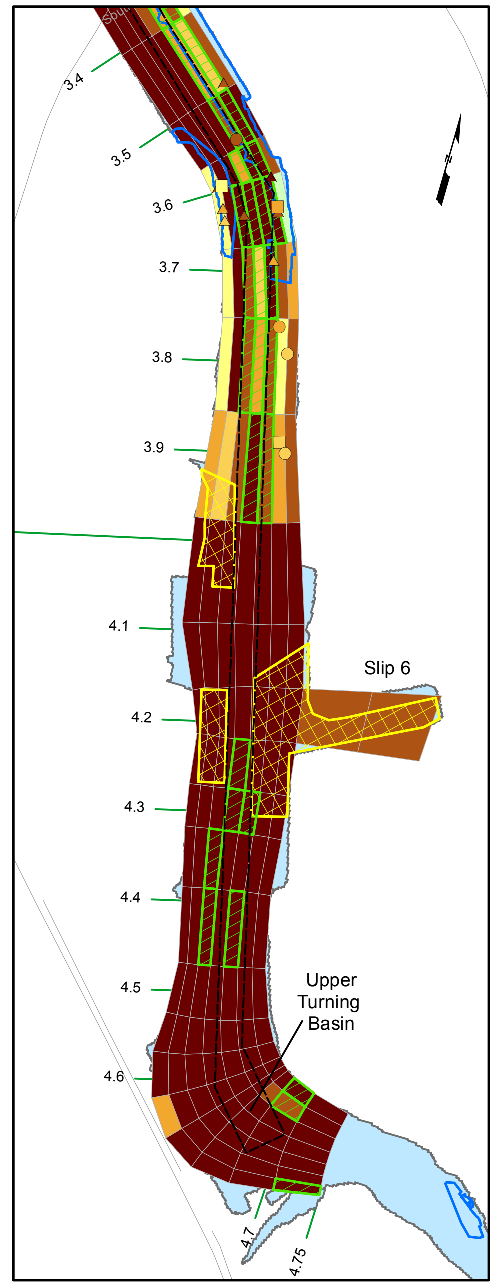
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			**Percentage of Bed Sediment from Lateral Sources after 10 Years**
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	**FIGURE 5-2**

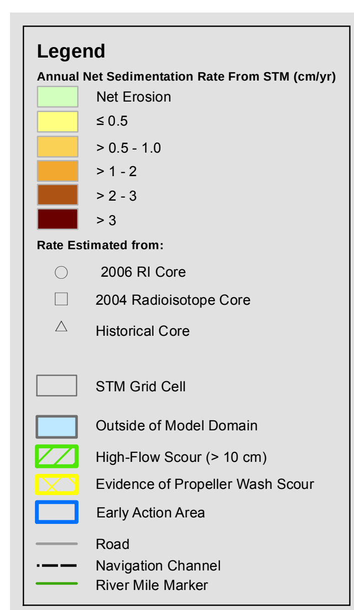
Figure 5-3 Sediment Loading to, within, and through the LDW over Two STM Time Periods



Notes:
1. Sediment loads are in metric tons over a 10- or 30-year model run, and include both suspended sediment and bed load material.
2. Upper end of modeled reach extends to RM 4.75.
3. Most of the incoming bed load (sand) settles in Reach 3. Of the incoming fines fractions, 10% of clay and 76% of the silt settle in the LDW.
LDW = Lower Duwamish Waterway; RM= river mile; STM = sediment transport model.







Notes:
1. Net sedimentation rates estimated from radioisotope core data provided by QEA LLC; and 2006 core chemistry data and historical core data provided by Windward Environmental LLC.
2. Numerous time markers used to estimate net sedimentation rates are from radioisotope, physical, and chemical geochronology profiles.
3. Ranges shown are calculated from recovered depths.
4. STM GIS shapefile from 30-year run (QEA Feb. 2009).
5. High-flow scour of 10 cm or more over 30-year simulation (QEA 2008).

0 400 800 Feet

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Final Feasibility Study
60150279-14.8

DATE: 10/31/12 | DWN. BY: MVI/sea

Potential Scour Areas and Estimated Net Sedimentation Rates

FIGURE 5-4

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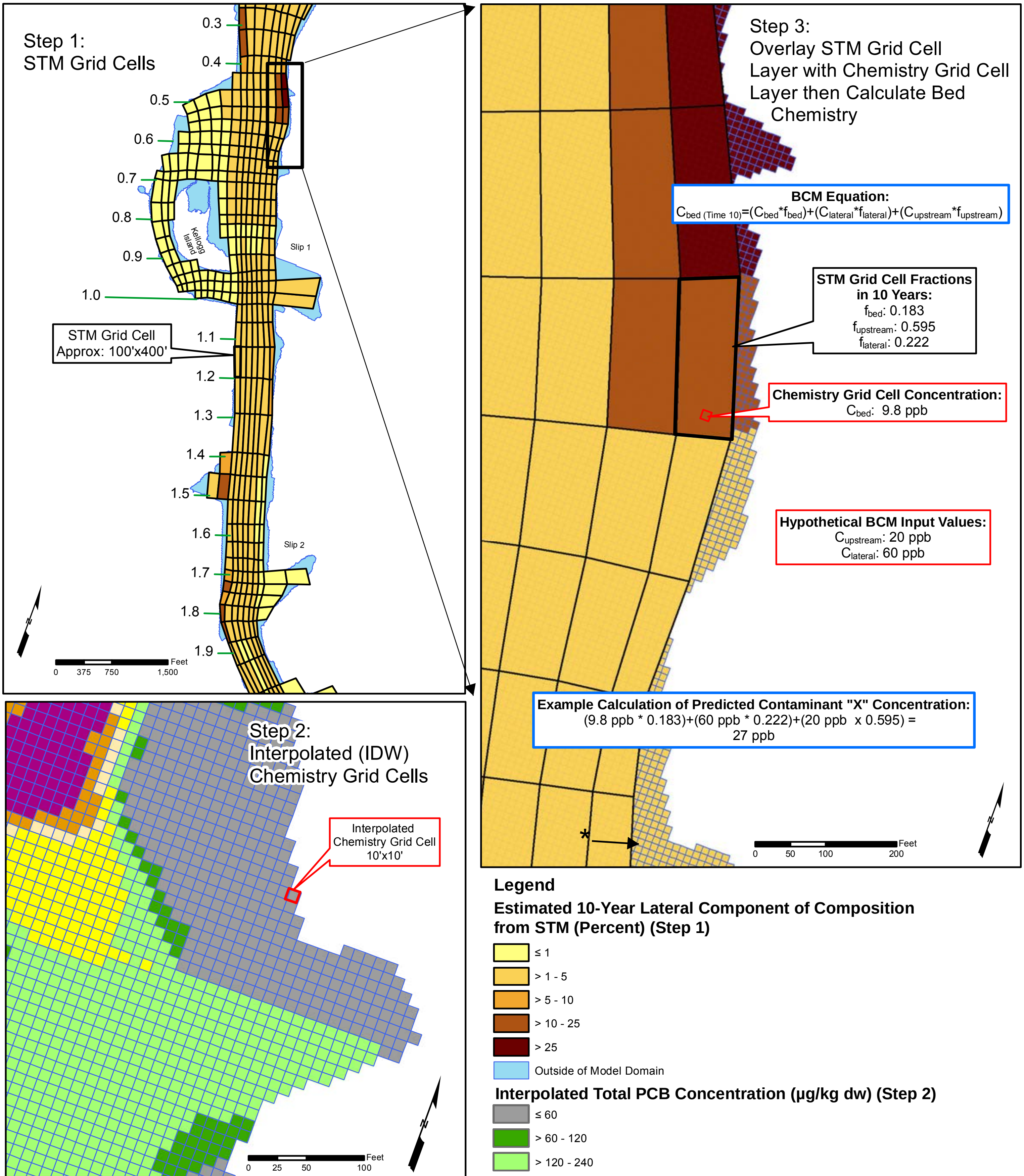

Legend

Estimated 10-Year Lateral Component of Composition from STM (Percent) (Step 1)

- ≤ 1
- > 1 - 5
- > 5 - 10
- > 10 - 25
- > 25
- Outside of Model Domain

Interpolated Total PCB Concentration (µg/kg dw) (Step 2)

- ≤ 60
- > 60 - 120
- > 120 - 240
- > 240 - 480
- > 480 - 720
- > 720 - 1,300
- > 1,300

River Mile Marker

Notes:
1. 10-year STM GIS shapefile (QEA June 2008) for illustrative purposes only.
2. The BCM equation is calculated for each 10'x10' grid cell.

* Data from closest STM grid cell was extrapolated towards shore to match chemistry grid cell layer footprint.
ppb= parts per billion.

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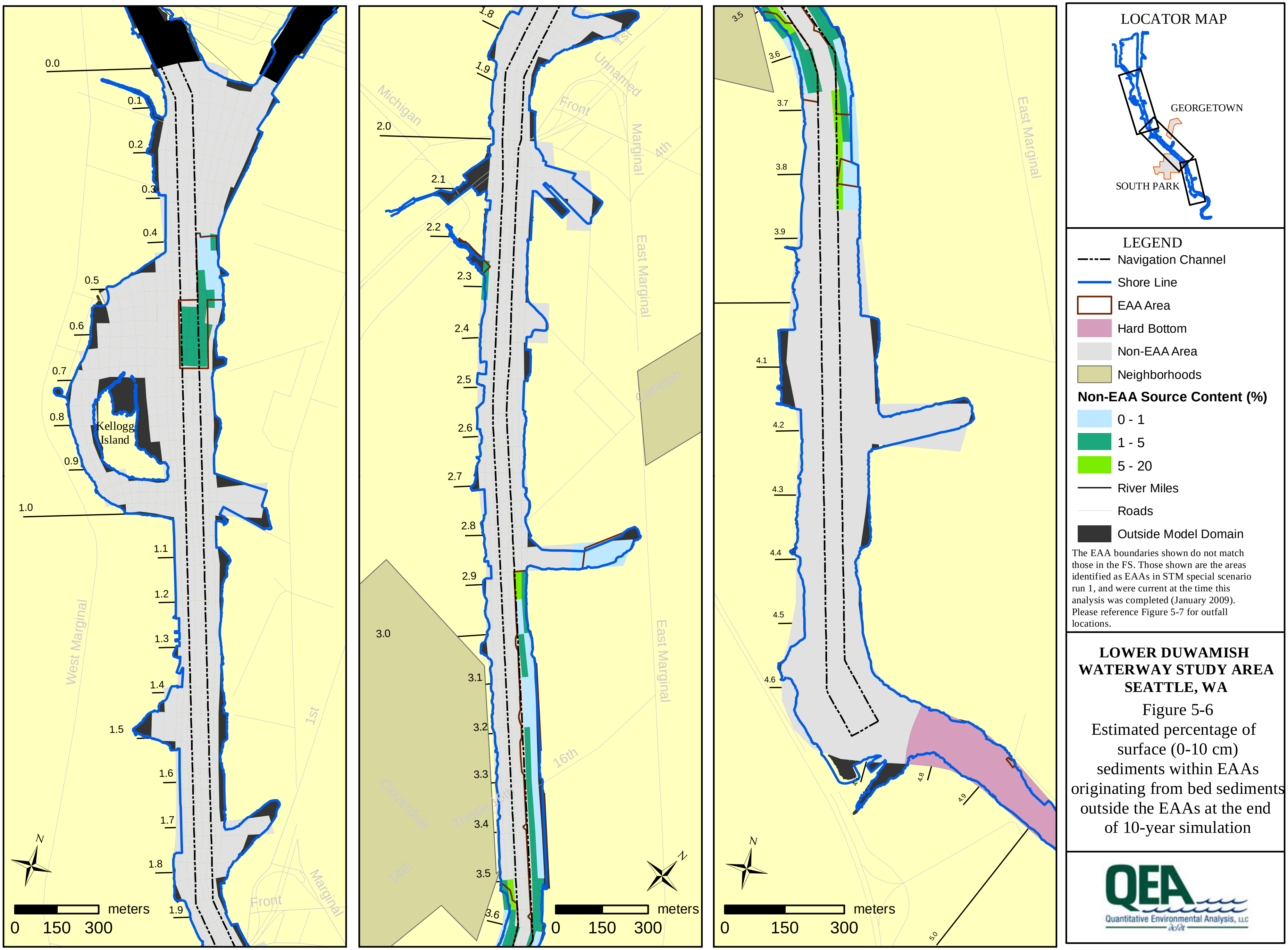
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			BCM Calculation in Model Grid Cells
DATE: 10/31/12	DWRN:MVI/Sea	Revision: 0	FIGURE 5-5



The EAA boundaries shown do not match those in the FS. Those shown are the areas identified as EAAs in STM special scenario run 1, and were current at the time this analysis was completed (January 2009). Please reference Figure 5-7 for outfall locations.

LOWER DUWAMISH WATERWAY STUDY AREA SEATTLE, WA

Figure 5-6
Estimated percentage of surface (0-10 cm) sediments within EAAs originating from bed sediments outside the EAAs at the end of 10-year simulation

QEA Quantitative Environmental Analysis, LLC

FC/JC/DN- \\Jill\D_drive\Jobs\RETltw\Documents\Scenario_Memo\RETldw_tech_memo_fig_1-3_FS_fig_5-9_090320.mxd

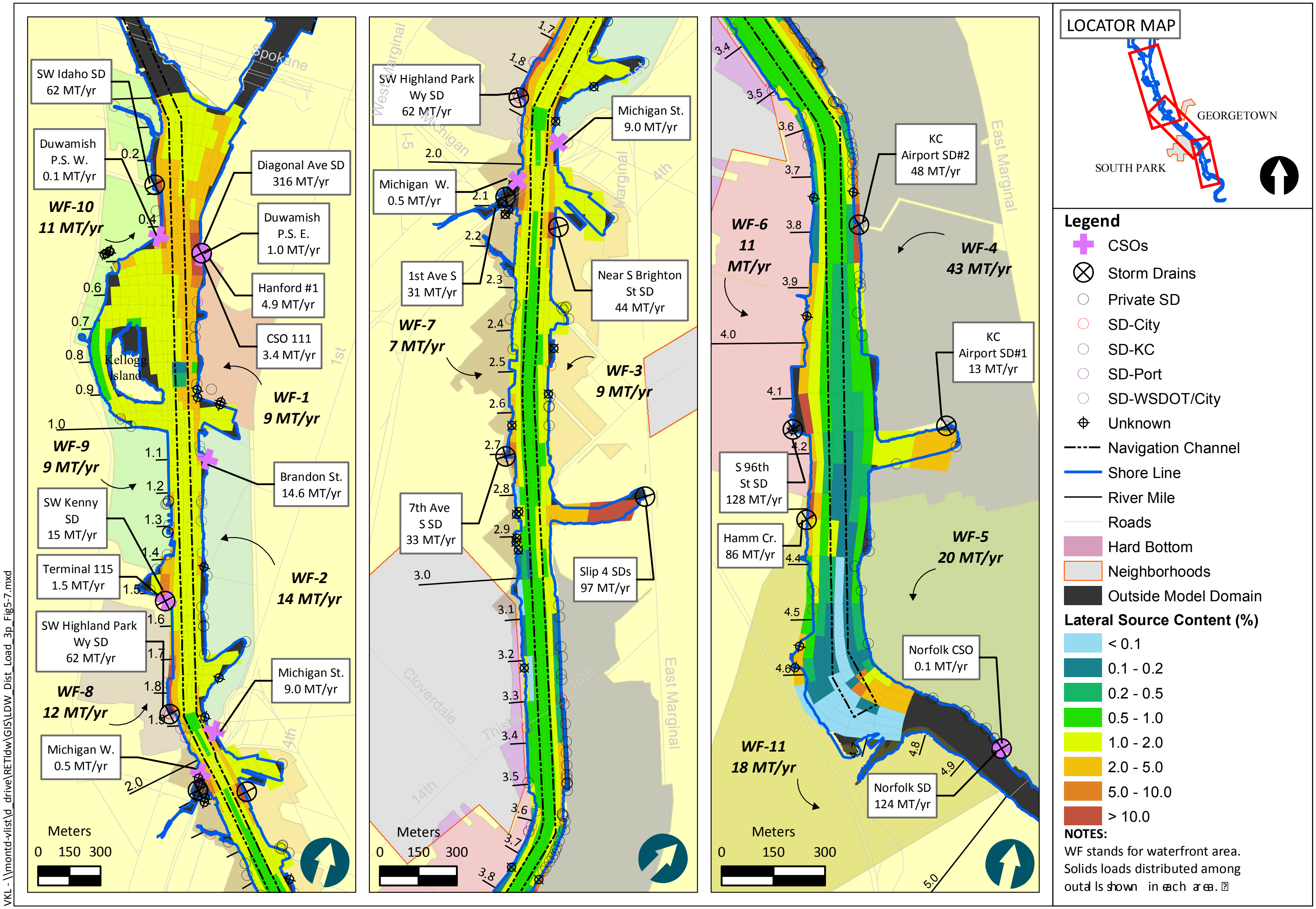
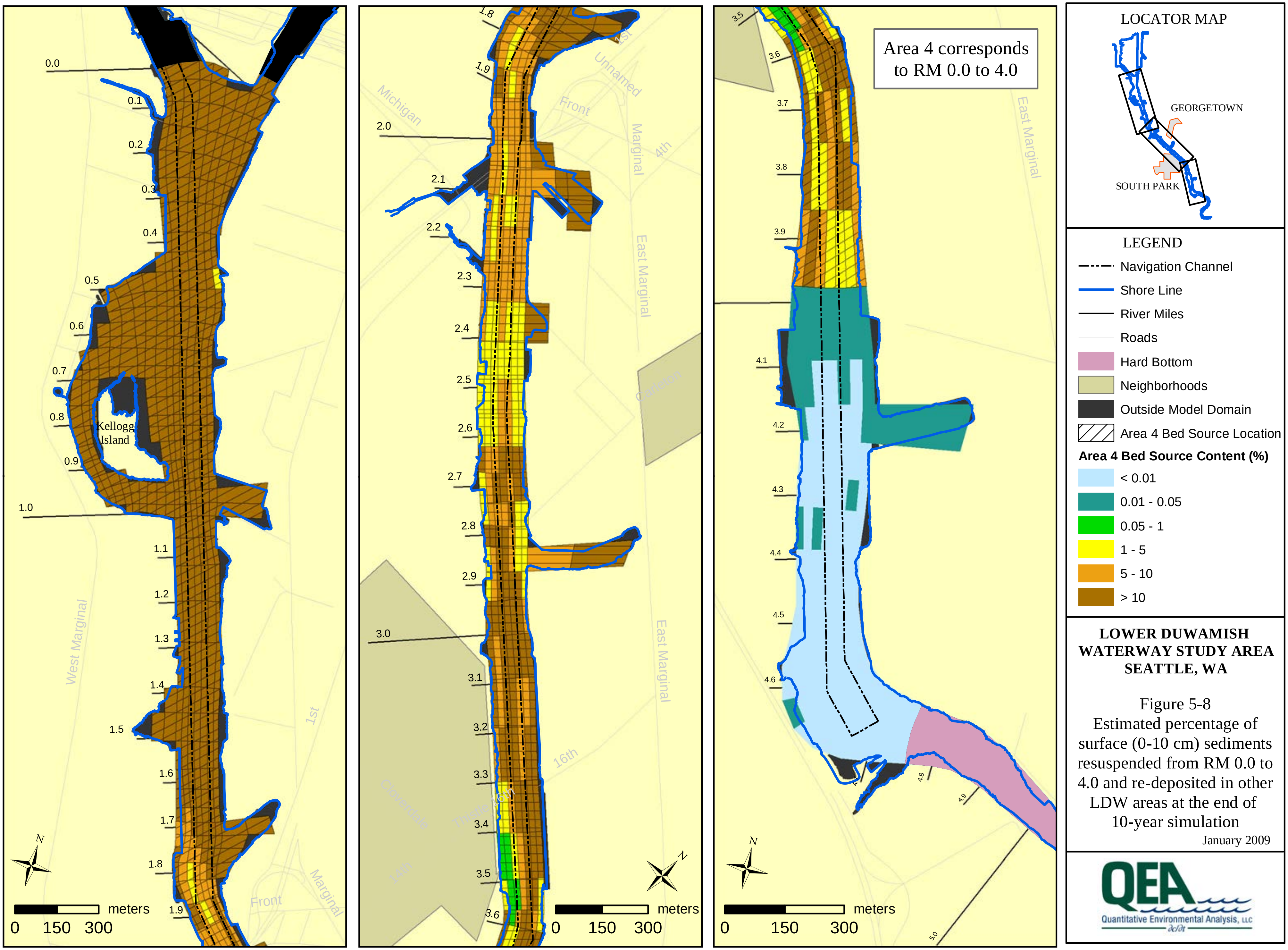


Figure 5-7

Distributed lateral annual loads and lateral source content in surface (0-10 cm) sediments at end of 10-year simulation

Quantitative Environmental Analysis, LLC



LOWER DUWAMISH WATERWAY STUDY AREA SEATTLE, WA

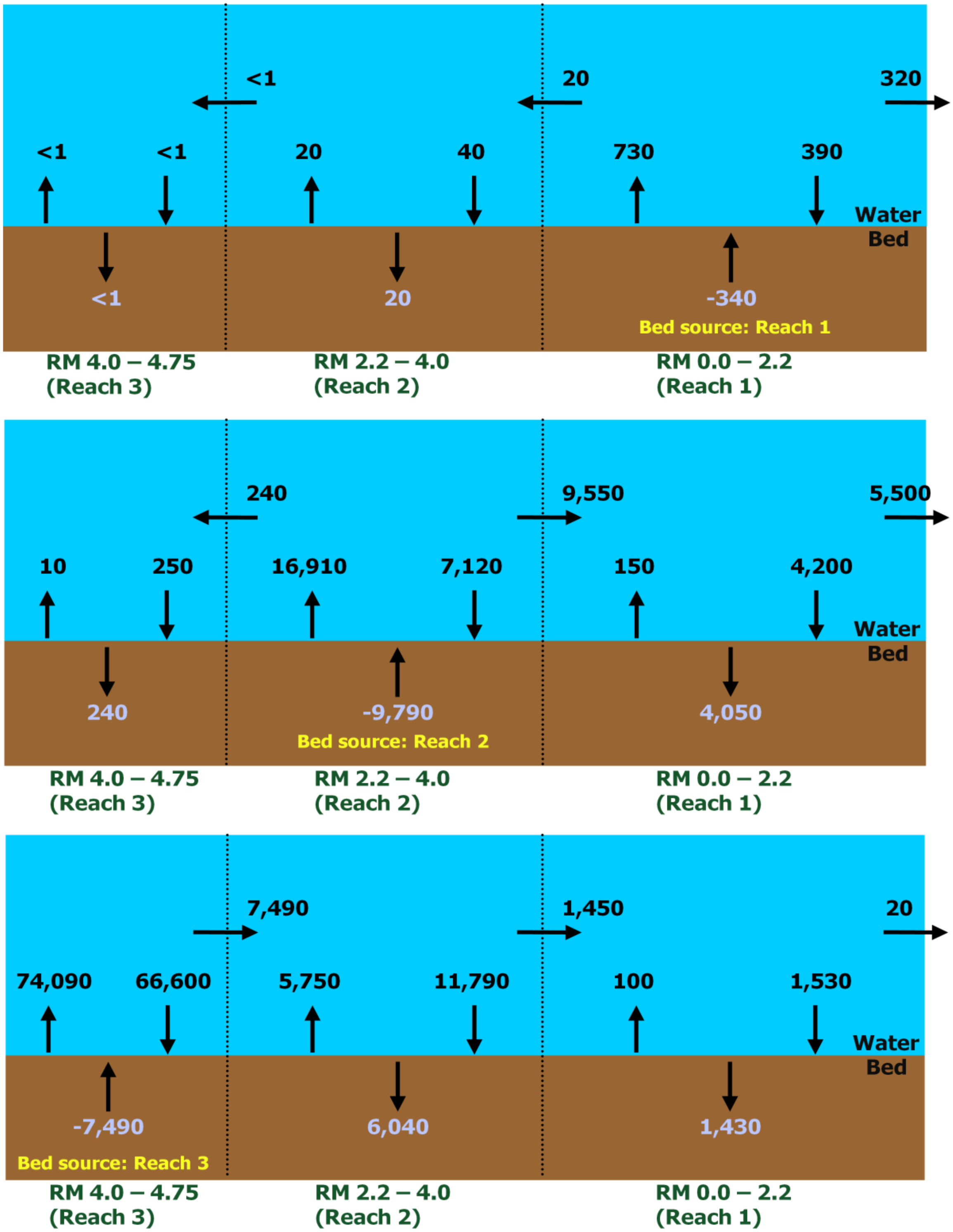
Figure 5-8
Estimated percentage of surface (0-10 cm) sediments resuspended from RM 0.0 to 4.0 and re-deposited in other LDW areas at the end of 10-year simulation

January 2009

QEA Quantitative Environmental Analysis, LLC

FC/JC- \\Jill\D_drive\Jobs\RETltw\Documents\Scenario_Memo\RETldw_tech_memo_fig_3-4_FS_fig_5-11.mxd

Figure 5-9 Mass Balances for Bed Sediment Originating from Reaches 1, 2, and 3 for 10-year STM Simulation



Note:
Sediment mass units are in metric tons, rounded to the nearest 10 metric tons.

Figure 5-10a Total Sediment Mass Balance for 100-year High-flow Event Simulation

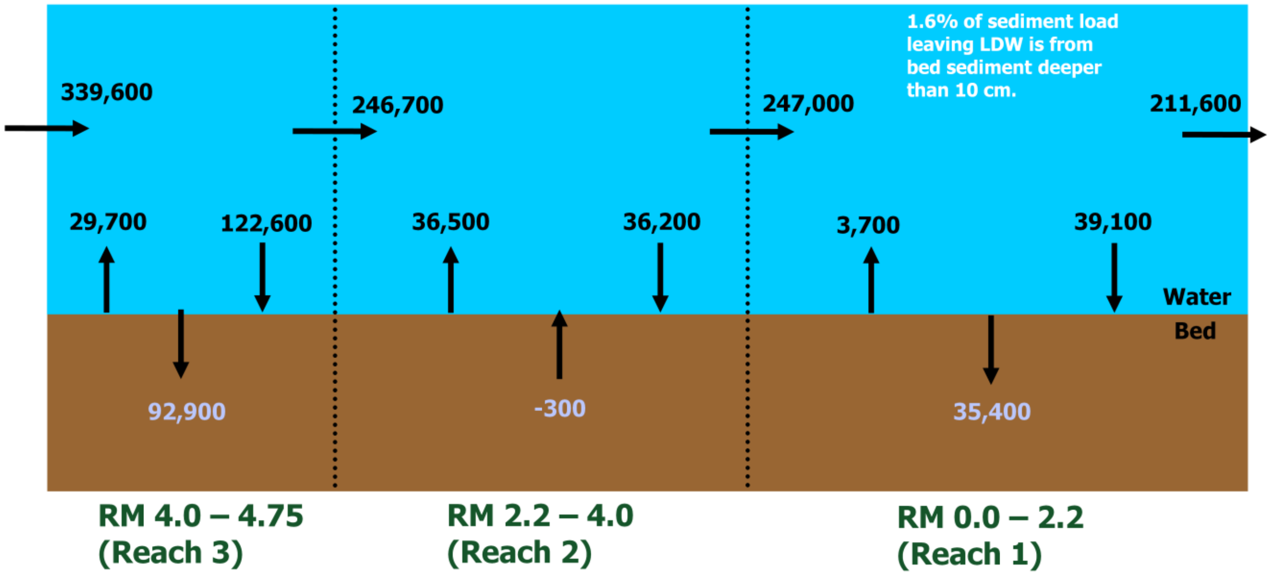
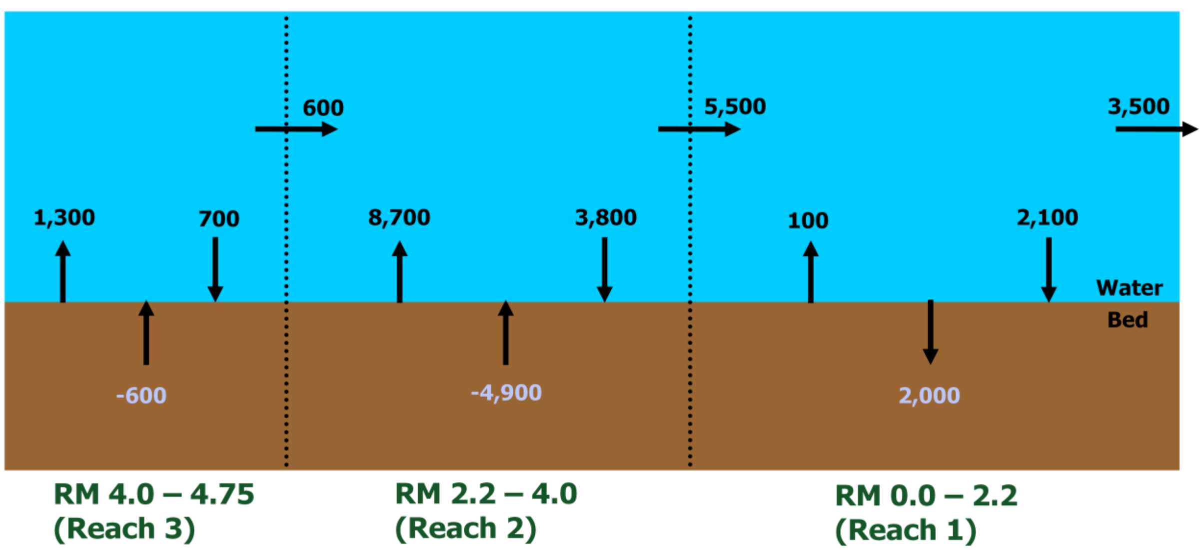
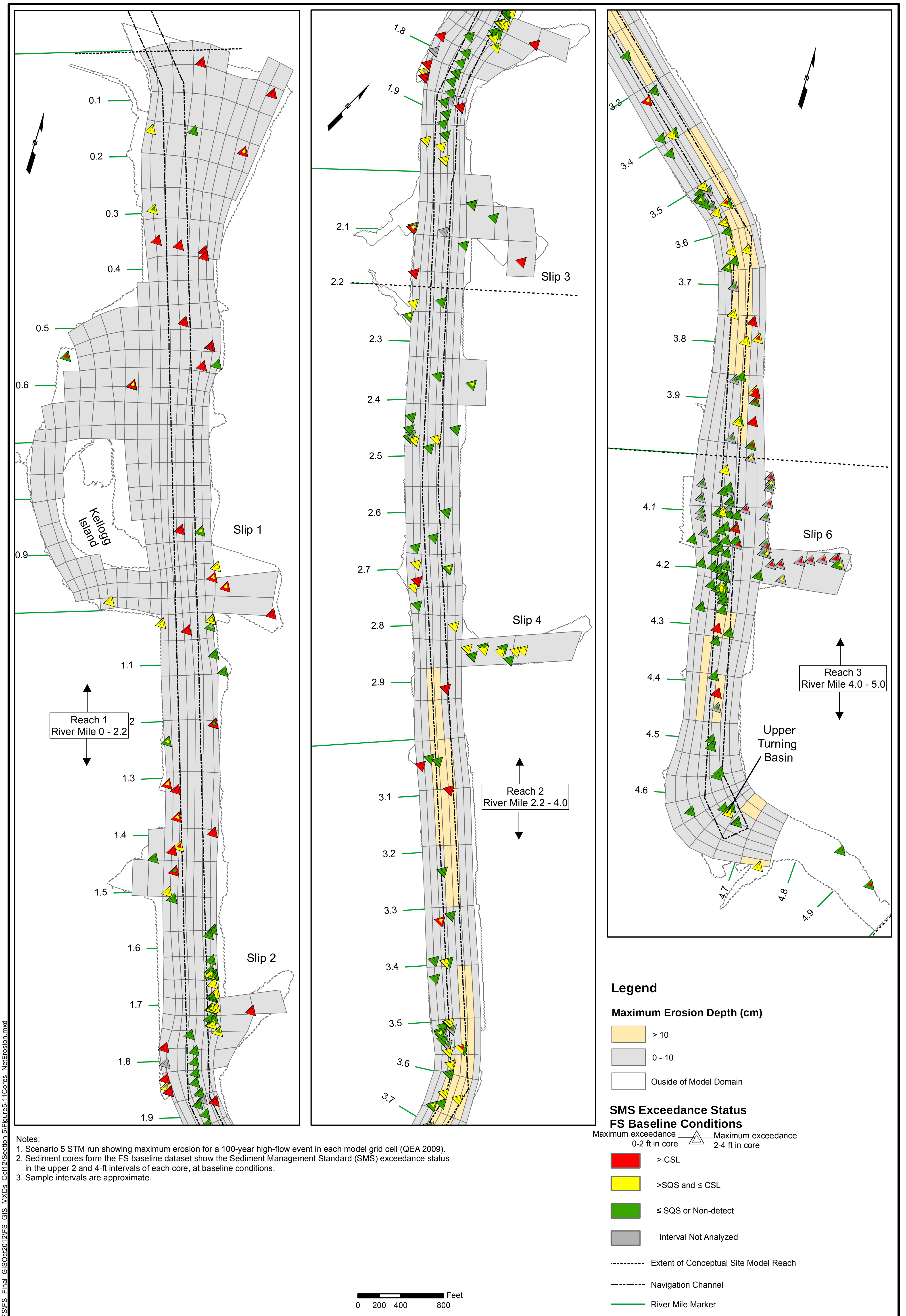


Figure 5-10b Mass Balance for Bed Sediment Originating from Deeper-than-10-cm Layer during 100-year High-flow Event Simulation





Notes:
1. Scenario 5 STM run showing maximum erosion for a 100-year high-flow event in each model grid cell (QEA 2009).
2. Sediment cores form the FS baseline dataset show the Sediment Management Standard (SMS) exceedance status in the upper 2 and 4-ft intervals of each core, at baseline conditions.
3. Sample intervals are approximate.

L:\Lower Duwamish FS\FS_Final_GIS\Oct2012\FS_GIS_MXDs_Oct12\Section 5\Figure5-11Cores_NetErosion.mxd

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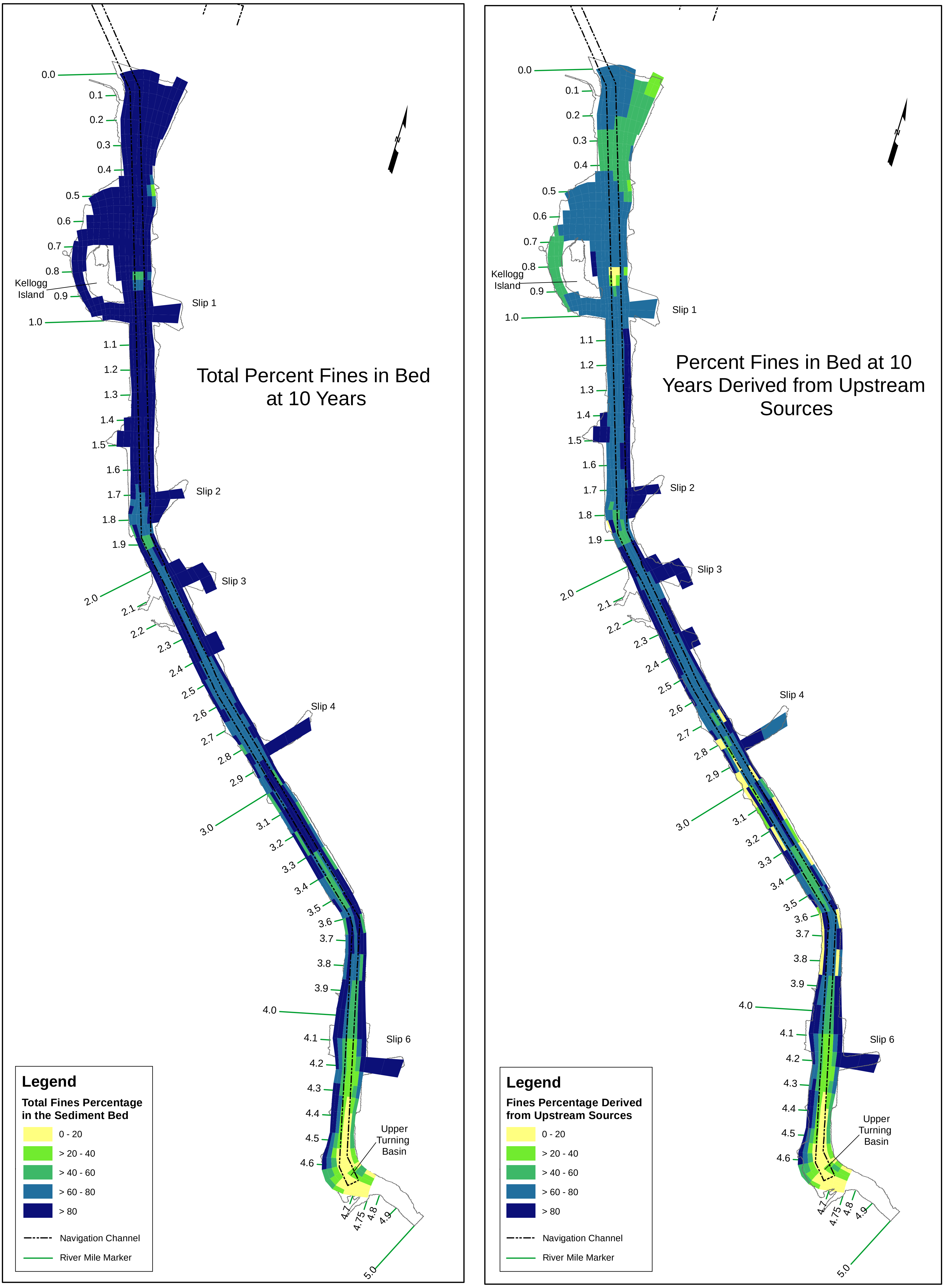
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49	Subsurface Sediment SMS Exceedance Locations with > 10 cm Erosion Depth During 100-year High-flow Event
DATE: 10/31/12 DWRN:MVI/sea Revision: 1	FIGURE 5-11



Notes:
1. 10-year STM GIS shapefile (QEA Feb. 2009).
2. Total fines represents the sum of fine-grained sediment from bed, lateral, and upstream sources in each grid cell at 10 years.
3. Upstream fines represents the percentage of fine-grained sediment from upstream sources in each grid cell at 10 years.
4. Fine-grained sediment is the sum of grain size classes 1A and 1B modeled in the STM. The fines are mostly class 1B, only about 10% of the class 1A material settles in the LDW.

Feet
0 750 1,500 3,000

AECOM

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			Estimated Distribution of Grain Size After 10 Years - Total Percent Fines and Fines Sourced from Upstream
DATE: 10/31/12	DWRN: MVI/sea	Revision: 1	FIGURE 5-12

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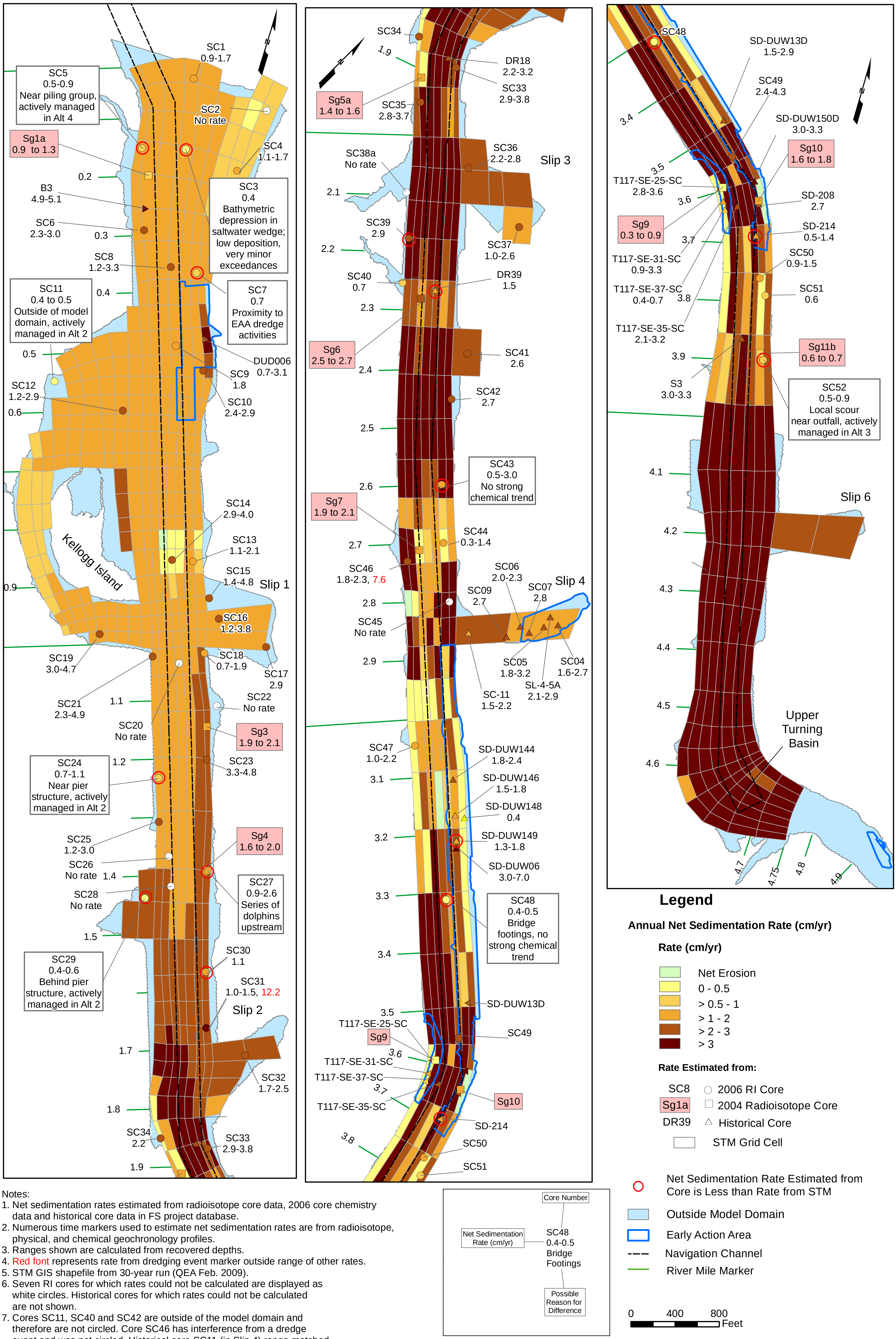

Notes:
1. Net sedimentation rates estimated from radioisotope core data, 2006 core chemistry data and historical core data in FS project database.
2. Numerous time markers used to estimate net sedimentation rates are from radioisotope, physical, and chemical geochronology profiles.
3. Ranges shown are calculated from recovered depths.
4. Red font represents rate from dredging event marker outside range of other rates.
5. STM GIS shapefile from 30-year run (QEA Feb. 2009).
6. Seven RI cores for which rates could not be calculated are displayed as white circles. Historical cores for which rates could not be calculated are not shown.
7. Cores SC11, SC40 and SC42 are outside of the model domain and therefore are not circled. Core SC46 has interference from a dredge event and was not circled. Historical core SC11 (in Slip 4) range matched model predictions. Core SC51 straddled two grid cells and therefore was not circled.

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Final Feasibility Study
60150279-14.49

Comparison of Net Sedimentation Rates Estimated from the STM and from Sediment Cores

DATE: 10/31/12 | DWN. BY: MVI/sea | FIGURE 5-13

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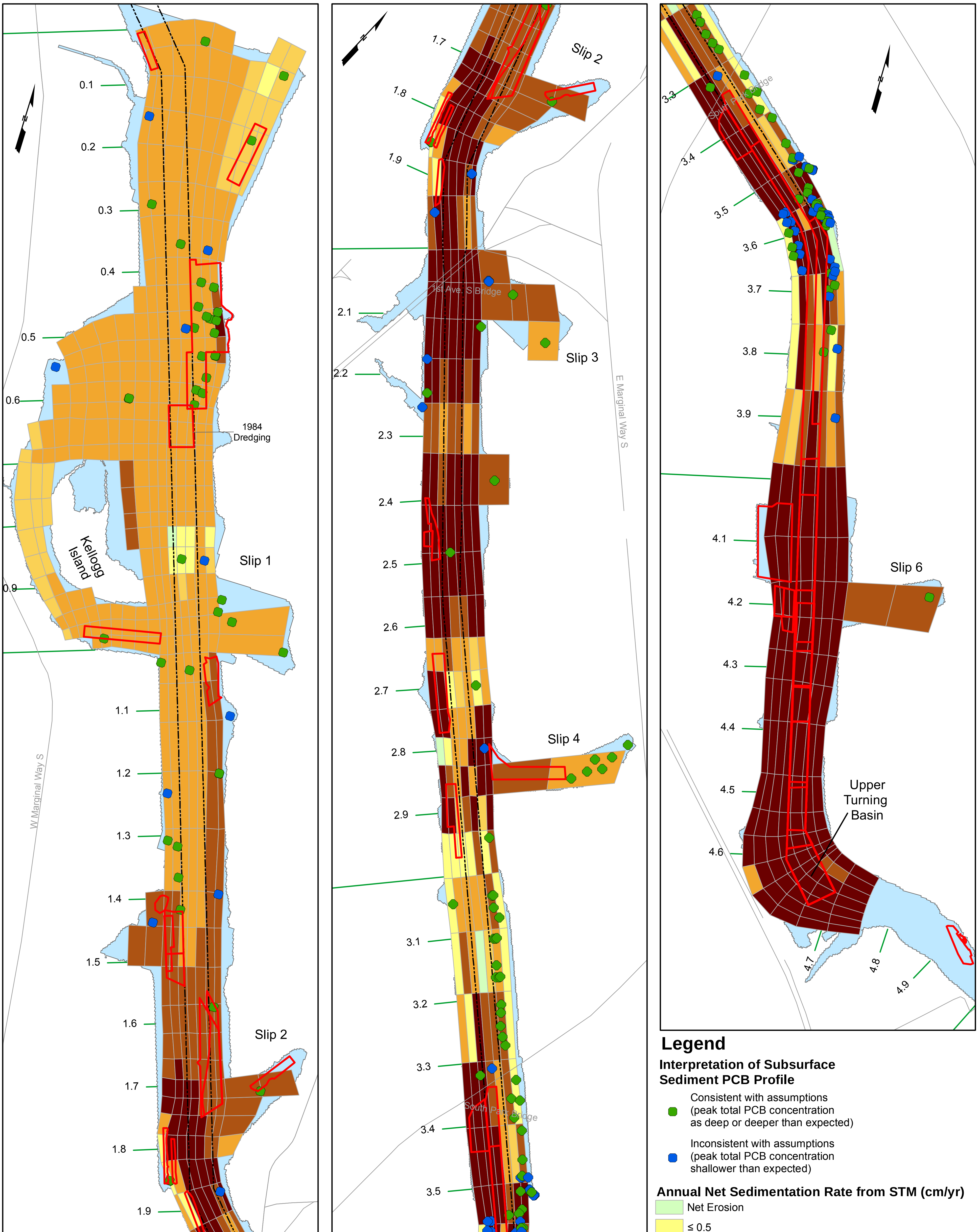

Legend

Interpretation of Subsurface Sediment PCB Profile

- Consistent with assumptions (peak total PCB concentration as deep or deeper than expected)
- Inconsistent with assumptions (peak total PCB concentration shallower than expected)

Annual Net Sedimentation Rate from STM (cm/yr)

- Net Erosion
- ≤ 0.5
- > 0.5 - 1
- > 1 - 2
- > 2 - 3
- > 3
- Outside of Model Domain
- Dredging Event within Last 30 Years
- Road
- Navigation Channel
- River Mile Marker

Notes:

1. Interpretation of total PCB profile from Final RI (Windward 2010). Interpretation identifies whether PCB peak assigned to 1965 is at a depth consistent with the net sedimentation rate from the STM annualized from a 30-year run.
2. Peak total PCB concentration at depth identified where concentration is at least two times concentration in surface interval.
3. 30-year STM GIS shapefile (QEA Feb. 2009).

L:\Lower Duwamish FS\FS_Final_GISOct2012\FS_GIS_MXDs_Oct12\Section 5\Figure5-14SedratesSubsurfEmpirical.mxd

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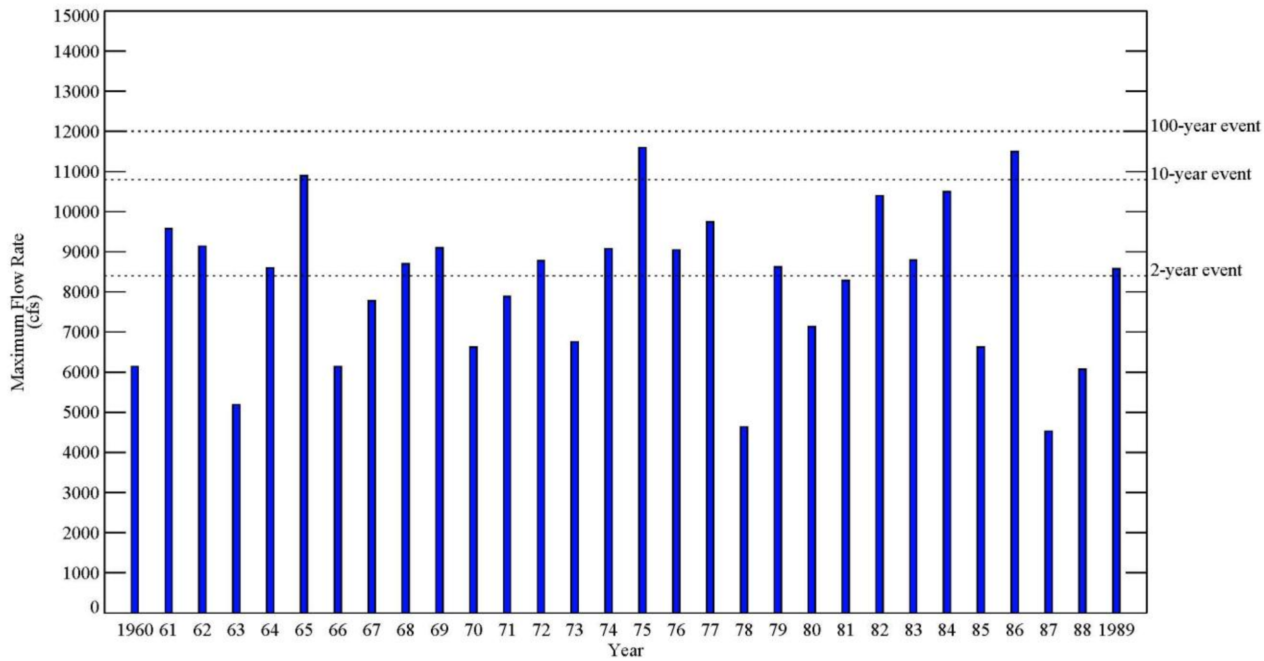
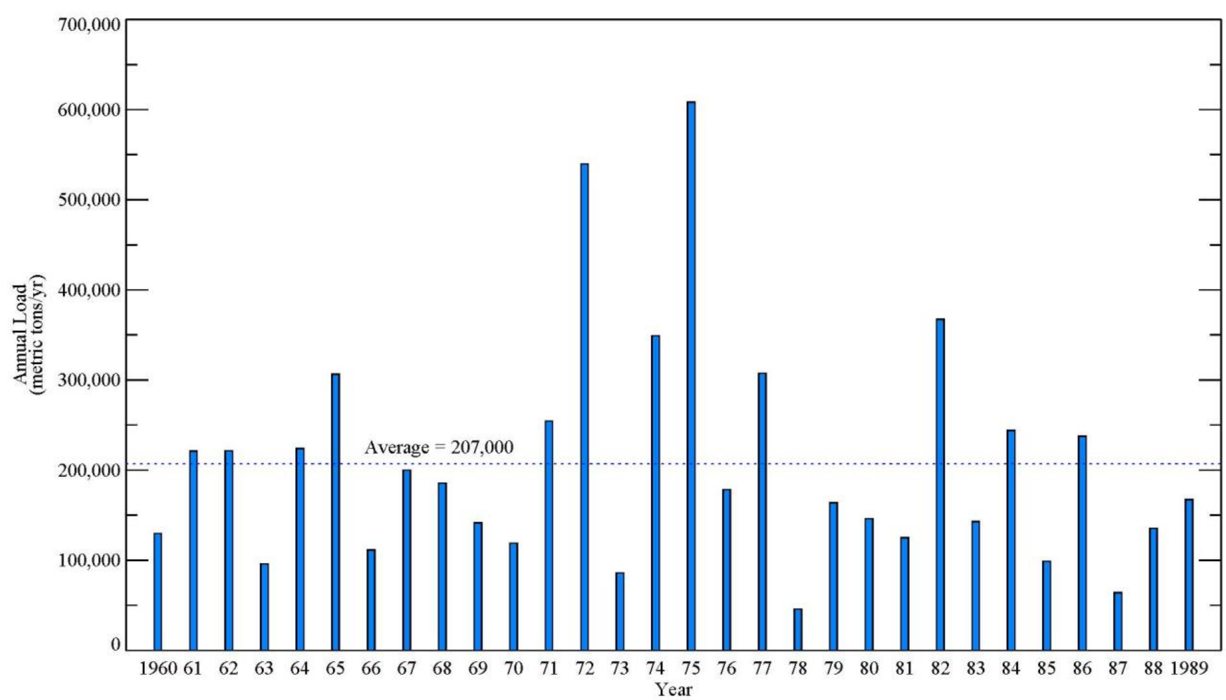
Lower Duwamish Waterway Final Feasibility Study 60150279-14.49			Annual Net Sedimentation Rates and Empirical PCB Trends in Subsurface Core Data
DATE: 10/31/12	DWRN:MVI/sea	Revision: 0	FIGURE 5-14

Figure 5-15a Maximum Flow Rate during Each Year from 1960 to 1989



Flow data: Fresh Water Discharge at USGS 12113000 (Green River).

Figure 5-15b Estimated Annual Total Sediment Load (suspended and bed load) in the Green River from 1960 through 1989



Note: 207,000 metric tons (MT) is annual average sediment load over 30-yr period. The annual average sediment load over the first 10 years (1960 – 1969) is 185,000 MT.

6 Areas of Potential Concern, Remedial Action Levels, and Recovery Potential

This section defines the areas of potential concern (AOPCs) with potentially unacceptable risks based on the findings of the baseline ecological and human health risk assessments (ERA and HHRA; Windward 2007a, 2007b). This section also presents the remedial action levels (RALs) designed to address these risks and used in developing the remedial alternatives. Lastly, this section presents categories of recovery potential for sediments in the Lower Duwamish Waterway (LDW) based on physical conditions and empirical trends in contaminant concentrations.

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Model Toxics Control Act (MTCA) require a feasibility study (FS) to identify volumes and areas of sediment where remedial action may be necessary and applied. Defining these areas requires

> *"…careful judgment and should include a consideration of not only acceptable exposure levels and exposure routes, but also site conditions and the nature and extent of contamination"* (EPA 1988).

Following U.S. Environmental Protection Agency (EPA) guidance (1988, 2005b), this section describes the relationship between location, extent, and concentrations of risk drivers relative to both hot spot areas and areas of lower level contamination. This information is used to delineate areas of sediment with potentially unacceptable risks. These areas are carried forward to Section 8, where technologies are assigned and remedial alternatives are developed. Further, the extent to which natural recovery is potentially viable is evaluated to guide the application of active and passive remedial actions in Section 8.

Hence, consistent with guidance, the steps in the FS process for mapping cleanup areas at the LDW include:

- Delineate AOPCs based on findings of unacceptable risks in the ERA and HHRA (Windward 2007a, 2007b). These areas will require consideration in this FS, and they are described in Section 6.1.
- Define a range of RALs that achieve or make progress toward achieving preliminary remediation goals (PRGs). RALs are contaminant-specific sediment concentrations that trigger the need for active remediation (e.g., dredging or capping). A RAL is equivalent to a "remediation level" under MTCA, which is defined as "…a concentration (or other method of identification) of a hazardous substance in soil, water, air, or sediment, above which a particular cleanup action component will be required as part of a cleanup action at a site" (Washington Administrative Code [WAC] 173-

340-200). A range of RALs, which trigger active remediation, is identified in Section 6.2. The remedial action objectives (RAOs; see Section 4) can be achieved through combinations of active remediation (triggered by the RALs), natural recovery, and institutional controls.

- Define areas within the AOPCs that have similar physical characteristics, engineering considerations, and recovery potential for which particular remedial technologies may be applied. These areas are referred to as recovery categories, which are discussed in Section 6.3.

Collectively, these evaluations are used in the assembly of the remedial alternatives in Section 8. Combinations of active and passive management of the AOPCs are evaluated relative to the RAOs. The AOPC boundaries and the recovery potential within those boundaries will likely need to be refined during remedial design and even, perhaps, during implementation of the remedy.

6.1 Delineating the Areas of Potential Concern (AOPCs)

The AOPCs represent the areas of sediment that have potentially unacceptable risks and will likely require application of active or passive remedial technologies. Defining the AOPC footprints requires: 1) an understanding of the types and levels of estimated risks in the LDW (see Section 3); 2) the RAOs to address those risks and associated PRGs (see Section 4); and 3) the conceptual site model, site conditions, and the data collection and analysis efforts over the past 20 years (see Section 2). The AOPC footprints defined for this FS are discussed in this section, along with a summary of the considerations used in deriving and evaluating these AOPCs. The contaminant concentrations used to develop the AOPC footprints include detected FS baseline surface sediment concentrations of risk drivers above the thresholds described below. The data used to define the AOPCs also include toxicity data and subsurface sediment data, when available (see Section 2).

The AOPCs do not include the five early action areas (EAAs; 29 acres), which are being addressed separately. However, the enhanced natural recovery (ENR) portion of the Duwamish/Diagonal EAA is included in AOPC 1. Evaluations used to define the AOPCs assume cleanup of the five EAAs will be completed prior to cleanup within the AOPCs. The two AOPC footprints developed for this FS are shown in Figure 6-1 and are described below.

Multiple thresholds were developed for each risk driver, and sediment areas were included in the AOPCs if any of the thresholds were exceeded. AOPCs are normally delineated by concentrations of contaminants of concern (COCs) or risk drivers above PRGs. For the LDW, the PRGs for total polychlorinated biphenyls (PCBs) and dioxins/furans (RAO 1) and for arsenic (RAO 2) are set at natural background for final cleanups, as required by MTCA. Model predictions indicate that natural background for these three risk drivers is unlikely to be achieved because of the concentrations of these risk drivers in incoming Green/Duwamish River suspended solids and because of practical limitations on control of lateral sources from the generally urban LDW drainage basin. For these reasons, it was not possible to use the RAO 1 PRGs for total PCBs or dioxins/furans or the RAO 2 PRG for arsenic to develop the AOPCs. Thus, a modified objective of getting those three risk-driver concentrations as close as possible to the natural background values (i.e., as low as practicable) was used to delineate the AOPCs. For the purposes of the FS, this is assumed to be the long-term model-predicted concentrations. These concentrations are believed to be the lowest technically achievable concentrations based on the available data and analyses conducted to date. These long-term model-predicted concentrations are uncertain, because future risk-driver concentrations in upstream- and lateral-source sediments are uncertain and may change in the future. The term "cleanup objective" in this FS is used to mean the PRG or as close as practicable to the PRG where the PRG is not predicted to be achievable. This FS uses long-term model-predicted concentrations as estimates of "as close as practicable to PRGs".[1] AOPC 1 was designed to achieve this objective using a combination of active cleanup and natural recovery, and AOPC 2 was designed to achieve this objective using only active cleanup.

6.1.1 AOPC 1 Footprint

As noted above, natural background is unlikely to be achieved, and both the sediment transport model (STM) and bed composition model (BCM) predict that, in the long term, the LDW will reach concentrations similar to those incoming from the upstream Green/Duwamish River system. For these reasons, the FS has adopted an incremental approach to delineate AOPCs and to develop remedial alternatives with varying degrees of active remediation and natural recovery.

The AOPC 1 footprint is based on the PRGs that are not set at natural background (i.e., the PRGs associated with RAO 2 for risk drivers other than arsenic and with RAOs 3 and 4). Natural recovery is assumed to be required following active remediation of the AOPC 1 footprint to reduce site-wide average total PCB, dioxin/furan, and arsenic concentrations to the cleanup objective as defined above.

Interpolated surface sediment concentration maps for total PCBs, arsenic, carcinogenic polycyclic aromatic hydrocarbons (cPAHs), dioxins/furans, and contaminants that exceed the Sediment Management Standards (SMS) were the primary sources of information used to delineate the AOPC 1 footprint. In addition, shallow subsurface sediment contaminant concentrations were considered in areas prone to scour and disturbance and in intertidal areas where the point of compliance for human health direct contact risk drivers (PCBs, arsenic, cPAHs, and dioxins/furans) is the upper 45 cm of sediment. As described in Section 2, inverse distance weighting (IDW) was used for interpolating total PCBs, arsenic, and cPAHs, and Thiessen polygons were

[1] For further information on cleanup objectives, see Section 9.1.2.3.

used to interpolate dioxins/furans and SMS exceedances in surface sediment. Each data layer was mapped independently. AOPC 1 was delineated where any of the layers exceeded the threshold concentrations described below.

RAO 3. AOPC 1 was first delineated for benthic community risk drivers with detected concentrations in surface sediments exceeding the sediment quality standards (SQS) (the RAO 3 PRGs). Each Thiessen polygon was classified as an SQS exceedance if one or more detected SMS contaminants exceeded this criterion. In addition, cleanup screening level (CSL) exceedances are also shown to indicate more highly contaminated areas. Toxicity test results, if available, were used in the final classification. If the Thiessen polygon exceeded the SQS, it was included in AOPC 1. Because total PCBs were spatially interpolated as dry weight concentrations (see Section 2 and Appendix A), the area with total PCB concentrations greater than 240 micrograms per kilogram dry weight (µg/kg dw; the dry weight equivalent of the 12 milligrams per kilogram organic carbon [mg/kg oc] SQS value, assuming 2% total organic carbon [TOC]) derived with IDW rather than Thiessen polygons was also used to delineate AOPC 1. Best professional judgment was used for mapping in cases where the total PCB IDW-based layer resulted in small, isolated areas exceeding 240 µg/kg dw. These small areas were not included in AOPC 1 if, using the sample-specific TOC data, they did not exceed the SQS on an organic-carbon normalized basis.

RAO 2. The AOPC 1 footprint was then evaluated for compliance with RAO 2. Active remediation of the AOPC 1 footprint achieves the total PCB PRGs (1,300 µg/kg dw for netfishing site-wide; 1,700 µg/kg dw for beach play areas; and 500 µg/kg dw for clamming areas). The footprint was expanded to achieve human health direct contact PRGs on a SWAC basis for cPAHs and dioxins/furans (380 µg toxic equivalent [TEQ]/kg dw and 37 nanograms [ng] TEQ/kg dw for netfishing [site-wide]; 90 µg TEQ/kg dw and 28 ng TEQ/kg dw for beach play; and 150 µg TEQ/kg dw and 13 ng TEQ/kg dw for clamming, respectively). The RAO 2 PRGs for arsenic are natural background over all three exposure areas (netfishing, clamming, and beach play), and therefore these PRGs are not likely to be achieved based on the model predictions. The AOPC 1 footprint was expanded to achieve site-wide and area-wide arsenic SWACs within the limits of what the long-term model predicts is achievable over time when natural recovery across the entire LDW is included. Also, to address beach play PRGs (RAO 2), individual beaches were included in AOPC 1 whenever the total direct contact excess cancer risks based on the beach play RME scenario (for all four human health risk drivers) exceeded 1×10^{-5}.

In intertidal areas, the point of compliance for human health risk drivers for clamming and beach play is assumed to be the upper 45 cm of sediment, because of potential exposures to people through direct contact with sediments during clamming or beach play activities. Average sediment concentrations from this interval[2] were considered and compared to the PRGs for direct contact tribal clamming and beach play RME scenarios. However this did not affect the designation of the AOPC footprint because the existing footprint covered these areas.

RAO 4. Active remediation of the AOPC 1 footprint achieves a site-wide spatially-weighted average concentration (SWAC) for total PCBs less than the RAO 4 PRG range of 128 to 159 µg/kg dw, and therefore no adjustment to AOPC 1 was required to meet RAO 4.

RAO 1. The AOPC 1 footprint was evaluated for compliance with RAO 1 PRGs, which are natural background concentrations on a site-wide basis for total PCBs and dioxins/furans. The footprint was not expanded for RAO 1. The FS assumes that remediation of AOPC 1 makes progress toward RAO 1 goals by achieving the long-term model-predicted sediment concentrations for total PCBs and dioxins/furans over time. Neither arsenic nor cPAHs have seafood consumption PRGs[3] for RAO 1, but remediation of AOPC 1 also reduces sediment concentrations for these risk drivers. Refer to Section 9 for predicted outcomes of the remedial alternatives.

Subsurface Contamination in Potential Scour Areas. Lastly, subsurface contamination was considered in the delineation of AOPC 1. Areas with SQS exceedances in the top 2 ft of sediment that are potentially subject to 100-year high-flow scour deeper than 10 centimeters (cm; as predicted by the STM; see Figure 2-9) or that are subject to vessel scour (see Figure 2-10) were added to the AOPC 1 footprint. In an area with an SQS exceedance in the top 2 ft of a core, the spatial extent was defined by the extent of the predicted high-flow scour area or the potential vessel scour area around that core. The spatial extent of the SQS exceedance within potential scour areas was conservatively assumed to be the entire extent of the potential scour area if there was only one core within that area (in part because there are relatively few subsurface sediment cores compared with surface sediment samples). If more than one core was located in a scour area, the spatial extent of the RAL exceedance was governed by the nearest core.

Summary. Table 6-1 lists the lowest risk-driver concentrations identified in surface sediment that were used to delineate AOPC 1 and the estimated post-construction

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2 Sediment data used to evaluate this interval included the following: surface sediment grabs in the top 10-cm, which were assumed to represent the top 45-cm; 0 to 45-cm depth samples in beaches; and where available the top 6-in or 1-ft core interval from subsurface sediment cores in intertidal areas.

3 Based on data collected during the RI, relationships between clam tissue and surface sediment concentrations of arsenic and cPAHs were too uncertain to develop quantitative risk-based threshold concentrations in sediment; therefore, no seafood consumption (RAO 1) PRGs were developed for these risk drivers.

SWACs if the entire AOPC 1 footprint was actively remediated.[4] It also compares those SWACs to the PRGs.

In summary, outside of the EAAs, the considerations used to delineate AOPC 1 were:

- Surface sediments with:
  - Areas delineated by Thiessen polygons that exceed the SQS criteria detected in surface sediment. Sediment toxicity data override chemical SQS or CSL exceedances and chemical passes, as described in Section 2.
  - Total PCB concentrations greater than 240 µg/kg dw
  - Arsenic concentrations greater than 57 mg/kg dw
  - cPAH concentrations greater than 1,000 µg TEQ/kg dw
  - Dioxin/furan concentrations greater than 25 ng TEQ/kg dw
  - Arsenic concentrations greater than 28 mg/kg dw in intertidal areas
  - cPAH concentrations greater than 900 µg TEQ/kg dw in intertidal areas.
- Areas with SQS exceedances in the top 2 ft of subsurface sediment that are predicted to be subject to 100-year high-flow scour deeper than 10 cm or are potentially subject to vessel scour based on empirical evidence.

AOPC 1 represents the maximum extent of any exceedance delineated by the layers described above. Therefore, the AOPC 1 footprint is larger than the area defined by the concentration for any one risk driver. Overall, the AOPC 1 footprint (Figure 6-1) represents about 180 acres or about 41% of the entire LDW site (441 acres).

The AOPC 1 footprint encompasses the initial area designated in the FS for remedial alternative development. Cleanup of the EAAs and all of AOPC 1, through a combination of active cleanup, verification monitoring, and natural recovery, is predicted to achieve cleanup objectives for RAOs 1, 2, 3, and 4. PRGs based on natural background for RAO 1 (total PCBs and dioxins/furans) and for RAO 2 (arsenic) are not predicted to be technically practicable, and thus, the cleanup objectives are to achieve long-term model-predicted concentrations that are as close to natural background as technically practicable.

6.1.2 AOPC 2 Footprint

In addition to AOPC 1 shown on Figure 6-1, EPA and the Washington State Department of Ecology (Ecology) required that an incrementally larger remedial footprint (outside of AOPC 1) be evaluated, called AOPC 2. The goal for final cleanup is to achieve

[4] The resulting SWACs were calculated by replacing the risk-driver concentrations in the AOPC 1 footprint with a post-remedy bed sediment replacement value, which is provided in Table 6-1. The SWACs do not assume any natural recovery.

concentrations as close to the natural background concentrations as technically practicable for total PCBs and dioxins/furans (RAO 1) and arsenic (RAO 2). Natural background for these three risk drivers is unlikely to be achieved because of incoming contaminant concentrations from the Green/Duwamish River and practical limitations on control of lateral sources. Instead, AOPC 2, when actively remediated along with AOPC 1, achieves the lowest long-term model-predicted SWACs for total PCBs, dioxins/furans, and arsenic[5] immediately after construction. AOPC 2 also addresses all areas outside AOPC 1 with subsurface contamination above the SQS. The AOPC 2 footprint is 122 acres. The AOPC 1 and AOPC 2 footprints combined encompass 302 acres (or approximately 68% of the LDW study area).

The AOPC 2 footprint was explored through a step-wise evaluation in which active remediation was first assumed for AOPC 1 plus every point with a total PCB concentration above 100 µg/kg dw. Second, site-wide SWACs for dioxins/furans and arsenic were calculated by changing the surface sediment concentrations in this larger footprint to the post-remedy bed sediment replacement values[6] and assuming no natural recovery. Based on these SWACs, the AOPC 2 footprint was then expanded to capture areas with:

- Arsenic concentrations greater than 15 mg/kg dw to achieve the long-term model-predicted site-wide SWAC.
- Dioxin/furan concentrations greater than 15 ng TEQ/kg dw to achieve the long-term model-predicted site-wide SWAC.

Finally, the footprint was again expanded to include remaining sediment cores with detected SQS exceedances at any depth (regardless of scour potential).[7]

The results of this analysis indicated that active remediation of AOPCs 1 and 2, using the post-remedy bed sediment replacement values for total PCBs, arsenic, and dioxins/furans, yields site-wide SWACs within the range of the long-term model-predicted concentrations (Table 6-1) immediately after construction. This analysis indicates:

1) Active remediation of the entire 302-acre AOPC 1 and AOPC 2 footprints would result in the lowest long-term model-predicted concentrations, and

[5] A cPAH threshold was not needed for AOPC 2 delineation because all areas where remediation is needed to meet cPAH PRGs are included in AOPC 1.

[6] Post-remedy bed sediment replacement values in AOPCs 1 and 2 (respectively) for each risk driver are: total PCBs = 60 and 20 µg/kg dw; arsenic = 10 and 9 mg/kg dw; and dioxins/furans = 4 ng TEQ/kg dw (mid-range and low values, respectively, from Table 5-1c).

[7] The exception is three cores collected from the Upper Turning Basin in 2009. Sediment in this area had not exceeded the SQS in previous samples, and data were not received in time to include in the AOPC 2 delineation. The sediment represented by these cores was dredged in 2010.

the model predicts that further changes over time after the cleanup through natural recovery would be minimal.

2) Any further active remediation would not yield additional sustainable SWAC reduction or risk reduction, because sediments from upstream and lateral sources would continue to deposit onto remediated areas.

It is important to recognize that, as with other input parameters, values used as post-remedy bed sediment replacement values for this analysis are uncertain. A range of replacement values was developed for each human health risk driver in this FS. The sensitivity of post-remedy sediment concentration predictions to the range of replacement values is described in Section 9. Based on this analysis, active remediation of the AOPC 1 and 2 footprints is predicted to reach long-term model-predicted concentrations. Cleanup of the EAAs and active remediation of the AOPC 1 and 2 footprints is predicted to achieve the maximum technically practicable degree of SWAC risk reduction. The areas beyond the AOPCs are not considered for active cleanup in this FS (but may be subject to sampling and verification monitoring during remedial design).

In summary, active remediation of AOPC 1 achieves the PRGs for RAOs 2 (for all human health risk drivers except arsenic), 3, and 4. The combined footprint of AOPCs 1 and 2 results in the lowest model-predicted SWACs for RAO 1 (total PCBs and dioxins/furans) and RAO 2 (arsenic) immediately after construction without consideration of natural recovery. Therefore, the AOPC 1 and 2 footprints are considered appropriate to identify alternatives that achieve the PRGs or make substantial risk reduction toward achieving the PRGs. The footprints have been defined with enough rigor to facilitate a detailed evaluation of remedial alternatives (in Section 8) for the purposes of this FS.

6.2 Remedial Action Levels

RALs are contaminant-specific sediment concentrations that trigger the need for active remediation (i.e., dredging, capping, or ENR). RALs define the active remediation footprint within the AOPCs for each remedial alternative (Section 8).

RALs are very different from PRGs. PRGs are the long-term cleanup levels and goals for the project, whereas RALs are point-based values that define where active remediation is to occur for a given alternative. PRGs are the same for all alternatives, whereas RALs vary among alternatives. RALs are also used as the compliance concentration to verify that active remediation for an alternative is complete, or successful, before equipment is demobilized from an area.

The development and use of RALs for this FS is based on the premise that once active remediation is complete (in areas where the RALs are exceeded), SWACs for human health risk drivers immediately following construction will be considerably lower than those for baseline conditions. The cleanup objectives are achieved either immediately after construction or over time through natural recovery. Higher RALs are associated with higher post-construction SWACs and larger areas that rely on natural recovery to achieve cleanup objectives. The evaluations of risk reduction over time and the time to achieve cleanup objectives are presented in Section 9.

For this FS, ranges of RALs are developed for the risk drivers (total PCBs, arsenic, cPAHs, dioxins/furans, and SMS contaminants [i.e., detected risk drivers that exceeded the SQS in surface sediments]) for which PRGs were presented in Section 4 (see Figures 6-2a through 6-2d for the human health risk drivers). RALs are developed with the understanding that remediation of these risk drivers will also address the remaining COCs (see Table 3-16) that do not have PRGs.

6.2.1 Methods Used for Development of RALs

This section briefly summarizes the methods used to develop a range of RALs that serve to define a range of active remedial footprints and a corresponding range of expected outcomes. The range of RALs allows a broad array of remedial alternatives to be defined in Section 8, each with differing:

- Areas/volumes of sediment to be actively remediated
- Levels of risk reduction immediately after construction
- Time frames for achieving cleanup objectives.

The residual risks remaining immediately after construction of each remedial alternative and additional risk reduction predicted over time through natural recovery are discussed in Sections 9 and 10 of this FS.

RAL development considers only individual COCs and does not consider the extent to which COCs are commingled. Because many of the LDW COCs have some commingling and co-occurrence, it is reasonable to expect that by remediating an area to address one risk driver exceeding a RAL, some reduction in other COCs will also occur. Thus, the remediation of sediments exceeding RALs may result in risk reduction not accounted for when only individual COCs are evaluated. Section 9.11 describes how the remedial alternatives address COCs other than the risk drivers. In addition, natural recovery is predicted to further reduce sediment concentrations over time below the reduction achieved by active remediation alone.

The approaches used to select RALs and to develop an array of remedial alternatives require best professional judgment. The RALs for this FS were selected based on the following considerations:

- **Achievement of PRGs.** Certain sediment PRGs can directly translate into RALs, such as SMS criteria applied on a point basis, which directly relate to protection of benthic receptors (RAO 3). RALs for RAO 3 were defined using two time points: at the end of construction and 10 years after construction, in accordance with SMS guidelines. Although not defined in the RAL development process, some RALs may require more than 10 years after construction to achieve PRGs. Area-based PRGs (SWACs) for certain direct contact scenarios (RAO 2) are the basis for point-based RALs for this FS.

- **Range of RALs**. By definition, the RALs are point concentrations that exceed PRGs and require active remediation. However, a direct comparison of point concentrations (at specific sample locations) to PRGs is not appropriate for RAO 1 (seafood consumption), RAO 2 (direct contact), and RAO 4 (wildlife consumption of prey) because these RAOs have SWAC-based PRGs. Therefore, each SWAC-based PRG needs to be "converted" to a not-to-exceed point concentration (RAL). To accomplish this "conversion" from SWACs to point concentrations, human health risk drivers were evaluated in an iterative fashion (called "hilltopping") by ranking their concentrations from highest to lowest (using interpolated grid cells). The highest values were sequentially replaced with a post-remedy bed sediment replacement value until the appropriate site- or area-wide PRG was achieved. The highest concentration remaining then becomes the RAL for the SWAC-based PRG.

  A range of RALs was selected for each human health risk driver by comparing the highest remaining concentration to the resulting SWAC. The RALs were selected to represent a range of acres remediated and the resulting SWACs. Figures 6-3a through 6-3d present the hilltopping curves for the four risk drivers. The RALs (point values) are identified on the curves relative to the estimated SWACs they achieve based only on active remediation and no natural recovery.

- **SWAC Reduction for PRGs Set at Natural Background.** Certain PRGs, such as those for total PCBs and dioxins/furans for RAO 1 and for arsenic for RAO 2, cannot be used directly as RALs because they are set to natural background (Table 4-7). It is not technically possible to implement a RAL set at natural background because although sediments continually entering the LDW from upstream have COC concentrations considerably lower than those in LDW sediments, these concentrations are still above natural background concentrations. For PRGs set at natural background, a range of RALs was selected to achieve the long-term model-predicted concentrations over time and immediately after construction.

  As incrementally lower RALs were considered and more acres were identified for active remediation, a point of minimal change in SWAC was predicted. The estimated curves, shown in Figures 6-3a through 6-3d,[8]

[8] Section 9 contains SWAC-over-time curves based on future site-wide SWACs predicted using the BCM.

approach a value (the asymptote) driven by continual upstream inputs from the Green/Duwamish River as well as urban inputs from lateral drainage to the LDW. The estimated rate of change (SWAC reduction per acre) is predicted to be so small that, immediately after construction, the site would be considered to have reached the lowest model-predicted post-construction SWAC. Through continued natural recovery over time, the site would reach the long-term model-predicted concentrations (shown as the asymptote on the curve). It is worth noting that predicted changes in the post-remedy SWACs (shown in Figures 6-3a through 6-3d) are largely driven by the post-remedy bed sediment replacement values, while the long-term model-predicted concentrations are largely dependent on concentrations associated with upstream sources and to a lesser extent, lateral sources (see Tables 5-1a through 5-1c).

6.2.2 Range of Selected RALs

The array of RALs and how they relate to each RAO are summarized in the following subsections and in Table 6-2.

6.2.2.1 RAO 1 (Human Health Seafood Consumption) RALs

For this FS, progress toward achievement of RAO 1 (reduction of human health risks from seafood consumption) is assessed based on estimated reductions in the site-wide SWAC of total PCBs, arsenic, cPAHs, and dioxins/furans. The RALs for each risk driver are described below.

The total PCB PRG for RAO 1 is not expected to be achieved because it is set at natural background. Therefore, the goal is to set an array of RALs that result in incrementally lower site-wide SWACs after construction and shorter model-predicted natural recovery periods to reach cleanup objectives. (However, at very low RALs, time to achieve cleanup objectives increases due to longer construction times.) A total PCB RAL of 2,200 µg/kg dw was selected to address hot spots. The remaining RALs of 1,300, 700, 240, and 100 µg/kg dw comprise a range resulting in incrementally larger areas of active remediation and corresponding reductions in the site-wide SWAC immediately after construction (Table 6-2). The SWAC reduction is in turn predicted to result in a commensurate incremental reduction in human health risks. A RAL of 1,300 µg/kg dw is based on the CSL.[9] A RAL of 700 µg/kg dw is based on providing a well-spaced range of RALs for evaluation. A RAL of 240 µg/kg dw is based on the SQS.[10] The lowest total PCB RAL (100 µg/kg dw) is predicted to yield minimal change in the average

[9] Assuming a TOC content of 2% (the site-wide average), the total PCB dry weight equivalent of the CSL (65 mg/kg oc) is 1,300 µg/kg dw. If selected, actual implementation of this RAL would be based on the organic carbon-normalized CSL.

[10] Assuming a TOC content of 2%, the total PCB dry weight equivalent of the SQS (12 mg/kg oc) is 240 µg/kg dw. If selected, actual implementation of this RAL would be based on the organic carbon-normalized SQS.

concentration immediately after construction, and to achieve the long-term model-predicted concentration range. As discussed in Section 6.1.2, further active remediation is not predicted to appreciably lower the site-wide SWAC for total PCBs.

For arsenic and cPAHs, 95% or more of the risk associated with seafood consumption is attributable to the consumption of clams. A relationship between the concentrations of arsenic and cPAHs in clam tissue and sediment would be required to estimate sediment risk-based threshold concentrations (RBTCs) for RAO 1. However, RI data showed a poor relationship between clam arsenic and cPAH concentrations and associated sediment concentrations (i.e., clam tissue-to-sediment relationships for both arsenic and cPAHs were too uncertain to develop quantitative sediment RBTCs). Because of this, neither arsenic nor cPAHs have seafood consumption PRGs. RALs were selected for each to provide for overall reductions in sediment concentrations of these two risk drivers. Co-occurrence with the other risk drivers will also reduce site-wide sediment concentrations. For arsenic, a RAL of 93 mg/kg dw (the CSL) is used to address hot spots, and two other RALs, 57 (the SQS) and 15 mg/kg dw, are used to provide a range. For cPAHs, a RAL of 5,500 µg TEQ/kg dw is used to address hot spots, and two other RALs, 3,800 and 1,000 µg TEQ/kg dw are used to provide a range.

The dioxin/furan PRG for RAO 1 is not expected to be achieved because it is set at natural background. Therefore, the goal is to set a range of RALs that result in incrementally lower site-wide SWACs following active remediation. A RAL of 50 ng TEQ/kg dw was selected to address hot spots. Other dioxin/furan RALs of 35, 25, and 15 ng TEQ/kg dw comprise the range resulting in incrementally larger areas of active remediation and corresponding reductions in the site-wide SWAC immediately after construction (Table 6-2). The lowest dioxin/furan RAL (15 ng TEQ/kg dw) is predicted to result in minimal change in the site-wide SWAC and to achieve the long-term model-predicted concentration immediately after construction is complete. Further active remediation is not predicted to appreciably lower the site-wide SWAC for dioxins/furans.

6.2.2.2 RAO 2 (Human Health Direct Contact) RALs

Achievement of RAO 2 is assessed on three spatial scales, based on the three direct contact exposure scenarios: site-wide for netfishing, area-wide within potential clamming areas, and area-wide within beach play areas. In addition, future-use scenarios for beach play are evaluated in all intertidal areas (see Figure 3-1).

Netfishing

The netfishing exposure area is site-wide (441 acres) and the point of compliance is surface sediment (0 to 10 cm). For total PCBs, cPAHs, and dioxins/furans, the netfishing site-wide PRGs are predicted to be achieved immediately following remediation of the EAAs. All arsenic direct contact PRGs are set to natural background; therefore, they are unlikely to be achieved. The goal is to achieve the long-term model-predicted concentration. An arsenic RAL of 93 mg/kg dw is used to address hot spots.

The remaining RALs (57 and 15 mg/kg dw) provide a range, with the lowest RAL set to achieve long-term model-predicted concentrations at the end of construction. Further active remediation is not predicted to appreciably lower the site-wide SWAC for arsenic.

Beach Play Areas

As described in Section 3, the LDW has eight beach play areas; note that these are not all necessarily areas where beach play currently occurs but they were identified as such because public access is possible. The beach play scenario is evaluated on an average basis at individual beaches and across all beaches combined (exposure areas). The point of compliance for the beach play scenario is 0 to 45 cm. Intertidal RALs were developed for arsenic, cPAHs, and dioxins/furans. For total PCBs, an intertidal RAL was not needed in these areas because the tribal clamming and beach play direct contact PRGs for total PCBs are predicted to be achieved following remediation of the EAAs and hot-spot areas.[11]

The PRGs for the beach play areas are the 10^{-6} RBTCs for the individual risk drivers (with the exception of arsenic where the PRG is set at natural background). Total PCB beach play PRGs are predicted to be achieved at all of the individual beach play areas using the highest RAL of 2,200 µg/kg dw. The PRG of natural background for arsenic is unlikely to be achieved. For cPAHs, the PRG falls within the range of upstream inputs and post-remedy bed sediment replacement values, and therefore may not be achieved at all beach play areas, although some of the individual beaches are predicted to achieve the PRG. A dioxin/furan intertidal RAL was set to the 10^{-6} RBTC for beach play.

The beach play RALs for both arsenic and cPAHs are set to the 10^{-5} RBTCs as points to ensure that, at a minimum, 1) the total 10^{-5} risk goals required by MTCA are achieved, and 2) progress is made toward achieving 10^{-6} RBTCs (or natural background for arsenic) on an average basis over the beaches. For arsenic, cPAHs, and dioxins/furans, RALs of 28 mg/kg dw (10^{-5} RBTC), 900 µg TEQ/kg dw (10^{-5} RBTC), and 28 ng TEQ/kg dw (10^{-6} RBTC), respectively, are applied in all intertidal areas, and hence, all potential current and future beach play areas.

Clamming Areas

The tribal clamming scenario is evaluated on an area-wide basis across the potential clamming exposure areas. The same point of compliance considerations that applied to beach play, as described above, also applied to clamming areas. The direct contact tribal clamming PRG for total PCBs is predicted to be achieved after the EAAs have been actively remediated (Figures 6-2a through 6-2d). An arsenic RAL of 93 mg/kg dw, applied on a point basis, is expected to achieve the tribal clamming 10^{-5} RBTC; the 10^{-6}

[11] In intertidal areas, compliance for total PCBs was evaluated based on surface sediment and limited to the 10 cm depth (the biologically active zone). The site-wide RAL for total PCBs (in the top 10 cm) achieved the cleanup objectives for direct contact clamming and beach play areas.

RBTC is below natural background. The lower arsenic RALs (57, 28, 15 mg/kg dw) are designed to achieve incrementally lower SWACs and the long-term model-predicted concentrations in potential clamming areas. The RALs discussed above for cPAHs and dioxins/furans in beach play areas are also predicted to result in SWACs that achieve the PRGs in clamming areas, so no RALs based on tribal clamming were set for these two risk drivers.

6.2.2.3 RAO 3 (Protection of Benthic Invertebrates) RALs

The RALs for any risk-driver SMS contaminant for RAO 3 are:

- **CSL10 –** achieves the CSL within 10 years after construction is complete. The locations exceeding the CSL within 10 years were predicted using the recommended BCM input parameters. The BCM methods are described in Section 5, and predicted outcomes are shown in Section 9 and Appendix F.
- **CSL –** achieves the CSL by the time construction is complete.
- **SQS10 –** achieves the SQS within 10 years after construction is complete. The locations exceeding the SQS within 10 years were predicted using the recommended BCM input parameters. The BCM methods are described in Section 5, and predicted outcomes are shown in Section 9 and Appendix F.
- **SQS –** achieves the SQS by the time construction is complete.

SMS criteria for total PCBs and the other non-polar organic compounds are on an oc-normalized basis. Total PCB RALs for RAO 3 are 12 and 65 mg/kg oc for the SQS and CSL, respectively, but may be expressed as dry weight values in the FS for mapping purposes and ease of discussion (240 and 1,300 µg/kg dw for SQS and CSL, respectively, assuming 2% TOC). The SMS criteria for metals are expressed on a dry weight basis. For arsenic they are 57 and 93 mg/kg dw, for the SQS and CSL, respectively.

Implementation of the time-dependent RALs (SQS10 and CSL10) requires prediction of location-specific future concentrations using the BCM (methods are described in Section 5, and predicted outcomes are presented in Section 9 and Appendix F).

6.2.2.4 RAO 4 (Ecological Receptor Seafood Consumption) RALs

For RAO 4, total PCBs is the only risk driver. Achievement of the PRG (hazard quotient less than 1.0) is assessed on a site-wide basis. Separate RALs were not defined for RAO 4 because the total PCB range of RALs described above for RAO 1 (2,200, 1,300, 700, 240, and 100 µg/kg dw) is predicted to achieve RAO 4 immediately after construction or through a combination of active remediation and natural recovery.

6.3 Evaluating Recovery Potential of Sediments within the AOPCs

This section presents an evaluation of recovery potential intended to guide the final assembly of remedial alternatives (Section 8) within the AOPCs (outside of EAAs) and to prioritize areas that will likely require active remediation. This evaluation considers several factors, including proximity to potential contaminant sources, net sedimentation rates, scour potential, and empirical trends, that affect the ability of areas to recover through natural processes.[12]

The entire LDW was grouped into three categories with regard to recovery potential (Figures 6-4a and 6-4b). A recovery category represents areas of the LDW that share similar characteristics that could affect how well different remedial technologies would achieve the RAOs and how feasible they would be to implement. The recovery categories are:

- **Category 1** includes areas where recovery is presumed to be limited. It includes areas with observed and predicted scour, net scour, and empirical data demonstrating increasing concentrations over time.
- **Category 2** includes areas where recovery is less certain. It includes areas with net sedimentation and mixed empirical contaminant trends.
- **Category 3** includes areas where recovery is predicted. It includes areas with minimal to no scour potential, net sedimentation, and empirical trends of decreasing concentrations.

6.3.1 Mapping the Lines of Evidence for Evaluating Recovery Potential

To delineate the areas in each of these recovery categories, the following physical and chemical lines of evidence were considered (Table 6-3):

- Scour and deposition patterns:
  - Annual net sedimentation rates estimated by the STM and averaged over the 30-year STM period
  - 100-year high-flow event scour areas predicted in the STM (maximum scour depth observed over the 30-year model period)
  - Areas with empirical evidence of vessel scour, as interpreted from 2003 bathymetric survey sun-illumination maps.
- Land and water use functions:

[12] When reviewing empirical trends, proximity to contaminant sources, depth of contamination, and type of contaminant exceedance were considered. When source control is complete, recovery may be viable but not yet observed empirically.

- Berthing areas, former dredging events, and potential for disturbance by future dredging
- Proximity to the toe of the slope along the navigation channel
- Shoreline land use, public access, and outfall locations
- Overwater structures
- Vessel traffic patterns, based on knowledge of navigational operations, operator interviews, and bridge opening logs
- Habitat restoration areas, recreational shoreline access areas, and historical cleanup areas.

- Empirical evidence of recovery through total PCB and other risk-driver concentration trends (excluding dioxins/furans) in:
  - Surface sediment from resampled stations
  - Subsurface sediment from the top two intervals (the shallowest 2 ft) of cores.

Table 6-3 lists the key lines of evidence and the specific criteria used to delineate each recovery category, which are discussed below. The GIS maps showing the extent of these features are presented in Section 2 and Appendix F. Other bulleted items (not listed in Table 6-3) were secondary considerations used as lines of evidence to help interpret and evaluate empirical trends and to delineate the layers in Table 6-3. For example, overwater structures and former dredging events were used to define active berthing areas. The following subsections describe how these features were overlaid to map recovery category areas. Recovery categories are defined only for the purposes of developing site-wide remedial alternatives and assigning remedial technologies (Section 8). Location-specific design considerations and new empirical data for these areas will be evaluated during remedial design.

Figure 6-4a presents the three recovery categories. Figure 6-4b includes the empirical contaminant trends with the recovery categories. A detailed analysis of this process by subarea is provided in Appendix D.

6.3.1.1 Net Sedimentation

Natural recovery processes in the LDW include the natural deposition of cleaner sediment from upstream that is expected to reduce surface sediment COC concentrations.[13] Recovery is not considered viable if the STM estimates a potential for net scour (no sedimentation under average flow conditions); such areas are considered Category 1. Any positive rate of sedimentation indicates that an area may potentially be

[13] Important mechanisms that reduce surface sediment contaminant concentrations are deposition of sediment sourced from upstream, followed by mixing and burial (see Section 2, Figure 2-11). These processes are described in greater detail in Section 5.

amenable to natural recovery, and thus this criterion places an area in Category 2 or 3 unless other lines of evidence suggest recovery is not occurring.

Additionally, changes in bathymetric data between 2003 and 2008 in the navigation channel were reviewed (Figure 6-5) as a qualitative check on STM-estimated net sedimentation rates. Pre-dredge bathymetric data collected in 2008 in the navigation channel by the USACE were paired with bathymetric data collected in August 2003 by LDWG.[14] Figure 6-5 displays the differences in elevation at points along the 2008 transects in the navigation channel. Where data from both surveys were available, differences observed over this 5-year period suggest that sedimentation had occurred in much of the navigation channel. While not used as a primary line of evidence for assigning recovery categories, many areas of empirically-estimated deposition roughly match the model predictions. However, differences in survey methods and limited documentation of the bathymetric surveys have produced some uncertainties in the data, which may inaccurately show some areas as having scour (e.g., RM 1.7 to RM 1.9 near Slip 2).

6.3.1.2 High-flow Events

High-flow events increase the rate of erosion in certain areas of the LDW, which could reduce recovery potential. Scour deeper than 10 cm, as estimated by the STM to occur any time during a 100-year high-flow event, is evidence that recovery may not be occurring (see Figure 2-9). A depth of 10 cm was selected because it is the depth of the biologically active zone and the depth of most of the surface sediment samples in the FS baseline dataset.

6.3.1.3 Vessel Scour Areas

Vessel scour areas were identified based on observed ridges and furrows (as determined using the sun-illuminated image of the 2003 bathymetric data) assumed to be caused by vessel traffic along established vessel traffic routes. These bed form areas are assigned to Category 1 because deposited sediment may be eroding or sedimentation may be restricted. The mapping of this layer was restricted to areas where active berthing (vessels and overwater structures) was observed because vessels maneuvering into these areas may be causing scour or because spud placement during vessel mooring may be disturbing the sediments. Bed forms identified outside of berthing areas could represent spud mounds (from vessels moored outside of mapped berthing areas), depressions from vessels resting on the bottom in shallow water, debris, or shallow track lines from transiting vessels. However, these bed forms outside of known vessel use areas are relatively shallow and localized and are not expected to expose buried contamination or impede recovery. Therefore, the mapping of vessel

[14] The August 2003 data collection effort predated the January 2004 maintenance dredging in the navigation channel from RM 4.3 to 4.65 (the last navigation channel dredging event prior to the 2003 data collection was in January 2002; Table 2-9).

scour areas was restricted to higher-traffic areas based on the presence of a pier/wharf face, documented maintenance dredging events, and/or operator interviews indicating that the area supports frequent vessel traffic (see Figures 2-10 and 6-5).

6.3.1.4 Berthing Areas

Berthing areas are locations in the LDW adjacent to existing overwater structures that are not part of marinas, such as piers, wharves, pile groups, and dolphins (Figure 2-28 displays both overwater structures and berthing areas). These areas are assumed not to be viable for natural recovery if evidence of vessel scour was observed or empirical trends show increasing concentrations of risk drivers (excluding dioxins/furans). Berthing areas without evidence of vessel scour are assumed to exhibit recovery potential and thus were placed in Category 2. Berthing areas with evidence of vessel scour were placed in Category 1. Empirical contaminant trends, when available in berthing areas, were used as a final check to either confirm a recovery category designation or as an override to assign an area to another recovery category depending on the observed trend (see next subsection).

6.3.1.5 Empirical Contaminant Trends

Empirical trends in risk-driver (excluding dioxins/furans) concentrations were used as a final check to either confirm or override recovery category assignments based on physical criteria on a case-by-case basis. The identification of a sample location as belonging to an empirical trend category followed a three-step process. First, sample locations with the appropriate data (resampled surface sediment locations within 10 ft of one another or cores with two sample intervals in the top 2 ft) were identified (Table 6-4; Part 1). Second, each detected risk driver exceeding the SQS was assigned to one of three categories (Table 6-4; Part 2):

- Increase: contaminant concentration increasing more than 50% over previous or deeper concentration
- Equilibrium: a small (less than 50%) change in concentration
- Decrease: contaminant concentration decreasing more than 50% from previous or deeper concentration.

Third, the trend assignments for the risk drivers exceeding the SQS were grouped into a summary designation for each location (Table 6-4; Part 3 and Figure 6-4b). Dioxins/furans were not evaluated because of a lack of temporal data. Figure 6-4b shows two symbols per location, one for total PCBs alone and another for all other risk-driver contaminants:

- Increase (red): All contaminants evaluated increased by more than 50%. A location with two red symbols was in Category 1.

- Equilibrium or mixed (gray): A location with mixed results by contaminant (risk drivers other than total PCBs having any combination of assignments in bulleted list above) or concentration changes in equilibrium (less than 50% change) was in Category 2. If a location's trend assignment of "mixed" was based on a combination of decreasing trends and equilibrium (but no increasing trends) that location was in Category 3.
- Decrease (blue): All contaminants evaluated had concentrations decreasing by more than 50%. A location with two blue symbols was in Category 3.
- Below SQS (green): Total PCBs or all other contaminants were not detected above the SQS.

The shape of the symbol denotes whether it is a co-located surface grab sample or a sediment core. Empirical overrides of the physical criteria (Table 6-3) occurred on a case-by-case basis (described in Table D-2). The empirical data are discussed in greater detail in Appendix F.

6.4 Uncertainty Analysis of AOPCs and Recovery Potential

Uncertainties in the process of developing AOPC footprints and recovery potential categories are discussed below.

6.4.1 AOPC Uncertainty

This section examines the degree of confidence that exists with the estimate of the AOPC footprints using the criteria discussed in Section 6.1. The primary factors contributing to uncertainty in the AOPC footprints are:

- Age of the data
- Data mapping and interpolation
- Use of SWACs instead of 95% upper confidence limit (UCL95) on the SWAC.

These uncertainties are discussed below.

6.4.1.1 Age of Data

The FS baseline surface sediment dataset was used to map the AOPCs. Older data at stations that were resampled (collected within 10 ft of newer data) were excluded from the FS baseline dataset on a contaminant-by-contaminant basis. The intent was to use the most recent data available for defining the nature and extent of contamination. LDWG conducted sampling in 2005, 2006, 2009, and 2010 to expand and update the existing dataset. However, because the FS study area is large (441 acres), some data that are more than 10 years old remain in the dataset.

The FS baseline surface sediment dataset is comprised of over 1,400 surface sediment samples spanning 20 years of data collection (1990 through 2010). Between 1990 and 2004, approximately 1,200 surface sediment samples, 340 subsurface sediment cores, and 90 fish and shellfish tissue samples were collected from the LDW by parties other than LDWG. These samples and cores were analyzed for metals and organic compounds. Data that were deemed acceptable based on a review of analytical methods and quality assurance reports became part of the RI and FS baseline datasets. Additional data were collected from 2004 to 2006 by LDWG for the RI to characterize contamination and physical properties of the LDW. These data included approximately 900 samples of the following media: fish, clam, crab, and benthic invertebrate tissue; seep water (water seeping from banks along the LDW); surface sediment (the top 10 cm); subsurface sediment (below the top 10 cm); and porewater (water in spaces between sediment particles). In 2009 and 2010, LDWG collected an additional 41 surface sediment samples and 6 composite sediment samples for the FS to characterize beach play areas and to expand the dioxin/furan dataset.

Many of the sediment samples are now over 10 years old, and surface conditions may have changed in these sampled areas. In mapping the AOPCs, however, this level of uncertainty is considered to be acceptable for the FS by assuming all data points represent baseline conditions. Remedial alternatives are assembled around these predictions along with other lines of evidence described in Section 8. Sampling conducted during remedial design will be conducted to help reduce any outstanding uncertainties. To account for uncertainties associated with older data being used to evaluate RAL exceedances, areas of AOPC 1 meeting all or most of the following characteristics are assumed to be candidates for verification monitoring during remedial design:

- Relatively old data (i.e., sampled prior to 1998)[15]
- Risk-driver concentrations exceeding but close to the AOPC 1 RALs, specifically SQS exceedances less than 1.5 times the SQS or total PCB concentrations slightly over 240 µg/kg dw
- Isolated points (i.e., only 1 point with an SQS exceedance in a 0.5-acre or larger area or where a point is surrounded by passes)
- Not in Recovery Category 1
- BCM predictions of recovery within 10 to 20 years from baseline.

Verification monitoring during remedial design should confirm whether the sediments in these areas exceed the RALs. Areas designated as candidates for verification

[15] The AOPC footprint was first delineated in 2008 for the draft FS. Samples collected prior to 1998 were more than 10 years old at that time (2008).

monitoring are shown in Appendix D and are mapped separately in the remedial alternatives (Section 8). No empirical time trend data were available for these 23 acres.

6.4.1.2 Data Mapping and Interpolation

The FS baseline dataset contains data from numerous site investigations conducted over the past 20 years. These investigations have been used to determine the nature and extent of sediment contamination associated with past hazardous substance releases. This extensive dataset was used to build the conceptual site model, map the nature and extent of contamination, and understand site processes for evaluating remedial alternatives. However, as with every environmental investigation, some uncertainty remains associated with the horizontal and vertical extent of sediment contamination, as discussed in the following points:

- **Laboratory Reporting Limits:** A portion of the uncertainty is related to reporting limits that exceed the screening criteria, especially in older data. Therefore, only detected SQS exceedances (expressed spatially as Thiessen polygons) were used to delineate the AOPCs for RAO 3. Samples with only undetected data (i.e., reporting limits) exceeding the SQS criteria were not considered exceedances. In the ERA (Windward 2007a), an evaluation of the reporting limits that exceeded the SQS concluded that there was a low probability that these exceedances would be of concern.

- **Sampling Design**: Another portion of the AOPC uncertainty is related to the uneven distribution of sampling in historical datasets. Good spatial coverage exists throughout the LDW, but the sampling density is not evenly distributed. For example, some investigations targeted specific areas (e.g., Boeing Plant 2) and these areas have much denser sampling coverage than other areas of the LDW. For this reason, the spatial extent of contamination remains somewhat uncertain, which is common in the feasibility study phase of any large site. Sampling coverage and density will be refined through the addition of new data collected during remedial design.

- **Interpolation Methods:** Two interpolation methods were used to map surface sediment data (IDW and Thiessen polygons; see Appendix A). Each of these methods has inherent uncertainties, including the sampling density, influence of geomorphology on the distribution of contaminants, and influence of surrounding data. The uncertainty in these methods was minimized by conducting an extensive exploratory analysis and by optimizing the IDW parameters used for interpolating total PCBs, arsenic, and cPAHs. This parameterization simulates a "best-fit" estimate of the true concentration gradients (Appendix A). The selected mapping techniques (i.e., IDW interpolation and Thiessen polygons) are well documented and widely used in managing contaminated sediments. The spatial extent of

COC concentrations is expected to be refined during the remedial design phase when additional samples are collected.

- **Vertical Compositing**: The subsurface sediment dataset includes many sediment cores that extended down to "native sediments," where most contaminant concentrations were below the SQS. This was documented in the logs for the cores collected in 2006 for the RI. However, many cores collected for other sampling events did not have logs, were composited over broad depth intervals (e.g., 4 ft), or were too shallow to reach the native sediments and/or the bottom of contamination. For these shallower cores, the interpreted bottom of contamination may not be the true bottom. Some of the vertical core samples were composited over 2-ft or longer intervals, such that either the bottom of contamination is not completely understood within the sample interval or the depth within the core for the highest contaminant concentrations is not completely understood.
- **Vertical Extent of Contamination:** On a site-wide scale, the vertical extent of contamination (greater than SQS) has been interpolated into an isopach layer representing the bottom of this contamination (described in Appendix E). The native alluvium contact, which has also been interpolated into an isopach layer, can be used as a surrogate for the uncertainty in the extent of the bottom of contamination for this FS (see Appendix E). The top of the native alluvium isopach layer is also assumed to be the maximum depth of any subsurface sediments with total PCB concentrations greater than 100 µg/kg dw (below this contact, sediments are assumed to exhibit native, pre-industrialized conditions). Because cores are much less numerous than surface sediment samples, the interpolation of the subsurface contamination may not represent actual conditions as effectively as it does for surface sediments. These estimates will need to be refined during remedial design.

  Additionally, the cores were collected by many different parties using various sampling methods and compositing schemes. The data were also not always accompanied by field and core processing logs that could be used to adjust recovered depths to *in situ* depths or to provide other useful information. Finally, not all intervals within each core were sampled, and within those intervals sampled, not all COCs were analyzed. If a sampling interval was not analyzed and the interval immediately above was contaminated, then the bottom of the contamination is assumed to be the bottom of the skipped interval. For cores that did not reach the bottom of contamination (detected SQS exceedances), 1 ft was added to the depth of the bottom of the core, and this depth was assumed to be the bottom of contamination.

6.4.1.3 95% Upper Confidence Limits (UCL95) on SWACs

The UCL95 on the mean is a statistically derived quantity associated with a representative sample from a population (e.g., sediment or tissue chemistry results) such that 95% of the time, the true average of the population from which the sample was taken will be less than the quantity statistically derived from the sample dataset (e.g., 95% of the time, the true average sediment contaminant concentration will be less than the UCL95 based on sediment chemistry sample results). The UCL95 is used to account for uncertainty in contaminant concentration measurements and to ensure that contaminant concentrations are not underestimated.

The AOPCs were delineated in part by estimating when a post-remediation site-wide SWAC achieves a target concentration. Therefore, mean values, not UCL95s, were used to delineate the AOPCs and evaluate predicted results in Section 9. However, in accordance with EPA and Ecology policy for evaluating compliance and estimating exposure concentrations, an upper confidence limit of the true mean (UCL95 on the SWAC) will be developed for each compliance monitoring dataset and compared to the target goal to account for sampling variability. The UCL95 from a well-designed post-remediation sampling program is expected to exceed the true SWAC by some increment.

Because the delineation of the AOPCs and the evaluation of the remedial alternatives are based on SWACs instead of UCL95 values, the footprints could potentially be larger. However, remediation of incrementally larger footprints manages contaminated sediment to incrementally lower concentrations, decreasing variability in the dataset. Footprints based on achieving the long-term model-predicted concentrations (SWACs) are likely not much different than those based on UCL95 values because, over time, natural recovery, coupled with remediation of hot spots, will reduce variability such that SWACs and UCL95 values become similar.

Appendix H discusses methods for calculating the UCL95 on the SWAC using the total PCB RI baseline dataset.

Overall, the nature and extent of sediment contamination is sufficiently understood to characterize risks, and develop reasonable estimates of the AOPCs and LDW-wide remedial alternatives for the FS. Uncertainty in the horizontal and vertical extent of sediment contamination above selected RALs will be refined during remedial design.

6.4.2 Recovery Potential Uncertainty

The recovery categories synthesize a large amount of information into a simple construct that can be used for managing uncertainty in technology assignments for this FS-level analysis. However, each criterion used in this analysis contains both uncertainties and assumptions. Remedial design-level analysis will provide additional information that will supersede many of the assumptions in this analysis. A few of the major assumptions that may affect an FS and remedial design-level analysis include:

- Berthing areas, navigation channel operations, and elevations necessary for berthing and navigation may change.
- Further observations and analysis of location-specific vessel scour and its effect on recovery may change. Location-specific analysis of impacts may result in different conclusions on scour potential and may change technology selection.
- STM estimates may be combined with location-specific empirical data to refine sedimentation rates and scour potential.
- Additional data could refine location-specific contaminant trends over time.
- Source control changes could affect the rate of observed natural recovery.

A point to be considered in decision-making for source control implementation, remedy design, and remedy implementation is whether areas of AOPC 1 located near certain outfalls may be subject to recontamination. A premise of EPA's sediment remediation guidance is that active remediation should generally not be implemented until sources have been controlled to the extent necessary to reduce the risk of recontaminating the remediated area (EPA 2005b). Whether active or passive, the success of any remediation may be affected by source control. This FS analysis is consistent with these principles. The FS accounts for recontamination potential in the technology assignments (Section 8) and in the predicted outcomes (Section 9) using the range of BCM input parameters (Section 5).

Estimates of recovery potential should also include: 1) physical conditions that may preclude recovery; 2) predictive modeling that assumes lateral sources will be controlled, at least to some extent, in the future; 3) empirical trends demonstrating that recovery is underway, but that "final" recovery will require additional source control measures and time; and 4) recontamination potential from external sources (see Appendix J). All of these factors have been considered in this FS. However, remedial design-level sampling and further evaluation of source control effectiveness will be necessary in certain areas before any remedial action is initiated. These data and model predictions will be essential in reassessing future recovery or recontamination of surface sediments after source controls are in place.

Table 6-1 Lowest Point Concentrations Used to Delineate AOPCs and Associated SWACs

Risk Driver	Lowest Point Concentrations Used to Delineate AOPC[a]	Preliminary Remediation Goals (PRGs)				Long-term model-predicted concentrations (SWAC)	Estimated SWACs after Active Remediation of AOPC[b]			Are Cleanup Objectives Achieved?			
		RAO 1 (site-wide SWAC)	RAO 2[c] (site-wide netfishing; beach play; clamming SWACs)	RAO 3 (point)	RAO 4 (site-wide SWAC)		Site-wide	Beach Play	Clamming	RAO 1	RAO 2	RAO 3	RAO 4
AOPC 1 (180 acres). Active remediation of AOPC 1 would achieve PRGs for RAOs 2, 3, 4 immediately after construction (with the exception of RAO 2 for arsenic)													
Total PCBs (µg/kg dw)	240 (site-wide)	bg: 2	**1,300; 1,700; 500**	**12 mg/kg oc**	**128-159**	n/a	92	62	69	**T[e]**	✓	✓	✓
Arsenic (mg/kg dw)	57 (site-wide) 28 (intertidal)	n/c	bg: 7	**57**	n/a	n/a	11	9	9	n/a	**T[c]**	✓	n/a
cPAHs (µg TEQ/kg dw)	1,000 (site-wide) 900 (intertidal)	n/c	**380;** 90; **150**	n/a	n/a	n/a	210	150	150	n/a	✓[d]	n/a	n/a
Dioxins/furans (ng TEQ/kg dw)	25 (site-wide) 28 (intertidal)	bg: 2	**37; 28; 13**	n/a	n/a	n/a	6	4	5	**T[e]**	✓	n/a	n/a
SMS contaminants	SQS (site-wide)	n/a	n/a	**SQS**	n/a	n/a	n/a			n/a	n/a	✓	n/a
AOPCs 1& 2 (302 acres). Active remediation AOPCs 1 and 2 would achieve long-term model predicted concentrations (the lowest technically achievable SWACs) immediately after construction.													
Total PCBs (µg/kg dw)	100 (site-wide)	bg: 2	**1,300; 1,700; 500**	**12 mg/kg oc**	**128 to 159**	40-50	46	46	48	✓[e]	✓	✓	✓
Arsenic (mg/kg dw)	15 (site-wide) 28 (intertidal)	n/c	bg: 7	**57**	n/a	9-10	10	9	9	n/a	✓[c]	✓	n/a
cPAHs (µg TEQ/kg dw)	same as AOPC 1	n/c	**380;** 90; **150**	n/a	n/a	100 - 150	140	140	140	n/a	✓	n/a	n/a
Dioxins/furans (ng TEQ/kg dw)	15 (site-wide) 28 (intertidal)	bg: 2	**37; 28; 13**	n/a	n/a	4-6	4	4	4	✓[e]	✓	n/a	n/a
SMS contaminants	same as AOPC 1	n/a	n/a	**SQS**	n/a	n/a	n/a			n/a	n/a	✓	n/a

Table 6-1 Lowest Point Concentrations Used to Delineate AOPCs and Associated SWACs (continued)

Notes:

1. AOPC 1 is also delineated where cores having SQS exceedances in the top 2 ft occur in scour areas. AOPC 2 is also delineated where any core exceeds the SQS at any depth.
2. Site-wide point concentrations used to delineate AOPCs are applied to concentrations in the upper 10 cm of sediment and intertidal point concentrations used to delineate AOPCs are applied to concentrations in the upper 45 cm of sediment.

a. Site-wide point concentrations used to delineate AOPCs are applied in the upper 10 cm of sediment throughout the LDW, and in the upper 60 cm of potential scour areas (i.e., Recovery Category 1 areas; see Section 6.3). Intertidal point concentrations used to delineate AOPCs are applied in the upper 45 cm of sediment in intertidal areas (above -4 ft MLLW).

b. SWACs are estimated by replacing grid cells in AOPCs 1 and 2, respectively, with the following post-remedy bed sediment replacement values: total PCBs = 60 and 20 µg/kg dw; arsenic = 10 and 9 mg/kg dw; cPAHs = 140 and 100 µg TEQ/kg dw; and dioxin/s/furans = 4 ng TEQ/kg dw. AOPC 2 SWACs are based on replacing grid cells in both AOPCs 1 and 2. SWACs are based on the cumulative effect of removing all points/areas above the site-wide and intertidal point concentration shown for each risk driver (the entire AOPC footprint).

c. Because natural background PRG is unlikely to be achieved, this RAO is being evaluated by surface sediment reaching the long-term model-predicted arsenic concentrations. These concentrations are achieved with time after remediation of AOPC 1 and are achieved immediately after remediation of AOPCs 1 and 2.

d. Although the combined beach play area cPAH SWAC is not below 90 µg TEQ/kg dw, this PRG is considered to be achieved because most of the individual beaches achieve this PRG or a 1×10^{-6} excess cancer risk threshold.

e. Because natural background PRGs are unlikely to be achieved for total PCBs and dioxins/furans, RAO 1 is being evaluated by surface sediment reaching the long-term model-predicted concentrations for these two risk drivers. These concentrations are achieved with time after remediation of AOPC 1 and are achieved immediately after remediation of AOPCs 1 and 2.

✓ = Achieves cleanup objective (PRG or long-term model-predicted concentration) immediately following construction.

T = Achieves cleanup objective over time. Institutional controls will be required to further reduce RAO 1 risks regardless of the selected RAL. For RAOs 1 and 2 (arsenic) the goal is to reduce sediment concentrations to as close as practicable to the PRG, estimated in this FS as long-term model-predicted concentrations.

Bold = PRG achieved

AOPC = area of potential concern; bg = background; cPAH = carcinogenic polycyclic aromatic hydrocarbon; dw = dry weight; kg = kilograms; µg = micrograms; mg = milligrams; n/a = not applicable; n/c = not calculated; ng = nanograms; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAL = remedial action level; RAO = remedial action objective; SMS = Sediment Management Standards; SQS = sediment quality standard; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent.

Table 6-2 Array of Remedial Action Levels

Risk-Driver Remedial Action Level[a]	Rationale	Cleanup Objective Achieved[b] (✓ = achieved immediately after construction; T = achieved with time)			
		RAO 1[c]	RAO 2	RAO 3	RAO 4
Total PCBs (µg/kg dw)					
2,200 (site-wide)	• Manage hot spots	**T**	Total PCB direct contact PRGs are achieved following remediation of EAAs and hot spots.[e]	**T** (achieves CSL with time)	**T**
1,300 (site-wide)	• Dry weight equivalent of CSL[d]; achieved immediately after construction	**T**		**T**	**T**
700 (site-wide)	• Provides a well-spaced range of RALs for evaluation	**T**		**T**	✓
240 (site-wide)	• Dry weight equivalent of SQS[d]; achieved immediately after construction	**T**		✓	✓
100 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	✓		✓	✓
Arsenic (mg/kg dw)					
93 (site-wide)	• Achieve CSL immediately after construction / Manage hot spots	n/a	**T**	**T**	n/a
57 (site-wide)	• Achieve SQS immediately after construction and part of a well-spaced range of RALs	n/a	**T**	✓	n/a
28 (intertidal)	• 10^{-5} beach play RBTC (applied as point basis; 45 cm point of compliance) and part of a well-spaced range of RALs	n/a	**T**	✓	n/a
15 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	n/a	✓	✓	n/a
cPAHs (µg TEQ/kg dw)					
5,500 (site-wide)	• Manage hot spots	n/a	**T**	n/a	n/a
3,800 (site-wide)	• 10^{-5} netfishing RBTC (applied as a point basis) and part of a well-spaced range of RALs	n/a	✓	n/a	n/a
1,000 (site-wide)	• Site-wide SWAC within range of upstream values	n/a	✓	n/a	n/a
900 (intertidal)	• Beach play 10^{-5} RBTC (applied as point basis; 45 cm point of compliance)	n/a	✓	n/a	n/a

Table 6-2 Array of Remedial Action Levels (continued)

Risk-Driver Remedial Action Level[a]	Rationale	Cleanup Objective Achieved[b] (✓ = achieved immediately after construction; T = achieved with time)			
		RAO 1[c]	RAO 2	RAO 3	RAO 4
Dioxins/Furans (ng TEQ/kg dw)					
50 (site-wide)	• Manage hot spots	T	T	n/a	n/a
35 (site-wide)	• Provides a well-spaced range of RALs for evaluation	T	✓	n/a	n/a
28 (intertidal)	• 10^{-6} beach play RBTC (applied as point basis; 45 cm point of compliance)	T	✓	n/a	n/a
25 (site-wide)	• Provides a well-spaced range of RALs for evaluation	T	✓	n/a	n/a
15 (site-wide)	• Site-wide SWAC within range of upstream values and long-term model-predicted concentrations • Point of minimal change in SWAC	✓	✓	n/a	n/a
SMS Contaminants (apply throughout the LDW)					
CSL at Year 10 (site-wide)	• Achieve CSL within 10 years after completion of construction	n/a	n/a	T	n/a
CSL at Year 0 (site-wide)	• Achieve CSL immediately after completion of construction	n/a	n/a	T	n/a
SQS at Year 10 (site-wide)	• Achieve SQS within 10 years after completion of construction	n/a	n/a	T	n/a
SQS at Year 0 (site-wide)	• Achieve SQS immediately after completion of construction	n/a	n/a	✓	n/a

Notes:

a. A remedial action level is a contaminant-specific sediment concentration that triggers the need for active remediation (i.e., dredging, capping, or ENR with or without *in situ* treatment). It is a point-based concentration that can be targeted to achieve an area-based goal (SWAC). Site-wide remedial action levels are applied to concentrations in the upper 10 cm of sediment throughout the LDW and in the upper 60 cm in Recovery Category 1 areas. Intertidal remedial action levels are applied to concentrations in the upper 45 cm of sediment in intertidal areas (above -4 ft MLLW).

b. See Section 9 for predicted outcomes and RALs by remedial alternative.

c. Risks associated with RAO 1 are reduced through a combination of active remediation, natural recovery, and institutional controls. The goal is to reach the long-term model-predicted concentration, which is as close to natural background as technically practicable (equilibrium).

d. Dry weight equivalents of the SQS and CSL SMS criteria of 12 and 65 mg/kg oc, assuming 2% TOC (average site-wide TOC value). If selected, actual implementation of this RAL would be based on organic carbon-normalized criteria defined by the SMS.

e. An intertidal RAL for PCBs in the upper 45 cm of sediment was not developed because the PRGs for direct contact scenarios are achieved after remediation of the EAAs and other hotspot areas (using the highest RAL sets shown above).

Year 0 = the point in time immediately following completion of construction.

Year 10 = the point in time 10 years after completion of construction.

✓ = Achieves cleanup objective immediately following construction. For RAO 1, institutional controls are also needed.

T = Achieves cleanup objective over time. Institutional controls will be required for RAO 1 regardless of the selected RAL. For RAOs 1 and 2 (arsenic) the goal is to reduce sediment concentrations to achieve the long-term model-predicted concentrations.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; CSL = cleanup screening level; dw = dry weight; kg = kilograms; µg = micrograms; mg = milligrams; n/a = not applicable to the RAO; ng = nanograms; PCB = polychlorinated biphenyl; PRG = preliminary remediation goal; RAL = remedial action level; RAO = remedial action objective; RBTC = risk-based threshold concentration; SMS = sediment management standards; SQS = sediment quality standard; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; TOC = total organic carbon.

Table 6-3 Criteria for Assigning Recovery Categories

Criteria		Recovery Categories: Category 1 Recovery Presumed to be Limited	Recovery Categories: Category 2 Recovery Less Certain	Recovery Categories: Category 3 Predicted to Recover
Physical Criteria				
Physical Conditions	Vessel scour[a]	Observed vessel scour	No observed vessel scour	
	Berthing areas[b]	Berthing areas with vessel scour	Berthing areas without vessel scour	Not in a berthing area
Sediment Transport Model	STM-predicted 100-year high-flow scour (depth in cm)[c]	> 10 cm	< 10 cm	
	STM-derived net sedimentation rate[b] (cm/yr) using average flow conditions	Net scour	Net sedimentation	
	Rules for applying criteria	*Any one criterion in Category 1 results in the area achieving a Category 1 designation.*	*Conditions achieve a mixture of Category 2 and 3 criteria*	*All conditions must achieve the Category 3 criteria.*
Empirical Contaminant Trend Criteria – used on a case-by-case basis to adjust recovery categories from the criteria above				
Empirical Contaminant Trend Criteria[d]	Resampled surface sediment locations	Increasing total PCBs or increasing concentrations of other detected risk drivers exceeding the SQS (> 50% increase)[e]	Equilibrium and mixed (increases and decreases) results (for risk drivers exceeding the SQS)	Decreasing concentrations (> 50% decrease) or mixed[f] results (decreases and equilibrium)[e]
	Sediment cores (top 2 sample intervals in upper 2 ft)			

Notes:

1. Empirical trends were evaluated for two contaminant groups: total PCBs and other risk drivers exceeding the SQS. Dioxins/furans were not evaluated because the small dioxin/furan dataset does not include resampled surface sediment locations and has very few subsurface sediment samples.

a. Observed vessel scour areas are shown on Figure 2-10.
b. Berthing areas are shown on Figure 2-28 and modeled net sedimentation rates are shown on Figure 2-11.
c. High-flow scour areas are shown on Figure 2-9.
d. Empirical trend data are described in Appendix F and summarized in Figure 6-4b. See Table 6-4 for description of empirical trend methodology.
e. ±50% decrease is reasonable considering that analytical variability alone is 25%, and the difference in co-located field replicates ranged from 8% (arsenic) to 48% (cPAHs).
f. A location with mixed results in which risk drivers exceeding the SQS have decreasing trends and concentration changes in equilibrium (but no increasing trends) can be in Recovery Category 3.

cPAH = carcinogenic polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyl; SQS = sediment quality standard; STM = sediment transport model

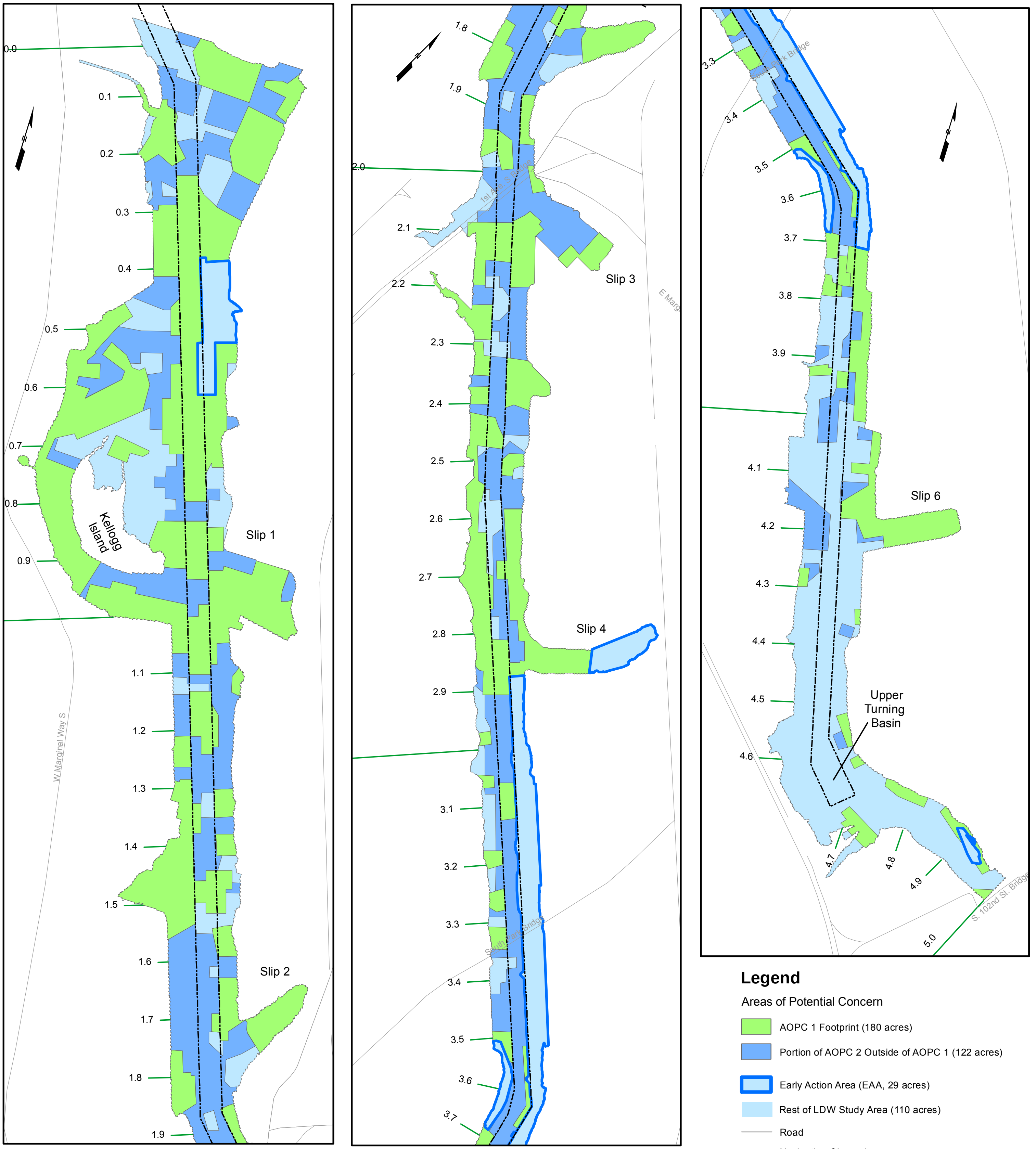
Table 6-4 Empirical Data Methodology Used in Natural Recovery Trend Evaluation

Part 1: Selection of Locations and Data	**Part 2: Trend Criteria Evaluated by Risk Driver**	**Part 3: Natural Recovery Classification by Station** (Figure 6-4b symbol colors; 1 symbol for total PCBs; 1 symbol for other risk drivers)
Resampled surface stations (not more than 10 ft apart)	Increase (> 50% change in concentration)	Increase – all evaluated risk drivers increase (**red**); all red symbols = Category 1
Top 2 sample intervals of cores within top 2 ft of core[a]	Equilibrium (less than 50% change in concentration in either direction)	Equilibrium – all evaluated risk-driver concentrations change by less than 50% (**gray**); Category 2.
Detected risk drivers exceeding the SQS evaluated for concentration changes	Decrease (>50% change in concentration)	Mixed – Risk drivers other than total PCBs have some mixture of any of 3 classifications (**gray**); Category 2 if mixture includes increases; Category 3 if mixture is decreases and equilibrium.
		Decrease – all evaluated risk drivers decrease (**blue**); Category 3.
		Detected total PCBs or other risk drivers do not exceed the SQS[b] (**green**); not specifically used for recovery assignments; area is likely below RALs.

Notes:

1. Two groups of contaminants evaluated: (a) total PCBs detected above the SQS, and (b) risk drivers other than total PCBs detected above the SQS. Figure 6-4b has one symbol for total PCBs and one symbol for other risk drivers.
2. Empirical data evaluation included: 53 to 67 resampled surface sediment locations and 165 cores with appropriate depth intervals (118 samples with an SQS exceedance for total PCBs, 58 samples with an SQS exceedance for other risk drivers). Evaluated the top two intervals of cores if both intervals were within the top 2 ft (can use co-located surface samples).

a. Core trends were also evaluated by comparing the data from the uppermost core interval to that from a co-located surface sediment location, if available.



Notes:
1. AOPC 1 was delineated using beaches with total excess cancer risk (all risk-driver contaminants combined) greater than 10^{-5}, subsurface SQS exceedances in scour areas, and surface sediments exceeding the following RALs: SQS for any contaminant, total PCBs 240 µg/kg dw, arsenic 28 mg/kg dw in intertidal areas, cPAHs 1,000 µg TEQ/kg dw sitewide and 900 in intertidal areas, and dioxins/furans 25 ng TEQ/kg dw.
2. AOPC 2 was delineated using subsurface SQS exceedances at any depth and surface sediments exceeding the following RALs: total PCBs 100 µg/kg dw, arsenic 15 mg/kg dw, and dioxins/furans 15 ng TEQ/kg dw.
3. Verification monitoring areas (23 acres) are included in AOPC 1, but are presumed to be below the RAO 3 PRGs at the time of construction.

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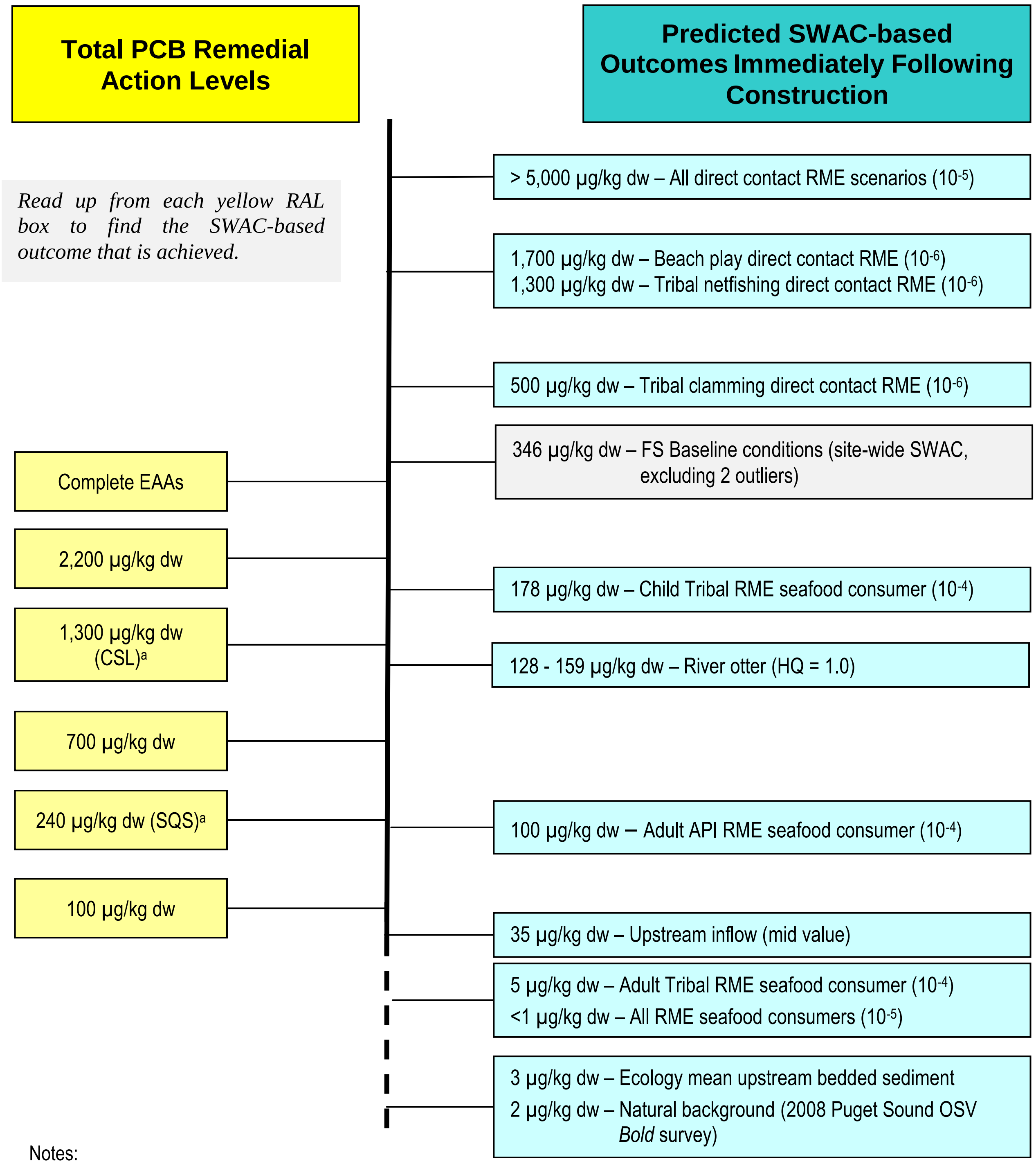
Lower Duwamish Waterway Final Feasibility Study 60150279-14.38	The AOPC Footprints
DATE: 10/31/12 DWRN:mvi/sea Revision: 0	FIGURE 6-1

Figure 6-2a Total PCB Remedial Action Levels for Human and Ecological Health



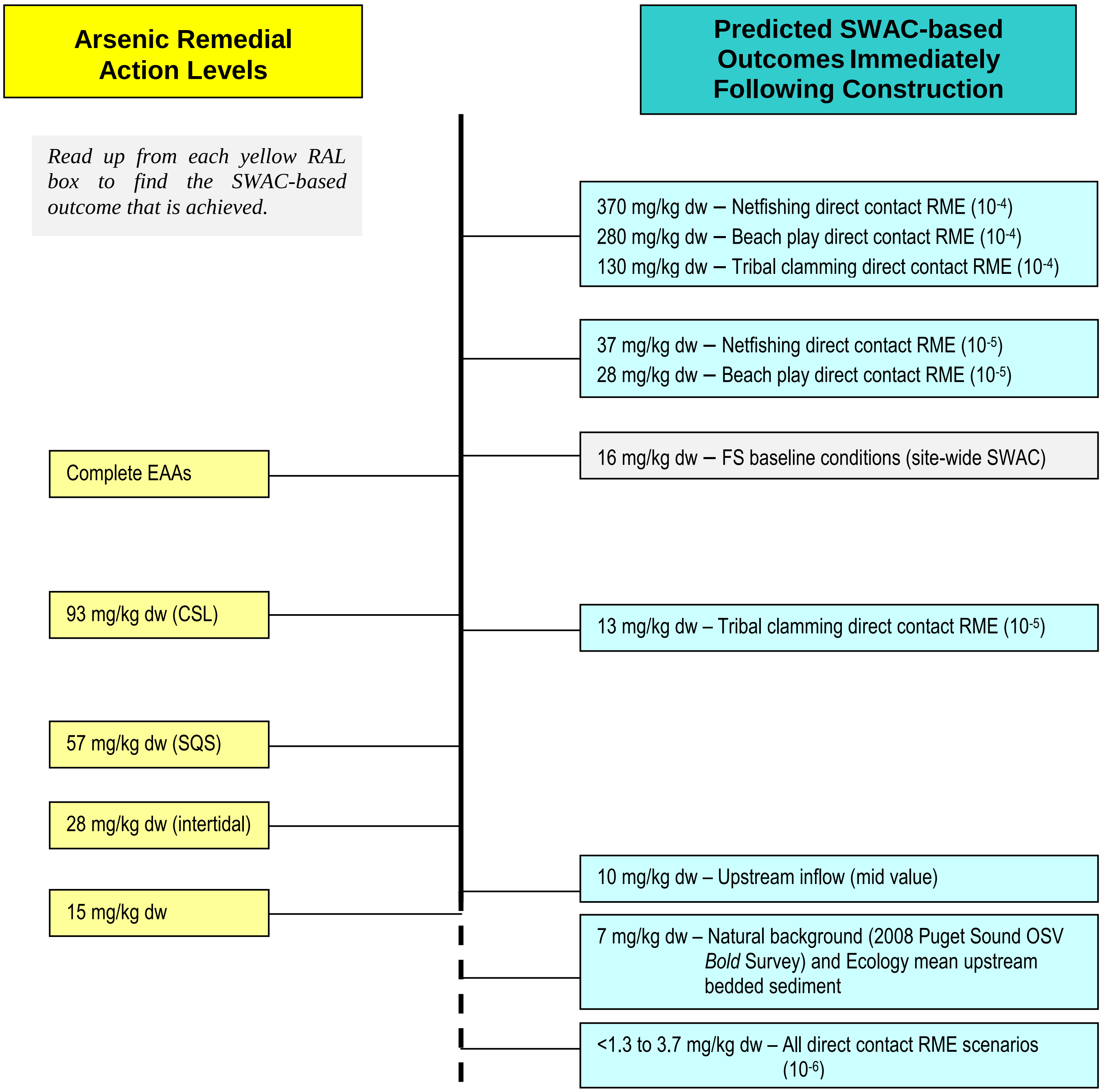
Notes:

a Dry weight equivalents of the SQS and CSL SMS criteria of 12 and 65 mg/kg oc, assuming 2% total organic carbon (average LDW-wide TOC value).

10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CSL = cleanup screening level; EAA = Early Action Area; HQ = hazard quotient; PCB = poly-chlorinated biphenyl; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; SQS = sediment quality standard; TOC = total organic carbon; UCL = upper confidence limit.

– – – – = not achievable

Figure 6-2b Arsenic Remedial Action Levels for Human and Ecological Health

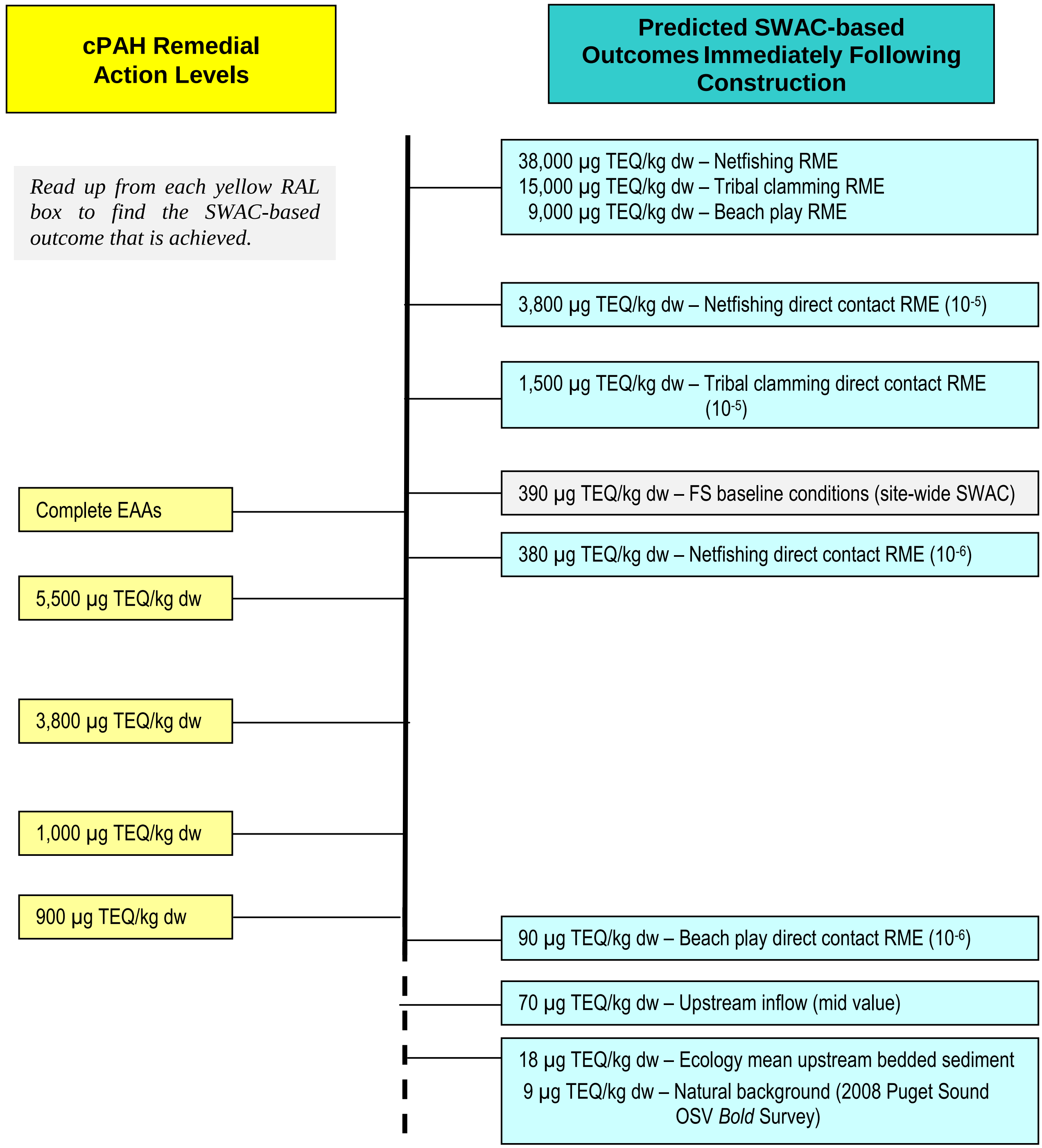


Notes:

10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CSL = cleanup screening level; EAA = Early Action Area; HQ = hazard quotient; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; SQS = sediment quality standard; UCL = upper confidence limit.

▬ ▬ ▬ ▬ = not achievable

Figure 6-2c cPAH Remedial Action Levels for Human Health

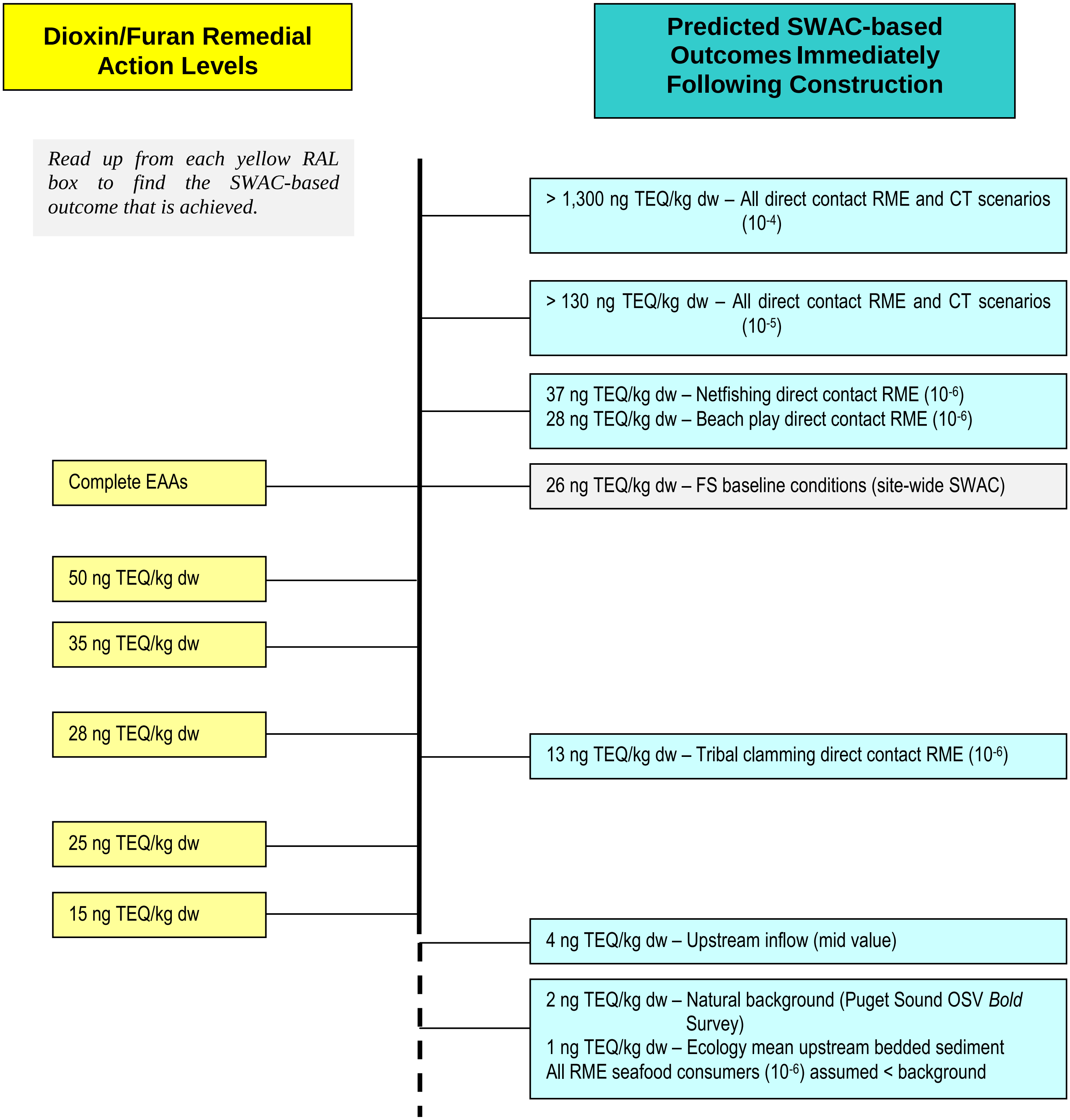


Notes:

10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; cPAH = carcinogenic polycyclic aromatic hydrocarbon; EAA = Early Action Area; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent; UCL = upper confidence limit.

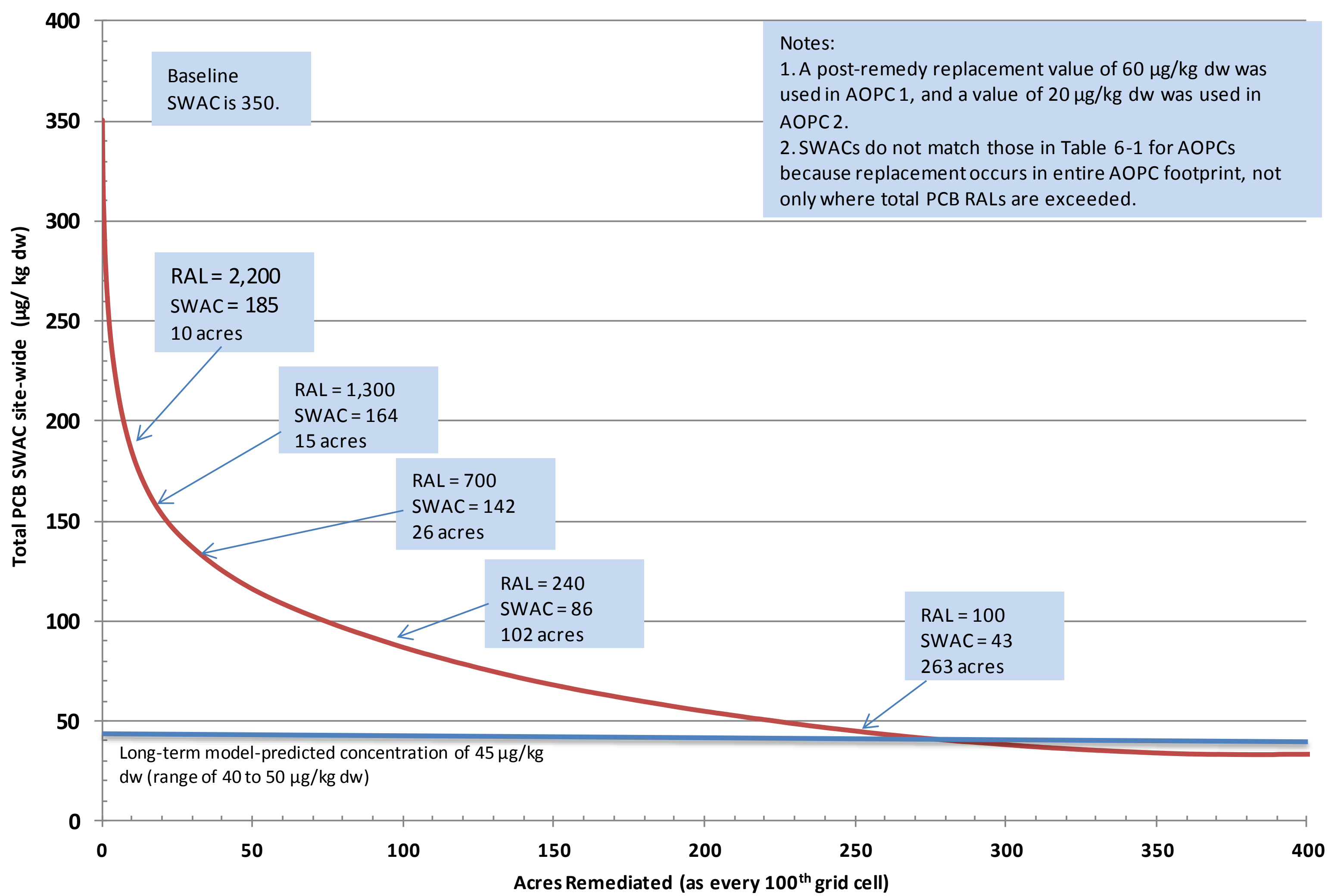
▬ ▬ ▬ ▬ = not achievable

Figure 6-2d Dioxin/Furan Remedial Action Levels for Human Health



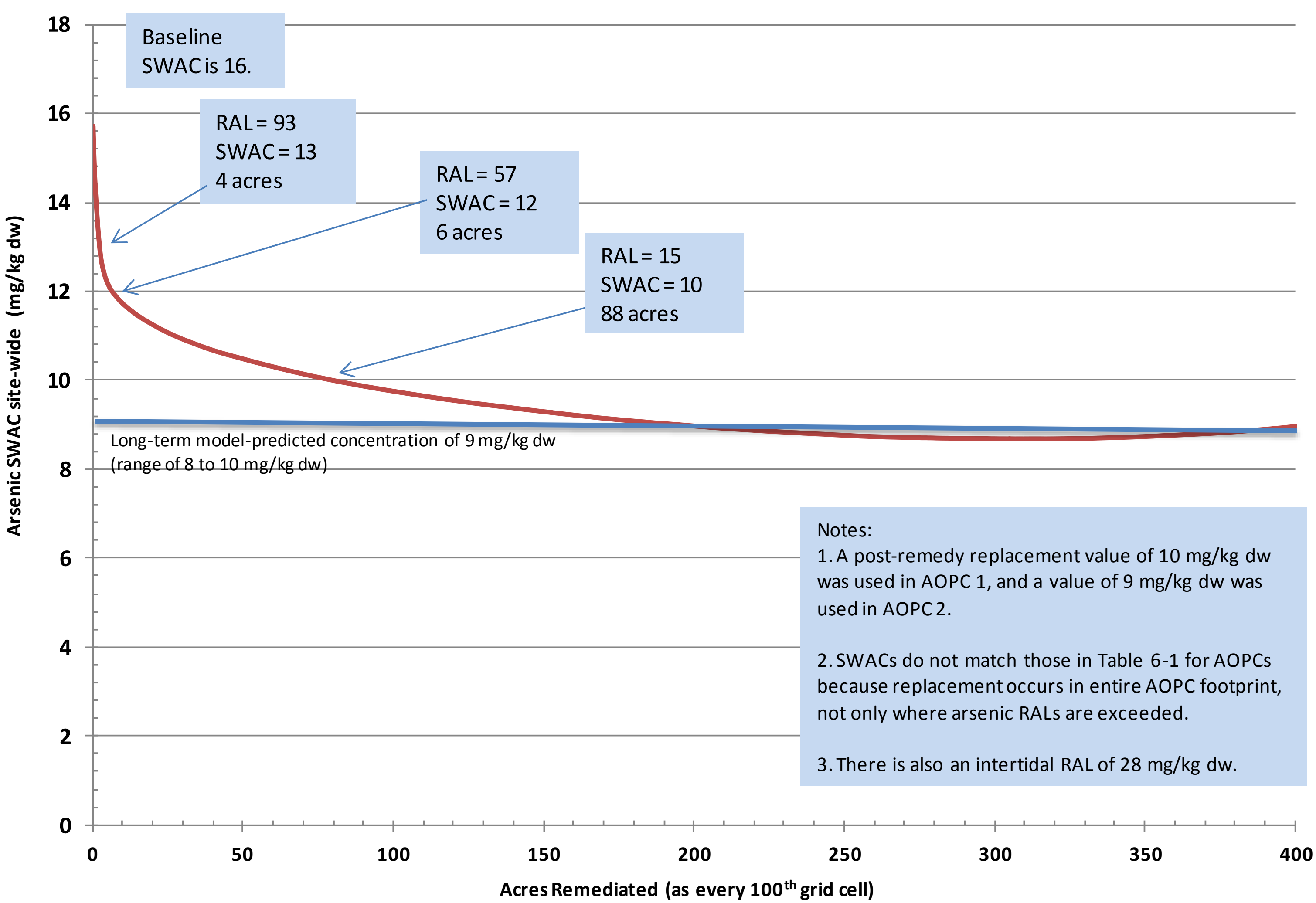
Notes:
10^{-5} = Risk of 1 additional cancer in 100,000 people over a lifetime; CT = central tendency; EAA = Early Action Area; RME = reasonable maximum exposure; SWAC = spatially-weighted average concentration; TEQ = toxic equivalent.
— — — — = not achievable

Figure 6-3a Site-wide SWACs vs. Remediated Acres – Total PCBs



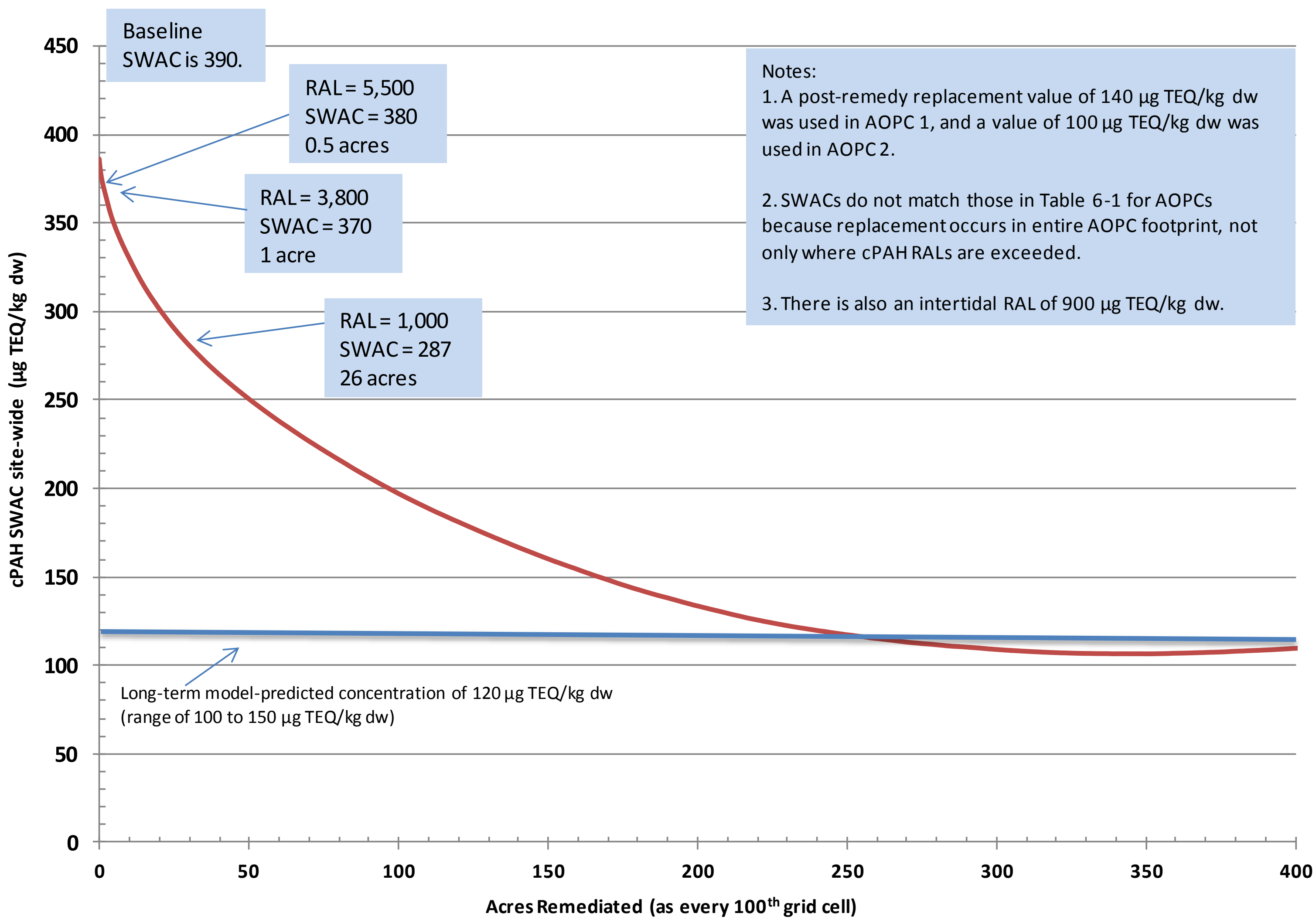
All units are µg/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3b Site-wide SWACs vs. Remediated Acres – Arsenic



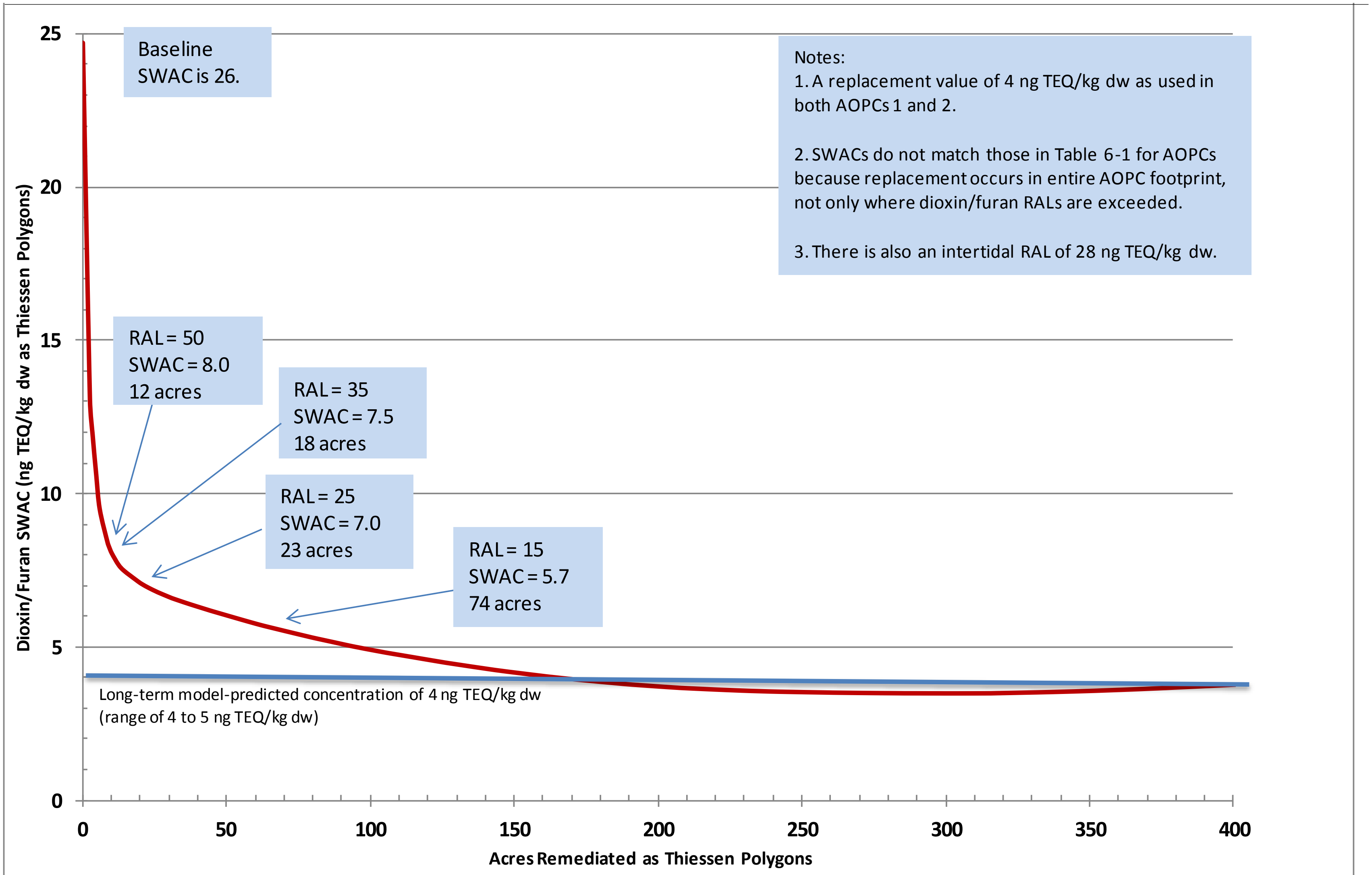
All units are mg/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3c Site-wide SWACs vs. Remediated Acres – cPAHs



All units are µg TEQ/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Figure 6-3d Site-wide SWACs vs. Remediated Acres – Dioxins/Furans



All units are ng TEQ/kg dw; RAL = remedial action level; SWAC = spatially-weighted average concentration.

Legend

Recovery Category

- Category 1: Recovery Presumed to be Limited (77 acres)
- Category 2: Recovery Less Certain (44 acres)
- Category 3: Predicted to Recover (281 acres)
- Early Action Area (29 acres)
- Outfall Location
- River Mile Marker
- Navigation Channel

Feet
0 200 400 800

Notes:
1. See Table 6-3 for recovery category criteria.
2. The entire FS study area downstream of RM 4.75, except the EAAs, is grouped into recovery categories (402 acres).
3. Surface sediment concentrations are evaluated separately (during technology assignments).
An area may be remediated because of elevated COCs regardless of the recovery category.

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.38	**Recovery Categories**
DATE: 10/31/12 DWRN:de/SEA Revision: 0	**FIGURE 6-4a**

Notes:
1. Resampled surface sample locations include data trumped from both the RI and FS datasets.
2. See Table 6-3 for recovery category criteria and Table 6-4 for the methodology used to evaluate the empirical contaminant trends.
3. Only detected risk drivers exceeding the SQS were evaluated for concentration trends.
4. Mixed results can occur at a location when the risk drivers other than total PCBs do not all have the same trend (mixture of increase, decrease, and/or equilibrium).

Natural Recovery Empirical Data

Resampled Surface Sediment LocationTotal PCBs — Resampled Surface Sediment Location Other Risk Drivers

Top Two Intervals in Cores Total PCBs — Top Two Intervals in Cores Other Risk Drivers

Below SQS

Data that do not Support Recovery (≥50% Concentration Increase)

Data that Support Recovery (≥50% Concentration Decrease)

Station in Equilibrium or Mixed Result

Legend

Recovery Category

Category 1: Recovery Presumed to be Limited (77 acres)

Category 2: Recovery Less Certain (44 acres)

Category 3: Predicted to Recover (281 acres)

Early Action Area (29 acres)

Outfall Location

River Mile Marker

Navigation Channel

Feet
0 200 400 800

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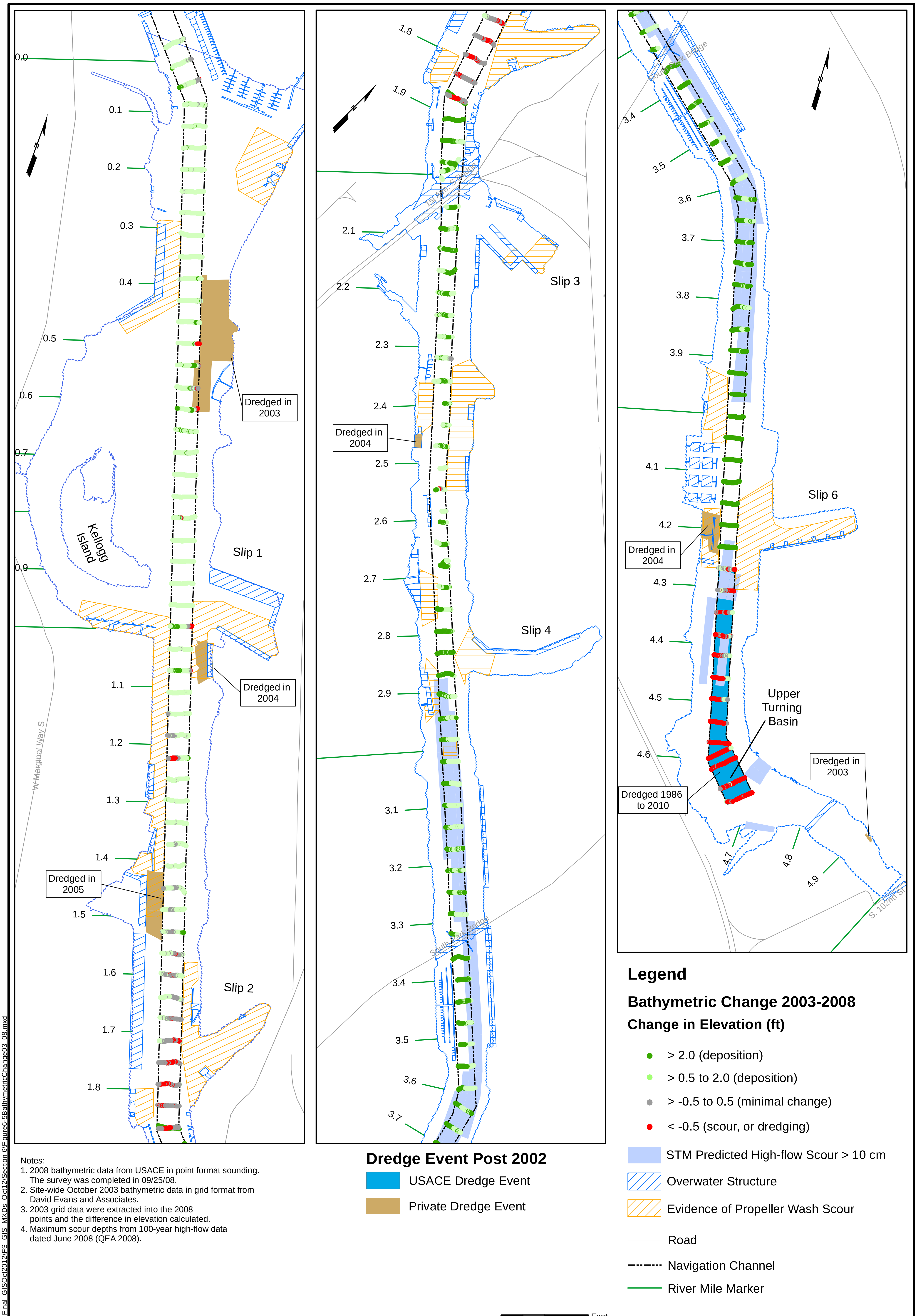
Recovery Categories and Empirical Contaminant Trends

FIGURE 6-4b



Notes:
1. 2008 bathymetric data from USACE in point format sounding. The survey was completed in 09/25/08.
2. Site-wide October 2003 bathymetric data in grid format from David Evans and Associates.
3. 2003 grid data were extracted into the 2008 points and the difference in elevation calculated.
4. Maximum scour depths from 100-year high-flow data dated June 2008 (QEA 2008).

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Lower Duwamish Waterway Final Feasibility Study 60150279-14.38			Bathymetric Change in Navigation Channel between 2003 and 2008
DATE: 10/31/12	DWRN:de/Sea	Revision: 1	FIGURE 6-5

7 Identification and Screening of Remedial Technologies

This section identifies and screens remedial technologies consistent with the U.S. Environmental Protection Agency's (EPA) *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988). This step toward development of the remedial alternatives parallels and is consistent with Washington State's remedial investigation and feasibility study requirements, Washington Administrative Code (WAC) 173-340-350.

The technology screening for the Lower Duwamish Waterway (LDW) was originally completed and issued as the *Candidate Technologies Memorandum* (CTM; RETEC 2005). The CTM identified and screened a comprehensive set of general response actions, technology types, and process options that are potentially applicable to cleanup of contaminated sediments in the LDW. These three categories or tiers provide a systematic structure and method to identify and evaluate various physical, chemical, and administrative "tools" available for implementing remedial actions. General response actions describe in very broad terms the types of actions potentially applicable to cleanup of contaminated media. Each general response action may contain one or more technology type. For example, one general response action is physical removal of contaminated materials from the site, and two common technologies that can accomplish sediment removal are dredging and excavation. Process options are a further subdivision or tier in the technology screening procedure, and define the specific type of equipment used within a technology. For example, dredging may use a clamshell dredge, hydraulic dredge, or upland-based excavation equipment, such as backhoes.

The CTM evaluated remedial technologies and process options that could be carried forward for additional consideration in the FS. The screening evaluation was conducted using the effectiveness, implementability, and cost criteria consistent with EPA guidance (EPA 1988). Effectiveness refers to whether or not a technology can contain, reduce, or eliminate contaminants of concern (COCs). Implementability refers to whether a technology can be operated under the physical and chemical conditions of the LDW, is commercially available, and has been used on sites similar in scale and scope to the LDW. The CTM contains complete descriptions of remedial technologies and process options and the supporting literature considered for alternative development in the FS.

In this section, technology recommendations from the CTM (RETEC 2005) are reviewed and updated to account for any recent technology developments or relevant experience at other cleanup sites. The Superfund Innovative Technology Evaluation (SITE) Program, the EPA Hazardous Waste Clean-up Information (CLU-IN) website, and the Federal Remediation Technologies Roundtable (FRTR) were reviewed for recent and relevant information about innovative treatment technologies, including their cost and performance, results of technology development and demonstration, and technology optimization and evaluation. The complete screening process is summarized in tables as follows:

- Table 7-1 lists all of the candidate remedial technologies and process options that were evaluated in the FS process, along with an initial screening for potential applicability. Remedial technologies retained as initially feasible are shaded.
- Tables 7-2a through 7-2e provide the detailed screening of process options shown as "retained as initially feasible" in Table 7-1, which were presented previously in the CTM and were updated to account for any recent technology developments. These tables were also updated to include new technologies reviewed for the FS (e.g., spray cap).
- Table 7-3 summarizes the assessment of the effectiveness, implementability, and relative costs of the retained remedial technologies and process options.
- Table 7-4 provides the technologies and process options carried forward into alternative development as representative technologies and process options.

Finally, this section selects representative, effective, and implementable process options to carry forward for developing remedial alternatives. The selections consider information on past and current sediment remediation projects in the Puget Sound region, elsewhere in EPA Region 10, and nationally where appropriate. Selecting representative process options for the FS is consistent with the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (EPA 1988) and Model Toxics Control Act (MTCA) (Ecology 2001) guidance. Reducing the number of process options does not preclude reexamination of other options during the remedial design/remedial action (RD/RA) phase of the cleanup project. Rather, it is a means to streamline the development and evaluation of the remedial alternatives (as described in Section 8) without sacrificing engineering flexibility. Representative technologies and process options used in the development of alternatives are shaded in Table 7-4.

Section 8 of this FS provides detailed descriptions of the technology types and process options that are assumed for cost estimating purposes under each remedial alternative.

7.1 Review and Selection of Representative Process Options

7.1.1 Dredging and Excavation

Removal is a common and frequently implemented general response action for sediment remediation nationwide and in the Puget Sound region. Mechanical dredging, mechanical excavator dredging, hydraulic dredging, and excavation using upland-based equipment (dry excavation) are the four representative process options available for removing contaminated sediments.

7.1.1.1 Removal Process Options

Mechanical Dredging

A mechanical dredge typically consists of a suspended or manipulated bucket that bites the sediment and raises it to the surface via a cable, boom, or ladder. The sediment is deposited on a haul barge or other vessel for transport to a disposal site. Under suitable conditions, mechanical dredges are capable of removing sediment at near *in situ* densities, with almost no additional water entrainment in the dredged mass and little free water in the filled bucket. Low water content is important if dewatering is required for sediment treatment or upland disposal.

Clamshell buckets (open, closed, hydraulic-actuated), backhoe buckets, dragline buckets, dipper (scoop) buckets, and bucket ladders are all examples of mechanical dredges. Clamshell dredges work best in water depths less than 100 feet (ft) to maintain production efficiency. Nominal bucket capacities (i.e., when full) for environmental applications typically range from less than 1 cubic yard (cy) to 10 cy. Clamshell buckets are most effective in consolidated sediments and are the devices of choice for sediments containing debris.

Environmental buckets, or specialty level-cut buckets, offer the advantages of a large footprint, a level cut, and the capability to remove even layers of sediment. A level-cut bucket reduces the occurrence of ridges and winnows that are typically associated with conventional clamshell buckets. Environmental buckets are effective in unconsolidated sediments. They are not effective when digging in heavier sand or where a significant amount of debris may be present.

Mechanical dredging results in sediment excavation with near *in situ* density (water content), thereby reducing the need for substantial ancillary facilities and equipment to process wet dredged material. Mechanical dredging tends to minimize water entrainment by maintaining much of the *in situ* sediment structure (water entrainment ratio of approximately two parts water to one part dredged sediment). Material tends to be dewatered on the barge and then can be transloaded, transported, and managed at permitted off-site facilities that are authorized to handle wet sediments (these facilities are available to projects in this region). As a result, upland sediment processing and water treatment facilities require less acreage to handle mechanically dredged sediments.

Hydraulic Dredging

Hydraulic dredges remove and transport dredged material as a pumped sediment-water slurry. Large debris is typically removed by clamshell buckets prior to hydraulic dredging of sediments. Then, sediment is dislodged into the water column by mechanical agitation, cutterheads, augers, or high-pressure water or air jets. In very soft sediment, it may be possible to remove surface sediment by straight suction or by forcing the intake into the sediment without first mechanically dislodging the sediment. The majority of the loosened slurry is then captured by suction from pumps into an intake pipe and transported through a dredge discharge pipeline to a handling/dewatering facility.

Hydraulic dredging requires substantial ancillary facility acreage (e.g., approximately 26 acres were utilized for Fox River Operable Unit 1 remediation) and equipment to process dredged sediments (dewatering) and to treat the wastewater before discharge. Hydraulic dredging entrains tremendous volumes of water, typically at 8 to 10 parts water to 1 part dredged sediment. As a result, the upland area requirements to support sediment and water handling for hydraulic dredging are significantly greater than for mechanical dredging to handle the same volume of dredged sediment. In addition, the facilities handling the slurry need to be placed as close as possible to the dredging operations to enable pumping from the site to occur effectively.

Land available to site sediment processing equipment adjacent to the LDW is limited and consists mostly of small parcels (i.e., less than 5 to 10 acres). Areas large enough to site a facility capable of dewatering hydraulically dredged sediment with meaningful dredging production rates are not available. Hydraulic dredging may be viable for location-specific circumstances where the total volume of water generated is relatively small and controllable.

A prime example is using a diver-operated, hand-held, hydraulic dredge to remove materials under or around piers, pilings, or in other under-structure places where conventional dredging equipment is unable to reach. Using this technology, an otherwise unreachable location may be feasible to dredge, depending on circumstances. However, one must consider the diver's limited visibility, the overall safety of the diver potentially exposed to physical hazards and resuspended contaminants, and the reduced production rate compared to overall project volume requiring removal. As with other hydraulic dredges, the presence of debris limits the effectiveness of a diver-operated hydraulic dredge. Because under-pier areas typically include riprap and debris, incomplete removal of contaminated sediments can be expected even with a diver-operated hydraulic dredge, and thus capping would likely still be required following dredging.

Dry Excavation

Dry excavation using barge-mounted or upland-based precision excavators refers to the removal of sediments in the absence or limited presence (e.g., a few feet) of overlying water. This involves removing intertidal sediment under naturally-occurring low-tide (exposed) or shallow-water conditions. The fixed-arm, articulated arrangement of the precision excavators pushes the bucket into the sediment to the desired cut level without relying on the weight of the bucket for penetration. Engineered dewatering of an excavation area can also be undertaken to enable dry excavation. Dewatering methods include the use of earthen dams or sheet piling, often in combination with dewatering pump operations.

Upland-based removal of sediment using precision excavators can be employed on exposed shoreline and intertidal areas during low-tide conditions where access is feasible. To avoid the need for extensive upland dewatering treatment facilities, this FS assumes that upland-based excavation is limited to elevations above –2 ft mean lower low water (MLLW) during low-tide conditions, and where access is practicable.

7.1.1.2 Dredge Residuals

All in-water removal operations result in the release of a portion of the contaminants in the material being dredged and will leave behind some level of residual contamination in the sediment after dredging is complete (USACE 2008a). Resuspension of sediments occurs when a dredge and associated operations dislodge bedded sediment particles and disperse them into the water column. These resuspended sediments either settle back near the point of dredging (known as "residual" contamination), or are transported by currents farther afield (known as "release"). Releases also occur as a result of dissolution of contaminants into the water column and, in some cases, through volatilization. Resuspension during dredging is affected by factors such as the type and size of dredging equipment, level of operator skill, positioning of equipment used during dredging, dredge sequencing, depth of dredge cut, type and volume of debris encountered, and the substrate type and bottom topography. Resuspension, residuals, and releases can be estimated and monitored.

Resuspension, releases, and residual contamination can result from various causes that can be grouped into two categories:

- Undisturbed residuals are contaminated sediments found at the post-dredging surface that were not fully removed. The causes of undisturbed residuals include:
  - Incomplete characterization of depth-of-contamination in the remedial design, resulting in previously undocumented contaminated sediment being left in place.
  - Inaccuracies in meeting target dredge design elevation, resulting in contaminated sediment being left in place.
  - Furrows or ridges created by incomplete horizontal removal also leaving contaminated sediment in place.
- Generated residuals are contaminated post-dredging surface sediments that are dislodged or suspended by the dredging operation and subsequently redeposited on the bottom of the water body. Causes include:
  - Material resuspended by the bucket (mechanical dredging) during its bite or by the dredge cutterheads (hydraulic dredging) during its pass.
  - Material resuspended outward by the auger or cutterhead beyond the influence of the pump suction and left behind.

- Vertical positioning of the auger or cutterhead at too great of a cut depth, resulting in material riding over the dredge head.
- Material adhering to the outside of the bucket and washed off on its upward travel through the water column, then settling back down to the bottom.
- Material dripping from a partially closed or overfilled bucket on its upward travel through the water column, then settling back down to the bottom.
- Turbid flow or sloughing of material from steep cut banks spreading sediment from adjacent areas on top of areas where dredging was completed.
- Release of sediment contaminants dissolved in porewater when sediment is disturbed during dredging.

The nature and extent of dredging residuals dislodged or suspended by a dredging operation are not easily predicted. Most projects have based their post-dredging residual concentration by monitoring a specified surficial sediment thickness (e.g., 0 to 10 centimeters [cm] below mudline). By comparing the monitored thickness to the average concentration in the final production cut profile, it is possible to estimate the amount of residuals that will be generated by the project (USACE 2008a). Palermo and Patmont (2007) performed mass balance calculations for 11 project sites, estimating that generated residuals represented approximately 2 to 9% of the mass of contaminant dredged during the last production cut. The available data suggest that multiple sources contribute to generated residuals, including resuspension, sloughing, fall back, and other factors. However, on a mass basis, sediment resuspension from the dredge operations appears to explain only a portion of the observed generated residuals, suggesting that other sources such as cut slope failure and sloughing could be quantitatively more important.

The study also indicated that the presence of hardpan/bedrock, debris, and relatively low dry density sediment results in higher generated residuals.

Numerous case studies have shown that the spatial extent of dredge residuals can extend beyond the footprint of the dredge prism. For this reason, residuals monitoring and management provisions will be included in the remedial design phase that address adjacent areas as well as the dredge prism.

Dredge monitoring studies conducted over the last 13 years have estimated the rate of resuspension at 2 to 5% of polychlorinated biphenyls (PCBs) by mass downstream (or as residuals) compared to the mass of material contained in a dredge prism. Most of the release is in the bioavailable dissolved form (USACE 2008a; TetraTech 2010a, Fox River; Connolly 2010, Hudson River; Steuer 2000; Anchor QEA and ARCADIS 2010). Some loss of material is expected at all dredging sites regardless of the specific dredging process options, engineering controls (e.g., silt curtains, barriers), and best management practices used during dredging. Estimates of sediment export downstream of the LDW from resuspension during dredging are presented in Section 9.1.2.3 and Appendix M, Part 2.

7.1.1.3 Recent Developments in Dredge Positioning Technology

Recent introduction and widespread use of real-time kinematic differential global positioning systems (RTK–DGPS), coupled with radio telemetry and data logging technology, have greatly improved the accuracy and operational flexibility of mechanical dredging. The latest generation of precision dredge and bucket guidance systems integrate RTK-DGPS, excavator and bucket inclinometer sensors, vessel motion sensors, electronic heading, and tide data to enable dredging accuracy generally to within less than 6 inches. Dredge operators are now able to visualize the location of the bucket cutting edge in relation to the target elevation, the bucket open/close status, and the horizontal position of the bucket through use of these advanced positioning and monitoring systems.

7.1.1.4 Dredging and Excavation Technology Summary

Mechanical dredging and excavation, the most commonly practiced forms of sediment removal in the Puget Sound region, are adopted in this FS as the representative primary removal process options for in-water work. Dry excavation using conventional earth-moving equipment is also retained for use in intertidal and embankment areas, but it is expected to be implementable only for a low percentage of the removal volume because of access limitations.[1] Representative dredging projects in the Puget Sound region are identified in Table 7-5. As shown in Table 7-5, approximately 90% of the projects completed in the Puget Sound region adopted mechanical dredging during implementation.

Mechanical dredging and excavation were selected as the primary in-water removal technologies because several factors within the LDW favor these over hydraulic dredging:

- The LDW is a working industrial waterway and significant amounts of debris may be present in the sediments, the result of approximately 100 years of commercial and industrial activity. The presence of debris is a significant problem for hydraulic dredging. Although mechanical dredging is also adversely affected by debris, it is better suited to manage and accommodate debris removal.

[1] Details regarding the range and type of dredge equipment available within the local/regional construction community are presented in Section 8 and Appendix I. Cost estimates prepared and presented in Appendix I are based on mechanical dredging, and barge-mounted excavators.

- Two Subtitle D landfills in the region are permitted to accept wet sediment generated from mechanical dredging (see Section 7.1.3), thereby avoiding the need to dewater mechanically dredged solids.
- The environmental dredging literature contains no documented quantitative evaluations that distinguish between the resuspension and recontamination characteristics of mechanical and hydraulic dredging under other than ideal debris-free site conditions (USACE 2008a).

The assumption of mechanical dredging and excavation for development of remedial alternatives does not preclude other options from being considered during remedial design.

For the FS, partial dredging (diver-operated hydraulic dredging) and capping are assumed as the representative primary process option for under-pier work (see Section 7.1.4) because full removal of contaminated sediment is often difficult in these areas. Under-pier areas have limited access, limited maneuverability, accumulated debris, and riprap structures. This assumption does not preclude other process options from being considered during remedial design. For example, a design decision could be made to remove a pier deck to allow access for mechanical excavation or capping, or to adopt diver-operated hydraulic dredging, or to apply a spray cap.

7.1.2 Treatment Technologies

Treatment technologies can potentially be applied to in-place sediment (*in situ* treatment) or to sediment after it has been physically removed from the aquatic system (*ex situ* treatment). The CTM (RETEC 2005) presented a detailed evaluation of treatment technologies and their applicability for sediment cleanup in the LDW. This section provides updated information about innovative technology developments and relevant experience at other cleanup sites. The CTM also reviewed the extensive regulatory and industry efforts in Washington State and elsewhere to determine the viability of treatment in the context of centralized sediment management facilities. The following discussion reviews viable *in situ* and *ex situ* treatment approaches and their applicability to the LDW.

In situ treatment options with potential applicability to the LDW are physical immobilization by amendment of materials to enhance sorption capacity of the natural sediments. To date, *in situ* treatment of sediments has been mostly by amendment of activated carbon or organoclays in pilot and full-scale sediment remediation projects.

In situ treatment techniques are less energy-intensive, less expensive, and less disruptive to the environment than conventional treatment technologies, and they can reduce ecosystem exposure by binding contaminants to organic or inorganic sediment matrices. The contaminant sorption capacity of natural sediments may be modified and enhanced by adding such amendments as activated carbon for adsorption of non-polar organics and certain metals such as mercury (various activated carbon products are available as powder, granules, or pellets, each with different sediment application characteristics); natural minerals such as apatite, zeolites, or bauxite and refined minerals such as alumina/activated alumina for sequestration of metals/metalloids; ion exchange resins (organoclays) for replacement of metals/ inorganic contaminants with amines or other functional groups; zero-valent iron for dechlorination of PCBs; and lime for pH control or degradation of nitroaromatic compounds. Multifunctional amendment blends may also be used to address complex contaminant mixtures in sediments, and subsequently may enhance overall sorption capacity. Usually activated carbon serves as the backbone (for hydrophobic partitioning) and either is impregnated with the target amendment or blended in a briquette-like composite using an appropriate and non-toxic binder (e.g., clays or other binder materials; Ghosh et al. 2011). Amendments can be engineered to facilitate placement in aquatic environments, by using an aggregate core (e.g., gravel) that acts as a weighting component and resists resuspension, so that the mixture is reliably delivered to the sediment bed, where it breaks down slowly and mixes into sediment by bioturbation.

One of the most advanced *in situ* treatment technologies in terms of its state of development is amending sediment with activated carbon. This treatment has the effect of adsorbing hydrophobic contaminants, reducing porewater contaminant concentrations, and reducing their bioavailability for uptake by benthic organisms. Direct placement of activated carbon to sediments has now been demonstrated in a wide range of bench-scale and pilot studies, and successfully deployed in large field efforts with promising documented monitoring results (Ghosh et al. 2011). Activated carbon has proven effective in reducing the bioavailability of a range of sediment contaminants, including PCBs, polycyclic aromatic hydrocarbons (PAHs), dioxins, DDT, and mercury. However, while the pilot studies are starting to provide valuable information, further research is needed to understand both transient and long-term changes that take place naturally in the environment, and also demonstrate the application of activated carbon at full-scale contaminated sediment areas. Further discussion of this technology is presented in Section 7.1.2.1.

Ex situ treatment options with potential applicability to the LDW are conventional soil washing/particle separation, advanced soil washing (Biogenesis™), solidification, and thermal treatment. To date, *ex situ* treatment of sediments, while a subject of considerable interest nationwide, has been mostly limited to soil washing and air (steam injection) stripping in full-scale sediment remediation projects.

Technologies that destroy or detoxify contaminants have been accepted at very few projects (e.g., Bayou Bonfouca) for cleanup at contaminated sediment sites for two reasons. First, it is difficult to balance treatment costs with a beneficial reuse outlet for the material; and second, upland and in-water disposal alternatives are much less expensive, particularly in this region. With the exception of the addition of cement-type materials to reduce free water content and mobility prior to upland disposal, only one contaminated sediment remediation project in this region (Area 5106 at Hylebos

Waterway in Commencement Bay) has utilized treatment (see Section 7.1.2.2) or incorporated beneficial reuse of treated sediments.[2]

7.1.2.1 Direct Amendment with Activated Carbon or Organoclays

The goal of *in situ* treatment, by amending or thin capping the bioactive surface layer of sediment, is to reduce the bioavailability of hydrophobic organic contaminants. The two most common material classes for amendment are activated carbon and organoclays. The transfer of organic contaminants such as PCBs from the sediment to the strongly binding activated carbon particles not only reduces contaminant concentration and the bioavailability to benthic organisms but also reduces contaminant flux into the water column, and thus accumulation of contaminants in the aquatic food-chain (Ghosh et al. 2011). Of the two amendments, activated carbon has received more testing and evaluation than organoclays, particularly with respect to sediment remediation, because the sorption capacities for PCBs and PAHs in activated carbon are at least an order of magnitude higher than in the other sorbents (Ghosh et al. 2011). Organoclays have received attention largely in the context of addressing localized deposits of dense non-aqueous phase liquids (DNAPLs; Bullock 2007, Reible and Lampert 2008).

Extensive bench-scale studies have confirmed the effectiveness of activated carbon for *in situ* treatment. For example, average doses of 2 to 4% (by dry sediment weight) of activated carbon applied to surface sediments have resulted in reductions greater than 95% in PCB bioavailability and sorption capacities of the activated carbon have been retained for as long as the bench-scale studies were continued (up to 10 years in some studies). Based on promising laboratory results, beginning in 2006, several pilot-scale field demonstrations of activated carbon placement were implemented in the United States and Norway (see Figure 7-1). These projects show how various engineering challenges were met for applying activated carbon and monitoring of its long-term effectiveness:

- Hunter's Point Naval Shipyard (San Francisco, CA), conducted in 2006, in estuarine application to address PCBs and PAHs
- Lower Grasse River (Massena, NY), conducted in 2006, in freshwater application to address PCBs
- Trondheim/Grenlandsfjords Harbors (Norway), conducted in 2006, in estuarine application to address PCBs, PAHs, and dioxins
- Grenlandsfjords Harbors (Norway), conducted in 2009, in estuarine application to address dioxins and furans

[2] Treatment to eliminate free liquids from dredged sediment is no longer required by two regional landfills servicing the Puget Sound area (see Section 7.1.3.2).

- Bailey Creek, U.S. Army Installation (VA), conducted in 2009, in freshwater wetland application to address PCBs
- Canal Creek (Aberdeen Proving Grounds, MD), conducted in 2010, in freshwater application to address mercury, PCBs, and DDT.

The primary objective of these demonstration projects was to verify that the bioavailability of PCBs, PAHs, DDT, dioxins/furans, and/or mercury can be effectively reduced at the field scale by placing activated carbon into surface sediments. While the specific approaches varied for each pilot project listed above, most of the projects focused on the following:

1) Evaluate efficient, low-impact delivery systems of activated carbon for amendments into in-place sediments (using large-scale equipment and a range of application methods).

2) Determine the extent of sediment resuspension and contaminant release during application.

3) Assess persistence, binding potential, and small-scale spatial variations of the activated carbon after application to sediments in the natural environment, and also assess mixing of activated carbon over time as a result of bioturbation processes.

4) Evaluate short- and longer-term changes in contaminant porewater concentrations, sediment-to-water fluxes, desorption kinetics, and/or equilibrium partitioning from sediments that result from activated carbon amendment.

5) Measure short- and long-term changes in contaminant bioavailability by biomonitoring deposit-feeding benthic organisms after applying the activated carbon amendment.

6) Evaluate activated carbon-sediment stability and erosion potential over time.

7) Evaluate contaminant bioavailability for uptake, transfer, or any changes to the benthic and/or submerged aquatic plant communities, as a result of activated carbon amendment.

Several types of activated carbon applications were evaluated at these sites, including slurry amendment (water and/or native clay mixtures) on top of the sediment surface, mixing or injection of slurry amendments into surface sediments, and pelletized applications (e.g., SediMite®, AquaBlok®).

The period over which ENR/*in situ* treatment remains effective will be an important consideration during remedial design. Design life will need to be evaluated at the location-specific level and will likely influence decisions on the type (e.g., source and type of carbon), amount of amendment used (i.e., design safety factor), and the potential need for replenishment. Physical stability and chemical activity (e.g., adsorption capacity) over the long term are the most important design life factors. Activated carbon and other charcoals created under high-temperature conditions are known to persist for thousands of years in soils and sediments, and both laboratory studies and modeling evaluations indicate promising long-term physical stability of the amendment material and chemical permanence of the remedy (Ghosh et al. 2011). Empirically-derived contaminant concentration data and modeling simulations show that *in situ* treatment can reduce bioavailability over the long term where contaminant loading (mass transfer) from groundwater, surface water, and newly deposited sediments is low.

The FS assumes that half of the ENR footprint would warrant amendment with a material such as activated carbon for *in situ* treatment. This assumption provides a basis for estimating costs and comparing the remedial alternatives; however, during remedial design, the emphasis on ENR or *in situ* treatment will depend on location-specific factors and additional testing of the implementability of these technologies. The composition of ENR/*in situ* treatment will depend on additional evaluation during remedial design; it may include carbon amendments, habitat mix, and/or scour mitigation specifications to increase stability and enhance habitat.

The following sections provide synopses of two of the most relevant field demonstrations.

Hunter's Point Naval Shipyard (San Francisco, California) – Carbon Amendment

Beginning in January 2006, a large field demonstration of activated carbon via direct amendment was conducted in a shallow tidal flat of the South Basin adjacent to the former Naval Shipyard at Hunters Point, in San Francisco Bay (CA) (Luthy 2005, Luthy et al. 2009, Cho et al. 2009). The former Navy installation was predominantly used for ship repair and maintenance, which resulted in the release of PCBs to the environment. The activated carbon was applied to two test plots (D and F) with a surface area of 34.4 m^2 each, located within the intertidal region of the former shipyard, and away from the shoreline. Two more plots (C and E) served as control and reference plots. A barge-mounted rotovator system (for plot D) and a crawler-mounted slurry injector system (for plot F) were used to mix activated carbon directly into the surface sediments at a target mixing depth of 30 cm below the mudline, to include the biologically active zone.

Baseline and post-amendment monitoring field assessments were conducted in December 2005, July 2006, July 2007, and January 2008, respectively. These assessments were performed to characterize surficial sediment concentrations, analyze the water column, test uptake, and study bioaccumulation. Prior to treatment, the PCB concentration in sediment among the plots varied between 1,350 and 1,620 micrograms per kilogram dry weight (µg/kg dw). Mixing of activated carbon into surface sediments was assessed using black carbon measurements. The measured activated carbon dose averaged 2.0 to 3.2% by dry sediment weight and exhibited small-scale spatial variability. The uneven activated carbon distribution was possibly induced by the unidirectional mixing motion of the large mechanical mixing devices, the relatively small dimensions of the test plots, and insufficient mixing time. In terms of variability, Plot F showed higher variability than Plot D, indicating that activated carbon-mixing via the slurry injection device on Plot F was less homogeneous than the rotovator device employed at Plot D. Ineffective homogenization of the activated carbon into the sediment would influence the short- and long-term performance of the technology.

No adverse impacts, such as sediment resuspension and PCB release, were observed in the water column over the treatment plots as a result of applying the activated carbon and mechanically mixing it into the sediments. In addition, the activated carbon amendment did not impact the structure of the macro benthic community (composition, richness, or diversity) (Luthy et al. 2009, Cho et al. 2009).

Both *in situ* clam bioassay and *ex situ* bioavailability for uptake studies confirmed that PCB bioaccumulation was reduced; an approximate 78% tissue concentration reduction in bioavailability was achieved when clams were exposed to sediment treated with an average 3.4% activated carbon. Although the *in situ* bioassay results were sometimes influenced by field conditions resulting from newly deposited sediment, heat stress, and shallow burrowing depth, the reduction in bioavailability was consistent with the results of earlier laboratory studies (Millward et al. 2005; McLeod et al. 2007, 2008). Reductions in congener bioaccumulation with activated carbon were inversely related to the congener octanol-water partitioning coefficient (K_{ow}), suggesting that the efficacy of activated carbon is controlled by the mass-transfer rate of PCBs from sediment into activated carbon (Millward et al. 2005). The semi-permeable membrane devices (passive samplers) were used to show that PCB uptake in activated carbon-treated sediment was reduced by 50%, with similar results in porewater. This reduction was evident 13 months post-treatment and even after a subsequent 7 months of continuous exposure, indicating activated carbon treatment efficacy was retained for an extended period (Cho et al. 2009). Although reductions in aqueous PCB concentrations in equilibrium with the sediment following activated carbon-amendment often correlate with reduced PCB bioaccumulation, the reduced availability of contaminants from ingestion of sediments appeared to be the actual cause of lower tissue concentrations (Janssen et al. 2010, 2011).

The two activated carbon-treated plots showed decreases in the fraction of PCBs desorbed with an increasing dose of activated carbon, which supports the finding of reduced PCB availability after activated carbon application. After 18 months, the field-exposed activated carbon demonstrated a strong stabilization capability to reduce aqueous equilibrium PCB concentrations by almost 90%. These results are promising and suggest the long-term effectiveness of activated carbon in the field (Luthy et al.

2009, Cho et al. 2009). Finally, based on the absence of significant differences between the 6-month and 18-month total organic carbon (TOC) values measured in cross sections of sediment cores taken from Plots D and F for sediment stability testing and based on hydrodynamic modeling, it was concluded that mixing activated carbon into cohesive sediment at selected locations within the South Basin at Hunter's Point neither reduced surface sediment stability nor resulted in significant erosion of treated sediments (Zimmerman et al. 2008). Surficial sediment of the two activated carbon-treated plots contained less black carbon/TOC 24 months after treatment, which was explained by continued sediment deposition.

Lower Grasse River (Massena, New York) – Carbon Amendment

Similar pilot field studies were initiated in September 2006 to evaluate the ability to deliver activated carbon slurries to in-place sediments and assess the effectiveness of this approach in reducing the bioavailability of PCBs in sediments and biota in the Lower Grasse River in Massena (NY). Alcoa Inc., with oversight from EPA, implemented the pilot demonstration project, which began with laboratory studies and land-based equipment testing, continued with field-scale testing of alternative placement methods, and culminated in a field demonstration of the most promising activated carbon application and mixing methods in a 0.5-acre pilot area within the Lower Grasse River (Alcoa 2007, EPA 2007b).

Based on the results of initial laboratory studies that evaluated bioavailability reductions achieved at different activated carbon doses, a target application concentration of 2.5% activated carbon (dry-weight basis) in the top 15 cm of sediment after treatment was used in the Lower Grasse River field demonstration. Three application techniques were implemented within the pilot study area as follows:

- A 7-ft by 12-ft enclosed device first applied (sprayed) the activated carbon slurry onto the sediment surface. The material was then mixed into near-surface (0 to 15 cm) sediments using a rototiller type mechanical mixing unit (tiller).
- A 7-ft by 10-ft tine sled device (tine sled) used direct injection of activated carbon into the upper 15 cm of the sediments.
- Application of activated carbon to the sediment surface using the tiller, but with the mixing devices removed. Monitoring of this "unmixed" treatment area allowed for an evaluation of the rate and extent of incorporation of the surficial layer of placed activated carbon into near-surface sediments over time through natural processes (e.g., bioturbation).

Baseline (summer 2006), construction (fall 2006), and post-construction (2007, 2008, and 2009) monitoring were conducted (Alcoa 2010). Water quality action levels for PCBs (0.065 micrograms per liter [μg/L]) were not exceeded adjacent to or downstream of the pilot project area during activated carbon application. Similarly, turbidity levels during construction never approached the action level of 25 nephelometric turbidity units (NTUs) above background. Turbidity measured downstream of the pilot project area was only slightly higher than that measured upstream, with average turbidity and total suspended solids (TSS) increases of roughly 0.2 NTU and 0.8 milligrams (mg)/L, respectively. The water column monitoring data indicated that construction activities did not have a significant impact on water quality in the river, and further suggested that silt curtains are not needed for either the tine sled or tiller equipment.

Sediment cores were collected immediately following the fall 2006 application and in the three post-construction monitoring years (2007, 2008, and 2009) and were analyzed for black carbon to verify the applied dose. The target dose of 2.5% activated carbon (dry weight basis) in the top 15 cm of sediment was achieved in nearly all test plots. Compared with the tine sled, application of activated carbon using the tiller (with or without mixing) resulted in greater small-scale spatial variability in activated carbon levels.

A detailed 3-year post-implementation physical, chemical, and biological monitoring program (i.e., 2007 through 2009) was completed to evaluate the long-term effectiveness of the activated carbon treatment. Monitoring results are summarized below:

- Measurements of activated carbon levels in the treated sediments (i.e., based on black carbon analysis and microscopy results) confirm that the applied carbon has continued to remain in place. Levels are based on mass balance calculations of activated carbon applied in 2006.

- Most of the activated carbon in the treatment areas was applied within the upper 10 cm of the sediment, declining to background levels at approximately 20 cm below the mudline. The 2008 and 2009 monitoring revealed that the activated carbon was slightly deeper in the sediment profile than observed in 2006 post-construction and 2007 sampling, due to natural sedimentation occurring on top of the activated carbon-treated sediments since 2006.

- PCB bioaccumulation in the tissue of test organisms (whole body worms; wet weight basis) in the activated carbon treatment areas was reduced in excess of 80% for the *in situ* tests and in excess of 90% for the *ex situ* tests. Greater than 90% reductions in porewater PCB concentrations were also observed in the test plots. PCB bioavailability was reduced even further over the 3-year post-construction monitoring period due to a combination of improved mixing (bioturbation) of activated carbon in surface sediments, and site-wide natural recovery over time.

- Batch equilibrium testing to evaluate the effect of activated carbon on PCB partitioning between the sediment and water phases showed reductions in the range of 93 to 99%.

- Two trends were observed in the results from *in situ* passive samplers deployed on top of the treatment areas:
  - Ambient PCB sediment levels declined from 2006 to 2009 (as a result of site-wide natural recovery); and
  - Aqueous PCB concentrations at the sediment surface in the treatment areas decreased by 90% (similar to reductions observed from biological testing), and even in 2009, the treated sediments continued to act as a "sink" for water column PCBs in the river (i.e., net flux of PCBs from surface water to sediments).
- Results of ecological monitoring activities show a benthic community adapted to fine-grained sediments both pre- and post-carbon application. Benthic habitat and community composition measures were similar (not statistically different) between the treatment areas and upstream background locations, suggesting that activated carbon application did not affect the benthic community. Additional studies of potential impacts to submerged aquatic vegetation at high activated carbon doses are ongoing.
- Erosion potential testing indicated that treated sediments had a slightly higher erosion potential than pretreatment sediments, but nevertheless were within the range of historic data for native sediments.

In summary, the Lower Grasse River pilot project demonstrated that activated carbon can be successfully applied to river sediments with minimal impact to water quality within the river. Post-construction monitoring revealed that the activated carbon is stable in the fine sediments and has significantly reduced PCB bioavailability. Batch equilibrium experiments showed that aqueous phase PCB concentrations in surface sediments have been reduced on average by more than 95% at activated carbon doses of 2% or greater. *In situ* and *ex situ* biological uptake studies showed 80 to 90% reductions with an activated carbon dose greater than 2%.

7.1.2.2 Soil Washing with Air Stripping

Soil washing can be classified as conventional or advanced form of *ex situ* treatment. Conventional soil washing is a form of primary treatment that uses conventional and readily-available material handling unit processes to separate sediment particles, typically into coarse (sand and gravel) and fines (silt and clay) fractions (Figure 7-2). This treatment process separates the sediment particles using conventional equipment. These equipment systems have been derived largely from the mining and mineral processing industries, and include screening, gravity settling, flotation, and hydraulic classification (e.g., using hydrocyclones) (USACE-DOER 2000). Advanced soil washing, such as Biogenesis™, combines the physical separation aspects of conventional soil washing with additional treatment such as agitation, or the addition of surfactants, chemical oxidants, or chelating agents to the finer fraction of material.

Soil washing is a wet process and therefore generates wastewater that requires treatment and discharge. Depending on site conditions, the washed coarse fraction may be suitable for in-water placement (see Section 7.1.3.4 for beneficial uses of sediment) as a cap, enhanced natural recovery (ENR), or habitat creation/restoration medium. The finer fraction, which has higher concentrations of contaminants, is typically dewatered, transported, and disposed of in a permitted upland landfill. Ideally, the net outcome of soil washing is a reusable coarse fraction and a reduced volume of contaminated material requiring additional treatment or direct disposal.

Sediments in portions of the LDW may be sufficiently coarse-grained to consider soil washing as a potentially viable treatment. One vendor has indicated that soil washing has the potential to be economical where the sediment contains greater than 30% sand (Boskalis-Dolman 2006). When the sediment contains less than 30% sand, treatment performance and economics deteriorate. Other factors affecting the economics and implementability of soil washing are:

- Physical and chemical properties of the sediment.
- Availability of an upland location for transloading sediment from barges.
- Availability of an upland location for sediment containment, storage, and operation of the soil washing facility. Although this facility may or may not be located at the transloading facility, this FS assumes that it will be located within the transloading facility footprint for the purpose of cost estimating.
- Disposal costs for the fines fraction.
- Ability to commit to long-term (and continuous) high-volume sediment throughput (economies of scale).
- Ability to reuse washed coarse fraction beneficially and at low cost.

The last two factors are the most difficult to reconcile in a manner that promotes economic viability.

The following sections describe conventional and advanced soil washing techniques recently used at several sites.

Area 5106, Hylebos Waterway, Commencement Bay (Washington) – Soil Washing

Unlike other parts of the Hylebos Waterway cleanup, the sediments at Area 5106 were treated before confined disposal (EPA 2004). The non-time critical removal action was conducted by Occidental Chemical Corporation at its former chlor-alkali plant facility along the Hylebos Waterway. About 36,000 cy of contaminated sediments containing volatile organic compounds and semivolatile organic compounds were hydraulically dredged and pumped to an upland treatment system. Treatment consisted of aeration and air stripping to separate out the volatile organic compounds (VOCs), which were, in turn, adsorbed onto activated carbon. The treated slurry was dewatered and the dewatered sediments were disposed of in the Blair Slip 1 confined disposal facility, because treated materials still contained relatively high concentrations of semivolatile organic compounds and metals.

Raritan River, Arthur Kill, and Passaic River (New Jersey) – Soil Washing

Biogenesis™ is an advanced soil washing process that was used in a recently completed full-scale demonstration, which treated approximately 15,000 cy of contaminated sediments from the Raritan River, Arthur Kill, and Passaic River, New Jersey (Biogenesis 2009, Malcolm Pirnie 2007). The Biogenesis™ process combines the physical separation aspects of conventional soil washing with high-pressure agitation, surfactants, chemical oxidants (e.g., hydrogen peroxide), and chelating agents. This process uses equipment including but not limited to: truck-mounted washing units, sediment processor, sediment washing unit, hydrocyclones, shaker screens, water treatment equipment, tanks, water blasters, compressors, and earth moving equipment.

Important Biogenesis™ process steps include:

1) Dredged sediment is screened to remove oversized material and debris before transfer to the holding tanks.

2) High-pressure water, proprietary solvent, and physical agitation are combined to separate contaminants from the solids.

3) Treated sediment is then dewatered using a hydrocyclone and centrifuge. Some effluent water may be recycled through the system, but significant quantities of wastewater are generated that require treatment and disposal.

The process results in residual waste products, including sludge and organic material, which require disposal at a regulated landfill. Depending on the nature of the sediment and cleanup levels required, the sediment washing process may need to be repeated for multiple cycles.

The Biogenesis™ proprietary process is designed to separate and to destroy organic contaminants partially (through oxidation); metals are conserved but concentrated in the fines fraction. Results for treated sediment from the three different dredged material sites demonstrated reductions in dioxin concentrations (from 517 nanograms toxic equivalent (ng TEQ)/kg dw prior to treatment to 71 ng TEQ/kg dw post treatment). While this washing technology achieved some measure of contaminant reduction, this appears to have been attributable primarily to solubilization of contaminants and separation of fine solids, rather than because of contaminant destruction through the cavitation/oxidation process. The mass of fine solids lost to the wastewater stream (centrate solids) ranged from approximately 9 to 18% of the incoming sediment mass, although dissolved concentrations were not evaluated (USACE 2011). Only slight decreases in PCB concentrations were documented (450 µg/kg dw prior to treatment and 380 µg/kg dw post treatment) (Biogenesis 2009). PAHs were not effectively removed or destroyed because of adsorption to, or sequestration within, the organic material mixed with the sediment. PAH concentrations in the treated sediment were approximately 52% of concentrations in the incoming sediment for the bench tests. Total PAH mass presumed destroyed or unaccounted for in the overall process ranged from zero to 49.9% (USACE 2011). Approximately 13,000 tons of processed dredged material was loaded onto trucks and transported off site for beneficial reuse as fill material.

Fox River (Wisconsin) – Soil Washing/Sediment Processing

In 2009, approximately 540,000 cy of PCB-contaminated sediments at Fox River (Operable Unit 1) were hydraulically dredged and pumped through a pipeline to a sediment processing facility equipped with particle-size separation, dewatering, and water treatment equipment (i.e., equivalent unit operations used in conventional soil washing). The sediment slurry passed over a vibrating screen enabling <0.5-inch material to pass through. The sand fraction of the slurry was then separated from the silt and clay fractions using a 150-micrometer (μm) coarse sand separation unit. The sand was polished in an up-flow clarifier, gravity dewatered, and temporarily stored on site for potential reuse. Average PCB concentration of dredged material was approximately 1,900 μg/kg dw (EPA 2009c). Total PCB concentrations in the treated sand fraction were on the order of 300 μg/kg dw.

The remaining fine grained sediment (<60 μm) was mechanically filter-pressed to dewater it. The resulting filter cake, typically containing between 1,000 and 10,000 μg/kg dw total PCBs was then land-filled. Process wastewater was treated by sand-filtration and granular activated carbon adsorption. Treated water was returned to the Fox River. Discharge water was monitored for PCBs, mercury, lead, pH, ammonia, biochemical oxygen demand, and TSS.

It is important to note that the process used at Fox River does not destroy organic contaminants. Further, while one of the project goals was beneficial reuse of the processed sand fraction, the sole beneficial reuse to date for this material was using a portion of the sand fraction as fill material (spread in the upland portion of the project site) and as a fill behind the sheetpile bulkhead wall constructed at the site. No beneficial uses outside of the project have been identified (TetraTech 2010a).

Hudson River (New York) – Soil Washing/Sediment Processing

Phase 1 of the dredging operations was conducted at the Hudson River during 2009 (Anchor QEA and ARCADIS 2010). Mechanical dredges with environmental clamshell buckets were used to remove approximately 278,000 cy of river sediments. Dredged material was transported by barges to a shore-based processing and transportation facility. Approximately 370,000 tons of PCB-contaminated sediments were processed to separate size fractions and dewater the solids in a similar fashion to that described above for the Fox River project. As a first step in processing the dredged material, debris and rock were removed and dredged sediments were processed through trammel screens and hydrocyclones to separate the material by size.

Approximately 40% of the sorted materials were fines and 60% were coarse material and wood. After coarse material separation, the slurry of fine sediments was mixed with a polymer in a gravity thickener and filter-pressed. Segregated debris and coarse solids and filter cake removed from the filter presses were temporarily stored in staging areas prior to rail transport and disposal at a permitted facility in Texas. Residual contaminant concentration in the coarse material precluded beneficial reuse of this material. All fractions of dredged material (debris, coarse, and fine) were therefore transported to and disposed of at a permitted facility in Texas. The fine fraction was separated from the coarse fraction and processed through mechanical dewatering to decrease the water content, thereby reducing the transport and disposal costs. A water treatment plant with the capacity to handle 2 million gallons of water per day was built to treat the water collected during the dewatering process. Treated water (approximately 88 million gallons per season) was discharged to the Champlain Canal.

Potential Environmental Review and Permitting Requirements

Permitting requirements for a prospective soil washing operation are currently undetermined and are dependent on the extent of the CERCLA and MTCA LDW site jurisdictional area. If the soil-washing location was determined to be on site, all substantive permitting requirements would be overseen by EPA and complied with as applicable or relevant and appropriate requirements (ARARs), and all procedural and environmental review requirements would be waived. The LDW site includes the upland areas (beyond the scope of this FS) that contributed contamination to the waterway; such upland areas would be considered "on site" for the purposes of siting a treatment facility. All necessary permits would need to be secured if the treatment location is not on site. Permits would also be required for any off-site disposition of treated CERCLA materials and waste streams, such as placement of treated material as off-site fill or off-site discharge of wastewaters to the King County sanitary sewer system.

7.1.2.3 Solidification

Solidification is a proven and effective *ex situ* technology that reduces the moisture content of dredged sediments and reduces the leachability (mobility) of metals. The process involves mechanical blending of the contaminated medium, in this case sediment, with an agent such as cement, cement kiln dust, or super-absorbent polymers. These agents react with moisture in the contaminated media and may produce a material that is much improved structurally (i.e., compressive strength) and can effectively reduce the leachability of contaminants. However, contaminants are not destroyed by solidification.

The major regional landfills (Allied Waste of Roosevelt, Washington, and Waste Management of Columbia Ridge, Oregon) are able to receive contaminated wet sediment at their sites in truck and rail containers (without requiring material to pass a Paint Filter Test [EPA 2008a]). These containers are lined to prevent loss of material (e.g., drainage) during transport.

Solidification does not adequately treat the COCs and solidified sediment would still require transport to a landfill for disposal. For this reason, solidification is not carried forward for alternative development in this FS, but it may be reconsidered during remedial design if moisture or leachability reduction is needed to comply with landfill operating permits.

7.1.2.4 Thermal Treatment

Thermal treatment involves the *ex situ* elevation of the temperature of dredged sediment to levels that either volatilize the organic contaminants (for later destruction in an afterburner) or directly combust the contaminants (e.g., incineration). A number of different system configurations and operating principles have been developed and are available in the marketplace, as described in the CTM. Thermal treatment systems are generally effective for destroying a broad range of organic compounds. Metals are not destroyed by thermal treatment systems.

Thermal treatment facilities are not available either locally or regionally. Therefore, dredged sediment would need to be transported out of state (either to Idaho or Utah) to utilize an existing facility. Alternatively, a temporary on-site (i.e., adjacent to the LDW) facility is technically feasible to consider. Implementability considerations include general siting considerations and obtaining local permits (e.g., air).

The primary drawback to thermal treatment is that treated sediment is unlikely to achieve metal concentration limits for beneficial reuse and may thus still require upland landfill disposal. Studies (e.g., toxicity testing) would also be needed to ascertain whether treated sediment would have properties suitable for supporting benthic productivity before in-water placement of the treated material would be allowed. Thermal destruction processes also require monitoring and management of air releases of hazardous constituents, such as dioxins/furans. Dioxins/furans can be created and released in air emissions from some thermal treatment processes, and fulfilling all substantive permit requirements for managing these air emissions can be difficult and can affect implementability of on-site thermal treatment.

Cement-Lock® Technology is a thermo-chemical manufacturing process that decontaminates dredged material and converts it into Ecomelt®, a pozzolanic material, which when dried and finely ground can be used as a partial replacement for Portland cement in the production of concrete. In the Cement-Lock® process, a mixture of material and modifiers is charged to a rotary kiln at high temperatures, which yields a homogeneous melt with a manageable viscosity. All nonvolatile heavy metals originally present in the sediment are incorporated into the melt matrix via an ionic replacement mechanism. The melt then falls by gravity into water, which immediately quenches and granulates it. The resulting material, Ecomelt®, is removed from the quench granulator by a drag conveyor.

Preliminary pilot-scale results have shown that organic contaminants are partially destroyed, and inorganics (e.g., metals) are encapsulated within the Cement-Lock®

matrix (i.e., Ecomelt®). Although the thermal technology is effective at destroying organic contaminants and immobilizing metals, some metals remain leachable (USACE 2011). The Cement-Lock® cement product passed the Toxicity Characteristic Leaching Procedure test for priority metals. The technology was recently demonstrated at a pilot-scale level for sediments dredged from the Stratus Petroleum site in upper Newark Bay (NJ) in 2006 and from the Passaic River (NJ) in 2006 and 2007. However, both demonstrations experienced equipment-related problems and were terminated (GTI 2008). In these studies, the Ecomelt product samples showed an average reduction in PCB concentrations from 2,800 µg/kg dw (pretreatment) to 0.2 µg/kg dw (post-treatment), with a PCB mass found in the off-gas stream of 0.01% of the incoming sediment PCB mass, for an overall 99.9% (not including the 30% of input mass adsorbed by the carbon bed) unaccounted for and presumed destroyed. The average reduction for 2,3,7,8-tetrachlorodibenzodioxin (TCDD) was from 0.17 µg/kg dw (pretreatment) to 0.008 µg/kg dw (post-treatment) (GTI 2008), with approximately 0.1% of the incoming total dioxin/furan mass being measurable in the Cement-Lock® product (USACE 2011). The fraction of metals leachable in the Ecomelt (Toxicity Characteristic Leaching Procedure [TCLP] mass/total metals in aggregate) ranged from zero to 20%, with average and median values of 3.0 and 0.28%, respectively. The fraction of metals leachable as a fraction of the total metals in the raw feed ranged from zero to 8.8%, with average and median values of 1.1 and 0.24%, respectively (USACE 2011).

Thermal treatment is not carried forward for further consideration in the FS because the process is unlikely to achieve the total metal concentration limits for beneficial reuse although a reduction in leaching potential could perhaps be achieved through use of one of the available technologies (e.g., Cement-Lock® technology).

7.1.2.5 Treatment Technology Summary

Application of activated carbon to sediments to reduce bioavailability is retained as a viable *in situ* treatment technology for the LDW. The technology can be considered in various ways from stand-alone applications to enhancements of other technologies (e.g., amending cap materials or incorporating into media used for ENR). Activated carbon amendment could also prove to be an essential tool of adaptive management (e.g., as a contingency action for underperforming remedial action areas).

Conventional soil washing/particle separation and advanced soil washing have sufficient merit to carry these processes forward in developing the LDW remedial alternatives. Soil washing is retained as an *ex situ* treatment process option because it has been applied at other contaminated sites in the United States and Europe, results in volume reduction of treated dredged material, and may result in a sand fraction suitable for beneficial use in the LDW, or possibly reduce or eliminate the cost of disposal for the sand fraction. Significant engineering design would be required to specify soil washing site location(s), special equipment needs (e.g., cyclones, filters, water treatment systems, etc.), operational procedures, and environmental review and permitting requirements to implement soil-washing treatment.

This FS assumes that soil-washing treatment would be located entirely within the transloading/dewatering facility and would consist of the following elements:

1) Physically wash the dredged material and separate the coarser grained (clean) sediment from the fine particle (contaminated) sediment.

2) Treat the wash water and discharge it to the LDW. Assume use of the following treatment train: collect and settle wastewater, flocculate, filter, analyze, and discharge.

3) Collect and stockpile the cleaned sediment in an on-site location separated from the soil-washing and wastewater treatment operations. Chemically analyze the sediment for COCs to confirm that remnant COC concentrations are less than sediment quality standards (SQS) or other applicable criteria and thereby are determined suitable for beneficial reuse.

4) Transfer the treated sands (processed material achieving target levels established for the project) off site and stockpile for assumed reuse as capping and ENR material for the project. Stockpile requirements need to address logistics and timelines for sand reuse. Specific requirements for sand quality and use need to be defined, including regulatory approvals.

5) Chemically analyze all remaining sediment to determine if treatment has magnified COC concentrations to be greater than landfill-designated hazardous waste concentrations.

6) Based on the chemical analytical results, load railcars with remaining sediment, transfer to the landfill, treat any excess wastewater, and dispose of the remaining sediment appropriately in either a Subtitle C or D landfill.

More advanced soil-washing technologies are not carried forward into the FS as the representative process option in the FS because conventional soil-washing techniques would likely produce the most value in terms of volume reduction for the cost. The expected post-treatment concentrations may preclude the material from beneficial reuse in Puget Sound.

Solidification and treatment technologies were screened out for full-scale consideration in the FS as described above.

7.1.3 Disposal/Reuse of Contaminated Sediment

Several disposal options for dredged sediment were identified in the CTM and are reconsidered here for their applicability to cleanup of the LDW:

- On-site disposal
    - Contained aquatic disposal (CAD)

  - Confined disposal facility (CDF)
- Off-site disposal
  - Existing Subtitle C landfill (40 CFR Part 265, Subtitle C of RCRA)
  - Existing Subtitle D landfill (40 CFR Part 258, Subtitle D of RCRA)
- Open water disposal
  - Dredged Material Management Program (DMMP) site
- Beneficial reuse.

The on-site disposal options retain the contaminated material in or very near the site in new, engineered facilities. The off-site disposal options pertain to upland disposal in existing regional landfills. Open water disposal is also a process option for dredged material that meets the DMMP's criteria for open water disposal. All of these disposal alternatives have demonstrated effectiveness and have been successfully used in the Puget Sound region.

Beneficial reuse is often preferred to disposal, when feasible, although application can be limited by physical characteristics or contaminant concentrations.

7.1.3.1 On-Site Disposal

CAD and CDF are two potential on-site process options for disposal of dredged sediment. As discussed in the CTM (RETEC 2005), both disposal options confine contaminated sediment within an engineered structure. These options differ primarily in location or setting: CAD facilities are located within a water body, and CDFs are located nearshore or upland.

CAD Sites

CAD implementation, although a proven technology, is constrained in the LDW. Material is typically placed in horizontal layers, which requires locating the CAD site in a relatively flat area or depression to minimize excavation quantities during construction, and to prevent spread of contaminated sediment downslope. Potential CAD sites in the LDW are located within or near the defined navigation channel. To ensure that the authorized channel depths are maintained, the top surface of the CAD must be positioned below the authorized channel depth to allow for maintenance dredging. The federally-authorized navigation channel requires maintenance of a specified depth; remedial alternatives within the channel cannot interfere with the authorized channel depth. Two locations in the LDW best satisfy these requirements:

- The deep area at the north end of the LDW directly south of Harbor Island, where the existing depth is well below the authorized navigation channel depth

- The southernmost portion of the LDW, defined by the Upper Turning Basin and adjacent navigation channel.

An advantage of CAD over upland disposal is that the overall project dredging production rate can be significantly accelerated because dredged sediment can be placed directly into bottom-dump barges for rapid movement to and placement into the CAD. Dredging would not be subject to the production rate constraints associated with transloading and transportation to a landfill. As result, the overall period of short-term dredging impacts could be reduced through use of CAD.

Numerous implementability issues would have to be addressed to implement CAD including:

- Logistical and timing considerations need to be planned for, including: 1) CAD construction (e.g., dredging and disposal of excavated sediment), 2) sequencing and timing to dredge and place contaminated sediment in the CAD, and 3) identification and coordination to secure and place capping material. In addition, capping (either interim or final) must be completed by the end of each in-water construction window to protect fish runs from disturbance by construction during migration.

- Barge dumping of contaminated sediment into a CAD site involves some dispersion of material as it falls through the water column and lands on the mudline. Unless care is taken, the dumped sediment can cause a "mud wave" when it strikes the bottom. This can cause contaminated sediment to move out of the CAD area and migrate onto adjacent surfaces. Models are available (e.g., STFATE) to assess this factor and engineering controls would need to be incorporated into the design to minimize or mitigate this factor. These engineering controls can include designing the CAD with features to limit mud waves, monitoring adjacent areas, and capping or implementing ENR for any affected adjacent areas.

- Propeller scour in the navigation channel as well as movement by tugs and other vessels accessing adjacent berthing areas could stir up exposed contaminants and move them into other areas before the cap is installed. Modeling of propeller wash, along with appropriate navigation controls during the construction season can be used to minimize this potential.

A CAD could also potentially be located outside of the LDW (e.g., elsewhere in Puget Sound). However, this would likely be an off-site disposal action subject to permitting requirements. Because the administrative implementability of an off-site CAD is considered low, these possibilities are not explored in this FS.

CAD is being carried forward, and will be evaluated as a disposal alternative with the understanding that CAD capacity may not match the total volume of contaminated dredged sediment under some alternatives. However, regardless of which remedial alternative is selected, CAD may be considered during remedial design on a smaller-scale, location-specific basis, subject to agency approval.

CDF Sites

A nearshore or upland CDF (e.g., construction of a CDF in a slip) is a technically feasible option for the disposal of LDW dredged material, but is not carried forward as a primary in-water disposal technology for the FS. During engineering design, if a small-scale CDF potentially could be applicable, numerous hurdles would need to be overcome. Some of these hurdles include: identifying suitable available land/water sites for acquisition, providing compensatory habitat mitigation for lost aquatic habitat, and demonstrating appropriate economic development purposes for the upland facility in accordance with the Clean Water Act Section 404(b)(1) guidelines.

7.1.3.2 Off-Site Landfill Disposal

Sediments removed from the LDW are not expected to require disposal in a landfill permitted to receive Resource Conservation and Recovery Act (RCRA) hazardous waste or Toxic Substances Control Act (TSCA) waste (i.e., Subtitle C landfill). Nevertheless, a regional Subtitle C landfill (Waste Management, Inc. located at Arlington, Oregon) is available to receive material that exceeds the relevant RCRA or TSCA limits should such material be encountered during remediation.

Two regional Subtitle D landfills (Waste Management, Inc. located at Columbia Ridge, Oregon, and Allied Waste, Inc. located at Roosevelt, Washington) receive both municipal waste and solid nonhazardous contaminated media. Both facilities have been used for the majority of contaminated sediment projects in the Puget Sound region, including several projects in the LDW (Table 7-5). Further, both facilities are permitted to receive wet sediment (i.e., sediment that does not pass the paint filter test and therefore contains free liquid). These existing Subtitle D landfills are retained as representative disposal process options for remedial alternatives that call for sediment removal with disposal in an upland landfill.

7.1.3.3 Open Water Disposal

In Puget Sound, the open water disposal of sediments is managed and monitored under the DMMP, which is jointly administered by the U.S. Army Corps of Engineers (USACE), the EPA, the Washington State Department of Natural Resources (WDNR), and Ecology. The DMMP User's Manual (USACE 2008b) details the sediment evaluation, testing, and disposal procedures for open water disposal of dredged material at DMMP-designated disposal sites in Puget Sound. The DMMP non-dispersive deep water disposal site nearest to the LDW is in Elliott Bay. This facility has approximately 6.6 million cy of remaining capacity.[3]

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[3] Approximately 2.4 million cy of dredged material have been placed at the Elliott Bay disposal site between 1989 and 2007. With a capacity of 9.0 million cy, the site will be operational for about 50 more years, assuming about 130,000 cy of placement per year (USACE 2007a).

Some of the LDW sediments that have been dredged from the navigation channel between river mile (RM) 3.8 and RM 4.8 and from private berthing areas outside of the navigation channel have previously been tested and accepted for open water disposal. This suggests that at least some of the sediment removed during remediation may meet DMMP criteria. However, the FS assumes that dredged sediments requiring remediation would not be clean enough to meet DMMP requirements, although they are not necessarily precluded from DMMP open water disposal.

Open water disposal may be considered in the remedial design phase for the following material if the sediment is demonstrated to achieve the DMMP criteria for open water disposal:

- The clean sand fraction from conventional soil washing
- Suitable material dredged from areas during construction of a CAD facility
- Suitable material, if any, dredged under some alternatives in this FS.

7.1.3.4 Beneficial Use of Sediment (Clean and Treated)

Beneficial use of dredged sediment is preferred to its disposal, when feasible. However, contaminated and untreated sediment is not suitable for direct beneficial use applications. This subsection examines the potential beneficial use of:

- Clean dredged material generated by local navigation channel maintenance dredging projects
- Treated sand fraction of dredged contaminated sediments from the LDW.

Any potential in-water beneficial use application would need to meet associated material specifications to ensure an appropriate match between physical, chemical, and biological material properties and functionality in the aquatic environment.

Beneficial Use of Dredged Material from Navigation Projects

Regular USACE maintenance dredging of regional navigation channels in the LDW, Snohomish River, Swinomish Channel, and other rivers generates large volumes of sandy and silty sediments. In the Puget Sound region, this dredged material has been used beneficially for both remediation and habitat enhancement projects. Examples of projects in Elliott Bay that have used sediment from LDW Upper Turning Basin maintenance dredging activities include:

- **Denny Way Combined Sewer Overflow (CSO) Capping –** In 1990, King County and the USACE sponsored the Denny Way CSO capping project to test the feasibility of capping contaminated sediments in Elliott Bay. A 3-ft layer of sediment dredged from the LDW Upper Turning Basin was placed over a 3-acre area at the Denny Way CSO. Monitoring results over the last 15 years demonstrate that the cap is stable, is not eroding, and has successfully isolated the underlying contaminated sediments (King County 2007b).

- **Pier 53-55 Capping –** In March 1992, about 22,000 cy of sediment dredged from the LDW Upper Turning Basin was placed offshore of Piers 53, 54, and 55 in Elliott Bay, to cap approximately 2.9 acres and ENR approximately 1.6 acres of contaminated sediments. Monitoring results indicate that the 3-ft cap and ENR areas are stable, and contaminants are not migrating from the underlying sediments up into the 3-ft cap or ENR area (King County 2010b).

- **Bell Harbor Capping –** In March 1994, the Port of Seattle placed a thin-layer cap of sediment dredged from the LDW Upper Turning Basin over 3.9 acres of contaminated sediments at the former site of Pier 64/65 in Seattle. The site was also designed to incorporate habitat enhancement components, including rock corridors on top of the cap and gravel below the slope and between corridors. These substrata were specifically designed to serve as habitat for brown algae and juvenile rockfish. Subsequent monitoring has demonstrated the success of both actions (Erickson et al. 2005a, 2005b).

- **Pacific Sound Resources Superfund Site in West Seattle –** Approximately 66,000 cy of sediment dredged from the LDW Upper Turning Basin, along with over 200,000 cy of sediment dredged from the Snohomish River, was placed as a cap at the Pacific Sound Resources contaminated sediment site in West Seattle in 2004 (USACE 2007b).

This FS assumes that upland-sourced materials (sand, gravel, and rock) will be purchased for use as cap materials and for ENR. However, the design process should consider the use of navigation channel and berthing area dredged materials determined suitable for beneficial use application as an alternative to upland-sourced materials. The EPA's Contaminated Sediment Technical Advisory Group (CSTAG) has recommended that the navigation channel and berthing area dredged material be considered for these uses in the remediation of the LDW (CSTAG 2006). However, significant administrative issues (including timing, contracting, and administrative approvals) are associated with procuring USACE and private party dredged materials.

Beneficial Use of Treated Contaminated Sediments

For contaminated sediments dredged as part of a cleanup action, treatment would be required before possible beneficial use. Treatment by soil washing followed by beneficial use of the sand fraction may be more cost-effective than treatment followed by disposal. The coarser (sand) product (processed material achieving target levels established for the project) from a soil washing process could potentially be reused within the LDW for capping, habitat restoration, or grade restoration (i.e., to meet final bathymetry requirements) as part of the remedial action. However, a review of existing literature and local knowledge did not identify any examples of treated sediments being used beneficially in the Puget Sound region.

The sand produced from a soil washing process could also be reused in the uplands as construction fill or as material feedstock for other industrial or manufacturing applications (e.g., concrete or asphalt manufacture). Depending on the end use and associated exposure potential, it is not known whether the treated sand fraction would achieve appropriate chemical criteria for all LDW contaminants. Upland beneficial use would also require resolution of legal issues related to material classification, antidegradation, and potential liability.

Remedial alternatives that include soil washing assume that the disposition of the washed material could result in a range of outcomes: 1) achieve the applicable chemical and physical requirements for in-water use and hence be used as on-site cap or ENR material with potential material cost savings; 2) be suitable for upland use as fill with no associated value or disposal cost; 3) be suitable for open water disposal with a comparatively low disposal cost; or 4) require landfill disposal at significant cost.

7.1.4 Capping

In the CTM (RETEC 2005), capping was evaluated and retained as a containment technology that is considered both effective and implementable in the LDW. Capping is a well-developed and documented *in situ* remedial technology for sediment that isolates contaminants from the overlying water column and prevents direct contact with aquatic biota (Figures 7-3 and 7-4). Depending on the contaminants and sediment conditions present, a cap reduces risks through the following primary mechanisms (EPA 2005b):

- Physical isolation of the contaminated sediment sufficient to reduce exposure through direct contact and to reduce the ability of burrowing organisms to move contaminants to the cap surface
- Stabilization of contaminated sediment and erosion protection of the sediment and cap sufficient to reduce resuspension and transport of contaminants into the water column
- Chemical isolation sufficient to prevent unacceptable risks of exposure to sediment contaminants that are solubilized and transported through the cap material and into the water column (e.g., via diffusion or groundwater advection).

7.1.4.1 Conventional Sand and Armored Caps

A large number of sediment caps have been successfully implemented in the Puget Sound region: One Tree Island Marina, Olympia 1987; St. Paul Waterway, Tacoma 1988; Georgia Pacific Log Pond, Bellingham 2000; East and West Eagle Harbor/Wyckoff, Bainbridge Island, 1993-2002; Middle Waterway, Tacoma 2003; General Metals, Tacoma late 1990s; and others (RETEC 2002).

Within the LDW, a sand cap was constructed in 2005 in conjunction with the Duwamish/Diagonal early action area (EAA) sediment remediation project (Anchor and EcoChem 2005a) (Figure 7-5). Preliminary monitoring results from 2007 to 2009 show trends indicating that the cap has successfully isolated underlying contamination. Following cap construction, total PCB concentrations in surface sediment have fluctuated around the SQS. However, because the Duwamish/Diagonal cap is located near an active storm drain and a CSO outfall, and is adjacent to other contaminated sediments, some degree of increase in contaminant concentrations on the cap surface has been noted, highlighting the importance of source control.

The ability to implement capping technology is influenced greatly by physical constraints and engineering design. Capping may be suitable where navigation or other public uses would not be physically impeded, or in areas where it is impractical to remove all of the contaminated material because of slope or nearby structure stability concerns. If capping is chosen as part of the selected remedial alternative for the LDW, then bathymetric, hydrodynamic, slope stability, and biological conditions, as well as commercial/public land use would need to be considered. An engineered cap design specifies material types, gradation, thickness, armoring requirements, design elevation ranges, placement requirements, and other design parameters. For example, the cap design for deep depositional waters would be different from designs for intertidal and shallow subtidal areas of high habitat importance and areas that have the potential for appreciable episodic erosion.

7.1.4.2 Composite and Reactive Caps

A composite or reactive cap may be an appropriate design solution in situations where:

- A reduced cap thickness is needed in navigation-constrained areas to avoid dredging.
- Standard sand capping would require excessive thickness for containment of a specific COC.
- Contaminant migration necessitates reducing contaminant flux over what is achievable with native capping materials.

Reactive cap technology refers to including reactive amendments in the granular cap material or in manufactured mats. The additives are selected based on their ability to adsorb or react with contaminants migrating through the cap strata. Activated carbon, bentonite, apatite, AquaBlok™ (a commercial product designed to enhance contaminant sequestering through organic carbon amendments to the cap, and to reduce permeability at the sediment-water interface), and coke are examples of reactive amendment materials that have been investigated at the demonstration level or in full-scale applications. The need for and type of amendment will be evaluated for specific project areas during remedial design; design data requirements may be different between conventional and thin-layer caps. Section 7.1.4.4 summarizes preliminary modeling results that indicate amendments may not be necessary as a component of cap design for reducing migration of hydrophobic organics through the cap (e.g., PCBs and cPAHs).

The following paragraphs describe examples of composite or reactive cap demonstration level or full-scale application projects.

Carbon Amendment of Cap Materials (Various sites, Washington)

Sand with a carbon amendment was used in caps at the Upriver Dam PCB Sediments Site, Spokane, WA (Anchor 2006a), Olympic View Resource Area, Tacoma, WA (Hart Crowser 2003), and Slip 4 EAA, Seattle, WA (Integral 2010).

Activated Carbon – Reactive Core Mat (Tukwila, Washington)

After sediment dredging and capping was conducted in 1999 by King County offshore of the Norfolk combined sewer overflow (CSO) outfall within what later became the LDW study area, surface sediment monitoring showed that additional sediment removal was needed in the vicinity of the nearby south storm drain outfall of the Boeing Developmental Center to prevent recontamination (PPC 2003). Approximately 60 cy of contaminated sediment were removed and backfilled in September 2003 by Boeing to eliminate the potential source of recontamination to the adjacent cap. The sediment removal was completed during low tide cycles over a one-week period; all work was completed above the water level (at low tide). Following each day's excavation work, a geotextile fabric layer (Mirafi filter fabric) was installed as a temporary cover to contain and limit any potential migration of silts and the associated contaminants from the excavation area. Turbidity was monitored daily as well as visual monitoring throughout the construction period. Based on turbidity measurements and visible appearance, the daily geotextile fabric cover worked well to prevent loose silt material from mobilizing within the LDW. The geotextile fabric was removed and disposed of before the cap was placed. The excavation area was capped with a fabric containing activated carbon, a layer of sand, and a cover consisting of quarry spoils in the channel segment (where higher velocities from the outfall discharge were expected). The activated carbon fabric was included in the cap permanently to adsorb and contain any residual PCBs in the channel area and prevent upward migration of PCBs in this area. Continued annual monitoring and sediment sampling have verified that no recontamination has occurred within the engineered cap and have demonstrated that the remaining contaminated area is limited to a small segment of the drainage channel located just below the south storm drain outfall (PPC 2003).

Activated Carbon – Reactive Core Mat (Stryker Bay, Duluth, Minnesota)

Stryker Bay in Duluth (MN) was heavily contaminated with tar and coke (Bell and Tracy 2007). Coal tar thicknesses under the water reached as much as 13 ft in some areas. Remediation involved placing six inches of sand cap and a reactive core mat (RCM), followed by six inches of sand cap over the contaminated sediments. The activated carbon-based geotextile fabric, a reactive cap, allowed the cap thickness to be less than a traditional sand cap, and provided stability and physical isolation. According to the *First Five-Year Review Report* (USACE 2003b), the remedial action was complete and was found to be protective of human health and the environment as intended by the 2000 Record of Decision (ROD) because soils above the direct exposure cleanup levels identified in the ROD for industrial use were removed.

Activated Clay Cap (Willamette River, Portland, Oregon)

In 2004, as part of the cleanup of the McCormick and Baxter Superfund site, the east bank and bed of the Willamette River in Portland (OR) were capped with an organoclay sediment layer to contain high concentrations of COCs, including pentachlorophenol (PCP), creosote, chromium, and arsenic (Aquatechnologies.com, Oregon Department of Environmental Quality [ODEQ] 2005). Over most of the site, the cap consists of a 2-ft-thick layer of sand. In more highly contaminated areas, a 1-ft organoclay layer was placed beneath a 5-ft-thick layer of sand. The organoclay consists of bentonite or hectorite clay modified to be hydrophobic, to have an affinity for non-soluble organics, and especially to prevent breakthrough of non-aqueous phase liquid through the cap. The design of the sediment cap incorporated different types of armoring to prevent erosion of the sand and organoclay layers. In the *Third Five-Year Review Report* (ODEQ 2011), the remedy for the sediment OU was determined to be protective of human health and the environment because the remedy required by the ROD is working as intended.

Granular Bentonite, Sand/Soil/Bentonite Slurry, and AquaBlok™ (Lower Grasse River, Massena, New York)

Pilot studies conducted in 2001 in the Lower Grasse River, Massena, (NY) evaluated capping with various materials as a cleanup alternative for remediating PCB-contaminated sediments (Quadrini et al. 2003). Materials such as a 1:1 sand/top soil mixture, granulated bentonite (clay), and AquaBlok™ were tested as single components or mixtures. Optimal results were achieved with a 1:1 sand/top soil cap applied via a clamshell attached to a barge-mounted crane. Few apparent short-term impacts were noted during the pilot project, as well as negligible water quality impacts. However, in 2003, cap monitoring data indicated significant loss of cap material, and in some cases, significant but localized scouring of underlying sediment (up to 2 ft), that translated into redistribution of the PCBs buried in the river sediments in the upper approximately 1.8 miles of the Lower Grasse River (Quadrini et al. 2003). The possible cause was an ice jam that formed on the river during the spring ice breakup. Consequently, an ice breaking demonstration project was conducted in 2007, the results of which were incorporated into the analysis of alternatives report to evaluate remedial options for the river (Alcoa 2007).

AquaBlok™/Sand (Anacostia River, Washington, D.C.)

A major demonstration of several active-addition reactive cap designs has been conducted on the Anacostia River in Washington, D.C. (EPA 2007c). The objective of this demonstration project was to provide information on the design, construction, placement, and effectiveness of these augmented caps. Various cap technologies were evaluated, including sand (as a demonstration control), AquaBlok™, coke breeze (with potential to sequester and retard the migration of organic contaminants through sorption), and apatite (which encourages precipitation and sorption of metals). The performances of these caps were evaluated in terms of physical stability, hydraulic seepage, and impacts on benthic habitat and ecology. Monitoring of the caps over an approximately three-year period using a multitude of invasive and non-invasive sampling and monitoring tools was used in assessing performance. Results indicate that the AquaBlok™ was highly stable, and likely more stable than traditional sand capping material even under very high bottom shear stresses. The AquaBlok™ material was also characteristically more impermeable, and it is potentially more effective at controlling contaminant flux, than traditional sand capping material. However, the low permeability AquaBlok™ cap showed evidence of heaving because of methane accumulation and release. AquaBlok™ also appeared to be characterized by impacts (lack of colonization) to benthos and benthic habitat similar to traditional sand capping material (EPA 2007c). Apatite results were not available for review in the EPA (2007c) report.

In another demonstration in the Anacostia River in 2004, a RCM was designed to accurately place a 1.25-cm thick sorbent (coke) layer in an engineered sediment cap (McDonough et al. 2006; Figure 7-4). Twelve 3.1-meter (m) x 31-m sections of RCM were placed in the river and overlain with a 15-cm layer of sand to secure it and provide a habitat for benthic organisms to colonize without compromising the integrity of the cap. Placement of the RCM did not cause significant sediment resuspension or impact site hydrology. The RCM was shown to be an inexpensive and effective method to accurately deliver thin layers of difficult to place, high value, sorptive media into sediment caps. It can also be used to place granular reactive media that can degrade or mineralize contaminants.

7.1.4.3 Capping and Overwater Structures

Overwater or floating structures (e.g., docks, piers, marina floats) preclude conventional means of installing a cap using a material barge and excavator or clamshell-based equipment. Various alternative methods are available and have been successfully implemented under these circumstances:

- A belt-conveyor system that can be controlled for angle and speed spray-deposits sand under piers and between pile bents (Figure 7-6).
- Small construction equipment (e.g., skid loader) that fits between pile bents can directly apply cap materials during low tide and where surface conditions are sufficiently stable and access is adequate for maneuvering. This approach was used successfully at the Wyckoff/Eagle Harbor West Operable Unit remediation site in 1997.
- A discharge pipeline can hydraulically deposit a sand-slurry underneath or through the overwater structure. The latter may require removing some of the pier decking. This approach was used successfully at the Wyckoff/Eagle Harbor West Operable Unit remediation site in 1997.

- Pier decks can be removed temporarily to improve access for mechanical placement, as employed at the Martinac Shipyard in the Thea Foss Waterway circa 2003.
- Grout-filled mats can be installed around pile bents, as employed in the Thea Foss Waterway circa 2003.

At intertidal locations where it is difficult to effectively place a sand cap by conventional means (e.g., where the slope is too steep or overhead obstructions exist), a shotcrete cap is an option. Shotcrete is typically composed of concrete or mortar and is pneumatically jettisoned from a nozzle at high velocity onto the surface to be coated at low tide. A shotcrete cap was installed during the Todd Shipyards sediment cleanup (McCarthy, Floyd|Snider 2005). The shotcrete application at Todd Shipyards effectively encapsulated existing debris (slag) mounds (Figure 7-7). Shotcrete can be applied to various material types and surface orientations, including steep embankments. However, shotcrete is not appropriate for use in habitat areas.

7.1.4.4 Modeling of Cap Recontamination

The potential for a conventional *in situ* isolation sand cap to be recontaminated over time by the movement of contaminants through the cap from underlying sediments was analyzed using a one-dimensional groundwater flux model (Lampert and Reible 2009) that also includes net sedimentation on top of the cap. The modeling approach and the results of the analysis are presented in Appendix C, Part 8 (Modeling Contaminant Transport through a Sediment Cap).

The analysis showed that PCB breakthrough above the assumed performance goals is not expected to occur.[4] This is true even where the assumed conditions are unfavorable (high groundwater flow, low sedimentation, and low organic carbon coefficient [K_{oc}]), because the sedimentation rate is always greater than the rate at which the contaminant front migrates through both the cap and the new sediment layer that is continually added over time. The analysis showed that cPAHs behave similarly to PCBs and therefore would also not exceed similar performance goals.

In the complete absence of sedimentation, the results show that capping is still feasible, but that minimum organic carbon requirements for cap materials may need to be specified to achieve a cap design life of more than 100 years. ENR is predicted to achieve assumed performance goals under average conditions, but may not be applicable in adverse conditions (high groundwater flow, no sedimentation, low K_{oc}).[5]

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[4] The assumed performance goals for cap modeling are: 1) sediment concentrations not exceeding 100 µg/kg dw total PCBs in the top 10 cm within 100 years, and 2) porewater concentrations below 0.03 µg/L at the sediment/water interface within 100 years.

[5] Analysis of ENR generally assumes that placed ENR sand mixes with underlying sediment. This analysis assumed that a thin ENR sand layer (15 cm) did not mix with underlying sediment. Therefore, the analysis is exploratory.

Cap or ENR material specification and applicability of ENR would be evaluated during remedial design.

For the 45-cm clamming point of compliance direct contact scenario, the results show that capping with a 3-ft sand cap is feasible, even in the absence of sedimentation. However, minimum organic carbon requirements for cap materials may need to be specified to achieve a cap design life of more than 100 years.

Specific locations within the LDW, such as Ash Grove Cement (RM 0.1E) and the Duwamish Shipyard (RM 1.35W), have historical high concentrations of metals (e.g., arsenic) in the subsurface. For this reason, remedial design should address the potential for dissolved metals (such as inorganic arsenic) to migrate through a proposed cap to surface sediment and surface water (Palermo et al. 1998). The potential for bioturbation and/or diffusion should also be considered during remedial design of caps.

Although cap modeling results presented in Appendix C (Part 8) indicate amendments may not be necessary as a component of cap design to reduce transport of hydrophobic organics (e.g., PCBs and PAHs), remedial design should identify whether the mobility and bioavailability of metals (such as arsenic) need to be reduced and incorporate any special needs into the design. Several studies (Pattanayak et al. 2000, Mohan and Pittman 2007) report the extensive research conducted on effective removal of arsenic through activated carbon adsorption mechanisms. Many other candidates appear interesting for arsenic adsorption, such as activated alumina, clay, silica sand, and organic polymers, which are known to be good adsorbents that can be regenerated *in situ*. Absorptive capacity should be considered in the design phase.

7.1.4.5 Capping Technology Summary

For developing and evaluating remedial alternatives in the FS, conventional sand cap and armored cap process options have been selected to represent the technology as a whole. Sand caps may be applied to net depositional areas, and armored caps may be applied to areas within the LDW subject to episodic erosion. Reactive caps, although not evaluated in this FS for LDW-wide application, may be appropriate and cost-effective depending on location-specific circumstances.

Section 8 of the FS identifies areas suitable for capping based on evaluating the potential for propeller scour, outfall scour, ship wakes, water depths required for vessel navigation and berthing, slopes, habitat requirements, and erosion associated with high-flow conditions in the LDW. Locations requiring armoring are also considered.

7.1.5 Monitored Natural Recovery (MNR)

Natural recovery of sediments refers to the ability of natural processes such as chemical and biological degradation as well as physical burial by incoming sediments to reduce contaminant concentrations over time (Figure 7-8). Where conditions support natural recovery and natural recovery is included in the remedial alternative, a monitoring program will be instituted as a key component of MNR to assess if, and at what rate, risks are being reduced and whether progress is being made toward achieving the cleanup objectives. The monitoring program associated with an MNR remedy generally combines physical, chemical, and possibly biological testing to track progress toward achieving the cleanup objectives. As with any risk-reduction approach that takes time to reach remediation goals, remedies that include MNR frequently rely upon institutional controls, such as seafood consumption advisories, to control human exposure during the recovery period (EPA 2005b). In the event that MNR does not achieve or progress sufficiently toward achieving performance objectives, contingency actions such as capping, ENR/*in situ* treatment, or dredging may be required. Establishing decision rules with targets and time frames for the performance of MNR is an essential component of an adaptive site management framework (Magar et al. 2009).

As discussed in Section 5, new material transported into the LDW from upstream will tend to settle and bury some of the contaminated sediments. This burial, combined with surficial mixing (both from bioturbation by benthic organisms and resuspension caused by physical processes), is the principal ongoing natural recovery process within the LDW. The majority of COCs in LDW sediments are resistant to chemical and biological degradation and dissolution. These mechanisms are not likely to make important contributions to natural recovery in the LDW. Thus, it is reasonable to expect that the primary factor in determining how quickly natural recovery will occur (assuming sources are adequately controlled) is the burial or sediment deposition rate. Recovery is expected to be more rapid in areas with intermediate to high net sedimentation rates and slow where net sedimentation rates are low or where the potential exists for either significant scour or episodic erosion. The bed composition model (BCM) (see Section 5) was developed as a tool to predict recovery as a function of both location within the LDW and of the concentrations of contaminants coming into the LDW from upstream and lateral (e.g., stormwater) sources.

7.1.5.1 Sediment Remediation Projects with an MNR Component

Examples of sediment remediation projects where MNR is a component of a combined remedy or where natural recovery trends have been observed are provided below.

Duwamish/Diagonal EAA (Seattle, Washington)

Data collected during the Duwamish/Diagonal EAA project (Anchor 2007) lend some empirical support to natural recovery potential in the LDW. This project involved a combination of removal (dredging), capping, and thin-layer sand placement. Surface sediment contaminant concentrations are being monitored on and adjacent to the actively remediated areas of the project site (Figure 7-5). Monitoring data associated with the cap and thin-layer sand placement are discussed below in Section 7.1.6. The data collected from stations peripheral to the actively remediated areas are plotted versus time in Figure 7-5 (center chart). The trends suggest that contamination from resuspension and dispersal during the dredging operation may have been responsible for total PCB concentrations remaining high and are consistent with data generated during the investigative phase of the project in the mid-1990s. Since that time, total PCB concentrations have declined by 50% or more in five of the eight perimeter locations, presumably as a result of natural recovery processes (see Appendix F). Net sedimentation rates ranging from 0.7 to 3.1 cm/yr were estimated from radioisotope core data in the Duwamish/Diagonal area, consistent with the STM model predictions (see Appendix F, Figure F-2). The average concentration of the perimeter stations (Figure 7-5) have already decreased (after 5 years) to below modeled predictions of recovery 10 years following remediation (Stern et al. 2009). However, dispersion of some of the newly placed capping material appeared to have initially influenced some immediately adjacent noncapped areas, thereby contributing to the decrease in PCB concentrations seen in the first post-capping year. Unpublished PCB data from 2010 sampling indicate that the total PCB concentration has decreased by approximately 67% from that observed in 2009 (Williston, personal communication, 2010) indicating the area is continuing to recover.

Slip 4 EAA (Seattle, Washington)

Additional empirical support for natural recovery in the LDW can be discerned from the Slip 4 surface sediment dataset, as shown in Figure 7-9, although the conditions in the slip are somewhat different than those in the LDW outside of the slip. This figure shows where surface sediment samples were collected and analyzed for total PCBs within the Slip 4 EAA. These data were divided into two groups representing conditions observed before 1999, and conditions observed in 2004 (see Figure 7-9). The two datasets were analyzed statistically and determined to be significantly different ($p<0.05$; Mann-Whitney two-sample test). The mean total PCB concentration in the 2004 dataset (830 µg/kg dw) is 24% of the mean concentration in the pre-1999 dataset (3,200 µg/kg dw). However, sampling of Slip 4 surface sediments in 2010 revealed increasing PCB concentrations within the EAA, which highlighted the need for additional source control actions (Ecology 2011a). Net sedimentation rates ranging from 1.6 to 3.2 cm/yr have been estimated from radioisotope core data in the Slip 4 area, contributing to the process of natural recovery; these estimated rates are consistent with the STM model predictions (see Appendix F, Figure F-2).

Sangamo Weston/Twelve-Mile Creek/Lake Hartwell (Pickens, South Carolina)

Lake Hartwell and its tributary Twelve-Mile Creek are heavily contaminated with PCBs, which were discharged by the Sangamo Weston Inc. facility between 1955 and 1977. MNR, in combination with institutional controls (fish consumption advisories), was selected by EPA as the main remedy for Operable Unit 2. Net sedimentation rates of 5 to 15 cm/yr, confirmed by radioisotope geochronology, and burial by progressively cleaner sediment over time is the dominant physical process for recovery. Field measurements show a gradual recovery of surface sediments from peak concentrations of approximately 40 mg/kg dw to around 1 mg/kg dw in more recent samples (Magar et al. 2003). In addition, sedimentation for the Twelve-Mile Creek arm of Lake Hartwell has been accelerated by the release of accumulated sediment from three upstream dams. Chemical transformation (i.e., PCB dechlorination) has also been observed via PCB congener analysis of sediment cores with depth and age. This natural process has been found to be slow and limited as a result of anaerobic subsurface sediment, but it has reduced the long-term risks associated with potential sediment resuspension (Magar et al. 2009).

Annual monitoring has been conducted through sediment sampling (at 21 locations within the tributary and lake), fish tissue sampling (at 6 lake locations), and bioaccumulation studies (in caged *Corbicula* clams) to track the progress toward achievement of cleanup objectives. Despite the substantial historical decrease in PCB sediment concentrations (below the 1 mg/kg dw cleanup level), fish tissue concentrations have not decreased accordingly (Magar et al. 2004, Magar et al. 2009). PCB concentrations in catfish fell below the Food and Drug Administration (FDA) tolerance level of 2 mg/kg wet weight (ww) for several years, but this trend has not been sustained since 2005. The other five fish species monitored show no clear trend of decreasing PCB concentrations. Fish consumption advisories remain in effect for Twelve-Mile Creek and Lake Hartwell, because PCB concentrations in fish continue to exceed the FDA tolerance level of 2.0 mg/kg ww.

James River (Hopewell, Virginia)

The chlorinated pesticide Kepone (chlordecone, a carcinogenic chlorinated hydrocarbon) was made and discharged between 1974 and 1975 through the municipal sewage system, surface runoff, and solid waste dumping into the James River estuary in Hopewell (Virginia). Average Kepone concentrations in the channel sediments ranged from 20 to 193 µg/kg dw.

MNR was selected as the main remedy for all areas of the site, and the dominant natural recovery processes were dispersion (in high-energy areas) and physical isolation through natural sedimentation (in low-energy areas). Radioisotope geochronology showed evidence of natural sedimentation within the estuary, ranging from less than 1 cm/yr to greater than 19 cm/yr, with an average of at least 8 cm/yr at 8 of the 21 sediment sampling locations (Magar et al. 2009).

Although Kepone tissue concentrations in James River fish reached as high as 5 mg/kg ww in 1975, the average tissue concentrations had fallen below the FDA action level of 0.3 mg/kg ww by 1986 (Luellen et al. 2006). The last exceedance of the action level in striped bass was measured in 1995, according to the Virginia Department of Environmental Quality (VA-DEQ 2011). However, Kepone continues to be detected in about 94% of fish tissue samples above reporting limits. Continued detections of Kepone are believed to be related to coastal disturbances related to severe weather (Luellen et al. 2006, Magar et al. 2009). The observed decline in fish contamination over the years is thought to be the result of the Kepone being sequestered in the tidal basin sediments of the James River and thus becoming less available to contaminate fish (Lawson 2004).

A fish consumption advisory is still in effect for Kepone, and the VA-DEQ continues to monitor Kepone levels in fish tissue and sediment to address concerns about contaminated sediment resuspension after high-energy events (Magar et al. 2009).

Bremerton Naval Complex (Puget Sound, Washington)

The cleanup of Puget Sound Bremerton Naval Shipyard Complex (PSNS), located on the Sinclair Inlet of Puget Sound at Bremerton (WA), included extensive dredging, capping, ENR, and long-term monitoring of surface sediments to assess natural recovery (EPA 2000c). The marine area of concern (Operable Unit B) in the PSNS is a subtidal section of the inlet, with water depths generally less than 15 m. Baseline total PCB concentrations in sediments within the area of concern were around 13 mg/kg organic carbon (oc) (with a maximum measured concentration of 61 mg/kg oc) (Merritt et al. 2010).

Three rounds of post-remedy monitoring (2003, 2005, and 2007) have been completed, including measures to verify the integrity of remedy components and assess progress toward cleanup goals. In addition, bathymetric surveys, sub-bottom profiling, and collection and analysis of sediment cores were performed. These activities have confirmed that dredging, capping and ENR remedy components are functioning as planned, and that ongoing sediment deposition and mixing (MNR) are occurring naturally (URS 2009).

Sampling of marine sediments throughout Operable Unit B and Sinclair Inlet were also conducted. In 2007, the geometric mean for Operable Unit B Marine sediment total PCB concentrations, estimated on an area-weighted average basis, was 4.5 mg/kg oc (URS 2009); this value exceeded the cleanup goal of 3 mg/kg oc, but it was less than the 2003 and 2005 area-weighted geometric mean values (6.7 and 6.1 mg/kg oc, respectively).

Total PCB concentrations in English sole tissue samples were also analyzed. The 2007 arithmetic mean English sole total PCB concentration was 0.033 mg/kg ww, above the remedial goal of 0.023 mg/kg ww (URS 2009) and well below the concentration of 0.085 mg/kg ww obtained in 2003.

Trend analyses for Operable Unit B Marine performed on the 2003, 2005, and 2007 sediment samples predicted a decreasing trend and indicated that the cleanup goals established in the ROD may be achieved within 10 years after remediation (<3 mg/kg oc for PCBs) and the long-term goal of <1.2 mg/kg oc for PCBs may be achieved by 2017 (EPA 2000, URS 2009, Leisle and Ginn 2009).

7.1.5.2 MNR Summary

NRC (2007) projected that MNR is likely to be a component of many large-scale sediment remediation projects with temporal goals. In the LDW, natural recovery is predicted to occur at varying rates at specific locations within the LDW, as supported by the LDW examples above, modeling, and comparison of co-located sediment samples collected over time (see Appendix F). For these reasons, MNR is retained as a remedial process option for developing the remedial alternatives in this FS. LDW-wide reductions in average concentrations of COCs such as PCBs are necessary to reduce resident fish and shellfish tissue concentrations. Hence, MNR is also evaluated as an LDW-wide “polishing step” for all of the remedial alternatives considered in this FS.

7.1.6 Enhanced Natural Recovery (ENR)

ENR refers to the application of thin layers of clean granular material, typically sand, to a sediment area targeted for remediation. Application thicknesses of approximately 6 inches are common, producing an immediate reduction in surface contaminant concentrations (Figure 7-7). Essentially, ENR reduces the time for sediment concentration reductions over what is possible by relying solely on natural sediment deposition where burial is the principal recovery mechanism (EPA 2005b). Thus, areas that are stable (not expected to erode) and are recovering naturally (albeit slowly) are candidates for ENR. Although ENR is best employed in areas not subject to scour, it may be appropriate in some cases to employ engineered aggregate mixes or engineered synthetic products to ensure stability (Palermo et al. 1998, Agrawal et al. 2007).

Unlike capping, which typically has a much greater application thickness, surface sediment contaminant concentrations in areas that undergo remediation by ENR are expected to be influenced by benthic recolonization and associated bioturbation. These processes result in the mixing of underlying contaminated sediment with the cleaner near-surface material. This is important for remedial design where a surface sediment concentration threshold is typically established below which MNR is appropriate (i.e., cannot be achieved in an acceptable time scale) and above which other active technologies (e.g., dredging or capping) should be considered.

The FS assumes that half of the ENR footprint would warrant amendment with a material such as activated carbon for *in situ* treatment. This assumption provides a basis for estimating costs and comparing the remedial alternatives; however, during remedial design, the emphasis on ENR or *in situ* treatment will depend on location-specific factors and additional testing of the implementability of these technologies. The composition of ENR/*in situ* treatment will depend on additional evaluation during remedial design; it may include carbon amendments, habitat mix, and/or scour mitigation specifications to increase stability and enhance habitat.

7.1.6.1 ENR Sediment Remediation Projects

Examples of ENR sediment remediation projects are provided below.

Ketchikan Pulp Company (Ketchikan, Alaska)

A thin-layer placement was successfully applied in 2001 over the sediments offshore of a former sulfite pulp mill (Ketchikan Pulp Company-KPC) in Ward Cove, Alaska (Merritt et al. 2009, Becker et al. 2009). The primary COCs were ammonia and 4-methylphenol. These COCs are not bioaccumulative. Diffusion of contaminants from underlying sediment was identified as the dominant mode of chemical transport responsible for toxicity to organisms in surface sediment.

The thin-layer cap of fine-grained to medium-grained sand was placed over 28 acres of native sediments to a thickness ranging from 15 to 30 cm (Merritt et al. 2009). In 2004 and 2007, the first and second monitoring events were conducted, and included evaluations of sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities. Concentrations of both COCs in the thin-layer strata were low in 2004, indicating ENR effectiveness. The clean sand placement material was not being noticeably affected by upward migration of the COCs from underlying native sediment; the concentrations of COCs remained low in 2007. For sediment toxicity, amphipod survival was about 93 to 96% in 2004 and remained high in 2007 (92 to 95%). Benthic communities had begun recolonization by 2004 and total abundance increased substantially in 2007 (Becker et al. 2009).

Duwamish/Diagonal EAA Project (Seattle, Washington)

In response to observed increases in surface sediment concentrations of total PCBs adjacent to a portion of the primary dredging and cap area at the Duwamish/Diagonal EAA, a thin layer of sand (9 inches, to ensure a minimum 6-inch coverage everywhere) was placed in February 2005 over 4 acres of sediment, providing immediate reduction in exposures, and reducing total PCB concentrations to between 1 and 32 µg/kg dw (Figure 7-5; Anchor 2006b). Prior to dredging and capping, this adjacent area had an average total PCB concentration of 46 mg/kg oc. Immediately following cap placement, that average tripled to 136 mg/kg oc. This increase in total PCB concentrations was attributed to resuspension and dispersal of contaminated sediment (i.e., dredging residuals) during the removal action. Within the ENR area, total PCB concentrations immediately following thin layer placement were well below the SQS (at a mean of 7 µg/kg dw[6]) because of the clean material placed, achieving its goal of immediately reducing PCBs to below predredge surface sediment concentrations. Subsequent years have shown a slight increase in the PCBs concentrations (Stern et al. 2009). The slight increase is likely due to resuspension of the surrounding sediments and by deposition of upstream and lateral load contributions according to the inputs to the area used in the STM. Modeling, supported by monitoring data and physical measurements of the sediment surface layer, has also shown that the thin sand layer is not significantly mixing with the underlying sediment, consistent with measured bioturbation depths (Stern et al. 2009).

A comparison of the 2008 and 2009 total PCB averages of 8 and 5 mg/kg oc, respectively, to the 2003 predredging/capping average of 46 mg/kg oc (almost a six-fold decrease) demonstrates that ENR continues to maintain exposures below the SQS.

Based on diver probing surveys conducted in April 2009, the thickness of the ENR sand layer exhibited a minor decrease from 2006 to 2009. The estimated thickness of the ENR sand layer ranged between 5 and 10 inches at 11 different sampling locations, while 1 to 8 inches of silt were observed to have accumulated on the surface of the ENR layer.

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[6] Total organic carbon content in the March 2005 sampling event was too low to calculate oc-normalized data.

When silt and sand are considered together, the average thickness was 12.8 inches (Anchor QEA 2009). These results are consistent with deposition and bioturbation processes as originally anticipated in the ENR area, but also indicate the presence of a stable surface over a period of time. Post-placement bathymetric monitoring was also conducted and nearly all of the Duwamish/Diagonal cleanup area exhibited accretion over the 5-year period following completion of the ENR remedy.

7.1.6.2 ENR Technology Summary

ENR has sufficient merit and has been sufficiently demonstrated in sediment remediation projects elsewhere to carry this technology forward in developing LDW remedial alternatives. ENR may be applied to broad areas of the LDW with lower levels of contamination, net sedimentation, and where significant erosion is not a concern.

7.2 Institutional Controls

Institutional controls are non-engineered measures that may be selected as remedial or response actions either by themselves or in combination with engineered remedies, such as administrative and legal controls that minimize the potential for human exposure to contamination by limiting land or resource use (EPA 2000e). The National Contingency Plan (NCP) sets forth environmentally beneficial preferences for permanent solutions, complete elimination rather than control of risks, and treatment of principal threats to the extent practicable. Where permanent and/or complete elimination are not practicable, the NCP creates the expectation that EPA will use institutional controls to supplement engineering controls as appropriate for short- and long-term management to prevent or limit exposure to hazardous substances, pollutants, or contaminants. It states that institutional controls may not be used as a sole remedy unless active measures are determined not to be practicable, based on balancing trade-offs among alternatives (40 CFR 300.430 [a][1][iii]).

EPA recommends that where it may provide greater protection, multiple institutional controls should be used in combination, referred to as "layering" by EPA. Institutional controls may be an important part of the overall cleanup at a site, whenever contamination is anticipated to remain following active remediation at concentrations that exceed cleanup levels. Institutional controls may be applied during remedy implementation to minimize the potential for human exposure (as temporary land use or exposure limitations). These controls may also extend beyond the end of construction (or be created at that time) or even after cleanup objectives are achieved to ensure the long-term protectiveness of remedial actions that leave contaminants on site above cleanup levels (as long-term or permanent limitations, e.g., protecting a contaminant barrier like a sediment cap from being accidentally breached).

Institutional controls potentially applicable to cleanup of the LDW site are identified and discussed below. This section describes specific individual controls in sufficient detail to allow for a comparison of remedial alternatives that includes various types and degrees of reliance on institutional controls. An integrated Institutional Controls

Implementation Plan is anticipated for the LDW after the ROD is issued that meets specific location, tribal treaty rights, and community needs. These considerations are discussed further in the FS as part of the development and evaluation of remedial alternatives (Sections 8 and 9).

EPA guidance broadly lists four types of institutional controls: governmental controls, proprietary controls, enforcement tools, and informational devices. However, governmental controls such as the permitting of some (point source but not non-point source) discharges to, or dredging and filling of the LDW, as well as some enforcement controls, such as consent decrees or administrative orders under which settling parties implement remedies including institutional controls, are not discussed at any depth in this FS because they do not inform the choices among alternative remedies. These governmental controls are, for remedy selection purposes, uniform across all alternatives and options (i.e., permitting requirements cannot be changed by remedy selection in the ROD), and consent decrees will be used if responsible parties implement any or all of any remedial action EPA selects in the ROD as required by Section 122(d) of CERCLA. Therefore, the most important institutional controls, or aspects of them, for the development of remedial alternatives are emphasized below. Enforcement tools, even though they are used, for example, to establish enforceable proprietary controls pursuant to consent decrees or orders, are discussed under the category of informational devices. It should be clear at this point that many categories overlap and that the agency guidance that created them was intended to be helpful in analyses rather than necessarily invent divisible categories (e.g., proprietary controls have government enforcement mechanisms to ensure their continuation, and some informational devices can be related to or enhanced by governmental enforcement programs):

- Proprietary controls
- Informational devices
  - Monitoring and notification of waterway users
  - Seafood consumption advisories, public outreach, and education
  - Enforcement tools
  - Environmental Covenants Registry.

These types of institutional controls are outlined below.

7.2.1 Proprietary Controls

Proprietary controls are recorded rights or restrictions placed in property deeds or other documents transferring property interests that restrict or affect the use of property. Covenants are a grant or transfer of contractual rights. Easements are a grant of property rights by an owner, often for a specific purpose (e.g., access, utility, and environmental, among other types of easements). Covenants and easements are essentially legally binding arrangements that allow or restrict usage of property for one or more specific objectives (e.g., habitat protection, protection of human health, etc.). They commonly survive the transfer of properties through real estate transactions and are binding on successors in interest who have not participated in their negotiation. This distinguishes covenants and easements from ordinary contracts or transactions between or among parties. At cleanup sites, covenants and easements commonly control or prevent current and future owners from conducting or allowing activity that could result in the release or exposure of buried contamination as long as necessary. Potential activities controlled or prohibited may include in-water activities (e.g., anchoring, spudding, vessel or tug maneuvering) and construction activities (e.g., pile driving and pulling, dredging, and filling) where buried contamination may become exposed as a result of the activity, as long as it is an activity the owner may legally control. Selecting a less expensive remedy in the form of a proprietary control that limits future property uses in ways a more expensive remedy would not, involves a complex balancing of interests by EPA and Ecology. For example, a proprietary control can lower remedial costs for a former owner at the expense of the redevelopment options of a current owner, who acquired the property after it was contaminated. For this reason, among others, EPA policy and guidance stress assessing reasonably anticipated future land use as an important part of remedy selection generally, and specifically stress limiting use of institutional controls.

Traditionally, covenants or easements were only enforceable by whomever they were granted to, and their successors, depending on how they were crafted. In Washington State, MTCA gave Ecology the right to enforce covenants created under MTCA. More recently, Washington passed its Uniform Environmental Covenants Act (UECA), which allows EPA, as well as the state (in addition to the parties to an UECA covenant), to enforce environmental covenants. For this reason, UECA covenants are anticipated to be the primary proprietary control used in LDW environmental cleanup actions.

Parties with sufficient ownership interests in shorelines and aquatic land could grant UECA covenants that would help ensure that remedial measures (such as sediment caps) are not disturbed. However, UECA covenants may not be implementable or practicable for the publicly-owned, working industrial waterway portions of the LDW where the balancing of interests is especially complex, where access and use are in any case difficult to control, and where the extent of the authority of public entities with ownership or management rights to grant covenants with the full range of controls commonly included in UECA covenants, is uncertain. Another uniquely important interest to consider is the extent to which public entity granted covenants may interfere with tribal treaty-protected seafood harvesting, in particular.

7.2.2 Informational Devices

7.2.2.1 Monitoring and Notification of Waterway Users

The LDW ROD could include an enhanced notification, monitoring, and reporting program for areas of the LDW where contamination remains following cleanup activities. Under such a program, the protection of areas where contamination remains above levels needed to meet cleanup objectives, including areas where capping or CAD containment technology have been utilized, could be enhanced. Such areas could be periodically monitored (by vessels and/or surveillance technology), with vessels performing the dual role of educating potential violators of the existence of activity restrictions, and promptly reporting violations of use restrictions to EPA or Ecology, or the U.S. Coast Guard (USCG) if the area were formally designated as a Restricted Navigation Area (RNA) by formal USCG rulemaking as described in Section 7.2.2.3, Enforcement Tools. Notification to waterway users could further be provided through enhanced signage and other forms of public notice, education, and outreach. A mechanism for the review of any USACE navigation dredging plans and other Joint Aquatic Resource Permit Application (JARPA) construction permitting activity could be established. The review would identify any projects that may compromise containment remedies (cap or CAD) or potentially disturb contamination remaining after remediation, which would include a requirement to promptly notify EPA and Ecology during the permitting phase of any project that could affect cleanup remedies. This mechanism would serve as a backup to an existing Memorandum of Agreement between EPA and USACE for coordinating such permitting, especially if that agreement were to lapse or be discontinued for any reason by either agency in the future.

Additional measures could include: establishing a LDW cleanup protection hotline private citizens could call or email to report potential violations, with a requirement that reports be investigated and conveyed to EPA and Ecology (and the USCG for any RNAs) under specified protocols; and developing and implementing periodic seafood consumption surveys to identify, by population group and geographical location, which seafood species are consumed, where they are consumed, and in what quantities they are consumed. This information would be used to update the Institutional Control Implementation Plan as appropriate and improve seafood consumption advisories and associated public outreach and education. Additional monitoring of the effectiveness of these tools can be used to adapt this approach, as discussed in the next section. The effectiveness of all these measures could be re-evaluated periodically to assess which ones should be continued or be modified.

7.2.2.2 Seafood Consumption Advisories, Public Outreach, and Education

The Washington State Department of Health (WDOH) publishes seafood consumption advisories in Washington. The WDOH currently recommends no consumption of resident seafood from the LDW. Salmon are not resident in the LDW; they are anadromous species that spend most of their lives outside of estuaries like the LDW. WDOH recommendations for Duwamish salmon are the same as for Puget Sound as a whole (e.g., no more than one meal per week of Chinook salmon). The WDOH maintains a web site that includes its advisories and provides publications and other educational forums that cover healthy eating and seafood consumption. In addition, WDOH seafood consumption advisories are posted on signs at public access locations around the LDW. Following these advisories is wholly voluntary, which makes advisories, as a necessity, a last resort. Advisories would also be fundamentally inconsistent with tribal fishing rights secured under treaties of the United States if they were relied on in lieu of cleanup measures intended to provide seafood suitable for consumption. More information can be found at http://www.doh.wa.gov/ehp/oehas/fish/rma10.htm.

The Washington State Department of Fish and Wildlife (WDFW) develops and enforces seasonal restrictions on recreational fishing and seasonal and daily catch limits per individual for various seafood species. WDFW licensing and LDW enforcement activities presumably limit resident LDW seafood consumption to some unknown degree. All recreational fishers over 15 years of age must have a fishing license and comply with specific size, species, and seasonal restrictions on fishing for fish and shellfish throughout the Puget Sound region. In the LDW, all resident fish and shellfish should not be consumed according to WDOH advisories. While WDFW regulations summarize the WDOH seafood consumption advisories, which may enhance their reach and effectiveness, they do not prohibit fishing or shellfishing within the LDW. It is lawful to seasonally collect and consume certain fish and shellfish from the LDW.

Some level of seafood consumption advisories will likely be necessary into the foreseeable future to reduce human health risks from seafood consumption. This is because of the technical impracticability of achieving the seafood consumption cleanup levels under any of the remedial alternatives. Concerns associated with the use of these ICs include the burden placed on tribes exercising their treaty rights and other fishers who use the LDW. Relying on seafood consumption advisories to further reduce human health risks may require fishers to change behavior or make cultural adjustments. This burden is difficult to value precisely given the broad range of needs different fishers may have. Given the diversity of the community that can access the LDW, including tribal members, recreational users, low-income, and non-English-speaking people, additional measures to enhance the effectiveness of seafood consumption advisories and thereby enhance confidence in relying upon them, should be fully and aggressively explored.

An enhanced approach called community-based social marketing was adopted at the Palos Verdes Superfund site in California to reduce the limitations of seafood consumption advisories (EPA 2009a, 2009b). This approach, pioneered by Doug McKenzie-Mohr of St. Thomas University in Canada in 1999, as cited in EPA (2009a), can be summarized broadly as:

- Researching to establish and quantify baseline behaviors and size/demography of different populations and to identify culturally-specific barriers and benefits.
- Defining desired behaviors and understanding barriers to achieving those behaviors; definition of incentives for overcoming barriers and achieving behavior change.
- Creating effective messages/incentives and effective delivery and monitoring mechanisms.
- Implementing culturally-appropriate outreach to all target populations using brief, clear, tested messages and incentives.
- Following up on research after a time period to monitor and evaluate levels of behavior change and to modify the approach as needed.

Application of community-based social marketing concepts in the LDW, modeled after the program and experience-base developed for the Palos Verdes site, could improve the effectiveness of existing seafood consumption advisories for protecting human health.

A collaborative advisory group could be convened to develop an LDW-specific framework and technical approach. Likely participants would include EPA, Ecology, WDOH, WDFW, and other interested federal, state, and local government agencies such as the National Oceanic and Atmospheric Administration, the Seattle Department of Neighborhoods, and ethnically-specific community group leaders, as well as non-governmental organizations and settling parties. A key mandate of the advisory group would be the founding of a small, credible, and knowledgeable core team to facilitate the effort (e.g., develop and complete surveys to better understand affected populations [demographics], and potential incentives for and barriers to improving the effectiveness of seafood consumption advisories).

The overarching goal of this effort would be to develop and implement a public outreach and education program that focuses on incentives and activities that research indicates have the greatest likelihood of adoption and would make the greatest substantive difference in environmental health. Ideally, the program would be coordinated with other health-based initiatives such as the City of Seattle's urban agriculture initiative.

Implementation of the outreach and education program could be accomplished in a number of ways, stressing culturally-appropriate teams, objective and credible participants, and a systematic approach to applying, documenting, and quantifying results of the approach. The advisory group would recommend program elements based on ideas generated by the group and the affected communities, and a review of approaches demonstrated to have caused positive behavior changes at other sites. It would also recommend appropriate programmatic changes as needed based on the evolution of monitoring and survey-based information. Example elements of the outreach and education program for enhancing the effectiveness of seafood consumption advisories include:

- Establish a website to provide up-to-date information on seafood contaminant concentrations and consumption advisories.
- Increase the use of signs containing advisory information at fishing locations.
- Conduct outreach efforts at fishing locations on a regular and periodic basis.
- Ensure all recreational anglers receive seafood consumption advisory information when purchasing licenses.
- Disseminate advisory-related information at community health facilities, schools, and at community-based functions such as health fairs.
- Encourage medical and other health professionals to communicate risks to the public.

A significant difference between the Palos Verdes site and the LDW is the presence in the LDW of tribal fishing rights secured by treaties of the United States. Nothing in this section or anywhere in this FS is intended to suggest that exercise of such rights, or the underlying cultural traditions, would be precluded by seafood consumption advisories and related programs to reduce contaminated seafood consumption as part of LDW remedial action. For this reason, the seafood consumption advisories, and public outreach education programs should be developed in consultation with affected tribes to develop accommodations for such tribes to the greatest extent practicable. A significant limitation of the Palos Verdes enhancement to conventional seafood consumption advisories is that individual responses remain entirely voluntary.

7.2.2.3 Enforcement Tools

As mentioned above in the context of the potential development of monitoring and notification programs as a selected component of remedial action for the LDW, RNAs are created by the promulgation of formal rules by the USCG. RNAs represent an enforceable means of protecting containment remedies and other areas where contamination remains from anchoring and other physical interference, particularly where UECA covenants or other proprietary controls may not be achievable, such as within Commercial Waterway District #1. To the extent that RNAs may potentially interfere with seafood harvest activities, particularly tribal harvests, engineered or other alternative means of accommodating fish harvest should be devised (e.g., alternative means of allowing anchoring or tying off a net within a RNA-created no-anchor zone).

Although this option has the significant potential to regulate potential impacts associated with anchorage, barge spudding, and tugboat propeller wash, it could restrict maritime commerce or preclude commercial activities generally necessary for construction, maintenance, and operation of commercial piers, depending on where the RNA was located. Like proprietary controls generally, even for sediment areas in private ownership, RNAs require a careful and often highly complex balancing of competing interests, and may only be useful in certain locations or circumstances. Whenever the government limits or adversely affects property rights, it may be subject to takings claims by affected persons based on the Fifth Amendment to the Constitution of the United States.

7.2.2.4 Environmental Covenants Registry

Placement and maintenance of LDW areas, with containment remedies (cap or CAD) or anywhere where contamination remains above levels needed to meet cleanup objectives, on Ecology's Environmental Covenants Registry in its Integrated Site Information System) would provide information regarding applicable restrictions (RNAs and proprietary controls) to anyone who uses or consults the state registry.

7.2.3 Institutional Controls Summary

In summary, it must be emphasized that all of the institutional controls described in this section are difficult to enforce. Privately owned sediments, like publicly owned sediments, in an urban commercial waterway are generally substantially more difficult to guard or restrict uses of than upland properties. Further, it is anticipated that some people, including tribal members with treaty-protected harvest rights, will choose to fish and consume what they catch regardless of seafood consumption advisories and robust public outreach and education programs. For these reasons, institutional controls will be relied on only to the extent necessary to develop practicable remedial actions for the LDW.

7.3 Monitoring

Monitoring is an important assessment and evaluation tool for collecting data and is a requirement of remedial alternatives conducted under CERCLA and MTCA. Monitoring data are collected and used to assess the completeness of remedy implementation, remedy effectiveness, and the need for contingency actions. The sampling and testing process options common to most sediment remediation projects are as follows:

- ♦ Sediment quality (e.g., chemistry, grain size distribution)
- ♦ Sediment toxicity
- ♦ Surface water quality (e.g., conventional parameters and contaminant concentrations)
- ♦ Contaminant concentrations in porewater

- Contaminant concentrations in fish and shellfish tissue
- Physical (e.g., visual inspections, bathymetry).

Typically, these sampling and testing process options are prescribed components of project monitoring plans which, in turn, focus on different aspects of the remedial action. For example, monitoring during the construction phase has different objectives than the operation and maintenance (O&M) monitoring that follows construction. Five different monitoring concepts that form the basis for individual or combined monitoring plans, depending on project-specific circumstances, are described below.

In addition, source control monitoring (addressed under Tier 4 of the source control strategy, see Section 2.4) and evaluation within upland drainage basins will be required by Ecology in parallel with in-water monitoring for remedial actions and may include parties other than those responsible for performing the remedial action. The goal of source control monitoring is to determine the potential for recontamination in areas that have already been remediated and become subsequently recontaminated above LDW cleanup levels. Type and scope of source control monitoring is not discussed in the FS since this varies on a site by site basis.

7.3.1 Baseline Monitoring

Baseline monitoring establishes a statistical basis for comparing physical and chemical site conditions prior to, during, and after completion of a cleanup action. Baseline monitoring for the LDW will likely entail the sampling and analysis of sediment, surface water, and tissue samples in accordance with a sampling design that enables such a statistical comparison of conditions.

7.3.2 Construction Monitoring

Construction monitoring during active remediation is area-specific and short-term and is used to evaluate whether the project is being constructed in accordance with plans and specifications (i.e., performance of contractor, equipment, and environmental controls). This type of monitoring evaluates water quality in the vicinity of the construction operations to determine whether contaminant resuspension and dispersion are adequately controlled.

Further, bathymetric monitoring data establish actual dredge prisms or the placement location and thickness of cap material.

7.3.3 Post-construction Performance Monitoring

Post-construction performance monitoring at the conclusion of in-water construction evaluates post-removal sediment conditions in dredging or containment areas. Both chemical and physical data are collected to determine whether the work complies with project specifications.

7.3.4 Operation and Maintenance (O&M) Monitoring

O&M monitoring refers to data collection for the purpose of tracking the technology performance, long-term effectiveness, and stability of individual sediment cleanup areas.

In capping areas, O&M monitoring typically consists of analysis including COCs, grain size, TOC, and cap thickness using sediment or porewater matrices. A combination of tools, including bathymetry soundings, surface grab samples, sediment cores, diver surveys, peepers, staking, and/or settlement plates is used to evaluate cap performance. Some of these tools are also used for ENR and MNR performance monitoring.

7.3.5 Long-term Monitoring

Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action to assess risk reduction and progress toward achievement of cleanup objectives. Data collected under long-term monitoring yields information reflecting the combined actions of sediment remediation and source control.

7.3.6 Monitoring Summary

Monitoring is an essential element of remedial alternatives developed in this FS. Appendix K set forth key assumptions and an overall framework for monitoring using the process options and monitoring objectives described above. Appendix K also cross references these monitoring terms and concepts with those used in MTCA.

7.4 Ancillary Technologies

7.4.1 Barge Dewatering

Dewatering mechanically dredged sediment on transfer barges prior to additional sediment handling (e.g., off-loading and disposal) is an important interim management step. Dewatering produces a more consolidated sediment load and reduces the volume of water that would otherwise need to be managed elsewhere (e.g., at a transloading facility or at a landfill). Typically, the dewatering step occurs on a transfer barge within the dredge operations area by gravity settling and separation. In the past, the separated water was decanted directly back to the receiving water without further treatment. This confines the discharge to the area that is already seeing elevated turbidity as a result of dredge operations. Barge dewatering in this manner is typical of sediment remediation projects conducted in the Puget Sound region and this FS assumes it will be part of the remedial alternatives for costing purposes. As discussed below, more recent projects have included treatment.

Examples of Puget Sound region projects that used this technology are provided below. Each was implemented in compliance with project-specific water quality certifications.

Todd Shipyards (Seattle, Washington)

A patented (General Construction Company) sloping drain barge was used on this project. The technique involved ballasting one end of the barge with ecology blocks to create a sloping deck surface, which in turn, promotes gravity drainage to the down-slope end of the barge (Figure 7-10). The down-slope end of the barge is equipped with an overflow weir. The separated water was released directly back into the receiving water without further treatment.

Denny Way and East Waterway Phase 1 (Seattle, Washington)

For these two projects, dredged material was placed on flat-deck barges equipped with fabric-lined scuppers to allow gravity drainage of sediment. Sediment was retained in the barge, while the separated water was decanted directly back into the receiving water through the scuppers without further treatment.

Hylebos Waterway Sediment Remediation (Tacoma, Washington)

Dredged material was placed in hopper barges for gravity dewatering. Excess water from the hopper barge was decanted, treated to the water quality standards set for the project, and released back into the waterway. During the initial project phase, water treatment consisted of adding flocculants followed by routing the water through a series of weirs to enable suspended solids removal prior to discharging the water to the water body. During the final phase, a combination of flocculants and mixing tanks were used to treat the water prior to release to the water body.

Slip 4 Non-Time Critical Removal Action (Seattle, Washington)

For the recently completed Slip 4 project (one of the EAAs), a barge-based process was used that filtered the decant water through geotubes and several layers of geotextile fabric, and then drained the filtered water through granular activated carbon. While not a required element of the Slip 4 project, this step reduced turbidity in the return water. The project was completed in compliance with the water quality permit issued for the project.

7.4.2 Wastewater Treatment Associated with Sediment Remediation

Remedial alternatives that involve the removal and upland handling of contaminated sediment invariably generate wastewater that must be managed, treated, and discharged in a manner consistent with ARARs. Wastewater treatment technologies (e.g., for treatment of stormwater or industrial wastewater) are standard, myriad, and ubiquitous in their application to a wide variety of site-specific conditions. Treatment trains using conventional equipment are capable of treating water generated during sediment remediation projects to levels consistent with ARARs.

Section 8 assumes wastewater treatment would be required at a transloading facility to manage water generated from dewatering of sediments. Selection of an appropriate treatment train for this wastewater would require characterizing the wastewater properties and, potentially, conducting some treatability testing. The process options likely to be employed are expected to be standard and commercially available. For example, a common treatment train consists of gravity separation to remove suspended solids, media (e.g., sand) filtration, and adsorption on granular activated carbon for removal of dissolved organic compounds. Depending on dissolved metals concentrations, a chemical coagulation/flocculation process step might also be required. Discharge of treated water, similar to the soil-washing water treatment discharge (Section 7.1.2.2), would likely be directly back to the LDW after treatment, and would be governed by a CWA 401 water quality certification.

Discharge to the King County Metro sewer system could also be considered where the discharge meets flow (i.e., capacity) and chemical parameter limits. This approach would be an off-site disposal action, potentially requiring pretreatment to achieve discharge criteria and comply with all permit requirements (e.g., daily discharge volume, etc.), so as not to contribute to an overflow event (e.g., holding tanks for monitored flow).

7.4.3 Best Management Practices

Implementation of best management practices (BMPs) is widely considered essential to sediment cleanup projects (NRC 2007). BMPs are particularly important for environmental dredging to minimize release to the environment of contaminated material (sediment, water, debris) from the dredging footprint, and during barge transport, off-loading, and upland rehandling.

Environmental dredging to remove COCs also causes some residual sediment contamination (Palermo 2008). Contaminated sediments that are dislodged or suspended by the dredging operation are subsequently redeposited on the bottom either within or adjacent to the dredging footprint. The primary causes for this residual contamination are described in Section 7.1.1.2.

Resuspended residuals generally accumulate (settle) above the dredging cutline in thin layers, and are characterized by fine-grained sediment, being unconsolidated, having a high moisture content, and possibly existing as a fluid mud layer. The constituent COC concentrations in the residual layer can be approximated using the average dredge prism concentration (Hayes and Patmont 2004). The residual layer can be present within and adjacent to the dredge prism.

Potential BMPs to evaluate during design for dredging residuals and water quality management include:

- Remove debris prior to dredging.
- Minimize residuals generation by dredge control and design, such as carefully controlling depth, location, and cutting action to maximize sediment capture and minimize sloughing and bottom impacts. Optimize the fill efficiency of a dredge bucket to minimize both free-water capture and overfill fallback.

- Control speed of bucket through the water column to minimize loss of adhered sediment.
- Allow sediment-filled bucket to drain before fully emerging above the water surface.
- Contain drippage during the overwater swing of a filled bucket (e.g., by placing an empty barge or apron under the swing path during offloading).
- Wash bucket prior to lowering back into the water column.
- Use environmental or sealed bucket if practicable and if proper sediment conditions exist.
- Start dredging in upslope areas and move downslope to minimize sloughing.
- Plan multiple dredge cuts: limit initial cut depths to avoid sloughing of the cut bank; plan initial cut(s) to remove most of the contamination; and design a final "cleanup" cut into subsurface "clean" sediment to lower the average dredge prism COC concentrations.
- Use floating and/or absorbent booms to capture floating debris or oil sheens.
- Use conventional construction stormwater BMPs to control and reduce the silt burden in runoff from barges or rehandling areas.
- Develop and implement a post-dredging residuals monitoring and management plan.
- Monitor natural recovery of dredged area.
- Place a thin-layer sand cover (ENR) to address residuals.

The use of silt curtains around the dredging operations to reduce the transport of suspended solids is an engineering control that can be employed under certain circumstances. However, the effectiveness of a silt curtain is primarily determined by the hydrodynamic conditions at the site (usually relatively shallow, quiescent water, without significant tidal fluctuations are preferred), the quantity and type of suspended solids, the mooring method, and the characteristics of the barrier. Often, strong currents (greater than 2.5 ft/second) are problematic, and any application and deployment of silt curtains for high velocities would require special design and engineering features (USACE 2008a). In the Puget Sound region, silt curtains are not frequently used in areas where there are large tidal excursions, high-flow velocities, conflicts between dredging activities and navigation, or other technical limitations.

The specific array of BMPs or engineering controls implemented during cleanup will be location-specific and will be determined during design of the remedial alternative. Often, the remedial design specifications define certain BMPs along with performance requirements (such as water quality standards) to which the contractor must adhere. The contractor typically is required to provide additional details on specific BMPs in their work plans. Monitoring and adaptive management are common practices that will be used to refine and optimize BMPs throughout the duration of the project to ensure compliance with the project performance requirements. Representative BMPs have been identified as part of the FS remedial alternatives to develop cost estimates.

7.5 Summary of Representative Process Options for the FS

The shaded rows of Table 7-4 show the representative technology process options carried forward to Section 8 for potential development and evaluation of remedial alternatives. Consistent with CERCLA guidance, alternate process options may be considered during remedial design.

The suite of technologies and institutional controls is consistent with most of the sediment feasibility studies and cleanup projects conducted to date within the Puget Sound region and around the country. Further, it is consistent with recent deliberations and reports that have emerged from the sediment remediation community nationwide (NRC 2007). These reports conclude that a limited number of engineering approaches are available to address sediment cleanup and that some combination of dredging, disposal, capping, ENR, and MNR will invariably be at the core of almost every future major project.

Table 7-1 Initial Screening of Candidate Remedial Technologies

GRA	Technology Type	Process Option	Description
No action	None	Not applicable	No active remedy or monitoring.
Institutional controls	Proprietary controls and informational devices (EPA 2000)	Proprietary controls	Mechanisms in deeds or other instruments transferring property that restrict or affect the use of property
		Seafood Consumption Advisories, Education and Public Outreach	Public advisories that consumption of resident LDW fish and shellfish (and sediment contact) may present health risks.
		Monitoring and notification of waterway users	Regulatory constraints on uses such as vessel wakes, anchoring, and dredging. Physical constraints, such as fencing and signs, placed on property access points that limit human access to areas that pose a health risk.
		Enforcement Tools	Agency consent decrees or orders overseeing implementation of institutional controls and monitoring. Restrictive Navigation Areas, per Coast Guard formal rulemaking, could be an enforceable means of protecting containment remedies and other areas from anchoring and other physical interference, particularly where UECA covenants or other proprietary controls may not be achievable.
		Site Registry	Placement and maintenance of site information on the State Registry (Ecology's Hazardous Sites list and Site Register) would provide information regarding restrictions on the property.
Monitoring	Physical and chemical assessment	Baseline Monitoring	Establishes a statistical basis for comparing site conditions before, during, and after the cleanup action.
		Construction Monitoring	Short-term monitoring during remediation used to evaluate whether the project is being constructed in accordance with specifications (i.e., water quality monitoring, bathymetric surveys, discharge monitoring, inspection surveys, sediment monitoring).
		Post-construction Performance Monitoring	Post-construction performance monitoring evaluates post-removal surface and subsurface sediment conditions in dredging or containment areas to confirm compliance with project specifications.
		Long-term Operation and Maintenance Monitoring	Long-term operation and maintenance monitoring of dredging areas, containment, and/or disposal sites (i.e., CAD sites, ENR, and capping areas) required to ensure long-term effectiveness and continued stability of the structure.
		Long-term Monitoring	Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action.
Monitored natural recovery	Chemical/physical transport and degradation	Combination	Desorption, dispersion, diffusion, dilution, volatilization, resuspension, and transport.
	Biological degradation	COC metabolism	Chlorine atoms are removed from PCB molecules by bacteria; however, toxicity reduction is not directly correlated to the degree of dechlorination. PAHs may be partially or completely degraded.
	Physical-burial processes	Sedimentation	Contaminated sediments are buried (by naturally occurring sediment deposition) to deeper intervals that are less biologically available. (Resuspension and transport are minor components of MNR.)

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Enhanced natural recovery	Thin-layer placement	Placement of thin layer to augment natural recovery	Application of a thin layer of clean sand and natural resorting, sedimentation, or bioturbation to mix the contaminated and clean sediments, resulting in acceptable contaminant concentrations.
Containment	Capping	Conventional sand cap	Placement of clean sand over existing contaminated bottom to physically isolate contaminants.
		Conventional sediment / clay cap	Use of dredged fine-grained sediments or commercially obtained clay materials to achieve contaminant isolation.
		Armored cap	Coarse granular material such as: cobbles, pebbles, or larger material are incorporated into the cap to prevent erosion in high-energy environments or to prevent cap breaching by bioturbators (example: membrane gabions).
		Composite cap	Soil, media, and geotextile cap placed over contaminated material to inhibit migration of contaminated pore water and/or inhibit bioturbators.
		Spray cap	Placement of capping materials (usually concrete) by spraying concrete or mortar from a nozzle at high velocity onto a surface via pressure hoses with either a dry or wet mix process.
		Reactive cap	Incorporation of materials such as granular activated carbon or iron filings to provide chemical binding or destruction of contaminants migrating in porewater.
Removal	Dredging	Hydraulic dredging	Hydraulic dredges use a cutter head, and suction provided by an on-board pump(s) to agitate, entrain, and hydraulically transport sediment via pipeline to a land-based sediment handling facility or slurry discharge location.
		Mechanical dredging	A barge-mounted floating crane on a derrick barge maneuvers a dredging bucket. The bucket is lowered into the sediment; when the bucket is withdrawn, the jaws of the bucket are closed, retaining the dredged material.
		Mechanical dredging (excavator)	Excavator dredges use a barge-mounted excavator with fixed arm linkages (boom and stick), instead of cables, to position the clamshell bucket at the target elevation for sediment removal.
	Excavating	Dry excavation	Sediment is removed by upland-based conventional excavation (backhoe) equipment. Removal during low tides may not require sheet-pile walls or cofferdams. This removal option may include erecting sheet-pile walls or a cofferdam around the contaminated sediments to dewater.
In Situ treatment	Biological*	*In situ* slurry biodegradation*	Anaerobic, aerobic, or sequential anaerobic/aerobic degradation of organic compounds with indigenous or exogenous microorganisms. Oxygen, nutrients, and pH are controlled to enhance degradation. Requires sheet piling around entire area and slurry treatment performed using aerators and possibly mixers.
		In situ aerobic biodegradation*	Aerobic degradation of sediment *in situ* with the injection of aerobic biphenyl enrichments or other co-metabolites. Oxygen, nutrients, and pH are controlled to enhance degradation.
		In situ anaerobic biodegradation*	Anaerobic degradation *in situ* with the injection of a methanogenic culture, anaerobic mineral medium, and routine supplements of glucose to maintain methanogenic activity. Nutrients and pH are controlled to enhance degradation.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
In Situ treatment (cont)	Biological	Imbiber Beads™*	A "cover blanket" of Imbiber Beads™ placed over contaminated sediments to enhance anaerobic microbial degradation processes and allow exchange of gases between sediments and surface water. The beads are spherical plastic particles that would adsorb PCB vapors generated.
	Chemical*	Aqua MecTool™ oxidation*	A caisson (18' by 18') is driven into the sediment and a rotary blade is used to mix sediment and add oxidizing agents such as ozone, peroxide, or Fenton's reagent. A bladder is placed in the caisson to reduce TSS and the vapors may be collected at the surface and treated.
		In situ oxidation*	Oxidation of organics using oxidizing agents such as ozone, peroxide, or Fenton's reagent.
		Electro-chemical oxidation*	Proprietary technology in which an array of single steel piles is installed and low current is applied to stimulate oxidation of organics.
	Physical-extractive processes*	Sediment flushing*	Water or other aqueous solution is circulated through contaminated sediment. An injection or infiltration process introduces the solution to the contaminated area and the solution is later extracted along with dissolved contaminants. Extraction fluid must be treated and is often recycled.
		In situ slurry oxidation*	An array of injection wells is used to introduce oxidizing agents such as ozone to degrade organics.
	Phy*sical-immo*bilization	Aqua MecTool™ stabilization*	A caisson (18' by 18') is driven into the sediment and a rotary blade is used to mix sediment and add stabilizing agents. A bladder is placed in the caisson to reduce TSS and the vapors may be collected at the surface and treated.
		Vitrification*	Uses an electric current *in situ* to melt sediment or other earthen materials at extremely high temperatures (2,900-3,650 °F). Inorganic compounds are incorporated into the vitrified glass and crystalline mass and organic pollutants are destroyed by pyrolysis. *In situ* applications use graphite electrodes to heat sediment.
		Ground freezing*	An array of pipes is placed *in situ* and brine at a temperature of -20 to -40°C is circulated to freeze soil. Recommended only for short duration applications and to assist with excavation.
		Activated Carbon Amendment **	Activated carbon (powder, granules, or pellets) serves as an amendment to the bioactive surface layer of sediment. Hydrophobic organic contaminants adsorb to activated carbon particles, reducing porewater contaminant concentrations and bioavailability for uptake by organisms.
		Organoclay Amendment **	Organoclay products for use in sediment remediation consist of mineral clay, polymer additives, and an aggregate core for densification. Organoclays bind contaminants through replacement of metal ions with amines or other functional groups, physically isolate the contaminated sediment from receptors (because of low permeability), and stabilize sediment by preventing resuspension and transport of contaminants.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Ex Situ treatment	Biological*	Landfarming/ Composting*	Sediment is mixed with amendments and placed on a treatment area that typically includes leachate collection. The soil and amendments are mixed using conventional tilling equipment or other means to provide aeration. Moisture, heat, nutrients, oxygen, and pH can be controlled to enhance biodegradation. Other organic amendments such as wood chips, potato waste, or alfalfa are added to composting systems.
		Biopiles*	Excavated sediments are mixed with amendments and placed in aboveground enclosures. This is an aerated static pile composting process in which compost is formed into piles and aerated with blowers or vacuum pumps. Moisture, heat, nutrients, oxygen, and pH can be controlled to enhance biodegradation.
		Fungal biodegradation*	Fungal biodegradation refers to the degradation of a wide variety of organopollutants by using fungal lignin-degrading or wood-rotting enzyme systems (example: white rot fungus).
		Slurry-phase biological treatment*	An aqueous slurry is created by combining sediment with water and other additives. The slurry is mixed to keep solids suspended and microorganisms in contact with the contaminants. Upon completion of the process, the slurry is dewatered and the treated sediment is removed for disposal (example: sequential anaerobic/aerobic slurry-phase bioreactors).
		Enhanced biodegradation*	Addition of nutrients (oxygen, minerals, etc.) to the sediment to improve the rate of natural biodegradation. Use of heat to break carbon-halogen bonds and to volatilize light organic compounds (example: D-Plus [Sinre/DRAT]).
	Chemical*	Acid extraction*	Contaminated sediment and acid extractant are mixed in an extractor, dissolving the contaminants. The extracted solution is then placed in a separator, where the contaminants and extractant are separated for treatment and further use.
		Solvent extraction(s)*	Contaminated sediment and solvent extractant are mixed in an extractor, dissolving the contaminants. The extracted solution is then placed in a separator, where the contaminants and extractant are separated for treatment and further use (example: B.E.S.T.™ and propane extraction process).
	Chemical/ Physical	Reduction/ Oxidation*	Reduction/oxidation chemically converts hazardous contaminants to nonhazardous or less toxic compounds that are more stable, less mobile, and/or inert. The oxidizing agents most commonly used are hypochlorites, chlorine, and chlorine dioxide.
		Slurry oxidation*	The same as slurry-phase biological treatment with the exception that oxidizing agents are added to decompose organics. Oxidizing agents may include ozone, hydrogen peroxide, and Fenton's reagent.
		Dehalogenation*	Dehalogenation process in which sediment is screened, processed with a crusher and pug mill, and mixed with sodium bicarbonate (base catalyzed decomposition) or potassium polyethylene glycol. The mixture is heated to above 630 °F in a rotary reactor to decompose and volatilize contaminants. Process produces biphenyls, olefins, and sodium chloride.

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Ex Situ treatment (cont)	Chemical/ Physical (cont)	Soil washing	Contaminants sorbed onto fine soil particles are separated from bulk soil in an aqueous-based system on the basis of particle size. The wash water may be augmented with a basic leaching agent, surfactant, pH adjustment, or chelating agent to help remove organics and heavy metals.
		Radiolytic dechlorination*	Sediment is placed in alkaline isopropanol solution and gamma irradiated. Products of this dechlorination process are biphenyl, acetone, and inorganic chloride. Process must be carried out under inert atmosphere.
	Physical	Particle Separation	Contaminated fractions of solids are concentrated through gravity, magnetic, or sieving separation processes.
		Solar detoxification*	Through photochemical and thermal reactions, the ultraviolet energy in sunlight destroys contaminants.
		Solidification	The mobility of constituents in a “solid” medium is reduced through addition of immobilization additives.
	Thermal	Incineration*	Temperatures greater than 1,400°F are used to volatilize and combust organic contaminants. Commercial incinerator designs are rotary kilns equipped with an afterburner, a quench, and an air pollution control system.
		High Temperature Thermal Desorption*	Temperatures in the range of 600-1,200°F are used to volatilize organic contaminants. These thermal units are typically equipped with an afterburner and baghouse for destruction of air emissions. Wastes are heated to volatilize water and organic contaminants. A carrier gas or vacuum system transports volatilized water and organics to the gas treatment system (examples: X*TRAX™, DAVES, Tacuik Process, and Holoflite™ Dryer).
		Low Temperature Thermal Desorption*	Temperatures in the range of 200-600°F are used to volatilize and combust organic contaminants. These thermal units are typically equipped with an afterburner and baghouse for treatment of air emissions.
	Thermal (cont)	Pyrolysis*	Chemical decomposition is induced in organic materials by heat in the absence of oxygen. Organic materials are transformed into gaseous components and a solid residue (coke) containing fixed carbon and ash.
		Vitrification*	Current technology uses oxy-fuels to melt soil or sediment materials at extremely high temperatures (2,900-3,650°F).
		High-pressure oxidation*	High temperature and pressure are used to break down organic compounds. Operating temperatures range from 150-600°C and pressures range from 2,000-22,300 MPa (examples: wet air oxidation and supercritical water oxidation).

Table 7-1 Initial Screening of Candidate Remedial Technologies (continued)

GRA	Technology Type	Process Option	Description
Disposal	On-site disposal	Level-bottom cap*	Relocation of contaminated sediment to discrete area and capping with a layer of clean sediments. Provides similar protection as capping, but requires substantially more sediment handling that may cause increased releases to surface water.
		Contained Aquatic Disposal (CAD)	Untreated sediment is placed within a lateral containment structure (i.e., bottom depression or subaqueous berm) and capped with clean sediment.
		Confined Disposal Facility (CDF)	Untreated sediment is placed in a nearshore CDF that is separated from the river by an earthen berm or other physical barrier and capped to prevent contact. A CDF may be designed for habitat purposes.
	Off-site Disposal	Subtitle D landfill	Off-site disposal at a licensed commercial facility that can accept nonhazardous sediment. Regional landfills can accept both dewatered and wet sediments.
		Subtitle C landfill	Off-site disposal at a licensed commercial facility that can accept hazardous dewatered sediment removed from dredging or excavation. Dewatering required to reduce water content for transportation.
		TSCA-licensed landfill*	Off-site disposal at a licensed commercial facility that can accept TSCA sediment. Dewatering required to reduce water content for transportation.
		DMMP open water non-treated (if acceptable) disposal	Treated or separated sediment is placed at the Elliott Bay DMMP disposal site. Requires that the placed sediment be at, or below, DMMP disposal criteria for priority pollutants and potentially bioaccumulative contaminants.
		Upland MTCA confined fill (commercial/industrial – beneficial use)*	Treated or untreated sediment is placed at an off-site location. Requires that sediment be at, or treated to, MTCA cleanup levels at an off-site location and meet nondegradation standards. Location may require cap or other containment devices based on analytical data.
		Upland MTCA fill (residential/clean – beneficial use)	Treated or untreated sediment is placed at an off-site location. Requires that sediment be at, or treated to, a concentration at or below MTCA cleanup levels for unrestricted land use and meet nondegradation standards.
		In-water beneficial use	Sediments treated to below DMMP guidelines may be beneficially reused for habitat creation, capping, or residual management.

Notes:

Shaded technologies and process options are retained at end of initial screening as potentially feasible at the end of the Table 7-2 series, where more detailed screening information is provided. These process options were retained at the conclusion of the detailed screening and are evaluated in Table 7-3 for applicability in the LDW with the exception of institutional controls, which do not lend themselves to comparison on the same terms as other technologies. Institutional controls are discussed only within Section 7.2 and are not included in Tables 7-2 and 7-3.

A detailed description of these process options is not included in the FS text. Details regarding these technology and process options are provided in the document *Identification of Candidate Cleanup Technologies for the Lower Duwamish Waterway* prepared by The RETEC Group Inc. (2005). These process options were eliminated in the detailed screening process shown in Table 7-2 series. The *in situ* treatment (activated carbon and organoclay amendments) have been added to this table as a result of recent advances in these technologies and project case studies now available for review.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; DMMP = Dredged Material Management Program; ENR = enhanced natural recovery; GRA = general response action; MTCA = Model Toxics Control Act; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; TSCA = Toxic Substances Control Act; TSS = total suspended solids; UECA = Uniform Environmental Covenants Act

Table 7-2a Detailed Screening of Process Options: No Action, Institutional Controls, and Monitoring

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Screening Decision	Cost
No Action	None	Not Applicable	—	Retained per NCP requirement	Technically implementable for conditions within the LDW.	—	—	Retained per NCP requirement	Low
Institutional Controls				All retained				All retained	Low
Monitoring	Physical and Chemical Assessment	Baseline Monitoring	Can be effective for evaluating changes.	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate
		Construction Monitoring	Can be effective for evaluating changes.	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Low
		Post-construction Performance Monitoring	Can be effective for evaluating changes	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Low
		Operation and Maintenance Monitoring	Can be effective for evaluation and maintenance of LDW following remedial actions	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate
		Long-term Monitoring	Can be effective for evaluating sediment, tissue and water quality over an extended period of time following remedial actions	Retained for further evaluation	Technically implementable for conditions in the LDW.	Available and demonstrated	—	Retained for further evaluation in the FS	Moderate to High

Note:
COC = contaminant of concern; CTM = Candidate Technologies Memo; FS = feasibility study; GRA = general response action; NCP = National Contingency Plan

Table 7-2b Detailed Screening of Process Options: Monitored Natural Recovery and Enhanced Natural Recovery

GRA	Technology Type	Process Option	Effectiveness Screening: LDW COCs	Effectiveness Screening: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Implementability: Screening Decision	Cost
Monitored Natural Recovery	Chemical Degradation	Natural Désorption, Diffusion, Dilution, Volatilisation	Potentially effective for immobilizing COCs through TOC or sulfide sorption.	Retained for further evaluation	Technically implementable for conditions within the LDW	—	—	Retained for further evaluation in the FS	Low
Monitored Natural Recovery	Biological Degradation	COC Metabolism (aerobic and anaerobic)	Effective for SVOCs and PAHs but does not result in complete destruction of PCBs or TBT in acceptable time frame. Not applicable to metals.	Retained for further evaluation	Technically implementable for conditions within the LDW	—	—	Retained for further evaluation in the FS for SVOCs only	Low
Monitored Natural Recovery	Physical/Burial Processes	Natural Sedimentation and Burial (resuspension and transport are minor components of MNR)	Potentially effective for LDW COCs via deposition and reburial. Requires demonstration of long-term deposition and burial.	Retained for further evaluation	Technically implementable for conditions within the LDW	Preliminary results at some projects show some success.	—	Retained for further evaluation in the FS	Low
Enhanced Natural Recovery	Thin-layer Placement	Thin-layer Placement	Effective for all LDW COCs. Applicable: 1) at areas where MNR processes are demonstrated, but faster recovery is required; or 2) as a residual management tool after completion of a removal action.	Retained for further evaluation	Technically implementable for conditions within the LDW.	Thin-layer placements for ENR and residuals management have been applied in multiple locations in Puget Sound and nationally.	—	Retained for further evaluation in the FS	Low to Moderate

Note:

CTM = Candidate Technologies Memorandum; COC = contaminant of concern; ENR = enhanced natural recovery; FS = feasibility study; GRA = general response action; MNR = monitored natural recovery; PAHA = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; SVOC = semivolatile organic compound; TOC = total organic carbon; TBT = tributyltin

Table 7-2c Detailed Screening of Process Options: Containment Process Options

GRA	Technology Type	Process Option	Effectiveness		Implementability				Cost
			LDW COCs	Screening Decision	Site Conditions	Available and Demonstrated	Innovative Technology	Screening Decision	
Containment	Capping	Conventional Sand Cap	Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained for consideration throughout the LDW	Applicable to LDW conditions. Easily applied *in situ;* however, scouring must be considered. Decreased water depth may limit future uses of waterway and may impact flooding, stream bank erosion, navigation, and recreation.	Conventional sand caps have been applied in multiple locations in Puget Sound and nationally.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Conventional Sediment/Clay Cap	Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact. Sediment with silt and clay is effective in limiting diffusion of contaminants. Sediment caps are generally more effective than sand caps for containment of contaminants with high solubility and low sorption	Retained for consideration throughout the LDW	Generally applicable to LDW conditions. Placement of clay caps is considered in shallow water depth areas where minimal cap thickness is required. Special engineering controls will be needed to place clay cap in the LDW.	Conventional sediment caps using river-dredged sediments have been applied in multiple locations in Puget Sound and nationally. Application of clay caps is relatively new, but demonstrated.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Armored Cap	Applicable to LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained for limited use in high-energy sections of the LDW	Applicable to areas of LDW where increased velocities from river flow, or potential scouring associated with propeller wash might be expected. Decreased water depth may limit future uses of waterway and may impact flooding, stream bank erosion, navigation, and recreation. Limited use in intertidal areas that support clamming and recreational activities.	Armored caps have been implemented at several sites in Puget Sound and nationally.	—	Retained for limited use in the FS for high-energy sections of the LDW.	Low to Moderate
		Composite Cap (geotextile, HDPE)	Effective for LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants. Can be used: 1) to limit cap thickness, 2) for low solids underlying sediments where additional floor-support is required, 3) as a bioturbation barrier, or 4) as a barrier for areas where methane generation may be an issue.	Retained for consideration throughout the LDW	Applicable to LDW site conditions. Application must consider that decreased water depth may limit future uses of waterway and impact flooding, stream bank erosion, navigation, and recreation. Limited use in intertidal areas that support clamming and recreational activities.	Application of composite capping is relatively new, but commercially demonstrated for projects with similar size and scope.	—	Retained for consideration in the FS for all areas of the LDW.	Low to Moderate
		Spray Cap	Confines COCs by encapsulating with shotcrete (usually concrete) placed over underlying surface.	Retained for consideration throughout the LDW	Applicable to hard to access areas under piers and wharves. Shotcrete cap reduces the habitat value of the intertidal sediment bed.	Shotcrete was used at the Todd Shipyards effectively encapsulating existing debris (slag) mounds under dock structures from the aquatic environment.	Demonstrated effective at recent Puget Sound region remediation project.	Retained for consideration in the FS for application in hard to access areas under piers or wharf structures.	Low to Moderate
		Reactive Cap	Effective for LDW COCs. Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants.	Retained	Reactive caps may be applicable to site conditions on the LDW. Limited use in intertidal areas that support clamming and recreational activities.	Addition of materials to increase sorptive capacity of cap has been implemented in Puget Sound. Long-term effectiveness data may be available during the LDW FS.	Reactive capping is an innovative technology that is in the demonstration phase on the Anacostia River. Results of those tests are expected during the LDW FS.	Retained for consideration in the FS as an innovative technology.	Low to Moderate

Notes:

COC = contaminant of concern; FS = feasibility study; GRA = general response action; HDPE = High-density polyethylene

Table 7-2d Detailed Screening of Process Options: Treatment Process Options

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
In Situ Treatment	Biological	*In Situ* Slurry Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		In Situ Aerobic Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		In Situ Anaerobic Biodegradation	Biodegradation has not been demonstrated to effectively remediate metals, PCBs, or TBT within a reasonable time frame.	Eliminated	—	—	—	—	—
		Imbiber Beads™	Potentially applicable to PCBs and SVOCs, not metals. No data on effectiveness with TBT. Not demonstrated for remediation of sediments. Removal and disposal of the blanket is not demonstrated.	Eliminated	—	—	—	—	—
	Chemical	Aqua MecTool™ Oxidation	Technology is effective for PCBs, SVOCs in soils. Process should be effective for TBT, but not metals.	Retained for further consideration	Could be applicable to conditions in LDW. Requires treating sediments in place using caisson and proprietary injectors.	Not demonstrated in pilot- or full-scale sediment projects. Technical difficulties in field trials injecting high air flows into caisson with standing water while preventing generation of TSS.	Not considered innovative or available during LDW FS.	Eliminated	—
		In Situ Oxidation	Has not been demonstrated to be effective for LDW COCs in sediments.	Eliminated	—	—	—	—	—
		Electro-chemical Oxidation	Applicability for use in water is not known. No demonstrated sediment application.	Eliminated	—	—	—	—	—
	Physical-Extractive Processes	Sediment Flushing	Bench scale effectiveness for all LDW COCs.	Retained for further consideration	Potentially applicable to LDW. Requires in-water steel piling around treatment area and extensive water quality monitoring outside piles.	No known pilot or full-scale applications.	Not considered innovative or available during LDW FS.	Eliminated	—
		In Situ Slurry Oxidation	Not demonstrated in full-scale applications effective for LDW COCs. Requires in-water steel piling around treatment area and extensive water quality monitoring outside piles.	Eliminated	—	—	—	—	—
	Physical-Immobilization	Aqua MecTool™ Stabilization	Proprietary technology that has been effective in stabilizing metals, PCBs and SVOCs in soil. No data available on TBT, but physical process likely to be effective on butyltins.	Retained for further consideration	Could be applicable to conditions in LDW. Requires treating sediments in place using caisson and proprietary injectors.	Proprietary technology that was tested in a pilot-scale application in Wisconsin with coal tar-contaminated sediments, and found to be not implementable. Previous trials with this technology created water treatment problems inside the caisson.	Not considered innovative or available during LDW FS.	Eliminated	—
		Activated Carbon Amendment	Effective at adsorbing organic contaminants in sediment applications. Pilot studies (in five pilot-scale demonstration projects in the United States and Norway) and research indicates technology has promising long-term effectiveness. Carbon-amended sediment provides a suitable habitat for benthic communities.	Retained for further consideration	Potentially applicable to LDW. Easily applied *in situ*; may require armoring in scour areas.	Demonstrated effective in recent pilot-scale remediation projects (San Francisco-CA, Lower Grasse River-NY, Canal Creek-MD, and Trondheim-Norway) in various aquatic environments (tidal mudflat, freshwater river, marine harbor, deep-water fjord, tidal creek, and marsh).	Activated carbon amendment is considered innovative and available during the LDW FS.	Retained for consideration in the FS	Low to Moderate
		Organoclay Amendment	Effective at adsorbing organic contaminants in sediment applications. Long-term effectiveness shown in pilot-scale demonstration projects in Anacostia River (Washington, DC).	Retained for further consideration	Potentially applicable to LDW. Easily applied *in situ*; may require armoring in scour areas.	Demonstrated effective at the Anacostia River in a recent pilot-scale remediation project that used AquaBlok® (proprietary clay polymer composite).	Organoclay amendment is considered innovative and available during the LDW FS.	Retained for consideration in the FS	Low to Moderate

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
In Situ Treatment	Physical-Immobilization	Vitrification	Effective at stabilizing COCs in soil applications, but requires less than 60% water content. Remaining sediment surface may not provide suitable habitat. No known sediment applications.	Eliminated	—	—	—	—	—
In Situ Treatment	Physical-Immobilization	Ground Freezing	Not permanently effective for LDW COCs. Long-term effectiveness in presence of standing water has not been demonstrated. Standing water likely provides a significant sink for cold temperatures and would substantially increase cost.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Landfarming/ Composting	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Biopiles	Not effective for metals, PCBs, dioxin or TBT. Used for reducing concentrations of petroleum constituents in soils. Applied to treatment of nonhalogenated VOCs and fuel hydrocarbons. Requires large upland area.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Fungal Biodegradation	Not effective for metals, PCBs, dioxins or TBT. No known full-scale applications. High concentrations of contaminants may inhibit growth. The technology has been tested only at bench scale.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Slurry-phase Biological Treatment	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation. Large volume of tankage required. No known full-scale applications.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Biological	Enhanced Biodegradation	Not effective for metals, PCBs, dioxin or TBT. PAHs and some SVOCs are amenable to aerobic degradation.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Chemical	Acid Extraction	Suitable for sediments contaminated with metals, but not applicable to PCBs or SVOCs. No data on TBT.	Eliminated	—	—	—	—	—
Ex Situ Treatment	Chemical	Solvent Extraction	Potentially effective for treating sediments containing PCBs, dioxins, or SVOCs. Not applicable to metals. No data on TBT. Extraction of organically-bound metals and organic contaminants creating residuals with special handling requirements. At least one commercial unit available.	Retained for further consideration	Potentially applicable to dewatered (dry) sediments on the LDW containing primarily organic contaminants such as PCBs. Extracted organic contaminants from the process will need to be treated or disposed. Requires pre-treatment that involves screening of sediments.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale.	This technology has been used to demonstrate under the EPA SITE program, but there are no data for similar implementation of this technology for large-scale PCB-impacted sediment. No current or planned projects.	Eliminated	—
Ex Situ Treatment	Chemical	Solvent extraction: Solvent Electron Technology (SET™)	Effective for SVOCs and PCBs, but not metals. No data on TBT. Full scale system commercially available for treatment. Mobile units can be set up to meet project requirements. Nationwide TSCA treatment permit for SET™ issued by EPA for mobile PCB chemical destruction in soils.	Retained for further consideration	Potentially applicable to dewatered (dry) sediments on the LDW. This technology results in destruction of PCBs and other organic contaminants. Operates on a closed loop system and does not produce secondary hazardous waste or off-gas.	Not demonstrated in pilot- or full-scale sediment projects.	—	Eliminated	—
Ex Situ Treatment	Chemical	Solvent Extraction; Peroxide and Ferrous Iron Treatment	Oxidation using liquid hydrogen peroxide (H_2O_2) in the presence of native or supplemental ferrous iron (Fe^{+2}) produces Fenton's Reagent which yields free hydroxyl radicals (OH^{-}). These strong, nonspecific oxidants can rapidly degrade various organic contaminants.	Retained for further consideration	Potentially applicable to LDW.	Technology is neither commercially available nor demonstrated on a project of similar size and scope.	This technology has been used for pilot studies for treating PAH-impacted sediment from Utica Harbor, but there are no data for similar implementation of this technology for PCB-impacted sediment. No current or planned projects.	Eliminated	—

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
Ex Situ Treatment (continued)	Chemical (continued)	Solvent Extraction: High Energy Electron Beam Irradiation	Full-scale system commercially available for treatment of PCBs and SVOCs, and process is limited to slurried soils, sediments, and sludges. Slurrying is a required pre-treatment for this technology. Not demonstrated to be effective in sediments. Pilot-scale testing has been performed to treat wastewaters with organic compounds. Metals are not amenable to treatment. No data on TBT.	Retained for further consideration	Potentially applicable to slurried sediments in the LDW consisting primarily of organic contaminants such as PCBs.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale.	This technology demonstrated under the EPA SITE program to treat wastewater with organic compounds, but no data for similar implementations are available for PCB-impacted sediment. No current/planned projects.	Eliminated	—
		Reduction/ Oxidation	Target contaminant group for chemical redox is inorganics. Less effective for nonhalogenated VOCs, SVOCs, fuel hydrocarbons, and pesticides. Not cost-effective for high contaminant concentrations because of large amounts of oxidizing agent required.	Eliminated	—	—	—	—	—
		Dehalogenation	PCB and dioxin-specific technology. Generates secondary waste streams of air, water, and sludge. Similar to thermal desorption, but more expensive. Solids content above 80% is preferred. Technology is not applicable to metals.	Eliminated	—	—	—	—	—
		Slurry Oxidation	Applicable to SVOCs, but not PCBs or metals. TBT treatment unknown. Large volume of tankage required. No known full-scale applications. High organic carbon content in sediment will increase volume of reagent and cost.	Eliminated	—	—	—	—	—
		Soil Washing with Air Stripping	Full-scale testing of Biogenesis™ Advanced washing process showed demonstrated effectiveness for metals, SVOCs and PCBs in sediments. Limited data suggests not effective for TBT. High recalcitrant (e.g., PCB) contaminant concentration, increased percentage of fines, and high organic content increases overall treatment costs.	Retained for further consideration	Potentially applicable to dewatered sediments on the LDW. Would require upland processing space, storage capacity for dredged sediments, wastewater treatment, and discharge. Treated residuals would still require disposal.	Equipment is commercially available, but has not been demonstrated on a project of similar scope and scale. Tests to date have been on 15,000 cy.	Full-scale testing has been performed. Mobile units available for setup. Continuous flow process designed to process up to 40 cy of sediments per hour for the full-scale system.	Retained as innovative technology to consider further in the FS.	Moderate to High
		Radiolytic Dechlorination	Only bench-scale testing has been performed. Difficult and expensive to create inert atmosphere for full-scale project.	Eliminated	—	—	—	—	—
	Physical	Particle Separation	Reduces volumes of COCs by separating sand from fine-grained sediments. Some bench scale testing has suggested that at high PCB concentrations, the sand fraction retains levels that still require landfilling.	Retained for further consideration	Potentially applicable dredged sediments in the LDW.	Separation technologies available and have been used in several programs of similar size and scope.	—	Retained to consider further in the FS.	Low
		Solar Detoxification	The target contaminant group is VOCs, SVOCs, solvents, pesticides, and dyes. Not effective for PCBs, dioxins or TBT. Some heavy metals may be removed. Only effective during daytime with normal intensity of sunlight. The process has been successfully demonstrated at pilot scale.	Eliminated	—	—	—	—	—
		Solidification	Bench-scale studies have added immobilizing reagents ranging from Portland cement to lime cement, kiln dust, pozzolan, and proprietary agents with varying success. Dependent on sediment characteristics and water content. Lime is particularly effective at volatilizing PCBs in wet sediment (by a phase transfer mechanism).	Retained for further consideration	Potentially applicable to LDW.	Lime has been successfully added to dredged material at other projects. Considered for use during the dewatering operation to remove excess water and prepare material for disposal.	—	Retained to consider further in the FS.	Moderate

Table 7-2d Detailed Screening of Process Options: Treatment Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Final Screening: Site Conditions	Final Screening: Available and Demonstrated	Final Screening: Innovative Technology	Final Screening: Screening Decision	Cost[1]
Ex situ Treatment (continued)	Thermal	Incineration	High temperatures result in generally complete decomposition of PCBs and other organic contaminants. Effective across wide range of sediment characteristics but fine grained sediment difficult to treat. Not effective for metals.	Retained for further consideration	Technically applicable to LDW site conditions. Especially effective and potentially required where COCs exceed TSCA limits (e.g., PCB >50 ppm). Only a small portion of LDW sediments are above TSCA.	Only one off-site fixed facility incinerator is permitted to burn PCBs and dioxins. Metals not amenable to incineration. No data on TBT, but should be effective. Mobile incinerators are available for movement to a fixed location in close proximity to the contaminated sediments.	—	Eliminated	—
		High-temperature Thermal Desorption (HTTD) then Destruction	Target contaminants for HTTD are SVOCs, PAHs, PCBs, TBT and pesticides, which are destroyed by the heating process. Metals not destroyed.	Retained for further consideration	Technically applicable to LDW site conditions. Especially effective and potentially required where COCs exceed TSCA limits (e.g., PCB >50 ppm).	Technology readily available as mobile units that would need to be set up at a fixed location in close proximity to the contaminated sediments. Cement-Lock® Technology demonstration projects partially destroyed organics and encapsulated metals in the product matrix. The Cement-Lock® product passes the TCLP test for priority pollutants.	Cement-Lock® Technology -Two demonstration projects started. Both experienced equipment related problems and were shut down.	Eliminated	—
		Low-temperature Thermal Desorption (LTTD)	Target contaminants for LTTD are SVOCs and PAHs. May have limited effectiveness for PCBs. Metals not destroyed. Fine-grained sediment and high moisture content will increase retention times. Widely-available commercial technology for both on-site and off-site applications. Acid scrubber will be added to treat off-gas.	Retained for further consideration	Potentially applicable to LDW.	Demonstrated effectiveness at several other sediment remediation sites. Vaporized organic contaminants that are captured and condensed need to be destroyed by another technology. The resulting water stream from the condensation process may require further treatment.	—	Eliminated	—
		Pyrolysis	High moisture content increases treatment cost. Generates air and coke waste streams. Target contaminant groups are SVOCs and pesticides. It is not effective in either destroying or physically separating inorganics from the contaminated medium	Eliminated	—	—	—	—	—
		Vitrification	Thermally treats PCBs, SVOCs, TBT, and stabilizes metals. Successful bench-scale application to treating contaminated sediments in Lower Fox River, and in Passaic River.	Retained for further consideration	Potentially applicable to LDW.	Not commercially available or applied on similar site and scale.	No known pilot or full-scale applications in sediments planned.	Eliminated	—
		High-pressure Oxidation	Predominantly for aqueous-phase contaminants. Wet air oxidation is a commercially-proven technology for municipal wastewater sludges and destruction of PCBs is poor. Supercritical water oxidation has demonstrated success for PCB destruction.	Eliminated	—	—	—	—	—

Notes:

1. Costs indicated here are relative to incineration costs.
2. Institutional controls are retained as potentially feasible and applicable to the LDW, and carried forward in the detailed screening; however, they do not lend themselves to comparison on the same terms as other technologies. Therefore, they are discussed only within Section 7.2 and are not included in Tables 7-2 and 7-3.

COC = contaminant of concern; CTM = Candidate Technologies Memorandum; cy = cubic yards; EPA = U.S. Environmental Protection Agency; FS = feasibility study; GRA = general response action; HTTD = high-temperature thermal desorption; LTTD = low-temperature thermal desorption; MNR = monitored natural recovery; NCP = National Contingency Plan; PAH = polycyclic aromatic hydrocarbon; PCB = polychlorinated biphenyls; SET™ = Sediment Electron Technology; SVOC = semivolatile organic compound; TBT = tributyltin; TCLP = Toxicity Characteristic Leaching Procedure; TOC = total organic carbon; TSCA = Toxic Substances Control Act; TSS = total suspended solids; VOC = volatile organic compound

Table 7-2e Detailed Screening of Process Options: Removal Process Options

GRA	Technology Type	Process Option	Effectiveness: LDW COCs	Effectiveness: Screening Decision	Implementability: Site Conditions	Implementability: Available and Demonstrated	Implementability: Innovative Technology	Implementability: Screening Decision	Cost
Removal	Dredging	Hydraulic Dredging	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Best suited to low density, high water solids with little debris. Requires nearshore dewatering facilities and right-of-way for slurry pipeline. Water treatment and disposal required.	Hydraulic environmental dredging is available and demonstrated in similar size projects, but is less frequently used for projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
		Mechanical Dredging	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Better suited for higher density, low water solids, and more effective at handling debris. Environmental buckets suitable for softer materials with low debris; clamshell buckets suitable for harder, dense sediments.	Mechanical environmental dredging is available and demonstrated in similar size projects, and is commonly employed for projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
		Mechanical Dredging (Excavator)	Applicable to all LDW COCs	Retained for consideration throughout the LDW	Generally applicable to LDW in-water site conditions. Better suited for higher density, low water solids, and more effective at handling debris. Environmental excavators are suited for all materials (soft and dense), better able to handle debris, but may be depth limited.	In-water excavators are available and demonstrated in similar size projects, including projects in Puget Sound.	—	Retained for consideration in the FS for all areas of the LDW.	Moderate
	Dry Excavation	On-land or Intertidal excavator, backhoes, specialty equipment	Applicable to all LDW COCs. Effective for nearshore and/or intertidal areas where depths limit conventional dredging equipment	Retained for further consideration for intertidal or nearshore areas in the LDW	Limited in application to nearshore shallow and/or intertidal areas that can be reached from shore or by specialty equipment designed to work on soft unconsolidated sediments.	Equipment is commercially available and has been applied on projects of similar scope in Puget Sound.	—	Retained for consideration in the FS for shallow and/or intertidal areas of the LDW.	Moderate

Note:
COC = contaminant of concern; FS = feasibility study; GRA = general response action

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
No Action	None	Required by NCP	Applicable to all LDW COCs.	Applicable to all COCs. Effective where risk assessment demonstrates low to no risk to human health and environment.	COCs remain in place.	Applicable throughout LDW where COC concentrations are low.	1) Readily implemented with no construction or monitoring requirements; 2) Minimal impact on industrial and shipping uses of waterway.	1) Requires source controls to be in place.	Low
Monitoring	Physical and Chemical Assessment	Monitoring	Applicable to all LDW COCs.	Can be effective for evaluating changes during implementation phase and over the long-term	1) A lot of variability in data results, difficult to discern trends; 2) Relationships not well understood for some contaminants.	Applicable to all subtidal areas of LDW.	1) Readily implementable; 2) Minimal impact on industrial and shipping uses of waterway; 3) Good for risk communication to public.	1) Requires long-term financial commitment to ensure maintenance of engineered structures (i.e., cap, CAD) and monitoring/sampling.	Moderate
Monitored Natural Recovery	Chemical Degradation	Combination of natural desorption, diffusion, dilution, volatilization, resuspension, and transport	Effective principally to LDW organic COCs including SVOCs and PCBs. Inorganics not subject to degradation.	Effective where chemical degradation of COCs is demonstrated to occur in the short- and long-term.	1) Effective where risk assessment demonstrates low to no risk to human health and environment; 2) Physical/chemical degradation demonstrated for SVOCs, but less effective for metals, PCBs, TBT and pesticides; 3) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 4) Potentially low level of short-term effectiveness for ecological receptors because COCs remain in place, but can provide adequate long-term protection; 5) Requires implementation of long-term monitoring study and risk assessment objectives.	Applicable to all areas of the LDW.	1) Readily implemented with no construction requirements; 2) Minimal impact on current or future industrial and shipping uses of waterway; 3) May be used in conjunction with other technologies in a combined alternative.	1) Must be implemented in conjunction with a well-designed, long-term monitoring program; 2) May require future active remediation where MNR risk-expectations are not achieved.	Low
	Biological-Degradation	COC Metabolization (aerobic and anaerobic)	Effective principally to SVOCs. PCBs and TBT will degrade, but not within an acceptable time frame. Metals will not degrade.	Biodegradation is a demonstrated and proven remedial technology for volatiles and SVOCs. Effective where degradation of COCs are demonstrated to occur in the short- and long-term.	1) Biological degradation less effective for PCBs and TBT; 2) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) Less effective for ecological receptors because COCs remain in place; 4) Requires implementation of long-term monitoring study and risk assessment objectives.	Applicable in areas with low concentrations of SVOCs in well-mixed sediments.	1) Readily implemented with no construction requirements; 2) Minimal impact on current or future industrial and shipping uses of waterway; 3) May be used in conjunction with other technologies in a combined alternative; 4) Implemented in areas with biodegradable COCs.	1) Must be implemented in conjunction with a well-designed long-term monitoring program; 2) May require future active remediation where MNR risk-expectations are not achieved.	Low
	Physical Burial Processes	Sedimentation/ Burial Resuspension and Transport (minor components of MNR)	Effective for all LDW COCs where concentrations are low.	1) Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants; 2) Effective for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact.	1) Requires implementation of long-term monitoring study and risk assessment objectives; 2) Short-term impacts to human health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) Less effective for ecological receptors because COCs remain in place; 4) COCs not actively removed and remain in place. 5) Facilitates PCB contamination of the marine food chain when resuspension and transport occur	Applicable where geochronological studies and hydrodynamic modeling demonstrate long-term sedimentation and burial processes are in-place.	1) Readily applied and demonstrated process; 2) Can be combined with institutional controls until long-term risk-objectives are demonstrated; 3) Minimal impact on industrial and shipping uses of waterway.	1) Requires long-term monitoring and continuing financial commitment until risk-objectives are achieved; 2) Associated institutional controls may limit future uses of waterway.	Low

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: COCs	Effectiveness: Advantages	Effectiveness: Disadvantages	Implementability: Site Conditions	Implementability: Advantages	Implementability: Disadvantages	Cost[1]
Enhanced Natural Recovery	Enhanced Physical Burial	Thin-layer placement to augment natural sedimentation	Effective for all LDW COCs where MNR processes are demonstrated.	ENR dilutes COC concentrations while not resulting in the resuspension and transport of contaminants that occurs with dredging.	1) Requires implementation of long-term monitoring study and risk assessment objectives; 2) Short-term impacts to human and ecological health may continue, and require use in conjunction with seafood consumption advisories and/or other site restrictions; 3) COCs not actively removed, but attenuated by addition of clean sediments.	Applies where data and modeling indicate placement of a thin-layer of material, combined with natural recovery processes will result in achievement of risk-based sediment objectives. Particularly useful for critical habitat areas, and/or shallow intertidal areas where active remedial methods could result in unwanted habitat loss. Potentially suitable for management of dredge residuals.	1) Puget Sound-demonstrated technology with local construction knowledge; 2) Sediment for thin-layer placement readily available.	1) Requires long-term monitoring, institutional controls and continuing financial commitment until cleanup objectives are achieved; 2) Institutional controls may limit future uses of waterway.	Low
Containment	Capping	Conventional Sand Cap	Applicable principally to PAHs, other SVOCs, metals, and PCBs; Limited applicability to VOCs.	1) Demonstrated effectiveness for isolating contaminants in the LDW; 2) Isolates contaminants from the overlying water column and prevents direct contact between aquatic biota and contaminants; 3) Capping does not result in the resuspension and transport of contaminants that occurs with dredging.	1) Sand cap may be subject to bioturbation and release of buried COCs; 2) Sand caps may be susceptible to propeller and/or high-flow scour, methane generation, and earthquakes; 3) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 4) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable to subtidal areas where sediments have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COCs into the clean cap surface.	1) Readily applied and demonstrated technology. Local construction experience; 2) Capping materials readily available from navigation dredging at the Upper Turning Basin.	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway; 4) Impacts to flooding, stream bank erosion, navigation, and recreation must be addressed in design.	Low to Moderate
		Conventional Sediment/Clay Cap	Applicable principally to COCs with potentially higher solubilities and lower sorption.	1) Sediment with high fines (silt and clay) and or TOC is effective in limiting diffusion of contaminants. Sediment caps are generally more effective than sand caps for containment of contaminants with high solubility and low sorption; 2) Natural TOC present in conventional sediments more effective at adsorbing COCs such as PCBs.	1) Clay liners in caps are potentially more susceptible to breaches caused by methane generation through the cap; 2) Caps may be susceptible to propeller and/or high-flow scour, methane generation, and earthquakes; 3) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 4) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable in sections of LDW with low erosion potential and where placement of finer-grained material can be managed. May be useful in nearshore, or intertidal applications where thinner caps with higher sorbtive capacities are required. Sediments must still have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COCs into the clean cap surface.	1) Readily applied and demonstrated technology; 2) Placement of high TOC and/or high fine sediments minimizes thickness of cap in areas with shallow water depth; 3) Materials readily available through upland sources or from navigation dredging at other systems.	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway; 4) Impacts to flooding, stream bank erosion, navigation, and recreation must be addressed in design; 5) Utilization of navigation dredged material for capping has potential logistical issues.	Low to Moderate
		Armored Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	Effective in combination with conventional caps to isolate contaminants and protect cap against physical erosion and/or bioturbation.	1) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 2) Armor rock may be less productive habitat for benthic organisms. 3) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable in conjunction with other cap configurations in areas of LDW, but can be applied where erosion potentials are higher.	(1) Readily applied and demonstrated technology; 2) Armor placement can be used to minimize thickness of cap in areas with shallow water depth; 3) Armor materials can be combined with habitat-enhancing materials (e.g., "Fish Mix").	1) Requires long-term maintenance and financial commitment; 2) May not be implementable for shallow, intertidal areas where elevation changes would result in unacceptable ecological impacts; 3) May require permanent institutional controls and limit future uses of waterway.	Low to Moderate
		Composite Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	1) Provides physical isolation of COCs from the overlying water column; 2) Assists in preventing bioturbation breaches of caps and prevents direct contact between aquatic biota and contaminants; 3) Rigid HDPE layers used in small areas to assist in NAPL containment, control hydraulic gradient, and methane containment and diffusion.	1) Composite caps at other sites have resulted in catastrophic breaches as a result of methane generation under the cap; 2) Rigid HDPE layers do not have long-term demonstrated effectiveness; 3) Use of geotextiles may not be necessary for contaminants with low solubility and high sorption where the main concern is resuspension and direct contact; 4) Geotextiles by themselves do not limit advective or diffusive flux of COCs; 5) Requires long-term monitoring, institutional controls, and financial commitment.	Composite caps with impermeable layers such as HDPE are generally applicable where control of NAPL or groundwater movement is needed in a limited area. Composite caps may also be potentially applicable in intertidal areas where physical separation between receptors and COCs are required, but where minimal change to the slope or bathymetric configuration is needed.	1) Increasingly applied technology; 2) Placement of geotextile or rigid HDPE can be used to minimize thickness of cap in areas with shallow water depth.	1) Requires specialty equipment for placement, sinking, and securing to the sediment floor; 2) Tidal ranges in the LDW can affect ability to place materials; 3) Requires long-term monitoring and financial commitment.	Low to Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness: COCs	Effectiveness: Advantages	Effectiveness: Disadvantages	Implementability: Site Conditions	Implementability: Advantages	Implementability: Disadvantages	Cost[1]
Containment	Capping	Spray Cap	Applicable to all LDW COCs as described for sand and/or conventional caps.	Good for application under hard to access areas such as piers and wharves. Provides good physical barrier between contaminants and overlying surfaces.	1) Creates a hard surface. If habitat surface values are required, habitat-suitable material would need to be placed on top of the shotcrete. 2) Must be applied in the dry with time to set. Areas of application are limited to high intertidal areas. 3) Requires long-term monitoring and maintenance, institutional controls, and a potential requirement for replacement habitat.	Labor intensive process to implement in difficult working conditions under docks and piers.	Good for application under hard to access areas such as piers and wharves.	1) Potentially dangerous work because of obstructions, slippage, and presence of contaminants next to workers applying the shotcrete. 2) Requires specialty equipment to place the shotcrete. 3) Tidal ranges can affect placement location. 4) Not applicable in habitat areas.	High
Containment	Capping	Reactive Caps	Potentially applicable to all LDW COCs as described for conventional sand and/or conventional sediment caps.	Similar to advantages described for other caps. Provides an additional level of contaminant-sorbing materials to caps.	Long-term effectiveness not demonstrated. Retained as innovative technology. Requires long-term monitoring, institutional controls, and financial commitment. Probably not acceptable in beach areas.	Applicable in conjunction with other cap configurations in areas of LDW.	Adds an additional level of environmental protection with contaminant sorbing materials. May allow for construction of thinner caps.	1) Requires specialty equipment for placement, sinking, and securing to the sediment floor; 2) Tidal ranges in the LDW can affect ability to place materials; 3) Requires long-term monitoring and financial commitment; 4) Long-term implementability not demonstrated. Retained as innovative technology.	Low to Moderate
Removal	Dredging	Hydraulic Dredging	Applicable to all LDW COCs at higher concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	1) Effective removal with lower resuspension and recontamination/residual rate relative to mechanical dredging; 2) Can be readily incorporated into treatment trains such as chemical and/or physical separation.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of low solid sediments, generally less than 20 ft. of water depth and low levels of debris.	(1) Various hydraulic dredges readily available on the West Coast and at least one dredging contractor has equipment on the LDW; (2) More effective lateral and vertical cut control may be achieved, relative to mechanical dredges; (3) High utility when used in conjunction with CDFs; (4) Local experience of use for the Sitcum and Blair Waterway projects.	1) Hydraulic dredges limited in heavy-debris environments; 2) Environmental hydraulic dredges are depth limited, and difficult to size to accommodate steady solids flow under varying tidal regimes; 3) Requires separation of solids from water, resulting in large volumes of water that may require treatment prior to discharge back to LDW; 4) Treatment facilities must be located near-waterway with enough land space to accommodate retention basins, mechanical dewatering equipment, sand and carbon filtration, and transfer of dewatered material to trucks or trains for transfer to regional landfill.	Moderate to High
Removal	Dredging	Mechanical Dredging	Applicable to all LDW COCs at concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	Effective for removal in areas with high debris and sediments with high sand or heavy clay content that require digging buckets.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of high percentage solids sediments, including areas with heavy debris, sand, and clay. Mechanical dredging is not depth restricted, and not affected by tidal exchange.	1) Various mechanical dredges, including environmental buckets and clamshells readily available on the LDW and in Puget Sound; 2) Recent construction experience in LDW and Puget Sound with skilled operators; 3) Environmental buckets useful in softer, unconsolidated materials with low debris; 4) Digging buckets (e.g., clamshells) useful in harder clays or compacted sediments, or where debris is high; 5) Existing infrastructure for barge transport, off-loading, and transfer to railcars for transport to regional landfills; 6) Depth and tidal limitations within the LDW do not restrict use of mechanical buckets.	1) Not all river segments may be accessible to a barge-operated mechanical dredge; 2) Can result in potentially higher resuspension and residual rates than hydraulic dredges; 3) Lower vertical and horizontal operational control relative to hydraulic dredges.	Low to Moderate
Removal	Dredging	Mechanical Dredging (Excavator)	Applicable to all LDW COCs at concentrations that either pose unacceptable risks to human health and the environment, and/or serve as sources for downstream recontamination.	Effective for removal in areas with high debris and sediments with high sand or heavy clay content that require digging buckets.	Requires management of contaminant residuals after dredging.	Applicable in areas with high volumes of high percentage solids sediments, including areas with heavy debris, sand, and clay. Mechanical dredging is not depth restricted, and not affected by tidal exchange.	1) Equipment is available to the Puget Sound region but to lesser extent than standard clamshell dredges; 2) Recent construction experience in LDW and Puget Sound with skilled operators; 3) Offer high level of vertical and horizontal control during dredging.	1) Not all river segments may be accessible to a barge-operated mechanical dredge; 2) Can result in potentially higher resuspension and residual rates than hydraulic dredges; 3) Lower vertical and horizontal operational control relative to hydraulic dredges.	Low to Moderate
Removal	Excavating	Dry Excavating	Applicable to all LDW COCs. Effective for nearshore and/or intertidal areas where depths limit conventional dredging equipment	1) Contaminated sediments removed; 2) Residuals can be minimized or eliminated by dry excavation.	Effective only in relatively small and narrow shoreline areas of limited intertidal bands. Requires either only working during low tides, or using cofferdams or sheet pile walls to create a contained, dry area.	Limited in application to nearshore shallow and/or intertidal areas that can be reached from shore or by specialty equipment designed to work on soft, unconsolidated sediments.	Equipment and construction experience in Puget Sound.	1) Construction costs may involve contingencies to address potential spills and leaks	Low to Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness			Implementability			Cost[1]
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	
In Situ Treatment	Physical-Immobilization	Activated Carbon Amendment	Applicable to certain LDW COCs at concentrations that pose unacceptable risks to human health and the environment	1) Contaminants adsorb to activated carbon particles; 2) porewater concentrations (sediment-to-water fluxes), contaminant concentrations, and bioavailability for uptake by organisms are reduced; and 3) promising pilot-scale results.	May require armoring in areas susceptible to propeller and/or high-flow scour. Requires long-term monitoring, institutional controls, and financial commitment. Retained as innovative technology. Long-term effectiveness not demonstrated at full scale.	Easily implementable, and applicable to most areas of the LDW. Sand could be mixed with the activated carbon as a form of modified ENR.	1) Recently demonstrated implementable technology; 2) activated carbon for placement readily available, and 3) commercial products have been developed to improve the deployment of the activated carbon, by using a weighting particle (sand, gravel, etc.) coated with an inert binder and activated carbon.	1) Can require specialized equipment depending on application method; 2) requires long-term monitoring.	Low to Moderate
		Organoclay Amendment	Applicable to certain LDW COCs at concentrations that pose unacceptable risks to human health and the environment	1) Chemically binds metal ions, replacing them with amines or other functional groups; 2) physically isolates the contaminated sediment from receptors (because of low permeability of clay); 3) stabilizes sediment preventing resuspension and transport of contaminants, and 4) promising pilot-scale results.	May require armoring in areas susceptible to propeller and/or high-flow scour. Requires long-term monitoring, institutional controls, and financial commitment. Retained as innovative technology. Long-term effectiveness not demonstrated at full scale	Easily implementable, and applicable to most areas of the LDW.	1) Recently demonstrated implementable technology; 2) organoclays for placement commercially available.	1) Can require specialized equipment depending on application method; 2) requires long-term monitoring.	Low to Moderate
Ex Situ Treatment	Chemical/Physical	Soil Washing	Applicable to all LDW COCs. Principal application would be for high volumes of organic-contaminated sediments.	1) Full-scale testing demonstrated ability to take high concentrations of COCs and treat to equivalent of MTCA soil standards; 2) Potential beneficial reuse for residuals.	1) Tests to date have treated hazardous waste-level materials. No data on treatment of lower concentrations of contaminants; 2) Effective treatment when starting with high sands materials—lower effectiveness when treating low solids and high fine-grained sediments; 3) Solid-waste classification in Washington state unclear, which may require disposal of treated materials at a Subtitle D landfill.	Applicable to potential dredge areas containing organic and coarse-grained sediment.	1) Readily implementable, resulting in reduced contaminated sediment volume; 2) System could be coupled with hydraulic dredging for continuous treatment train; 3) Mobile units are available 4) Continuous flow process designed to process up to 40 cy of sediments per hour for the full-scale system; 5) May be available for potential beneficial reuse.	1) Waste streams include hydraulic-dredge decant water, reagents used in soil washing, and the treated residuals; 2) Water will require filtration and treatment prior to discharge; 3) Treated residuals may require off-site disposal; 4) Volume/long-term supply of sediments to be treated and local market for beneficial use products affect the economics of implementing this technology.	Moderate
	Physical	Particle Separation	Only applicable to adsorptive COCs that would adhere to the fine-grained soil. Offers greatest utility and cost saving benefits where concentrations of COCs would otherwise require incineration or Subtitle C disposal.	1) Demonstrated effectiveness for reduction in volume of highly contaminated sediments with a high percentage of sand-content; 2) Used to increase effectiveness of dewatering dredged material.	1) Not effective for contaminants with high concentrations and high organic content; 2) Previous work at other sites with PCB-contaminated sediments has shown that PCBs are retained on sand particles (as emulsion), requiring Subtitle D disposal.	Applicable to potential dredge areas containing higher sand content.	1) Readily implementable, resulting in reduced contaminated sediment volume; 2) Can be combined with soil washing to improve contaminant separation and/or destruction; 3) Mobile units are available; 4) Separated sand may be available for potential beneficial reuse, capping, or disposal at DMMP Elliott Bay site.	Will require disposal of separated waste stream at a Subtitle D landfill. Fines could also require Subtitle C disposal or incineration.	Moderate
		Solidification	Applicable to all LDW COCs. Principal application would be for high volumes of PCB-contaminated sediments that exceed hazardous waste criteria and would otherwise require incineration or Subtitle C disposal.	1) Lime has been successfully added to dredged material at other projects; 2) Effective during the dewatering operation to remove excess water and prepare material for disposal.	High contaminant concentration and high water content results in higher project costs.	Applicable to all dredge areas of LDW.	1) Readily implementable; 2) Reagent materials readily available.	1) Immobilizing reagents, ranging from Portland cement to lime cement, kiln dust, pozzolan, and proprietary agents, have been applied with varying success. Dependent on sediment characteristics and water content; 2) Contaminants remain in place. Stabilized product requires disposal in regulated landfill.	Moderate

Table 7-3 Summary Assessment of Effectiveness, Implementability, and Cost for Retained Remedial Technologies and Process Options (continued)

GRA	Technology Type	Process Option	Effectiveness			Implementability			
			COCs	Advantages	Disadvantages	Site Conditions	Advantages	Disadvantages	Cost[1]
Disposal	On-Site	Contained Aquatic Disposal (CAD)	Applicable to all LDW COCs below hazardous waste designations.	1) Demonstrated local experience and effectiveness in the LDW and Puget Sound; 2) Effective containment of metals, organics, and PCBs; 3) Can be designed to include habitat enhancement for salmonids.	1) CADs must be engineered to withstand bioturbation, advective flux, and release of buried COPCs, propeller and/or high-flow scour, and earthquakes; 2) Changes in bed elevation may result in unacceptable ecological impacts to salmonid habitat; 3) Requires long-term monitoring, institutional controls, and financial commitment.	Applicable to subtidal areas where sediments have sufficient bearing strength to support cap, and have low erosive potential. Not suitable for areas where groundwater can advect COPCs into the clean cap surface.	1) Technically readily implemented within the LDW with contaminated sediments contained on-site; 2) Local construction experience; 3) Excavated clean pit-materials can be used for beneficial uses and/or to cap CAD; (4) Does not interfere with current industrial uses of LDW.	1) Volume-limited on LDW as a large area would be required to accommodate dredged sediments; 2) Requires long-term commitment to monitoring with the potential for additional actions if CAD fails; 3) Requires permanent institutional controls (e.g., deed restrictions, dredging moratorium) that may affect future development and uses of the LDW; 4) Requires concurrence with land owner.	Low to Moderate
		Confined Disposal Facility (CDF)	Applicable to all LDW COCs below hazardous waste designations.	1) Demonstrated local experience and effectiveness in Puget Sound; 2) Effective containment of metals, SVOCs and PCBs; 3) A subtidal CDF could be designed to include habitat enhancement for salmonids.	1) CDFs must be engineered to withstand advective flux and release of buried COCs, propeller, and/or high-flow scour, and earthquakes; 2) Filling of nearshore lands would result in unavoidable loss of aquatic lands that will require mitigation.	Requires large suitable near-shore or upland containment site. Former slips or similar in-water areas would be best suited to construct a CDF.	1) Puget Sound-demonstrated technology with local construction knowledge; 2) Cap sediments or soils readily available; 3) Could contain large volumes of contaminated sediments, depending upon site availability; 4) Beneficial upland industrial and/or residential reuse of filled site.	1) Site-limited on LDW. Few potential locations without other current uses; 2) Requires long-term commitment to monitoring with the potential for additional actions if CDF fails; 3) Requires permanent institutional controls (e.g., deed restrictions, dredging moratorium) that may affect future development and uses of the LDW; 4) Requires concurrence with land owner.	Moderate to High
	Off-Site	Subtitle D Landfill	Applicable to all LDW COCs below hazardous waste designations.	Subtitle D landfills highly effective for long term, permanent containment of contaminated materials.	COCs contained, but not permanently destroyed.	Applicable throughout LDW for both dewatered and wet sediments.	1) Several licensed landfills in Washington exist that can receive dredged materials in Puget Sound; 2) Transfer facilities for moving sediments from LDW to the landfills exist on-site; 3) Transport infrastructure in-place on the LDW; 4) Options exist for moving wet sediments - eliminating need for on-site dewatering facilities.	1) Transfer and barge offload facilities may not be present at the time the project is completed so a separate offload facility may need to be constructed; 2) Landfills in Eastern Washington and Eastern Oregon requires train transport with potential for spillage.	Moderate
	Off-Site	Subtitle C Landfill	Applicable to all LDW COCs exceeding hazardous waste designations.	Subtitle C landfills are federally-regulated facilities and are highly effective for long-term, permanent containment of highly contaminated materials.	1) COCs contained, but not permanently destroyed; 2) Requires dewatering of dredged sediments.	Applicable throughout LDW for dewatered sediments	Option for disposal of listed, hazardous wastes	Transport of hazardous materials to facility expensive.	High
		DMMP Open Water Disposal	Applicable to all LDW COCs in sediments that are separated or treated to below the DMMP disposal standards.	DMMP is a well-established and effective program with a long-term track record of monitoring to verify environmental protectiveness.	None	Applicable throughout LDW	The DMMP disposal site is located in nearby Elliott Bay	Sediments that require remediation are not likely to meet the open water disposal criteria.	Low
		MTCA Reuse (upland or in-water beneficial reuse)	Applicable to all LDW COCs in sediments that are either below, or treated-to below the reuse standards for uplands and in-water.	Beneficial reuse of sediments	Some residual COCs may remain after treatment	Applicable throughout LDW	Potential use of sediments that meet the MTCA Level A soil requirements as upland fill, or other beneficial upland uses including daily landfill cover. Potential beneficial reuse as in-water ENR, capping material, and habitat enhancement. May be implementable for high volumes of materials with low concentrations of COCs, or for treated sediments.	No specific beneficial upland reuse has been identified. As such, requires the additional costs for transport of material and/or tipping fee to send to landfill.	None

Notes:

1. Cost assessment is based on the relative cost of a process option in comparison to other process options within a given technology type.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; CTM = Candidate Technologies Memorandum; cy = cubic yards; DMMP = Dredged Material Management Program; ENR = enhanced natural recovery; FS = Feasibility Study; GRA = general response action; HDPE = High-density polyethylene; HTTD = high-temperature thermal desorption; LTTD = low-temperature thermal desorption; MNR = Monitored Natural Recovery; MTCA = Model Toxic Control Act; NCP = National Contingency Plan; PCB = polychlorinated biphenyls; SVOC = semivolatile organic compound; TBT = tributyltin; TOC = total organic carbon; TSCA = Toxic Substances Control Act; TSS = total suspended solids; UECA = Uniform Environmental Covenants Act

Table 7-4 Remedial Technologies and Process Options Retained for Potential Use in Developing Remedial Alternatives

General Response Action	Technology Type	Process Option	Comments Related to Technology Assumptions for the FS
No Action	None	Not Applicable	Per NCP requirements
Institutional Controls	Proprietary controls and Informational Devices (EPA 2000)	Proprietary Controls	Access to much of the LDW shoreline from the uplands is already restricted by general security measures put in place by private and public property owners. The LDW is a public waterway and public access to nearshore areas is generally not prohibited.
		Seafood Consumption Advisories, Education, and Public Outreach	Public advisories regarding fish and shellfish consumption are currently posted for the entire LDW. Public advisories regarding sediment contact risks are not currently posted. Advisories are a likely element of all remedial alternatives and will remain in place until monitoring data confirms that the advisories can be modified or removed entirely.
		Monitoring and Notification of Waterway Users	As needed, these will be tailored to specific remediation activities and site constraints.
		Enforcement Tools	CERCLA or MTCA consent decrees for settling potentially responsible or liable parties, or unilateral orders for non-settling parties, issued by EPA or Ecology are anticipated.
		Site Registry	Provides information on applicable restrictions associated with Restricted Navigation Areas and other proprietary controls.
Monitoring	None	Baseline Monitoring	Establishes a statistical basis for comparing conditions before and after the cleanup action.
		Construction Monitoring	Short-term monitoring during remediation used to evaluate whether the project is being implemented in accordance with specifications (i.e., water quality monitoring, bathymetric surveys)
		Post-Construction Performance Monitoring	Post-construction performance monitoring evaluates post-removal surface and subsurface sediment conditions in dredging or containment areas to confirm compliance with project specifications.
		Operation and Maintenance Monitoring	Operation and maintenance monitoring of dredging areas, containment, and/or disposal sites (i.e., CAD sites, ENR and capping areas) required to ensure long-term effectiveness and continued stability of the structure.
		Long-term Monitoring	Long-term monitoring evaluates sediment, tissue, and water quality at the site for an extended period following the remedial action.
Monitored Natural Recovery (MNR)	Natural Physical, Biological, and Chemical Recovery	Multiple potential mechanisms: burial (sedimentation), immobilization, desorption, dispersion, diffusion, dilution, volatilization, resuspension, biological degradation.	Surface sediment chemistry is monitored over time to track recovery by multiple physical, chemical, and biological mechanisms that operate naturally in the estuarine environment of the LDW. Burial by the comparatively cleaner sediments coming into the LDW from the Green River is the principal mechanism for recovery in the LDW. Natural recovery is operative in the waterway as supported by analysis of the empirical data and predicted by the STM. Areas potentially suitable for MNR must be depositional, not subject to significant physical disturbances from high river flows, vessel propeller scour, anchor drag, and routine dredging. Future construction activity in MNR zones is not precluded; however, the applicant/owner must be prepared to appropriately handle any contaminants that may be encountered as part of the project.
Enhanced Natural Recovery (ENR)	Thin-layer Placement	Placement of a thin layer of granular media (e.g., sand) to augment natural recovery	ENR differs from MNR with respect to the modification of initial conditions (i.e., placing clean material onto the contaminated sediment surface). In other respects, siting, monitoring, and future use restrictions and considerations are the same. Placement also can serve as a means of managing contaminated dredging sediment residuals, called thin-layer sand placement. The composition of ENR/*in situ* treatment may include carbon amendments and/or habitat mix.
Containment	Capping	Conventional Sand Cap	Conventional capping is restricted to net deposition areas that are not subject to appreciable sustained or episodic erosion. Cap thickness must be sufficient to prevent reintroduction of buried contaminants into biologically active zone (upper 10 cm).
		Conventional Sediment/Clay Cap	Cap thickness must be sufficient to prevent reintroduction of buried contaminants into biologically active zone (upper 10 cm).
		Armored Cap	If capping is considered in erosion areas, armoring will likely be required to maintain the cap integrity.
		Spray Cap (Technology not addressed by CTM)	Shotcreting is potential approach for confining, isolating contaminants under dock or overwater structures. The shotcrete application at Todd Shipyards effectively encapsulated existing debris (slag) mounds from the aquatic environment.
		Composite Cap	Application would be location- and contaminant-specific where space or pollutant constraints indicate conventional sand capping is not adequate.
		Reactive Cap	Application would be location- and contaminant-specific where space or pollutant constraints indicate conventional sand capping is not adequate.
Removal	Dredging	Hydraulic Dredging (including diver-assisted dredging)	Hydraulic dredging has several constraints that limit its project-wide application: the cost and logistics of managing large volumes of water including large land area adjacent the dredging area; potential for water quality impacts; debris leads to operational difficulties and dredging inaccuracies; interruption of waterway use caused by placement of the hydraulic discharge pipeline in the LDW. Application of hydraulic dredging in the LDW may be appropriate on a small scale (e.g., diver-assisted dredging in under dock/pier areas) or on location-specific basis.
		Mechanical Dredging	Demonstrated effective in the Puget Sound region and nationwide sediment remediation projects. Readily available and least-cost dredging option in the Puget Sound region.
		Mechanical Dredging (Excavator)	Excavator dredges offer a high level of control in the placement of the dredge bucket because it uses fixed linkages instead of cables. This yields a higher degree of accuracy resulting in less volume of dredged sediment and reduced water quality impacts as compared to a conventional derrick barge. Often used for debris removal and/or shallow in-water dredging operations.
	Excavating	Dry Excavating	Generally applicable to nearshore areas above elevation -2.0 ft MLLW or 25-ft reach from top of bank.

Table 7-4 Remedial Technologies and Process Options Retained for Potential Use in Developing Remedial Alternatives (continued)

General Response Action	Technology Type	Process Option	Comments Related to Technology Assumptions for the FS
In Situ Treatment	Physical/Immobilization	Activated Carbon Amendment	Demonstrated effective in nationwide sediment remediation projects at pilot-scale level. Readily available and low-cost *in situ* treatment technology. Sand could be mixed with the activated carbon as a form of modified ENR (see above).
		Organoclay Amendment	Demonstrated effective in nationwide sediment remediation projects at pilot-scale level. Readily available and low-cost *in situ* treatment technology. May require armoring in LDW areas susceptible to propeller and/or high-flow scour.
Ex Situ Treatment	Chemical/ Physical	Soil Washing	Mechanically dredged sediment is screened to remove oversize debris and is then processed through a series of unit operations resulting in the following products or waste streams: wastewater, sludge (fines fractions), and sand/gravel. Wastewater requires treatment, the sludge is typically disposed (upland landfill), and the sand/gravel component may be reused for in-water applications if it tests suitable for beneficial use pursuant to the Washington State Sediment Management Standards (i.e., less than SQS criteria). Soil washing/particle separation is potentially effective and implementable in the LDW where the percentage of sand in the sediment exceeds ~ 30% by weight. It is anticipated that most of the COCs will concentrate on the remaining sludge (fines fraction), which will then need disposal. This concentrating process, if too great, could cause the sludge to be designated as hazardous waste.
	Physical	Separation	Presented as unit costs in FS.
		Solidification	If future designs require further water reduction methods and to remove free water prior to landfilling.
Disposal	On-site Disposal	Contained Aquatic Disposal (CAD)	The overall space (volume) capacity for CAD is limited. However, adequate capacity may be available to contain substantial portion of the contaminated dredged sediment for those alternatives requiring the least amount of dredging. However, for most alternatives, CAD will not be adequate for project-wide application, but could serve to contain a portion of the contaminated sediment. Substantial implementability logistics issues need to be addressed with CAD. Also, constraints with long-term institutional controls (e.g., conflict if located within established dredging areas) and multiple agency approvals to authorize the site are a concern.
		Confined Disposal Facility (CDF)	Not applicable to LDW site-wide application because of limited locations (and capacity) without other current uses. May be applicable for smaller-scale location-specific application.
	Off-site Disposal	Subtitle D Landfill	Applies specifically to sediment that is characterized as non-hazardous in accordance with federal or state regulations. Regional landfills that can accept nonhazardous sediment are Allied Waste Inc. (Roosevelt, Washington) and Waste Management (Arlington, Oregon).
		Subtitle C Landfill	Applies specifically to sediment that is characterized as hazardous or dangerous in accordance with federal or state regulations. This condition is not expected to occur on a large scale and more likely will be limited to localized hot spot removal areas, if triggered at all.
		Dredged Material Management Program (DMMP) Open water Disposal	This is a potentially viable disposal option where the average concentration of COCs in the entire dredged material management unit is determined to be less than the DMMP disposal requirements.
		Beneficial Use (In-Water and Upland)	Sediment that tests suitable for beneficial use pursuant to the Washington State Sediment Management Standards (i.e., less than SQS criteria) may be beneficially reused for habitat creation, capping, or residual management. In case of treatment (e.g., soil washing), the sediment may qualify for beneficial reuse.

Notes:

Representative site-wide process options included in the development of the remedial alternatives and cost estimates for this FS. Other process options may have location-specific applicability; but not site-wide applicability.

1. These technologies and process options were screened and retained in Tables 7-2a through 7-2e, and summarized in Table 7-3 with the exception of institutional controls, which do not lend themselves to comparison on the same terms as other technologies. Institutional controls are discussed only within Section 7.2 and are not included in Table 7-3.

CAD = contained aquatic disposal; CDF = confined disposal facility; COC = contaminant of concern; CTM = Candidate Technologies Memo; DMMP = Dredged Material Management Program; Ecology = Washington State Department of Ecology; ENR = enhanced natural recovery; EPA = U.S. Environmental Protection Agency; MNR = monitored natural recovery; MTCA = Model Toxics Control Act; MLLW = mean lower low water; NCP = National Contingency Plan; SQS = Sediment Quality Standards; STM = Sediment Transport Model

Table 7-5 Sediment Dredging and Handling Methods Used on Representative Projects in the Puget Sound Region

Sediment Remediation Project	Completed	Dredge Method	Disposal Method	Predicted Volume of Dredged Sediment (cubic yards)	Actual Volume of Dredged Sediment (cubic yards)
Wyckoff/Eagle Harbor West Operable Unit	1997	Mechanical	CDF	1,300 to 9,200	6,000
Norfolk Sediment Remediation	1999	Mechanical	Subtitle D landfill and Subtitle C landfill	4,050	5,190
Cascade Pole Site	2001	Mechanical	CDF	n/a	40,000
Puget Sound Naval Shipyard	2001	Mechanical	CAD	300,000	n/a
Weyerhaeuser	2002	Mechanical	Landfill	n/a	n/a
Hylebos Waterway – Area 5106	2003	Hydraulic	CDF	20,000	n/a
East Waterway	2004	Mechanical	Subtitle D landfill	n/a	n/a
Lockheed Shipyard	2004	Mechanical	Subtitle D landfill	46,625	70,000
Todd Shipyard	2004	Mechanical	Subtitle D landfill	116,415	220,000
Duwamish/Diagonal	2004	Mechanical	Subtitle D landfill	42,500	66,000
Middle Waterway	2004	Mechanical	CDF	75,000	109,000
Hylebos Waterway – Segments 3-5	2004	Mechanical	CDF	n/a	>100,000
Pacific Sound Resources	2004	Mechanical	Subtitle D landfill	3,500	10,000
Head of Hylebos Waterway	2005	Mechanical	Subtitle D landfill	217,000	419,000
Thea Foss – Wheeler Osgood Waterways	2005	Hydraulic/Mechanical	CDF	620,000[a]	422,535
Denny Way	2007	Mechanical	Subtitle D landfill	13,730	14,400

Notes:

a. Volume from combined projects from Commencement Bay Nearshore/Tideflats Explanation of Significant Differences (EPA 2000e)

CAD = contained aquatic disposal; CDF = confined disposal facility; n/a = not available

Figure 7-1 Pilot-scale Demonstrations of Activated Carbon Amendment Delivery into Sediment

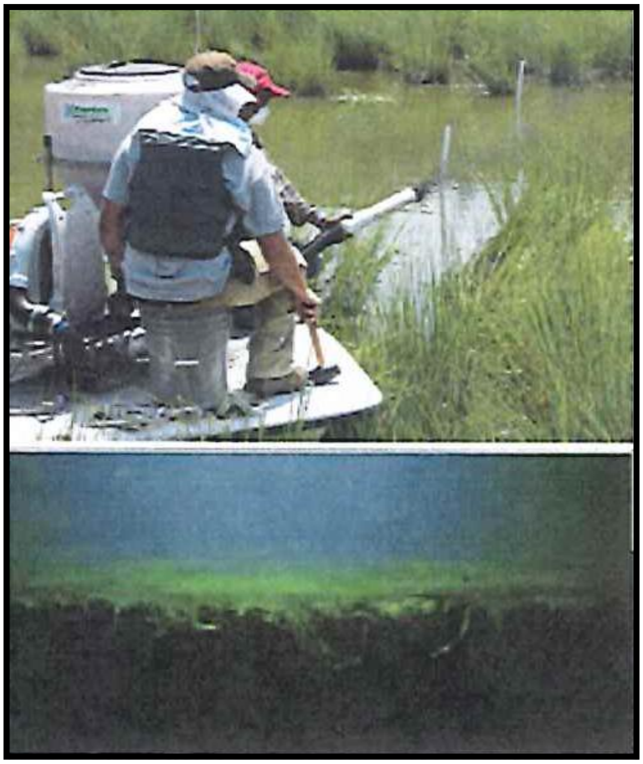
A) *Application of activated carbon in a tidal mudflat at Hunter's Point Naval Shipyard, San Francisco Bay, CA using two application devices (2004 and 2006). The Aquamog (top) using a floating platform approached the site from water and used a rototiller arm while the slurry injection system (bottom) was land based and applied a carbon slurry directly into sediment.*

Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech.*, 45, 1163–1168, 2011.

B) *Application of activated carbon under 15 feet of water at Lower Grasse River, NY (2006). The site was enclosed with a silt curtain and application was performed using a barge mounted crane. Placement and mixing of the activated carbon was achieved using two devices: 1) a 7-by 12-foot rototiller-type mixing unit (top); and 2) a 7-by-10-foot tine sled device (bottom).*

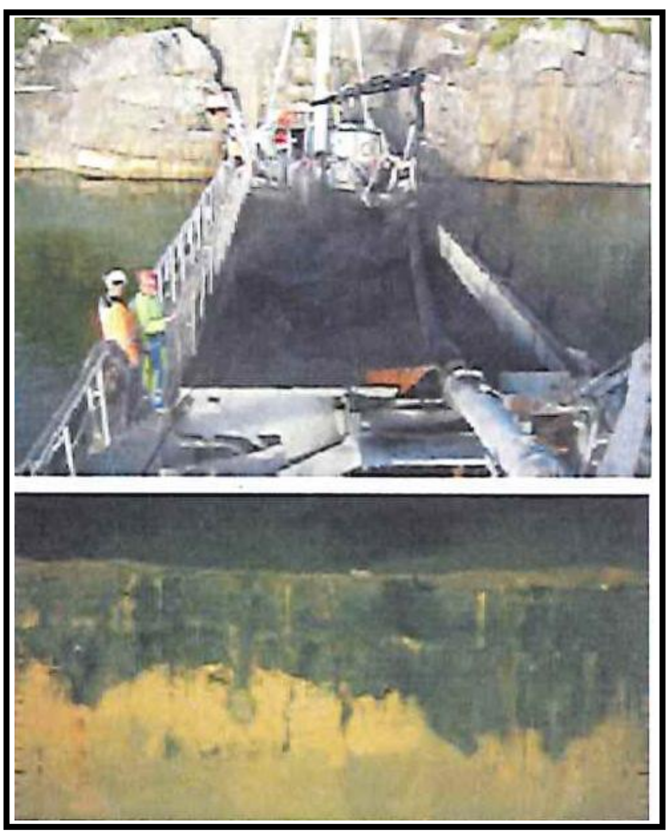
Source: 2006 Activated Carbon Pilot Study Project (thegrasseriver.com).

Figure 7-1 Pilot-scale Demonstrations of Activated Carbon Amendment Delivery into Sediment (continued)



D) *Application of activated carbon in a pelletized form (SediMite™) using an air blown dispersal device (top) over a vegetated wetland impacted with PCBs near the James River, VA (2009). Picture below illustrates bioturbation induced breakdown and mixing of pelletized carbon with a fluorescent tag in a laboratory aquarium (bottom).*

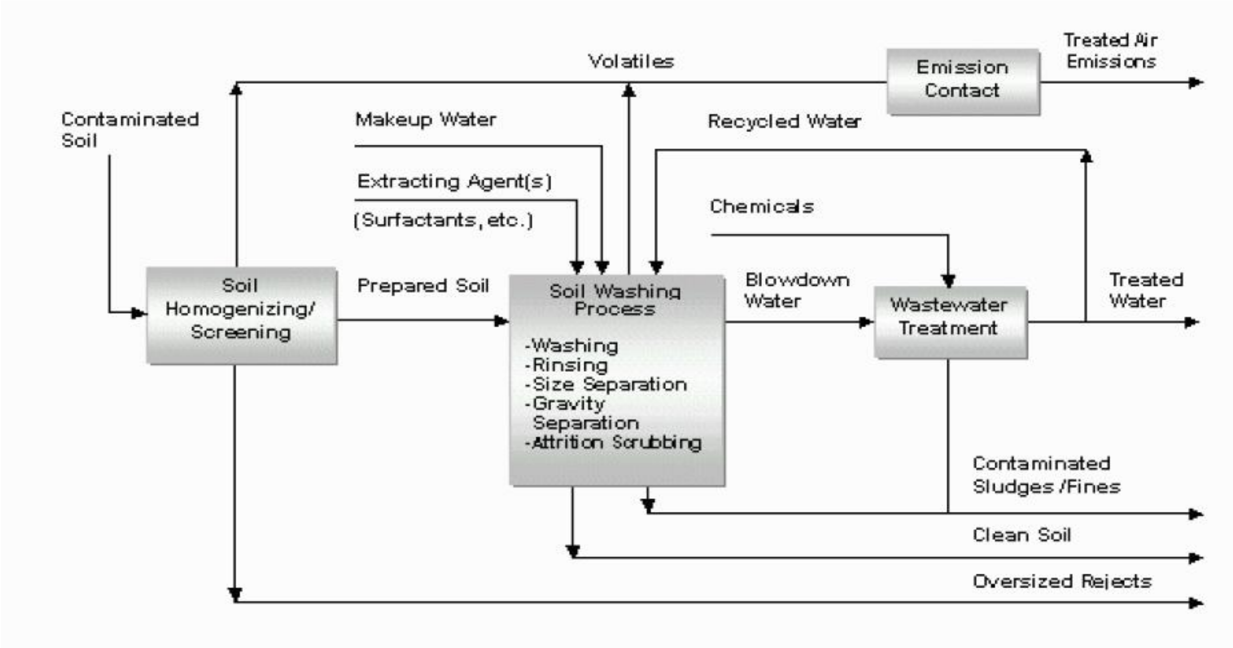
Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech*., 45, 1163–1168, 2011.



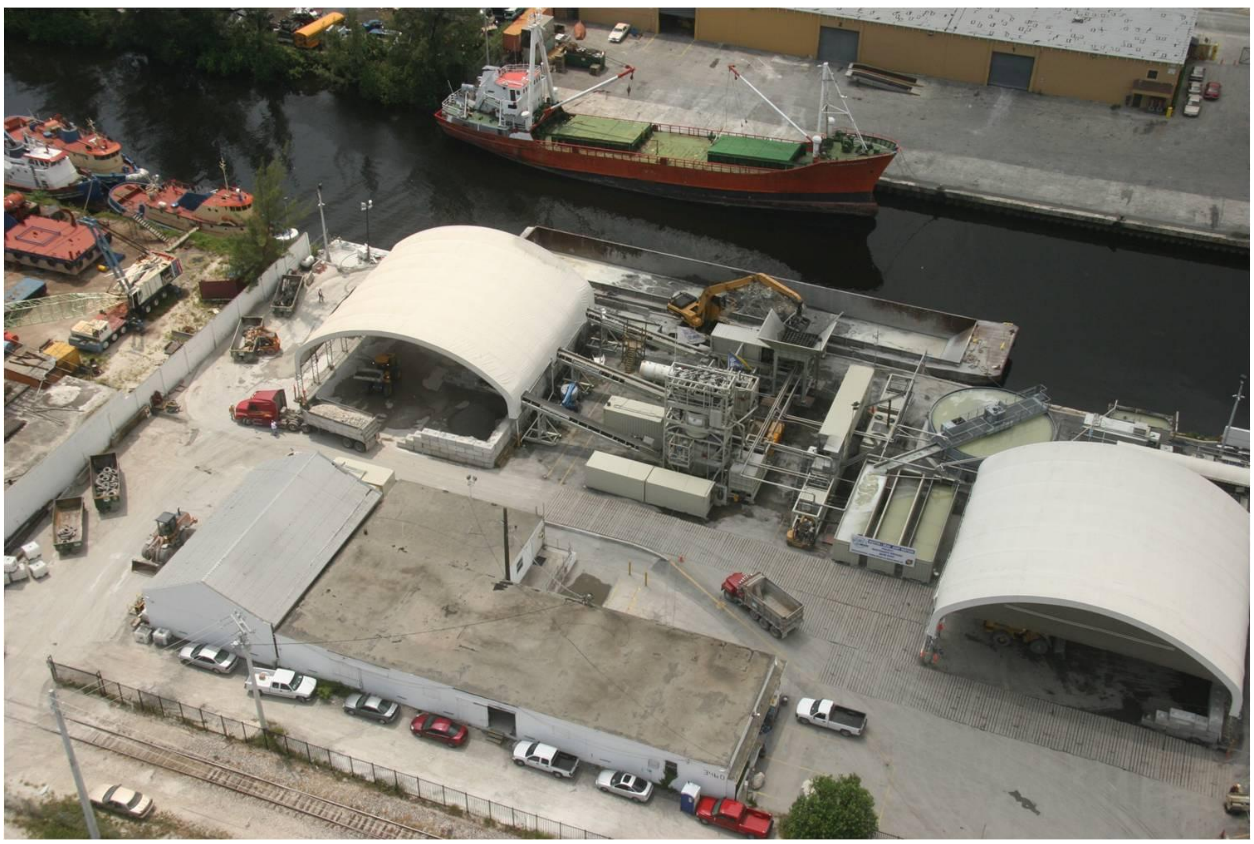
E) *Application of activated-carbon-clay mixture at 100- and 300-ft depth, Grenlandsfjords, Norway (2009), led by NGI and NIVA. A hopper dredger was used to pick up clean clay from an adjacent site. After activated-carbon-clay mixing, the trim pipe was deployed in reverse to place an activated-carbon-clay mixture on the sea floor. Sediment-profile imaging and sediment coring (bottom figure) showed that placement of an even active cap was successful.*

Source: Ghosh et al., "In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management", *Env.Sci&Tech*., 45, 1163–1168, 2011.

Figure 7-2 Soil Washing



Process Diagram *http://www.frtr.gov/matrix2/section4/4-19.html*

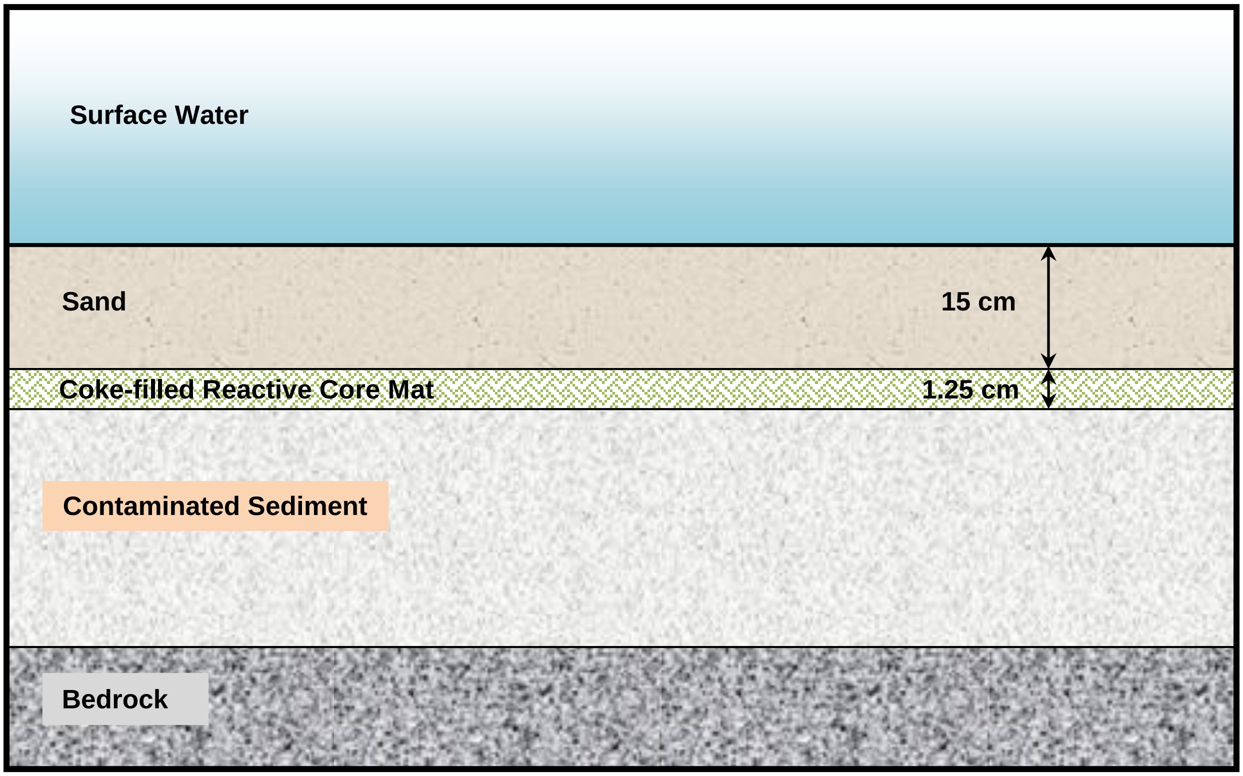


Soil Washing. Miami River Soil /Sediment Separation Plant.
Source: Boskalis-Dolman 2006

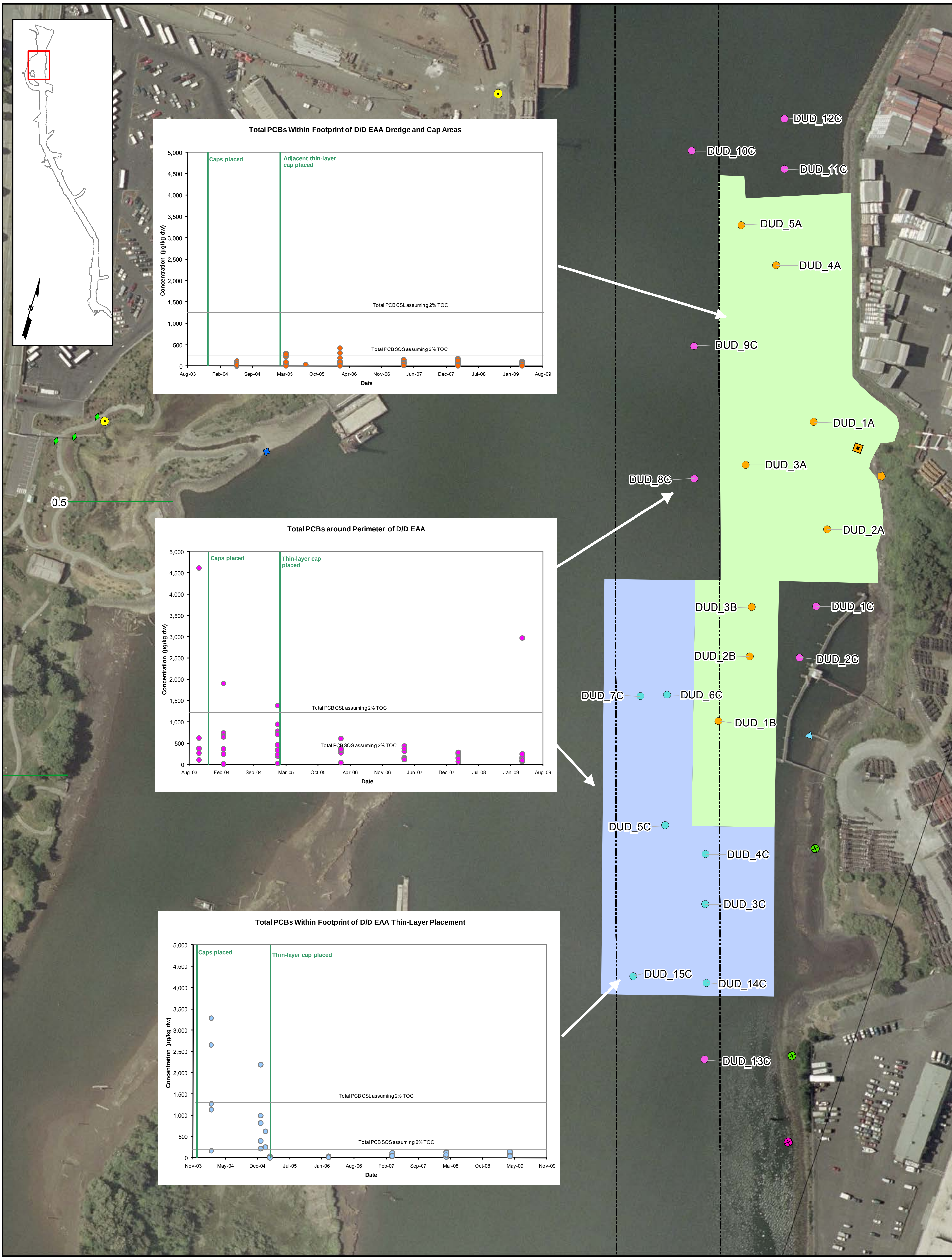
Figure 7-3 Mechanical Placement of Cap at Ward Cove, Alaska

Source: Candidate Technologies Memorandum, Retec 2005.

Figure 7-4 Schematic of Reactive Cap from Anacostia River



Note:
This reactive core mat (RCM) was designed to accurately place a 1.25-cm thick sorbent (coke) layer in an engineered sediment cap in twelve 3.1-m × 31-m sections. The RCM was overlain with a 15-cm layer of sand to secure it. It was placed in the Anacostia River (Washington D.C.) during the Anacostia River Active Capping demonstration project in April of 2004 (McDonough et al. 2006).



Notes:
1. USGS 2002 photograph provided by Windward Environmental.
2. Data provided by Windward Environmental in Access database accompanying Final Remedial Investigation (Windward 2010).
3. CSO= combined sewer overflow; EOF= emergency overflow.
4. Outfalls shown were identified during a City of Seattle low-tide survey in 2003 (Herrera 2004). Some locations were initially identified using drainage maps from Ecology's National Pollutant Discharge Elimination System (NPDES) permit files and other relevant agency databases. These locations were later surveyed in the field. Review of agency files and interviews with agency and LDWG personnel provided additional outfall-specific information. Some locations were field-verified by LDWG members; some additional outfall locations were identified during these subsequent verifications.

Legend

Dredge and Cap (2003-2004)
Thin-layer Sand Placement (2005)

Outfall Location

- Private storm drain
- Permitted private storm drain
- Public storm drain
- CSO/storm drain
- EOF
- Stream, channel, or swale
- Abandoned
- Pipe of unresolved origin and/or use

Sampling Location

- Dredging/Cap
- Thin-layer Sand Placement
- Perimeter

River Mile Marker
Navigation Channel

0 120 240 480 Feet

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Surface Sediment Total PCB Trends at Duwamish/Diagonal EAA

FIGURE 7-5

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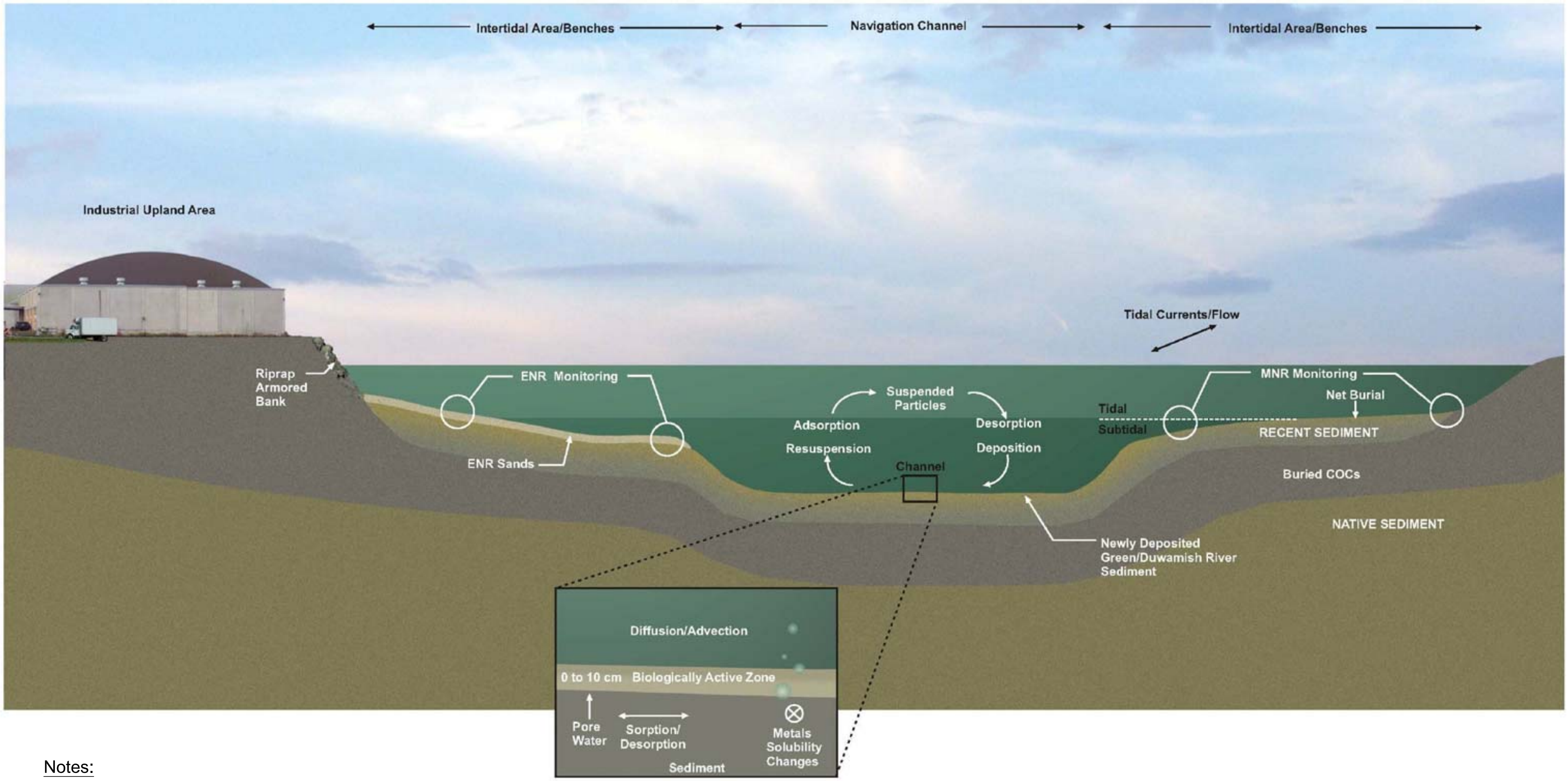
Figure 7-6 Placement of Under-pier Capping Sand between Bents by Sand Throwing Barge

Source: Interim Construction Inspection Report, Todd Shipyards (McCarthy and Floyd|Snider 2005)

Figure 7-7 Finished Shotcrete Surface on Debris Mound

Source: Interim Construction Inspection Report, Todd Shipyards (McCarthy and Floyd|Snider 2005)



Notes:

COCs = Contaminants of Concern
ENR = Enhanced Natural Recovery
MNR = Monitored Natural Recovery

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CONCEPTUAL DIAGRAM OF MONITORED NATURAL RECOVERY AND ENHANCED NATURAL RECOVERY

FIGURE 7-8

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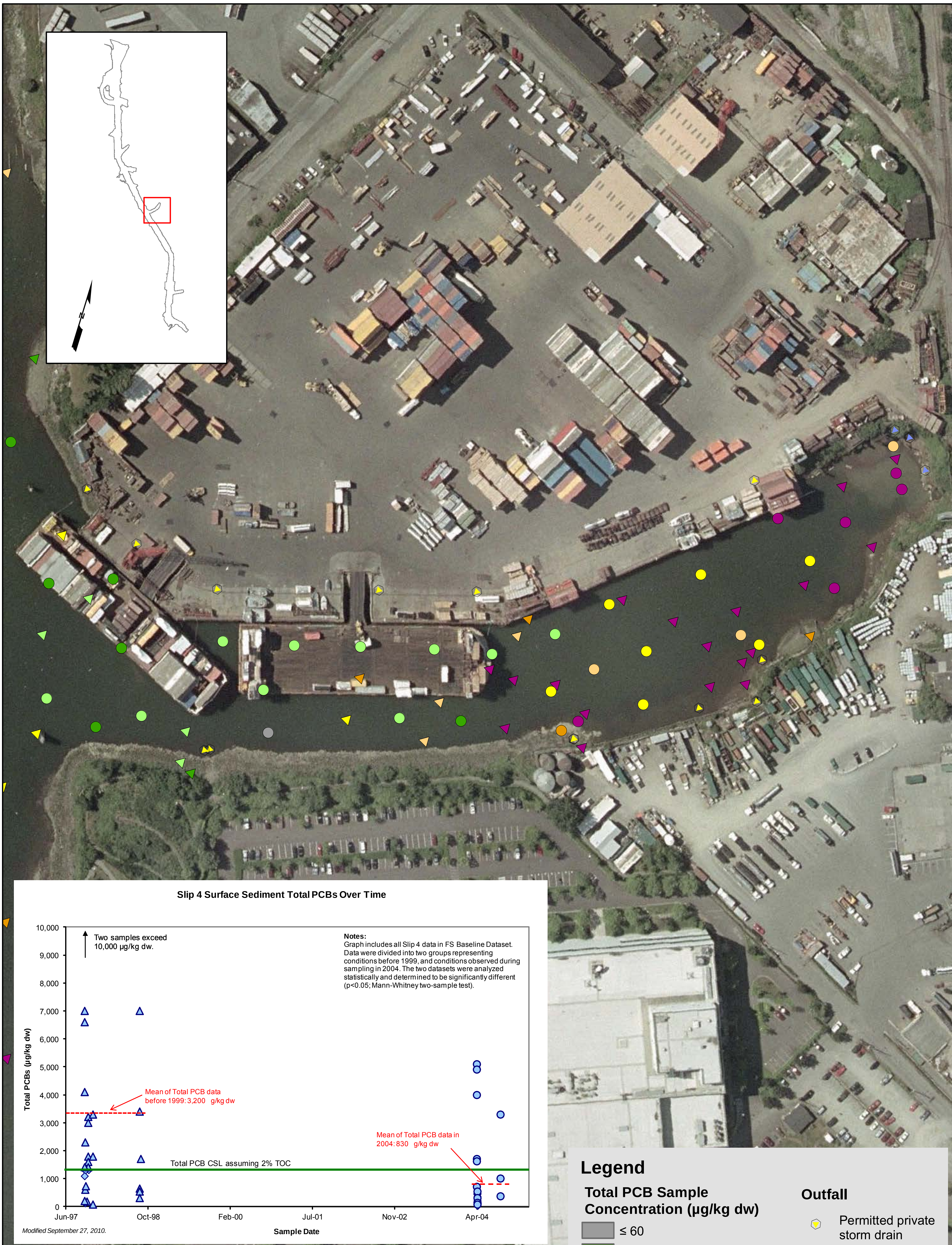
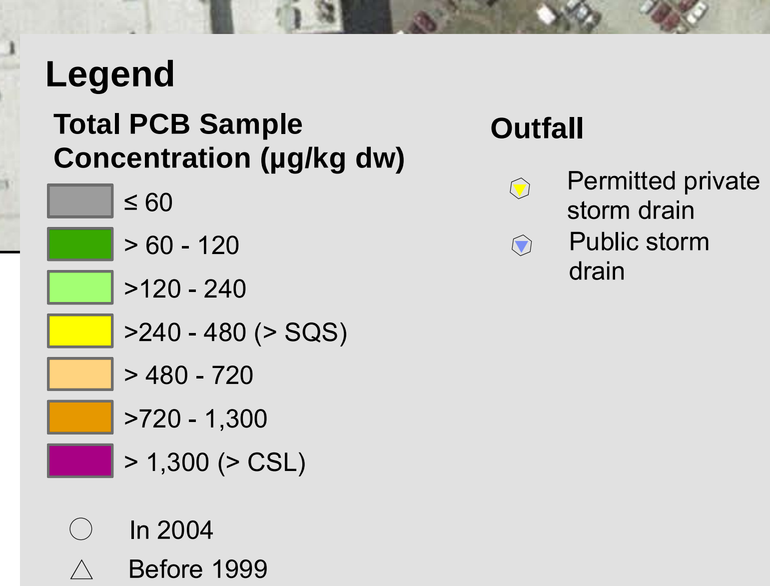
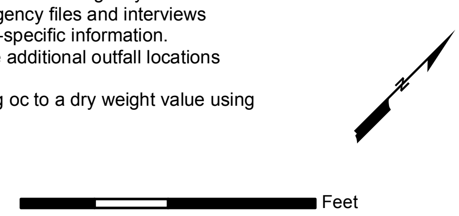




Notes:
1. USGS 2002 photograph provided by Windward Environmental.
2. Outfalls shown were identified during a City of Seattle low-tide survey in 2003 (Herrera 2004). Some locations were initially identified using drainage maps from Ecology's National Pollutant Discharge Elimination System (NPDES) permit files and other relevant agency databases. These locations were later surveyed in the field. Review of agency files and interviews with agency and LDWG personnel provided additional outfall-specific information. Some locations were field-verified by LDWG members; some additional outfall locations were identified during these subsequent verifications.
3. SQS value of 240 µg/kg dw based on conversion of 12 mg/kg oc to a dry weight value using 2% TOC.



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Lower Duwamish Waterway Final Feasibility Study 60150279-14.39			Surface Sediment Total PCB Trends in Slip 4
DATE: 10/31/12	DWRN:MVI/sea	Revision: 1	FIGURE 7-9

Figure 7-10 Sloping Drain Barge (Hylebos Waterway, Tacoma, WA)

Source: "Barge Dewatering on Contaminated Sediment Projects" WEDA Pacific Chapter 2007 Annual Meeting Honolulu, Hawaii. Presented by Integral Consulting Inc., October 26, 2007