EPA Review Comments on PDI Evaluation Report and Acoustic Fish Tracking Study 12-Month Addendum

Dated June 17, 2019 and July 25, 2019

Comments dated September 12, 2019

Following are the United States Environmental Protection Agency's (EPA's) comments on the documents titled *PDI Evaluation Report, Portland Harbor Pre-Remedial Design Investigation and Baseline Sampling, Portland Harbor Superfund Site, Portland, Oregon* (PDI Report) and *Appendix B.7c Acoustic Fish Tracking Study 12-Month Addendum* prepared by AECOM Technical Services (AECOM) and Geosyntec Consultants, Inc. (Geosyntec) on behalf of the Portland Harbor Pre-Remedial Design Group (Pre-RD Group).

General and specific comments on the PDI Report follow. The general comments pertain to EPA's overall assessment of the material presented in the PDI Report. Specific comments are focused on the sections, tables, and figures as referenced and can also be assumed to cover the material presented in the Executive Summary, where applicable. EPA's comments on the PDI Report appendices (Appendices) follow the specific comments and are organized by appendix.

Main Report Text

General Comments

1. The PDI work plan (Geosyntec 2017), attached as an exhibit to the Pre-RD Group Administrative Settlement Agreement and Order on Consent (ASAOC), listed the following data use objectives relevant to the investigation activities outlined in the work plan: 1. Implement investigation baseline sampling to update existing sitewide data; 2. Gather data to be used as part of baseline dataset for future long-term monitoring; 3. Assist in refining the scope and extent of the remedial actions that will be performed at the Portland Harbor Superfund Site (site), including refining sediment management areas (SMAs), informing technology assignments consistent with the technology application decision tree (TADT) in the 2017 Record of Decision (ROD) (Appendix I Figure 28) throughout the site and refining the horizontal and vertical extent of the dredging and capping areas; 4. Collect data to facilitate completion of the third-party allocation by potentially responsible parties (PRPs), this allocation process is independent of EPA oversight; 5. Collect additional data regarding upstream conditions and contaminant loading into the site; and 6. Update and evaluate site conditions to refine the conceptual site model (CSM) for all pathways consistent with the ROD, page 106 (Post-ROD Data Gathering). The PDI work plan acknowledged that the pre-design data collected pursuant to the ASAOC is a first step and that additional sampling will be necessary during the remedial design phase.

   EPA considered the data provided by the Pre-RD Group as presented in the PDI Report and Appendices. Based on EPA's review of the data, EPA finds the pre-design investigation and baseline sampling (PDI/BL) data to be of suitable quality and generally acceptable for remedial design and the long-term monitoring program, pending minor corrections described in the Appendix C Comments. Based on its review of the data, EPA finds that the PDI/BL sampling program achieved data use objectives 1, 2, 3, 5, and 6 as detailed in the PDI work plan (Geosyntec 2017). Data use objective #4 pertains to the allocation process which is independent of EPA oversight and EPA cannot confirm if this data use objective was achieved. EPA anticipates that the PDI/BL data, in addition to existing and design-level data, will inform implementation of the ROD decision tree and appropriate remedial technologies during design. Furthermore, the PDI/BL data will serve as an essential baseline dataset for the long-term monitoring program against which the success of the remedy will be measured.

In addition to the data provided by the Pre-RD Group as presented in the PDI Report and Appendices, the PDI work plan allowed for the Pre-RD Group to present its interpretations of the data to EPA for consideration. Although EPA agreed to consider the Pre-RD Group's interpretations, the agency made no advance representation of its acceptance of such interpretations. Specifically, the PDI work plan stated that the "EPA and the Pre-RD Group recognize that the data gathered in the PDI is not the complete dataset for final remedy design/implementation and that EPA's review of the data reports or data analysis may include an assessment as to whether the data relied on is sufficient to support final evaluations, refinements, recalculations and updates." Consistent with the terms of the ASAOC and PDI work plan, EPA has considered the Pre-RD Group's interpretations of the PDI/BL data. As stated above and further explained below, although EPA finds that the PDI/BL data to be generally acceptable for use during remedial design and long-term monitoring, the data and the Pre-RD Group's analysis does not support many of the conclusions presented in the PDI Evaluation Report.

Please note that the PDI/Baseline data are not meant to replace all existing data. Rather, performing parties in remedial design will develop their own site-specific data replacement strategy that meets reasonable statistical standards and considerations. Although it is typical for EPA to consider updated sampling data and information in the context of remedial design, once a remedy decision is made and supported by a comprehensive administrative record (AR), it is not typical for the agency to continue revisiting assumptions made during the risk assessment, remedial investigation or feasibility study in the absence of compelling new information. In the case of Portland Harbor, the Proposed Plan received 5,348 individual comment submissions which were reflected in the ROD and Responsiveness Summary (RS). The ROD is supported by an extensive AR that forms the basis for the remedy decision. The Selected Remedy outlined in the ROD anticipates consideration of updated information during remedial design and allows for flexibility when determining appropriate remedial technologies through use of the ROD decision tree and site-specific conditions. Sampling data collected during the Pre-RD sampling effort, in addition to more refined SMA sampling data collected during remedial design, is important to moving cleanup forward at this site. The Pre-RD Group's evaluations of the data that revisit many decisions made in the ROD are not well supported and do not provide evidence of significant changed circumstances or inaccuracies in the assumptions in the ROD. EPA's rationale for its positions are provided in the comments that follow.

2. The PDI Report and Appendices present evaluations and conclusions on temporal change in contaminant concentrations between the remedial investigation and feasibility study (RI/FS) and PDI/BL data for sediment, surface water, and fish tissue. As described in the final FS dispute (Lower Willamette Group [LWG] Dispute Issue 1d and Legacy Site Services, Inc. [LSS] Dispute Issue 5 [AR Doc # 100036161]) and the LWG's 2012 draft FS (LWG 2012), evaluating temporal changes in contaminant concentrations was not an objective of the RI/FS data collection efforts. The temporal change analyses in the PDI Report are based on a

statistically insufficient number of samples (surface water, sediment traps) or differences in sample design (surface sediment, fish tissue) that preclude robust quantitative statistical analyses and conclusions. Additionally, there is an unknown statistical bias in the RI/FS data due to the different study designs as described in FS Appendix I (EPA 2016b). Interpolation methods such as the natural neighbor interpolation can reduce the bias in the RI/FS data but still do not allow for robust statistical comparisons between the RI/FS and PDI/BL surface area weighted average concentrations (SWACs). Going forward, statistically robust, quantitative rates of temporal change can be developed from the PDI/BL stratified random sampling (SRS) surface sediment samples and future long-term monitoring (LTM) samples that replicate this unbiased study design.

3. The PDI Report and Appendices provide minimal presentation of the ROD Table 17 contaminants of concern (COCs) that are not the focused COCs. All the environmental media sampled during the PDI/BL study except for the subsurface sediment (via cores) and the porewater (using peepers) were analyzed for their applicable ROD Table 17 COCs.
4. EPA has not finalized the proposed Explanation of Significant Differences (ESD; EPA 2018). The ROD cleanup levels (CULs) and remedial action levels (RALs) for polycyclic aromatic hydrocarbons (PAHs) are the applicable standards until the ESD is finalized.

Specific Comments

1. Section 2.1: Systemwide recovery was evaluated quantitatively in the FS and is discussed in detail in FS Appendix D Section D8 (EPA 2016b). The ROD selected a remedial alternative that is reliant on natural recovery and source control, as evidenced by the 84% of the total site area selected for monitored natural recovery (MNR). The statements in this section that assert that the RI/FS and ROD did not consider the rate at which natural recovery would occur do not account for the discussion in the FS Appendix D Section D8.
2. Section 2.1.1, Table 2.1, and Figure 2.1 series: Net deposition over the entire site area or within the ROD SMAs does not account for the dynamics observed at smaller spatial scales. Most of the site (1,008 acres) is neutral and may gain or lose sediment in different seasons or different years. This was further evaluated in the FS (Appendix D Section D8) with an analysis called "Consistency of Depositional and Erosional Processes." EPA performed this analysis again with data from the bathymetry surveys conducted in 2002, 2003, 2004, 2009, and 2018 to incorporate the most recent survey data. Results from the consistency evaluation show that the areas outside of the ROD SMAs (i.e., MNR areas) are, by surface area, 56% consistently depositional, 13% consistently erosional, and 31% neutral or in dynamic equilibrium. By contrast, the ROD SMAs are 30% consistently depositional, 17% consistently erosional, and 53% neutral or in dynamic equilibrium. Consistency of deposition is one line of evidence considered during the FS that suggests that the ROD SMAs, which are predominantly located in the nearshore areas, are less subject to natural recovery and require active remediation. See Appendix D.1 Comments for further discussion.
3. Section 2.1.2, Figure 2.2 series, Figure 2.3, and Figure 2.4: No discussion of uncertainty is included in this section, and the results are not clearly qualified as estimates or lacking statistical certainty. SWACs are estimates of average concentrations, and those developed from the RI/FS data using interpolation methods such as natural neighbor or Thiessen

polygons are not directly comparable to the PDI/BL SWACs because of differences in study design. Furthermore, direct comparisons of concentration change over time between proximal samples cannot be considered a quantitative line of evidence. This is due to the heterogeneity in sediment concentrations and the inherent probability of collecting a sample with less contaminant mass than the historical sample (i.e., a 50% probability). Similarities in sediment core profiles is one line of evidence for stability of contaminated areas and is insufficient alone to declare overall stability of the river bed. EPA evaluated temporal change in sediment concentrations with numerous analyses including:

a. Estimating SWACs at different spatial scales
b. Calculating rolling river mile concentrations (FS Appendix D Section D10; EPA 2016b)
c. Estimating temporal change in sediment decision units (SDUs) and proximal samples using a paired-difference statistical method combined with debiasing the RI/FS data using the SRS grid cell areas
d. Regression analyses using the above debiasing method for the RI/FS data

EPA acknowledges that decreases in contaminant concentrations in sediment are occurring consistent with the ROD CSM as described in Sections 6.2.4 and 10.1.1.6 (EPA 2017a). However, these estimated decreases are predominantly in the MNR areas, as noted in the PDI Report and PDI Report Appendix I, and do not reflect significant decreases in contamination in the ROD SMAs such that the RALs or TADT (ROD Appendix I Figure 28) need to be updated. Further discussion is included in Appendix D.2 and D.3 Comments.

4. <u>Section 2.1.3 and Figure 2.5 series:</u> Different compositing methods were used for the RI/FS and PDI/BL surface water sampling events such that direct comparisons can only be considered qualitative in nature. Furthermore, samples were collected under different flow conditions despite targeting similar flow regimes for the three rounds of sampling. Surface water CULs in the ROD were selected after a review of applicable or relevant and appropriate requirements (ARARs) and represent Federal or State of Oregon statutes or regulations. These CULs are a long-term goal of the selected remedy as discussed in ROD Section 15 (EPA 2017a). See Appendix D.5 Comments for further discussion.

5. <u>Section 2.1.4 and Figure 2.6 series:</u> The previous (2002 and 2007) fish tissue sampling events were designed and conducted differently from more recent studies (2012 and 2018). The 2002 and 2007 fish tissue sampling events composited specimens on a river mile basis either with combined (2002) or separate (2007) sides of the river. Additionally, specimens were either analyzed as whole body or as fillet and offal fractions, with concentrations in fillet or whole body estimated by calculation. These differences in study design preclude direct comparisons and make temporal change evaluations semi-quantitative estimates. Despite the differences in study design, EPA evaluated temporal change in the RI/FS and PDI/BL fish tissue data to develop estimates of the rates of change of contaminant concentrations in smallmouth bass (SMB) fish tissue from 2002 to 2018. This was accomplished through the development of a first order decay mixed effects model that assumes a common sitewide rate of change with differing absolute concentrations; this data handling provides greater statistical power by incorporating all fish specimens into the analysis. This analysis suggests that sitewide fish tissue concentrations are decreasing at

rates less than 10% per year. However, the sitewide trends do not represent accurate predictors of temporal change at smaller spatial scales where more variability was observed, likely due to the heterogeneity of sediment concentrations. See Appendix D.6 Comments for further discussion. Furthermore, the ROD acknowledges that the risk-based tissue target levels will not allow for unlimited fish consumption at the site and there is uncertainty due to surface water inputs at a watershed scale (ROD Sections 11 and 15.1.1; EPA 2017a). Consequently, it is premature to suggest that the tissue target levels are unattainable based on one sampling event.

6. Section 2.1.5: This section draws inaccurate conclusions about the potential for the site to achieve the ROD background-based sediment CULs by conflating the Downtown Reach and the Upriver Reach. As described in ROD Section 14.4, there are known sources in the Downtown Reach and EPA is relying on the Oregon Department of Environmental Quality (DEQ) to address these sources (EPA 2017a). Furthermore, the results of the PDI/BL surface sediment and sediment trap data in and near the Upriver Reach are predominantly below CULs and do not suggest limitations for achieving these CULs in the site. See Specific Comment #10 and Appendix F Comments for further discussion.

7. Section 2.1.6: The statement in Section 2.1 that "expanded use of natural recovery is an appropriate part of the Site remedy" is unsupported by the relevant Appendices or sufficient statistical confidence. Consistent with the ROD, EPA expects the SMA footprints to be refined during remedial design with higher spatial density sampling, which in turn will inform whether additional use of natural recovery is warranted. That said, the PDI/BL surface and subsurface sediment data will be used to inform remedial design sampling, and along with these new remedial design data, may show that the lateral extent of the SMA footprints have decreased.

8. Section 2.2, Figure 2.8 series, Figure 2.9 series, Figure 2.10, and Table 2.2: This section summarizes a set of evaluations performed during the FS to develop the CSM and determine applicable remedial technologies. Evaluations summarized in the PDI Report and performed by EPA confirm the validity of the ROD CSM and conclude that the high concentration areas are predominantly limited to the ROD SMAs. However, this section misinterprets consistency for sediment bed stability and does not base its conclusions on the FS consistency evaluation (Appendix D Section D8; EPA 2016b). The section incorrectly states that only 31 out of 90 PDI/BL subsurface sediment cores have RAL exceedances and does not clarify which RALs were applied for which COCs (77 PDI/BL subsurface cores have ROD RAL exceedances). Additionally, the section provides no empirical evidence for the conclusion that "in-situ remedial technologies are likely to remain permanent and stable." Additional lines of evidence will need to be considered on a site-specific basis during remedial design to definitively demonstrate that in-situ remedial technologies will be suitable for a given area. These include but are not limited to geotechnical slope stability evaluations, vessel/wind/wake modeling, and cap modeling. However, the flexibility in the ROD TADT allows for areas that demonstrate stability and protectiveness with in-situ remedial technologies to select capping over dredging during remedial design.

9. Section 2.3: Comparisons of surface sediment concentrations and SWACs at various spatial scales are considered estimates due to the differences in study design and limitations of statistical analyses such as interpolations. See General Comment #2, Specific Comment #3,

and Appendix D.2 Comments for further discussion. EPA's evaluation of SWACs at various spatial scales suggest that concentrations have decreased since the RI/FS due to natural recovery and source control. However, the occurrence of site SWACs lower than those in the ROD is not a sufficient line of evidence alone for reevaluating the ROD RALs. Decreases in SWACs are driven predominantly by decreasing COC concentrations in the MNR areas, which account for approximately 84% of the total site area and receive greater sediment deposition than the ROD SMAs. Section 2.2 of the PDI Report acknowledges that the highest surface sediment concentrations in the PDI/BL data were measured in the ROD SMAs, thus "indicating stability of the high concentration areas." Additionally, the ROD MNR areas and RALs were selected based on multiple lines of evidence evaluated during the FS (FS Appendix D) including initial and post-construction target SWACs, deposition and erosion rates, consistency of depositional and erosional processes, sediment grain size, anthropogenic factors, subsurface to surface sediment concentration ratios, and wind and wake generated waves (EPA 2016b). Lastly, decreasing sediment concentrations over time is consistent with the ROD CSM, as described in ROD Sections 6.2.4 and 10.1.1.6 (EPA 2017a).

10. <u>Section 2.4, Table 2.3, Table 2.4, Table 2.5, Figure 2.7 series:</u> The PDI/BL data from the Upriver Reach do not support the conclusion in this section that the ROD background-based, 95% upper confidence limit (UCL) sediment CULs are unattainable. In surface sediment, 90% of the Upriver Reach detected concentrations were less than the focused COC CULs. Sediment trap sediment concentrations from the river mile (RM) 16.2 transect, which is in the Downtown Reach, were all less than CULs for total polychlorinated biphenyls (PCBs), total PAHs, and dichlorodiphenyltrichloroethane and its derivatives (referred to collectively as DDx) during all PDI/BL sampling rounds. The polychlorinated dibenzo-p-dioxin and furan (dioxin/furan) congeners had some results with minor CUL exceedances, which is expected due to the higher percentage of fine-grained material in sediment traps compared to surface sediment. Suspended sediments in surface water represent potentially settleable material, and their effect cannot be quantitatively evaluated due to the absence of a fate and transport model for the site. As detailed in ROD Section 6.6.2, sediments captured in sediment traps indicate "the effect of erosion and resuspension of bottom sediment, the presence of current sources, or both" and that "approximately 82% of the suspended sediment load passes through the site" (EPA 2017a). Therefore, the Upriver Reach surface sediments are the most representative sampling strategy for understanding reasonably attainable surface sediment concentrations in the site. Furthermore, the average and 95% UCLs described in this section utilize data from the Downtown Reach. This is incompatible with the ROD, which has defined the Downtown Reach as an active cleanup area under the purview of DEQ. The ROD did not estimate fish tissue background concentrations due to insufficient data; therefore, the risk-based fish tissue target levels in the ROD are not cleanup standards but will be used to inform institutional controls such as fish advisories. Equivalence testing was performed by EPA with the PDI/BL data as per the baseline sampling and LTM plan (EPA 2017b). Results from this evaluation confirm that sediment and fish concentrations in the site are statistically higher than, and therefore not equivalent with, those in both the Downtown Reach and the Upriver Reach. Progress toward attaining remedial action objectives (RAOs) will be assessed during the five-year reviews at which

point the background-based CULs may be adjusted up or down as appropriate pending future LTM data. See Appendix D.2, D.4, D.5, D.6, and F Comments for further discussion.

11. Section 2.5.1, Table 2.6, and Figure 2.11 series: Providing an update to the publicly reviewed and EPA-approved baseline human health risk assessment (BHHRA; EPA 2016a) is outside the scope of the PDI/BL sampling program as defined in the ASAOC and PDI work plan (Geosyntec 2017). The PDI/BL sampling program was not designed for risk assessment and is not an agreed upon data use. As stated in BHHRA Section 2.0, "data needs for the BHHRA were identified through the data quality objective (DQO) process described in Section 7 of the Programmatic Work Plan (Integral et al. 2004)" (EPA 2016a). Integral et al. (2004) developed DQOs, sampling locations, and sampling numbers for the BHHRA following EPA's risk assessment guidance (EPA 1991). The data use objectives and DQOs for the PDI/BL sampling program outlined in the PDI work plan and PDI QAPP, respectively, do not discuss risk assessment.

    Despite risk assessment not being an agreed upon data use, EPA assessed the exposure assumptions and risk estimates in the PDI risk update. The estimated reductions in cancer risk and noncancer hazard are inflated due to incorrect application of the PDI/BL data to the 2013 BHHRA exposure assumptions (EPA 2016a) and because of multiple issues with the "up-to-date exposure assumptions," rendering them invalid and inadequate for decision-making at the site. Sediment exposure point concentrations (EPCs) with the PDI/BL data are not directly comparable to those from the RI/FS due to differences in the DQOs, sampling locations, and sampling numbers between the two studies. The BHHRA evaluated consumption of multiple species of resident fish (SMB, common carp, brown bullhead, and black crappie) for recreational and subsistence fishers, not just SMB. Concentrations of chlorinated organic compounds such as PCBs, DDx, and dioxins/furans were measured up to an order of magnitude higher in carp compared to SMB during the RI (EPA 2016a). The PDI risk update only considered SMB tissue concentrations as this was the only species collected during the PDI/BL sampling. The fish consumption rates in the BHHRA for the recreational fisher were based on a regionally relevant survey in the Columbia Slough (Adolfson Associates, Inc. [Adolfson]1996), whereas the PDI update relied on a study from Idaho (Buckman et al. 2015) without justifying its applicability to the site. The subsistence fisher consumption rates in the PDI risk update (EPA 2014a) do not state that they replace rates from the previous national survey used during the BHHRA (EPA 2002). Tribal fish consumption rates and diet composition in the PDI risk update are based on surveys from three tribes (Polisar et al. 2016a, 2016b; SRC 2015), two of which are not regionally appropriate and relevant to the site area. The 95th percentile fish consumption rate from the one regionally appropriate and relevant tribe (Nez Perce) in the PDI risk update was 233.9 grams per day (g/d), which is greater than the 175 g/d assumed in the BHHRA (Polissar et al. 2016a; EPA 2016a). The BHHRA tribal fish consumption rate and diet composition was developed from a survey of regionally relevant and appropriate tribes (Columbia River Inter-Tribal Fish Commission [CRITFC] 1994). See Appendix G Comments for additional discussion.

12. Section 2.5.2 and Table 2.7: Risk assessment, either in the site or the Downtown Reach and Upriver Reach (referred to collectively as the D/U Reach), is not an approved data use in the PDI work plan (Geosyntec 2017). Furthermore, EPA disagrees with the exposure

assumptions and risk estimates in the PDI risk update, as detailed in Specific Comment #11 and Appendix G Comments. The ROD risk-based fish tissue target levels are not cleanup standards but will be used to inform institutional controls. Additionally, the appropriateness and scientific validity of the RI/FS background evaluations and the ROD background-based sediment CULs was formally disputed and decided on March 24, 2015 (AR Doc # 500011627) and December 27, 2016 (AR Doc # 100036161). The suggestion in this section that background sediment values were not appropriately calculated is contrary to the decisions in the site AR. Compliance with sediment CULs will be measured in the top 30 centimeters (cm) of sediment (i.e., surface sediment), which represents the biologically active zone as detailed in ROD Section 14.2 (EPA 2017a). The RI/FS background evaluations measured COC concentrations in surface sediments in the Upriver Reach, which were then used for background-based sediment CULs where appropriate.

13. <u>Section 2.5.3:</u> EPA's evaluations of the PDI fish tracking data confirm rather than undermine the validity of the 1-mile home ranges previously estimated by an Oregon Department of Fish and Wildlife (ODFW) study (Pribyl et al. 2004) and applied in the BHHRA. EPA disagrees with the conclusion that the mechanistic food web model (FWM) used in the RI/FS needs to be updated. See Appendix D.7 and H Comments for further discussion. Risk-based sediment CULs were selected in the ROD only in instances in which these values were higher than background concentrations. The PDI risk update does not address uncertainties identified in the BHHRA and is an insufficient update of human health risks at the site.
14. <u>Section 2.6:</u> EPA disagrees with the conclusions presented in this section as summarized in Specific Comments 1 through 13.
15. <u>Section 3.1 and Table 3.1:</u> EPA disagrees with the assertion that the PDI/BL data support updated background values for the site. See Specific Comment #10 and Appendix F Comments for further discussion.
16. <u>Section 3.2 and Table 3.2:</u> The background porewater study targeted areas with reducing conditions and shallow anoxic zones to measure the upper range of naturally occurring dissolved arsenic and manganese concentrations in the lower Willamette River. However, the number of samples collected (n=9) precludes robust statistical analyses, and none of the porewater peepers achieved equilibrium, leading to uncertainties in the data. Additional study is warranted before background groundwater CULs for arsenic and manganese are established to replace the ARAR and risk-based CULs. That said, the background porewater study was a successful pilot program serving as a first step toward understanding dissolved arsenic and manganese concentrations upstream of the site. See Appendix D.8 Comments for further discussion.
17. <u>Section 3.3.1:</u> EPA disagrees with the assertion that the FWM developed by the LWG during the RI/FS and approved by EPA for use in calculating sediment risk-based preliminary remediation goals (PRGs) cannot accurately predict or relate sediment and fish tissue concentrations. Section 3.3.1 incorrectly states that the mechanistic FWM does not account for surface water exposure to dissolved COCs, that dissolved water concentrations alone would result in unacceptable fish tissue concentrations, and that the background-based sediment CUL for total PCBs was selected due to the absence of a "realistic FWM-based value." Insufficient supporting information is provided in the PDI Report and Appendix H to

independently evaluate whether the FWM was appropriately calibrated for the PDI data. The validity of the FWM is supported by the AR final FS dispute, LWG Dispute Issue 1l (AR Doc # 100036161). EPA's evaluations with the PDI/BL data found statistically significant positive relationships between smallmouth bass tissue concentrations and surface sediment concentrations (collected near fish capture locations) for total PCBs, DDx, and dioxins/furans. Additionally, the PDI fish tracking data support the home range estimates from the previous ODFW study (Pribyl et al. 2004). Lastly, EPA disagrees with the premise that purported issues with the mechanistic FWM and updates to the SMB home ranges are appropriate rationale for updated sediment CULs. See Appendix D.7 and H Comments for further discussion.

18. Section 3.3.2 and Table 3.2: EPA disagrees with the assertion that the updated CULs represent the lowest achievable and sustainable post-remedial concentrations of total PCBs; 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD); 1,2,3,7,8-pentachlorodibenzo-p-dioxin (1,2,3,7,8-PeCDD); and 2,3,4,7,8-pentachlorodibenzofuran (2,3,4,7,8-PeCDF). Appendix D.2 Table 1 shows that between 37 and 52% of the PDI/BL surface sediment samples in the site are already below the ROD background-based sediment CULs for these four focused COCs. These percentages are expected to increase in subsequent LTM sampling rounds following the completion of active remediation at the site. Furthermore, CULs are long-term remedial goals that need to be met to achieve RAOs; background-based CULs may be adjusted up or down as appropriate during the five-year reviews based on future LTM data.

19. Section 3.4 and Table 3.3: EPA disagrees with the assertion that the 2013 BHHRA did not use the most current fish consumption rates or statistical analyses for use in risk assessment. As discussed in Specific Comment #11 and Appendix G Comments, there are assumption issues with the PDI risk update that make it inadequate for decision-making at the site. The ROD fish tissue target levels are not enforceable CULs and will be used to inform institutional controls such as fish consumption advisories.

20. Section 3.5, Table 3.4, Table 3.5, and Figure 3.1 series: EPA disagrees with the assertion that updated RALs are needed based on the PDI/BL data. Estimated decreases in the sitewide SWAC, SWACs outside of ROD SMAs (i.e., MNR areas) approaching background sediment concentrations (see Specific Comment #10 and Appendix F Comments), and the limited spatial extent of dioxin/furan samples in the RI/FS data are inadequate rationale for developing updated RAL curves using only the PDI/BL data. In its review of the Pre-RD Group's assertions, EPA developed RAL curves following the methodology in FS Appendix D Section D1 (EPA 2016b) using the PDI/BL data alone and also combined with the RI/FS data. These evaluations show that the ROD RALs are at the knee of the curve and still represent reasonable thresholds that will result in significant risk reduction before reaching diminishing returns. Further discussion is included in Appendix I Comments.

21. Section 3.6: Issues with the PDI risk update (Specific Comment #11 and Appendix G Comments) invalidate the claim that media concentrations are now below the 1×10^{-3} cancer risk that triggers toxicity-based principal threat waste (PTW) designations. EPA expects that areas of PTW that are highly mobile and not reliably contained (NRC) will be fully delineated during remedial design. A detailed description of EPA's designation of PTW is summarized in the final FS dispute, LWG Dispute Issue 2c, and LSS Dispute Issues 2 and 17 (AR Doc # 100036161). Section 14.2.9.1 of the ROD states that "cap design may require the

use of activated carbon and/or other reactive material, as necessary" (EPA 2017a). Furthermore, cap design will be determined during remedial design based on cap modeling performed with site-specific parameters such as area-specific seepage velocities, initial contaminant concentrations, organic carbon content, and other design parameters.

22. Section 4.1, Section 4.2, Tables 4.1 – 4.5, and Figure 4.1: The refined SMA footprint presented in the PDI Report is an FS-level type of evaluation. Refining SMA footprints can only occur during remedial design using all existing data, including new data collected during remedial design. The refined SMA footprint proposed by the Pre-RD Group incorporates RALs different from those in the ROD, eliminates the three dioxin/furan congeners as focused COCs, replaces historical samples within 100 feet of a PDI/BL sample, and does not include delineation of PTW. Estimates of post-construction SWACs and risk are therefore insufficient. Furthermore, footnote 43 on page 45 incorrectly states that report updates to Kleinfelder (2015) have confirmed sampling depths. An errata memorandum with this information was only recently submitted to EPA for its review and consequently EPA cannot at this time include of the Kleinfelder (2015) data in its analysis. The data from Kleinfelder (2015) have not yet been approved by EPA for unqualified use and conclusions drawn using these data cannot yet be corroborated by EPA. For purposes of review, EPA estimated the SMA footprint following the methodology in FS Appendix C (EPA 2016b), using the PDI/BL data alone (362 acres) and the PDI/BL data combined with the RI/FS data (375 acres). These SMA footprints are comparable to the ROD SMA footprint of 365 acres. See Appendix J Comments for further discussion. CUL exceedances in the site for 1,2,3,4,7,8-hexachlorodibenzofuran and 2,3,7,8-tetrachlorodibenzofuran are not a line of evidence that the background-based ROD CULs are too low. The validity of background-based CULs is evaluated using data from the Upriver Reach and may be adjusted up or down as appropriate during the five-year reviews based on future LTM data. Furthermore, the PDI/BL data show that concentrations exceeding the ROD PTW threshold for these two COCs is limited to approximately RM 5 to 7 and 4.5 east in the site.

23. Section 4.3: The uncertainties with J-flagged values in the PDI/BL data qualitatively described in Section 4.3 are incorrect. J-flagged values are concentrations less than the quantitation limit (QL) but greater than the method detection limit (MDL) and the *Department of Defense (DoD) Department of Energy (DoE) Consolidated Quality Systems Manual (QSM) for Environmental Laboratories* states that concentrations measured at the MDL are accurate with 99% confidence for DOD-accredited laboratories (DoD/DoE 2019). All the laboratories that analyzed the PDI/BL samples for dioxins/furans were DOD accredited. Furthermore, dioxin/furan data were validated according to EPA National Functional Guidelines (EPA 2016c), which allows for the use of professional judgment when qualifying results that do not meet the ion abundance ratio (IAR) criteria. These samples were qualified as JN rather than as U, signifying the Pre-RD Group data validators' confidence in the analytical result to represent a detected value. See Appendix E Comments for further discussion.

24. Section 4.4: EPA disagrees with the assertion that the ROD TADT needs to be modified to allow for the considerations discussed in Section 4.4. The PDI/BL data do not adequately characterize any one area of the site for remedial design as acknowledged in the PDI work plan and PDI Report. Revisions to the ROD TADT are not necessary because area-specific

design level data collected by performing parties will govern final technology application decisions; the flexibility inherent in the ROD TADT allows for area-specific decisions to be made during remedial design. See Appendix L Comments for further discussion.

25. Section 4.5 and Table 4.6: The remedy design considerations detailed in the PDI Report and Appendices do not provide significant quantitative evidence that the ROD RAOs will be achieved. The PDI Report with Appendices is essentially an insufficient FS, which is not the intent or purpose of the PDI/BL sampling program or the PDI Report (see General Comment #1). EPA disagrees with the summary conclusions presented in this section as detailed in the Specific Comments above.
26. Section 4.6: It is inappropriate to eliminate COCs from the LTM program based on the results of the PDI/BL data. See Appendix M Comments for further discussion.
27. Section 5: EPA disagrees with the conclusions presented in this section as detailed in the comments contained herein.

Appendix B

Appendix B.1 Comments

1. Section 4 presents the results of various field quality control checks on the sonar and vessel positioning systems. However, the cumulative effects of the elevation and depth differences detected during these quality control tests is not evaluated. Total Propagated Uncertainty (TPU), which is the total propagated magnitude of all random and systematic errors, is a metric referenced by the U.S. Army Corps of Engineers (USACE) to evaluate the accuracy of hydrographic measurement methods (USACE 2013). As described in USACE 2013, TPU is estimated using mean square error propagation techniques that result in a total "±" error estimate. Appendix B.1 noted that quality control checks met project requirements as specified in the bathymetric survey field sampling plan (FSP; AECOM and Geosyntec 2018b) but did not provide any information regarding the level of uncertainty in final elevations (e.g., TPU) and the potential impacts to conclusions drawn from these data.
2. The location of gaps in survey coverage are not provided, other than general descriptions of site features that could result in coverage gaps. The spatial distribution of coverage gaps and potential data gaps in SMAs is not included in Appendix B.1. Additionally, the impact of coverage gaps on bathymetric data evaluation is not described in the report.
3. The reasoning for comparison of bathymetric survey results to the 2004 bathymetric survey, and not other bathymetric surveys previously conducted at the site (i.e., 2002, 2003, and 2009), is not provided in Appendix B.1. The 2004 bathymetric survey only provides one snapshot of sediment elevations, and comparison to 2004 alone does not provide a complete characterization of sediment dynamics at the site. Evaluation of multiple bathymetric survey pairs, as described in Appendix D of the FS (EPA 2016b), is needed to evaluate whether areas are consistently depositional, consistently erosional, consistently neutral, or in dynamic equilibrium.

Appendix B.2 Comments

1. Section 2.4.3 describes that a revised protocol was implemented following discussion with EPA on May 18, 2018. However, as noted in Section 2.1, surface sediment sampling started on March 30, 2018. The impact of revising sampling protocol during the field sampling

effort is not described in Appendix B.2. It is unclear if all samples were collected in accordance with the revised protocol and whether there are potential data quality issues with samples collected before the revised protocol was implemented.

2. On May 8, 2018, EPA oversight staff observed that at some locations, collection and processing of surface sediment composite samples was taking up to 2 hours, resulting in the initial grab samples remaining uncovered for extended periods of time. This was identified as a potential concern because warm ambient temperatures and sun exposure could result in the loss of contaminant mass due to volatilization. On May 11, 2018, the AECOM and Geosyntec field team began covering the sediment bowls with aluminum foil (while leaving the bowls on deck), and on May 13, 2018, the field team began placing the covered bowls in coolers with ice while other grab samples were being collected. However, these practices were not implemented universally throughout the sampling program. Samples that were not covered and placed on ice are not identified in Appendix B.2, and potential impacts to data quality from processes including but not limited to volatilization are not discussed in the report.
3. Section 1.2 states that the D/U Reach samples targeted sediment areas with >35% fine-grained sediment and a target range of 0.04 to 27% total organic carbon (TOC). However, the final approved field sampling plan (FSP; AECOM and Geosyntec 2018g), did not include a targeted ranged of TOC as part of the decision criteria for selection of D/U Reach surface sediment sampling locations.
4. The surface sediment FSP and Appendix B.2 describe that samples should be collected within a 25-foot radius of the target location, if possible, and then collected within a larger radius (i.e., 50 feet for SRS locations) if sampling within the 25-foot radius is not feasible. However, Appendix B.2 does not provide details on whether grab samples were collected within the 25- or 50-foot radius (or beyond these radii). This information would be helpful to support data analysis and to demonstrate that samples were collected at randomly selected points and not at locations with sampling bias.

Appendix B.3 Comments

1. During its oversight of sediment coring activities, EPA noted important information that has not been summarized in Appendix B.3. For instance, from July 23 through July 25, 2018, EPA oversight staff observed the field team using an unacceptable method of core storage that resulted in the lower 3 to 4 feet of the core being placed in a plastic bin full of ice and the upper 2 to 3 feet of the core protruding out of the bin with the core tube exposed to sunlight and air temperatures as high as 96 degrees Fahrenheit. Per the subsurface sediment coring FSP, cores should be sectioned into 4- to 6-foot lengths for storing vertically on ice (AECOM and Geosyntec 2018e). Improper cold storage resulted in potential for volatilization of contaminant mass from core intervals. Elements pertaining to the FSP deviation are not included in Appendix B.3, and it is unclear as to how these deviations may have affected data quality.
2. Section 2 summarizes sampling equipment decontamination methods and procedures. However, Appendix B.3 does not include any information on the instances in which improper decontamination of dedicated sampling equipment occurred. It is necessary to provide information on deviations from the subsurface sediment coring FSP, including but

not limited to improper cooling of collected cores, improper decontamination of dedicated sampling equipment, and exceedances of suggested holding times prior to core processing and how these deviations may have affected data quality.

3. Section 1 states, “Core tubes were placed in a refrigerator until processing, which was typically done 24 to 48 hours after collection. Core tubes were kept in an upright position and placed on ice on the sampling vessel prior to transporting to the field laboratory.” The subsurface sediment coring FSP (AECOM and Geosyntec 2018e) recommends core processing occur within 24 hours of collection. However, during its oversight of sediment coring activities, EPA oversight staff noted several instances of cores being processed more than 48 hours after collection, with two cores processed 96 hours and 9 days after core collection. Elements pertaining to these FSP deviations are not included in Appendix B.3, and it is unclear whether these deviations may have affected data quality.

Appendix B.4 Comments

1. Instances of compromised sediment tubes were observed and recorded in each of the sediment trap deployments. Section 2.7 states that the compromised or broken tubes were not included in the final composite sample “under CDM Smith oversight.” CDM Smith personnel did not approve of or recommend this practice in the field but rather were only present to observe that the data were collected in accordance with the approved FSP.

2. The surface water and sediment trap FSP does not include a section on sample acceptance or what to do in the case of compromised samples (AECOM and Geosyntec 2018h). Therefore, the omission of material from compromised sediment trap tubes is a deviation from the FSP. There is no text in Appendix B.4 identifying compromised sample tubes as a deviation from the FSP. Similarly, there is no discussion of how the reduced count of tubes in cases where they were compromised could affect the representativeness of the composite sample.

Appendix B.7c Comments

1. EPA disagrees with the SMB home range estimates provided in Appendix B.7c as described in the Appendix D.7 Comments. Further discussion on material provided only in Appendix B.7c is included below.

2. The description of study design in the fish tracking FSP characterizes gate receivers as providing a presence/absence, river-mile level of locational detail (AECOM and Geosyntec 2018a). It is unclear from the information provided in the fish tracking FSP and Appendix B.7c how it was determined which individual receiver a tag was closest to when it came within range of a particular gate. The typical receiver detection distances reported in Appendix B.7c Section 3.2 range from 200 to 500 meters. This distance approximates the Willamette River channel widths in some areas as noted in this section. Therefore, all individual receivers comprising a gate would presumably log redundant, overlapping detections of a tag once it was within range of the gate until such time that it passed out of range. More information is needed to explain how a specific receiver within a gate complex was identified as the “closest” to a given tag when the tag is detected simultaneously by multiple gate-associated receivers.

3. Section 3.4 references a footnote indicating that several detections corresponding to non-moving tags were excluded from the updated 12-month kernel density estimation (KDE)

because they were suspected to be associated with a predation event. Neither the fate of these tags nor any tags related to the study were ever confirmed following the initial release of tagged fish. This introduces the potential for confounding data resulting from post-predation or post-harvesting detections to be included at any point during the study. Of the approximately 300 observations excluded from KDE analysis, 74% were attributed to "Likely post-predation detection." The removal of these data without first verifying tag fate is a potential source of exclusion bias and may affect the resulting accuracy of home range estimates.

Appendix B.8 Comments

1. The highest oxidation reduction potential (ORP) for anoxic processes to occur was identified as 50 millivolts (mV) in the porewater FSP, and a footnote in Section 2.3.4 states that the same criterion was used during the pre-screening (AECOM and Geosyntec 2018f). It is unclear why the pre-screening results shown in Table 1 of Appendix B.8 rely on an ORP of 100 mV to determine anoxic conditions. The FSP also stated that "an acceptable target depth would be about 20 cm; if that is not possible, it is recommended to move to another target station for better penetration depth." However, Section 2.3.2 of Appendix B.8 states that peepers were "pushed to a minimum target depth of 15 cm," which is inconsistent with the FSP and with the drive depths shown in Table 2.

Appendix C

Appendix C.4 Comments

1. The PDI/BL data need to be provided in the Scribe format as per the final data quality management plan (AECOM and Geosyntec 2018c) prior to final approval for unqualified use by EPA.

2. River miles were included in the PDI/BL database file only for some of the SMA subsurface sediment cores and fish tissue sample locations. This information should be included in the database for all samples collected.

3. The top approximately 3 feet of sediment core SC-S113(B) were spilled on the deck of the research vessel after core collection on 8/6/2018 and the remaining material was stored at 4 degrees Celsius for 9 days until being processed and sampled on 8/15/2018. The subsurface sediment coring FSP states that core processing should ideally occur within 24 hours of core collection (AECOM and Geosyntec 2018c). The analytical results for all intervals from subsurface sediment core SC-S113(B) should be J qualified due to issues with sample handling and the elapsed time before sample processing.

Appendix D

Appendix D.1 Comments

1. The Willamette River within the site is a dynamic river system that experiences episodic deposition and erosion over a range of spatial and temporal scales caused by natural and anthropogenic factors. Although the site is net depositional, sediment is not deposited uniformly, and some areas are net erosional or in "dynamic equilibrium" and subject to periods of oscillating deposition and erosion. It is therefore important to consider the patterns of deposition and erosion over time and finer spatial scales and not simply net erosion or deposition.

2. In addition to the 2004 bathymetric survey that is used for comparison in this report, sitewide bathymetric surveys were completed in 2002, 2003, and 2009 to understand seasonal and inter-annual sediment elevation changes. To understand the dynamic nature of sediment erosion and deposition, comparison is needed among the multiple bathymetric surveys and not just the 2004 bathymetric survey. Some areas that are net depositional are dynamic in nature, with episodes of erosion and deposition. Independent data analysis performed by EPA comparing all bathymetric surveys (i.e., 2002, 2003, 2004, 2009, and 2018) indicated that the ROD SMA areas are 30% consistently depositional and 70% erosional, neutral, or in dynamic equilibrium.
3. The process for identifying and mapping generalized substrate types in Figure 3 is not clear in the report. Several data inputs used for mapping are described (i.e., grain size and total organic carbon data, 2018 bathymetry, bathymetric changes from 2004 to 2018, difficulty collecting power-grab samples, number of sample attempts, Atterberg Limit results, and visual observations and classification of substrate during sampling). However, the process of applying these inputs to produce the map and the relative weight of the data inputs is not described. Additionally, supporting data other than bathymetric data are not provided or referenced in Appendix D.1. It is therefore difficult to understand the uncertainty and applicability of the map presented in Figure 3.
4. The conclusion that refinement is needed to the CSM is not supported by the data and analysis presented in Appendix D.1. As described in Appendix D.1, general sediment erosion and depositional patterns are consistent with those described in the RI. Despite being net depositional sitewide, deposition is not distributed equally throughout the site, and areas of deposition and erosion are present throughout the site. There are a variety of sediment textures across the site, and the sediment textures are influenced by river hydrodynamics. These characteristics of sediment dynamics and types were previously described in the CSM in the RI and the ROD.
5. The conclusion that "recovery" is occurring is not supported by the data and analysis presented in Appendix D.1. This conclusion seems to hinge on net depositional rates and percentages of areas within the site that are net depositional. However, as described in Comments #1 and 2, net deposition is an insufficient method to evaluate erosion and deposition processes and does not account for the spatial variability of contaminated sediment. Even within specific SDUs or SMAs, net volume gain does not suggest that deposition was uniform and complete throughout these areas nor does it suggest the relative contaminant burden of the depositing sediments. Most importantly, evaluation of "recovery" requires data on concentrations of COCs in surface and subsurface sediments, surface water, and fish tissue. Appendix D.1 only presents data and analysis on bathymetric survey results and does not provide supplementary chemical data that would support evaluations beyond sediment dynamics.

Appendix D.2 Comments

1. The purpose of the PDI/BL surface sediment sampling program that is described in Section 1.1 is not consistent with the overall PDI/BL study goals identified in Section 3.2.2 of the quality assurance project plan (QAPP), the goals specific to surface sediment sampling presented in Table 3 of the QAPP (AECOM and Geosyntec 2018d), or the six data use objectives in the PDI work plan (Geosyntec 2017). The distinction between study goals in

the approved QAPP/work plan and data interpretation methods presented in Appendix D.2 is important to note as the study goals informed the data use objectives in the PDI work plan and should inform evaluations with the PDI/BL data.

2. There is insufficient information presented in the PDI Report to dismiss the surface sediment dioxin/furan results as "uncertain." As described in EPA review comments to Appendix E, results were independently validated, and data were accepted. Accordingly, these data are considered usable and can be evaluated in a manner that is consistent with other analytes.

3. Statistically robust quantitative rates of temporal change for surface sediment cannot be developed by comparing the RI/FS and PDI/BL datasets because of the differences in sample design for the two studies. The RI/FS sampling included targeted nearshore locations with elevated concentrations to evaluate risk and nature and extent of contamination, and the PDI/BL dataset was designed as an unbiased sampling program for use as baseline conditions for eventual comparison to LTM data after implementation of the remedy. Therefore, direct comparison of SWACs represents a qualitative estimate of concentration changes rather than a statistically robust comparison. The impact on SWAC calculations due to study design is even acknowledged in Section 4.1.3 when discussing differences in SWAC estimates when using SRS only and combined SRS and SMA datasets: "This does not necessarily indicate a methodology weakness in the calculation approaches or datasets but is likely a result of inherent differences in the datasets being used."

4. Several actions have already occurred within the site that involved active cleanup of contaminated sediments. Some of these occurred between 2004 and 2018, significantly biasing the comparisons presented in Appendix D.2. A full description of these actions is presented in the ROD and FS. Actions that included active remediation of sediments between 2004 and 2018 are summarized as follows:

    a. Terminal 4 (RM 4.5 East): Dredging and off-site disposal of 12,819 cubic yards of contaminated sediment was completed in 2008. Additionally, contaminated sediment was capped with an organoclay-sand mix in the back of Slip 3, and the bank along Wheeler Bay was capped and stabilized.

    b. NW Natural (RM 6 West): Approximately 15,300 cubic yards of tar-like material and tar-like contaminated sediment were dredged from the river bank and nearshore area adjacent to the Gasco facility and disposed of off-site in a permitted facility between August and October 2005. An organoclay mat and sand cap were installed over the dredged area and a portion of the area surrounding the removal area.

    c. McCormick and Baxter Superfund Site (RM 7 East): A sediment cap was placed on approximately 23 acres of contaminated sediment in 2004–2005. The sediment cap was one component of a variety of upland and in-water remedial actions to address contaminated soil, groundwater, stormwater, and Willamette River sediment (DEQ 2005).

    d. BP Arco Bulk Terminal (RM 5 West): Approximately 12,300 cubic yards of near-shore petroleum-contaminated sediment was removed in 2007–2008 under DEQ oversight and disposed of off-site. Removed material was replaced with clean fill.

5. Appendix D.2 generally focuses on sitewide or rolling river mile evaluations and comparisons to previous surface sediment datasets (particularly from 2004). Portland Harbor is a large and heterogeneous Superfund site, and generalized sitewide analysis and conclusions have limited application for remedial design other than to serve as a point of comparison for future evaluations of remedy effectiveness. Evaluation of surface sediment data at the SMA-scale, which is absent from Appendix D.2, is needed to support preparation for pre-design sampling that will inform remedial design.
6. The exclusion of dioxin/furan data from the analysis in Section 4.3 is not sufficiently justified in Appendix D.2 (see Appendix D.2 Comment #2 and Appendix E comments). Independent evaluation performed by EPA identified 78 instances where a dioxin/furan sample collected during the PDI/BL was within 100 feet of a sample collected during the RI/FS. A paired difference method was used to estimate concentration changes from the RI/FS to the PDI/BL data at these locations. For each of the five dioxin/furan congeners with RALs or PTW thresholds identified in Table 21 of the ROD (EPA 2017a), there was a statistically significant mean increase ($p<0.05$) in the concentrations detected in the PDI/BL samples compared to the proximate samples collected in the RI/FS, suggesting that concentrations of these COCs in surface sediment have increased. These results are contrary to the conclusion of recovery presented in the PDI Report and suggest that surface sediment recovery is both analyte- and location-specific.
7. As described in Section 4.3.1, the proximate point-by-point comparisons of PDI/BL to RI/FS data have some spatial uncertainty. Accordingly, this analysis has inherent limitations and cannot be used to justify replacement of RI/FS sample data as is recommended in Section 5.

Appendix D.3 Comments

1. The conclusion of the "overall stability of the system" is not supported by the data presented in Appendix D.3. This conclusion relies only on sediment core profile similarities between the RI/FS and PDI/BL datasets and the Top:Max ratio with the PDI/BL samples. Similar contaminant profiles at depth and Top:Max ratios less than 1 represent a single line of evidence but are not sufficient to determine sediment stability or natural recovery potential. Natural recovery was evaluated quantitatively using multiple lines of evidence during the FS and is summarized in FS Appendix D Section D8 (EPA 2016b). The recovery analysis described in Section 4.1 demonstrates that contamination is present at depth and is not recovering naturally. As summarized in Appendix D.1 Comment #1, the Willamette River is a dynamic river system that experiences physical processes (erosion, deposition, dispersion, bioturbation, advection) over a range of spatial and temporal scales that would influence "system stability." The data presented in Appendix D.3 does not provide supplementary data on system stability for these physical processes. Appendix D.3 only presents data and analysis on sediment core profiles and recovery ratios. It is unclear as to how the development of "overall system stability" was made in the absence of other important factors such as those described in FS Appendix D Section D8 (EPA 2016b).
2. Section 3 states, "Total PAH ROD RAL exceedance frequencies were greatest in RM 6-7, with decreasing frequencies in the downstream direction." This statement is misleading as Table 1 presents several total PAH exceedances downstream of RM 6; nearly 50% of cores downstream of RM 6 exhibit total PAH ROD RAL exceedances.

3. Section 4.2 states, "Cores outside SMAs that exhibit robust patterns of natural recovery with ROD RAL exceedances only at depth justify the delineation of SMAs based primarily on surface sediment concentrations." As described in Section 14.2 of the ROD, the PDI/BL dataset will inform the implementation of the remedial design and construction; selected remedies will be site specific and consider the lateral and vertical extent of contamination (EPA 2017a). While informing remedial design and construction, the limited number of cores and spatial density of the PDI/BL data are not sufficient to justify delineation of SMAs based primarily on surface sediment concentrations.

Appendix D.4 Comments

1. Average concentrations of total PCBs measured at the RM 11.8 transect were higher during each sampling event than the average concentrations at the RM 16.2 transect. Sections 3.2 and 5 conclude that this difference is the result of contamination from the Downtown Reach. However, this conclusion does not account for the hydrodynamics of the lower Willamette River. During low-flow conditions the lower Willamette River experiences tidally induced flow reversals causing suspended sediments and surface water to flow upstream. The highest concentrations of PCBs, and the only sediment trap sample when total PCBs were greater than their CUL, were measured during the low-flow deployment when flow reversals were the greatest magnitude and most frequent. Therefore, these elevated sediment trap concentrations may be influenced by tidally-induced flow reversals and the contamination directly downstream of RM 11.8 at the RM 11E project area within the site.

2. Statistical evaluations presented in Appendix D.4 that lump together low-flow, storm-flow, and high-flow sediment trap deployments are not useful for comparisons to CULs or evaluation of sediment characteristics (e.g., percent fines). As described in Section 3.1, sediment accumulation rates are different among the three periods evaluated (i.e., low-flow, storm-flow, and high-flow), and the arithmetic mean will not account for these differences in accumulation rates. Calculating average COC concentrations and percent fines without accounting for differences in sediment volume creates averages that are inappropriately weighted and do not reflect mean characteristics of sediment that was deposited in the traps.

3. Section 3.3 concludes that sediment trap results are representative of background conditions in the D/U Reach, are not due to resuspension of nearby surface sediments, and are characteristic of fine-grain, high organic carbon sediment deposited within the site. This conclusion is based on a qualitative comparison of sediment trap and surface sample grain sizes and TOC. No literature references were provided to substantiate the claims. Surface sediment samples represent the top 30 cm of the bedded sediments homogenized across the depth profile. Resuspension and/or transportation of the top layer of sediment may occur under normal flow conditions. Therefore, there is no substantial evidence that material captured in the sediment traps varies significantly from the nearby surface sediment samples.

4. Section 4 states that the COC concentrations are generally consistent across sampling events from the RI/FS (2006/2007) to the PDI/BL (2018/2019) in the D/U Reach. Any temporal comparisons between RI/FS and PDI/BL sediment traps are qualitative in nature.

Because of the low sample size and differences in measured flows, sampling locations, and water depths, a statistically robust analysis is not possible.

Appendix D.5 Comments

1. Section 3 states, "During the RI sampling in 2006, total suspended solids (TSS) concentrations were lower during the storm-flow event than the low-flow event. This finding is inconsistent with the CSM, which suggests a positive correlation between TSS concentrations and flow rate." This conclusion is not supported by the data and analysis presented in Appendix D.5 nor the RI. Figure 3 in Appendix D.5 presents concentrations of TSS by RM for each sampling event in which several RM segments display higher TSS concentrations during the high-flow event compared to storm and low-flow events. This supports RI/FS conclusions, which indicate positive correlation between TSS concentrations and flow rate. Section 3.1.5.2.4 of the RI summarizes that the relationship is significantly stronger among the TSS data collected at flow rates above 50,000 cubic feet per second (cfs) when natural suspension of bed sediment is expected to occur compared to periods of low discharge. In both cases, non-precipitation-influenced TSS data are much more strongly correlated than precipitation-influenced TSS data (EPA 2016a). Average daily flow during RI sampling was generally greater than flow during PDI/BL sampling—nearly 17,000 cfs of difference between the events at Transect 5. Furthermore, the Round 1 (low-flow) surface water sampling occurred in late summer when the greatest extent of algal blooms is present in the lower Willamette River. Round 1 also exhibited the highest average chlorophyll measurements and higher turbidity compared to the storm-flow sampling event, likely contributing to higher average TSS measured during low-flow sampling. Appendix D.5 does not evaluate the differences in flow rates and sampling conditions/methods between the datasets nor does it consider precipitation-influenced TSS data in the evaluation.

2. Water quality metrics such as turbidity, TSS (see Comment #1 on Appendix D.5), and concentrations of COCs are often impacted by hydrologic conditions, including flow rate and relative hydrograph position at the time of sample collection (i.e., rising limb, peak, or falling limb). Sections 4 and 5 acknowledge these inherent differences between samples collected during the RI and PDI/BL sampling events. However, Appendix D.5 includes extensive comparisons between the RI and PDI/BL surface water sampling results. Many of these comparisons (e.g., Figures 25 through 39), lack hydrologic context to support evaluation of these data. Comparisons between surface water concentrations detected during a single sampling event in the RI and PDI/BL are qualitative in nature because of the inherent differences in hydrologic properties during sample collection.

3. Exhibit A presents methods for estimating surface water concentrations using data collected from in-situ monitoring probes for water quality and optical parameters. EPA understands the utility in using optical water quality monitoring data for estimating COC concentrations in surface water. Additional supporting information to validate the partial least squares regression model and principal components analysis could be used to further develop these concentration estimates.

Appendix D.6 Comments

1. Unlike appendices D.2 and D.3, spatial distributions of focused COC concentrations are not mapped to provide the reader with contextual understanding to compare fish tissue

concentrations to SMA boundaries. However, the Figure 4 series shows that several apparent spikes in fish tissue concentrations occur for each of the focused COCs. These high fish tissue concentrations appear to be correlated with elevated surface sediment concentrations of the same COCs (i.e., total DDx between RM 7 and 8; 2,3,7,8-TCDD between RM 4 and 5 and between RM 6 and 9; 1,2,3,7,8-PeCDD between RM 6 and 8; 2,3,4,7,8-PeCDF between RM 7 and 8; and total PCBs throughout the site with particularly elevated concentrations observed between RM 7 and 9 and RM 10 and 11). These data, along with the results from the fish tracking study (see comments to Appendix D.7), are lines of evidence to suggest that the following statement from Section 1 is unsubstantiated: "concentrations of COCs in fish are not expected to necessarily reflect only sediment concentrations in the particular segment, RM, or area where the fish was caught."

2. As described in Section 3.2, there are inherent differences in the study designs and fish tissue sample types (e.g., whole fish vs. fillet) that create significant uncertainty when evaluating temporal trends in fish tissue concentrations. These trends, therefore, are estimates rather than statistically robust comparisons. The PDI/BL fish tissue dataset will provide a baseline dataset for SMB to compare to during future studies to evaluate the progress and effectiveness of the remedy.

3. Conclusions regarding whether target fish tissue concentrations can be realistically achieved are premature. The PDI/BL dataset only includes SMB fish tissue concentrations and does not consider other species that were used to develop the risk-based fish tissue targets presented in Table 17 of the ROD (i.e., brown bullhead, black crappie, and common carp). Additional data collected during the implementation of the remedy will support evaluations of the effectiveness of the remedy with respect to reducing fish tissue concentrations below target levels.

4. Modification to background fish tissue concentrations or target tissue concentrations is not needed at this time and cannot be supported using the PDI/BL dataset that only includes SMB. As noted in the ROD and quoted in Appendix D.6, the background fish tissue concentrations for PCBs will be evaluated during design and construction of the Selected Remedy. Additionally, Section 2.3.7 of the ROD RS (EPA 2017a) describes that remedial levels for fish and shellfish tissue will not be enforceable CULs but rather target levels used to evaluate the progress and protectiveness of remedial actions. EPA expects that the selected remedy, along with upland source control and cleanups upstream of the site that are being conducted under DEQ oversight, will result in decreased COC concentrations in fish tissue.

Appendix D.7 Comments

1. The last quarter (mid-January 2019 through early May 2019) of the fish movement data was not provided concurrent with the PDI Report, preventing complete understanding of seasonal variation in the spatiotemporal movement, habitat use/preference, and home range estimates of SMB at the site. EPA received the addendum containing the 12-month data download from the fish tracking study on July 25, 2019 and will provide additional comments on this addendum in a future comment set.

2. SMB are known to take spawning migrations, establish summer feeding stations, and winter in offshore areas of deeper water when they are not feeding (Pribyl et al. 2004). Research

has shown that bioaccumulation in fish tissue is associated with dietary uptake of contaminated food particles and aqueous uptake of dissolved contaminants (Streit 1988). The focused COCs at the site are all organic contaminants that do not readily dissolve in water and instead bind strongly to sediment particles, leading to an overall greater importance of the dietary uptake route than the aqueous uptake route for bioaccumulation in fish (Agency for Toxic Substances and Disease Registry 2019). Pribyl et al. (2004) and the PDI fish tracking study demonstrated that SMB predominantly feed and spend their time in relatively small nearshore home ranges; therefore, their greatest exposure to the focused COCs occurs in these locations and at this spatial scale. EPA's independent evaluations with the PDI fish tracking data found that 23 fish had enough data to assess movement behavior during the summer feeding period (July 1 to October 1) and that these fish spent 93% of their days within a 1-mile home range. This suggests that fish tissue concentrations from specimens captured during the summer feeding period have a high likelihood of being associated with the sediment concentrations associated with the locations where they were caught. Further analyses showed that 50% of fish exhibited little movement within or outside of their release area. An additional 35% of fish spent two months or greater during the summer and fall (post-spawning) in the same general location after moving from their initial release point. Two fish (5%) that left the study area were initially released at the RM 11.5E array and may have been located just upstream of the RM 11.8 gate, but this cannot be corroborated due to the absence of movement data outside the study area. These results suggest that up to 90% of the tagged SMB spent most of their feeding time in the same relatively small area, consistent with the findings of Pribyl et al. (2004) that the ROD relied on.

3. Section 4 states, "The range of movement of SMB within the Site varies widely among individuals. A few stay within a localized area (e.g., a few acres), while others are highly mobile and travel distances of several miles or more within and beyond the Site." Furthermore, Exhibit F – Smallmouth Bass Tag Profiles includes figures depicting fish detections that extend beyond the study area (RM 11.8 to 16.8). The figure states, "For illustration only, travel direction for fish past study area receivers is unknown." As summarized in Comment #2 on Appendix D.7, the majority of SMB did not move from a small localized area. SMB exhibited movement associated with a summer feeding station, consistent with other studies of SMB movement behavior (Pribyl et al. 2004). Concluding that fish are "highly mobile and travel distances of several miles beyond the Site" is speculative, unsubstantiated by the data, and beyond the scope of possible inference. Furthermore, there is uncertainty in the data as to whether detected fish movement was due to fish movement behavior or predation/capture.

4. The conclusion that SMB are not a reliable metric for monitoring contaminant intake on a local scale is not supported by the data presented in Appendix D.7. No evaluations outside SMB mobility (i.e., feeding behavior or seasonal variations in movement) are presented in Appendix D.7 to support this claim. As SMB predominantly feed and spend their time in small nearshore home ranges, their greatest exposure to contaminants occurs in these locations and at this spatial scale. The spatiotemporal data presented in Exhibits E-1 and E-2 further support the findings of Pribyl et al. (2004) that SMB generally stay within localized nearshore areas, with occasional excursions out of the study area.

5. Section 4 states that the BHHRA and RI/FS assumptions of SMB residency in a single RM are not supported by the PDI fish tracking study. This conclusion seems to rely on the net distance an SMB traveled throughout the study period as opposed to actual range and is not supported by the analysis presented in Appendix D.7. Based on the results of Pribyl et al. (2004) and despite their findings that SMB favored nearshore areas, the BHHRA assumed a 1-river mile home range using both sides of the river. Furthermore, the uncertainties surrounding SMB home ranges as they pertain to risk estimates were evaluated in the BHHRA and determined to be of low magnitude/severity, low significance to risk management decisions, and unlikely to over or underestimate risk as summarized in BHHRA Table 6-1 (EPA 2016a). Data presented in Appendix D.7 demonstrates movement behavior consistent with Pribyl et al. (2004) and provides additional justification for the single RM SMB exposure areas assumed in the BHHRA.
6. One of the assumptions required for the implementation of KDE for home range analyses is that data points are independent from one another. The data used to generate the home range estimates provided in this study are spatially autocorrelated because their values are entirely determined by the location of the array associated with the detection. No explanation is provided for how this failure to meet a basic assumption is accounted for. A second assumption for KDE is that the data points being evaluated occur in Euclidean (i.e., conventional two-dimensional) space (Miller and Wentz 2003). If a dataset is the result of a network-based process (e.g., a river network) then using KDE to characterize habitat utilization patterns will produce misleading results (Downs and Horner 2007). The study provided as an example of KDE for estimation of linear home range (Vokoun 2003) explicitly describes how data were transformed to avoid pitfalls associated with the application of KDE to linearly constrained systems. No such explanation is provided in the PDI study.
7. There are several issues associated with the use of gate receiver detections to characterize SMB habitat utilization. These gate receivers lack positioning capability and only provide an approximate location for each detection event (i.e., the location of the gate receivers associated with the detection). Therefore, the number of observable locations is limited to the number of locations where gate receivers are installed (i.e., eight). Previous studies suggest a minimum sample size of 30 positions (but preferably ≥50) for accurate analysis (Seaman et al. 1999). Furthermore, gate receivers are widely spaced (i.e., a minimum of 1 river mile apart). These gate receivers provide a coarse, river-mile level representation of fish movement but lack the spatial resolution appropriate to characterize the finer-scale home range of a species that previous studies have found to exhibit relatively limited movement patterns (Pribyl et al. 2004). Additionally, this poor level of spatial resolution means that large areas contained between gate receivers are included in the home range estimate even though no actual data exist to describe fish utilization of this space. The second study cited as an example for the successful application of KDE for home range estimation used a similar study design (Lowe 2013) but only used data obtained from high-resolution arrays capable of triangulating exact locations to develop KDE for the reasons described above. The PDI study did obtain fine-scale SMB movement data utilizing high-resolution arrays installed at RM 11.5, Swan Island Lagoon, and Willamette Cove. For unknown reasons, these data were not selected for use in KDE estimation. These high-resolution data indicate that areal habitat use by SMB for 90% of positions recorded for

individual fish were within approximately 2.2 acres in the RM 11.5 high-resolution array, 2.5 acres in the Swan Island Lagoon high-resolution array, and 1.5 acres in the Willamette Cove high-resolution array, as described in Section 3.2.

8. The use of KDE for the estimation of home range has been widely criticized in cases where it is used to describe linearly constrained habitats (e.g., a river network) because these conditions typically result in an overestimation of home range (Blundell et al. 2001). One potential source for this positive bias lies in the selection of bandwidth as KDE is highly sensitive to this parameter. Bandwidths that produce an overly smooth estimated utilization distribution (i.e., the intensity or probability of use throughout an animal's home range) overestimate home range size. There is no explanation provided in the PDI Report for how the bandwidth is selected other than a footnote indicating that "bw_method = "silverman" method of determining bandwidth for KDE." However, documentation for the statistical function employed by the study indicates that bimodal or multimodal distributions tend to be over-smoothed and that as the data become more strongly bimodal, the formula will over-smooth more and more, relative to the choice of smoothing parameter. Frequency distributions for fish with detections at more than one gate receiver would be at the very least bimodal if not multimodal in shape and would therefore be subject to over-smoothing, resulting in an overestimate of home range. A second critical component of KDE calculation is the smoothing parameter. This value has a significant influence over the size of home range estimated by KDE. The larger the smoothing factor, the larger and less detailed the final home-range estimate will be (Silverman 1986; Worton 1989). The study methods do not provide any details for the smoothing factor applied to KDE calculations despite it being a primary determinant of home range estimates.
9. The text states, "KDE results were limited to the study area, plus the 75th percentile of the total SMB distance travelled as reported in Pribyl et al. (2004) of 8.0 km (5.0 mile), beyond each of the upstream and downstream limits." Adding distance to the bounds of the study area based on data collected from an independent research effort conducted using different methodologies is not scientifically sound, extrapolates beyond the study's spatial scope of inference, and artificially expands estimations of home range for fish detected by arrays located at study boundaries. As an example, the fish identified as tag 47075 had a range of 4.8 miles within the bounds of the study (i.e., most downstream detections occurred at RM 7, and most upstream occurred at RM 11.8, which represents the upstream study boundary) but is reported as having a KDE linear home range of 8.8 miles at a cumulative probability of 90%. For this reason and those described in the comments above, the home range estimates provided in the report are not scientifically defensible, fail to meet basic assumptions required for KDE validity, and extrapolate beyond the scope of the dataset.

Appendix D.8 Comments

1. As stated in Specific Comment #15, the limited number of samples collected (n=9) precludes robust statistical analyses. Additional study is warranted before background groundwater CULs for arsenic and manganese are established to replace the ARAR and risk-based CULs, which were developed using an extensive dataset. EPA considers the background porewater data a useful pilot study for understanding porewater arsenic and manganese concentrations in the D/U Reach.

2. The porewater FSP indicated that using desktop calculations of 90 and 75% equilibrium was expected to be achieved for arsenic and manganese, respectively (AECOM and Geosyntec 2018f). There is no discussion of why these expected rates of equilibrium were not achieved. It would be helpful to understand potential reasons for equilibrium not being achieved during the 28-day deployment period.
3. According to the ProUCL Version 5.1 Technical Guide (EPA 2015), an upper simultaneous limit (USL) is more appropriate for a "well-established background dataset representing a single statistical population without any outliers." EPA does not consider that this limited dataset constitutes a well-established background dataset (i.e., does not meet the criteria for developing USLs) and therefore does not provide a reasonable background threshold value (BTV) estimate.
4. The conclusion that the recommended BTVs "demonstrate that arsenic and manganese are naturally present in anoxic porewater typical of sediment in Portland Harbor" is not supported by the results of this study. EPA notes that this was not the objective of the study and the study does not focus on anthropogenic sources of arsenic and manganese; therefore, this conclusion cannot be substantiated by the available information.

Appendix E Comments

1. In the discussion about estimated maximum possible concentrations (EMPCs) in Section 2.1.3, the 2014 EPA Region 10 guidance is referenced as indicating lab-reported EMPC results less than the QL should be qualified U (EPA 2014b). The PDI/BL data were validated by the Pre-RD Group, and the data validators for the dioxin/furan data reference EPA *National Functional Guidelines for High Resolution Superfund Methods Data Review* (2016c). This document states the following:

    > "If the IAR criteria are not met, examine the other information provided to be sure the other criteria have been met. Check the calculation of EMPC results and/or ask the laboratory to recalculate and re-report these results. The isotope dilution method provides the ability to calculate ion ratios for the two ions monitored. If the IAR is outside the criteria, it does not unequivocally prove that dioxins/furans are not present; it indicates that either interference is present for one of the ions, or that another compound may be present. Use professional judgment to decide how to qualify EMPCs."

    The data validation reports note that "The National Functional Guidelines were modified to accommodate the non-Contract Laboratory Program methodologies. In the absence of method-specific information, laboratory QC limits, project-specific requirements, and/or professional judgement were used as appropriate." The data validators reported the EMPC values as JN-qualified results, which is a valid and reasonable decision. These data should stand as reported from the validator.

2. Appendix E, Section 2.1 states, "Measurements reported near the estimated detection limit (EDL) are more uncertain than estimated results near the QL because they are at low concentrations where a number of factors may interfere with accurate congener identification and quantification." As stated in Section 2.1.1, the MDL is defined as the minimum measured concentration of a substance that can be reported with 99% confidence that the measured concentration is distinguishable from the method blank. The EDLs are

calculated for each sample and vary for each sample because the EDL is affected by sample-specific factors such as signal-to-noise ratio and the presence of other organic compounds. However, the identification criteria are the same for detections close to the EDL, close to the QL, and above the EDL, and the 99% confidence level remains the same between the EDL and the QL. Therefore, there is the same degree of confidence in detected results near the EDL and near the QL. These data should stand as reported from the validator.

3. Appendix E, Section 2.2 states, "At low concentrations, when the sample reported concentration and detection limit are the same or similar and are also near the CUL, uncertainty makes it difficult to determine whether the CUL has been met." Because the 99% confidence level in detections is there for results at or above the EDL, these sample results are appropriate for direct comparison to the CUL.

4. Exhibit B misquotes FS Appendix B Section B2 (EPA 2016b) and incorrectly states that a relationship between dioxin/furan tissue concentrations and sediment concentrations has not been established. FS Appendix B Section B2 explains the process for determining whether a total dioxin/furan or 2,3,7,8-TCDD eq PRG would be appropriate for sediment and fish tissue. It was determined that the dioxin/furan congeners contributing most of the risk in fish tissue are not the same as those in sediment. Additionally, high concentrations at RM 7W were driving the relationship between sediment total dioxins/furans and fish tissue 2,3,7,8-TCDD eq. When these samples were removed, no relationship was present between tissue concentrations and sediment for these two summation methods. EPA performed a modified version of this analysis with the PDI/BL sediment and fish tissue samples at multiple spatial scales (individual sample comparison, river mile E and W, river mile both sides, proposed Pre-RD Group river segments) with the same results. High dioxin/furan concentrations at RM 7W drive the relationship between 2,3,7,8-TCDD eq in fish tissue and sediment. Therefore, the PDI/BL data support the development of sediment CULs and fish tissue target levels for individual dioxin/furan congeners. Furthermore, FS Appendix B Section B1 (EPA 2016b) and final FS dispute, LWG Dispute Issue 1l (AR Doc # 100036161) describe how the FWM was calibrated for individual dioxin/furan congeners and used to develop risk-based sediment PRGs.

Appendix F

Appendix F.1 Comments

1. The 2018 Upriver Reach surface sediment data do not support the claim that the background-based ROD sediment CULs for total PCBs and dioxins/furans are too low. As the ROD background-based CULs are derived from a statistical evaluation with 95% confidence, minor exceedances of CULs in surface sediment and sediment traps for some COCs in a small percentage of samples do not suggest that the ROD CULs are unattainable. This is supported by the fact that 44% of the PDI/BL surface sediment samples within the site are below the total PCBs CUL. Further discussion on the development of background, why the LWG's "equilibrium" theory is not credible, and EPA's decision on these items is included in the March 24, 2015 EPA dispute decision (AR Doc # 500011627) and the final FS EPA dispute decision (AR Doc # 100036161).

2. The Upriver Reach background reference area is defined in the ROD as RM 16.6 to 28.4 and does not contain the Downtown Reach (RM 11.8 to 16.6) (EPA 2017a). While the background-based CULs in the ROD were calculated using samples collected from RM 15.3

to 28.4, it is inappropriate to recommend updates to the upstream sediment background concentrations using samples collected from within the Downtown Reach. As described in ROD Section 14.4, the Downtown Reach contains known sources, and EPA is relying on DEQ to use its authority to address these sources (EPA 2017a). Despite the recent cleanups completed in the Downtown Reach, there are still known source areas, and it is inappropriate to include this area in estimates of background reference concentrations with cleanup ramifications. Differences between the Downtown Reach and the Upriver Reach are supported by the significantly higher concentrations of PCBs and PAHs in the Downtown Reach compared to the Upriver Reach.

3. The lower Willamette River experiences tidally influenced flow reversals (i.e., upstream flow) during periods of low river stage in late summer and early fall (EPA 2016a). Therefore, it is not possible to conclude that the PDI/BL upstream sediment trap and surface water data only represent conditions in the D/U Reach. Rather, upstream contaminant concentrations measured during the low-flow and storm-flow sampling events are influenced by contamination present in the site. Additionally, surface water particulate fraction concentrations represent potentially settleable sediments, and their effect cannot be quantitatively evaluated due to the absence of a fate and transport model for the site. The use of these data to develop updated background concentrations is contrary to the background evaluation in the RI (EPA 2016a), which was formally decided in EPA's March 24, 2015 dispute decision (AR Doc # 500011627).

4. Flow reversals occurred during the low-flow and storm-flow sediment trap deployments as evidenced by negative discharge at the Morrison Bridge river gauge (U.S. Geological Survey gauge #14211720) during this time. Sediment trap CUL exceedances for total PCBs only occurred at the RM 11.8 transect during the low-flow deployment as shown on Appendix D.4 Figure 7a. Therefore, these suspended sediments are influenced by the RM 11E project area within the site, and it is incorrect to conclude that the sediment trap CUL exceedances are an upstream source of contamination. Additionally, the Downtown Reach and by extension the RM 11.8 sediment trap transect are not representative of background as described in Appendix F.1 Comment #2. When considering only the RM 16 sediment trap transects, sample results from all three deployments (low-flow, storm-flow, high-flow) are less than the ROD CUL of 9 micrograms per kilogram (μg/kg), which further validates the ROD background-based sediment CUL.

5. The PDI/BL surface sediment sampling in the D/U Reach was not a stratified random approach. Random sample points were placed in targeted areas of fine-grained material, leading to contaminant concentration results that are biased high relative to the D/U Reach as a whole. The PDI/BL unbiased SRS surface sediment samples collected in the site utilized a stratified random sampling design with full spatial coverage of the site and did not include a threshold for fine-grained material.

Appendix F.2 Comments

1. The ROD Table 17 fish tissue target levels are risk-based concentrations, apart from mercury that is an ARAR-based target level, that will be used to inform fish advisories, evaluate construction impacts, and update best management practices and institutional controls (EPA 2017a). The fish tissue target levels are not enforceable CULs due to the complexities of bioaccumulation and uncertainties in achieving these levels, but they need

to be achieved for protectiveness purposes as outlined in the ROD. As stated in ROD RS Section 2.3.7, if the fish tissue targets are not achieved, EPA expects to reevaluate the remedy and determine what else may be needed (EPA 2017a).

2. The ROD fish tissue target levels were developed from risk estimates in the BHHRA that accounted for a multi-species diet of resident fish (SMB, brown bullhead, black crappie, and common carp) while the PDI/BL fish tissue study only sampled SMB. Any conclusions regarding the future attainment of the ROD fish tissue target levels needs to include data from multiple species collected during long-term monitoring after the completion of the selected remedy.
3. As stated in Appendix F.1 Comment #2, the Downtown Reach contains active cleanups currently being overseen by DEQ and is inappropriate for background comparisons.

Appendix F.3 Comments

1. Equivalence testing is a rigorously peer-reviewed statistical evaluation (Erickson and McDonald 1995; McBride 1999) with multiple United States regulatory applications as cited in the guidance documents and federal regulations listed below. It is currently being discussed at other sediment sites as a method to evaluate compliance with SWAC-based remedial objectives.
    a. EPA/Superfund attainment of cleanup standards: equivalence factor (EF) = 1.0; alpha = 0.05 (EPA 1989)
    b. EPA/National Pollutant Discharge Elimination System whole effluent toxicity testing: EF = 1.2 to 1.25; alpha = 0.05 (EPA 2010)
    c. EPA/Federal Insecticide, Fungicide, and Rodenticide Act pesticide registration: EF = 1.2; alpha = 0.05 (EPA 1988)
    d. U.S. Department of the Interior Office of Surface Mining Reclamation and Enforcement/Surface Mining Control and Reclamation Act: EF = 1.1; alpha = 0.1 (30 Code of Federal Regulations § 816.116)
2. The self-equivalence evaluation described in Sections 2.2 and 3.2 incorrectly used 1.5 as the factor for declaring self-equivalence and did not follow the procedures for selecting the self-equivalence factor based on the number of samples collected as described in EPA (2017b). Additionally, the Downtown Reach and the Upriver Reach are separate areas with statistically different contaminant concentrations as described in Appendices F.1 and F.2 of the PDI Report. Therefore, it is inappropriate to test for equivalence between these two reaches and describe it as “self-equivalence” testing. Finally, calculating an average of the 90% UCLs on the ratio of geomeans has no scientific or statistical basis and does not support the conclusion that the Downtown Reach and the Upriver Reach are not self-equivalent.
3. The conclusion that the equivalence factor is too low because none of the focused COCs in the site are equivalent with the Downtown Reach or the Upriver Reach is incorrect and shows a fundamental misunderstanding of the application of equivalence testing as described in EPA (2017b). Equivalence has not been achieved between the site and upstream because active remediation has not occurred across the site and the site is more contaminated than the upstream areas. After completion of the selected remedy, it is likely

that the site will approach equivalence with upstream and will achieve equivalence (with an appropriate equivalence factor) due to the active remediation, source control, and natural recovery. An appropriate equivalence factor may be selected based on a scientifically meaningful value, regulatory precedent, or the statistical variability of the LTM data.

Appendix G Comments

1. Providing an update to the publicly reviewed and EPA-approved BHHRA (EPA 2016a) is outside of the scope of the PDI/BL sampling program as defined in the ASAOC and PDI work plan (Geosyntec 2017). Furthermore, the PDI/BL sampling program was not designed for risk assessment, and this is not an approved data use. The evaluation in the PDI risk update assumed that all fish consumed were SMB, stating in Section 2.1.1 of Appendix G, "As the concentrations of COCs in SMB tissue were generally higher than crappie, lower than carp, and similar to bullhead, SMB is a representative surrogate for the mixed diet of resident species." However, the 2013 BHHRA evaluated a mixed diet of resident species for recreational and subsistence fishers, consisting of 25% each of black crappie, brown bullhead, common carp, and SMB. As noted in Section 5.2.6.2 of the 2013 BHHRA, "Differences among these species is reflected in the EPCs; specifically, the use of fillet SMB data on a river mile scale resulted in a greater relative reduction of PCB concentration than would be seen if fillet data from common carp and brown bullhead were included. As such, a diet that consists of some portion of carp and bullhead could result in relatively greater intake of PCBs, and the associated risk and hazard would be correspondingly greater as well." Thus, by only using the SMB data, the PDI risk update likely underestimated risks, especially when combined with the uncertainty in the composition of the mixed diet.
2. EPA's 2013 BHHRA used a range of exposure scenarios, including a recreational fish consumption rate of 49 g/d (based on a regionally relevant survey in the Columbia Slough [Adolfson 1996]), a non-tribal subsistence fisher at a fish consumption rate of 142 g/d consuming resident fish only (based on the previous national survey [EPA 2002]), and a tribal fish consumption rate of 175 g/d consuming a mixed resident and transient fish diet (based on a survey of regionally relevant and appropriate tribes (CRITFC 1994). In addition, ambient water quality standards established by the State of Oregon are based on a fish consumption rate of 175 g/d and considered ARARs for the site. Under CERCLA, the remedy is required to meet ARARs such as the state water quality standards. The PDI risk update relies on a study of fish consumption in Idaho (Buckman et al. 2015) for the recreational fish consumption rate without justifying its applicability to the site. As noted in EPA 2014a, patterns of fish consumption vary by geography, such as residents who live on or near the coast and those who live inland. Since residents in Oregon are near the coast, the fish consumption rates of recreational anglers in Oregon are likely to be higher than fish consumption rates of recreational anglers in Idaho, who live inland. For the subsistence fisher consumption rates, the PDI risk update uses the freshwater and estuarine finfish and shellfish usual fish consumption rate for the Pacific (EPA 2014a). While this report provides more current data for national and regional fish consumption rates from the National Health and Nutrition Examination Survey (conducted 2003–2010), the fish consumption rates that were used in the 2013 BHHRA were evaluated by several interested parties, and the approach and exposure values involved a formal dispute process. Arbitrarily revising exposure values used in the risk assessment is not appropriate during this stage of the project. Tribal fish consumption rates and diet composition in the PDI risk update are based

on surveys from three tribes (Polisar et al. 2016a, 2016b; SRC 2015), two of which are not regionally appropriate and relevant to the site area. As reported in Inset Table D in Appendix G in the PDI risk update, the one regionally appropriate and relevant tribe (Nez Perce) had a 99th percentile fish consumption rate of 233.9 g/d, which is greater than the 175 g/d assumed in the BHHRA. By averaging the fish consumption rates of the Colville, Nez Perce, and Shoshone-Bannock tribes for use as the PDI tribal fish consumption rate, the PDI risk update reduces the relevance of this fish consumption rate to the Portland Harbor site.

3. The PDI scenario assumed 24.2% of the tribal diet consists of resident species and 75.8% migratory species based on data from a fish consumption study of the Nez Perce (Polissar et al. 2016a). In contrast, the 2013 BHHRA assumed that half the tribal diet comes from resident fish in the lower Willamette River. EPA's tribal fish consumption scenario is based on a mixed diet that includes both resident and anadromous fish, which was done at the request of the tribes and consistent with the CRITFC study on tribal fish consumption (CRITFC 1994). In the evaluation of remedial actions, EPA noted that the exposure pathway that poses the greatest risk to human health is the fish consumption exposure pathway. Fish consumption is known to occur throughout the site. As such, parameters used to evaluate this pathway are considered carefully. EPA's risk assessment process evaluates cancer risk and noncancer hazards based on a reasonable maximum exposure consistent with EPA guidance that could occur at the site, and it is reasonable to look at a subsistence level of fish consumption at this site given the reserved tribal treaty rights and known amount of fishing that immigrant communities in Portland do. Additionally, the Polisar study (2016a) indicates that the level of consumption reflected in the survey may indicate suppression effects, and thus, the study does not reflect baseline (heritage) consumption rates. Reducing the consumption of resident species to 24.2% from 50% for the tribal diet likely resulted in underestimated risks for this receptor in the PDI risk update.

4. The PDI risk update assumed cooking loss factors equivalent to 10th percentile values for lipophilic contaminants in fish tissue to account for loss due to preparation and cooking of the fish before human consumption. The 2013 BHHRA did not make any adjustments to contaminant concentrations in raw fish tissue because of the uncertainties associated with accounting for specific preparation and cooking practices. As noted in the PDI risk update, PCBs tend to concentrate in fatty tissues, and trimming away fatty tissues, including the skin, may reduce the exposure to PCBs. Additionally, cooking can reduce the concentrations, depending on the method (Wilson et al. 1998). However, while several studies reported a decrease in concentrations, one study showed a net gain in PCB concentrations after cooking (EPA 2000). In addition, as noted in Section 5.2.6.2 of the 2013 BHHRA, "Consumption of other portions of the fish in addition to the fillet can result in greater relative exposure to PCBs and other persistent bioaccumulative chemicals and thus, greater relative risks. Using SMB data as an example, the increased risk associated with consumption of the entire fish could be as much as an order of magnitude greater than associated with consumption of fillet only." Given the potential variability in the impact of cooking methods and the possibility that some consumption practices make use of whole fish, EPA decided that possible reductions in PCB concentrations from cooking practices not be considered quantitatively. Instead, these cooking loss factors were assessed in the uncertainty section of the 2013 BHHRA for further consideration in risk management decisions and fish advisories.

5. The PDI risk update makes many references to interim and long-term risk management targets in the ROD (EPA 2017a). As stated in the ROD RS Section 2.18, interim targets for risks and hazard indices (HIs) were developed for FS purposes because a long-term model is not available to predict the time to meet the PRGs. The interim targets were used to evaluate each alternative's effectiveness in achieving cleanup goals in a reasonable time frame among other matrices. The interim targets were not intended to be a ceiling for how much risk reduction construction could or would achieve. Thus, while comparisons to interim targets for risks and HIs may be used to gauge remediation progress, comparison to these targets is not definitive. The focus is on determining long-term impacts with remaining risks capable of being achieved through MNR.
6. Sediment EPCs with the PDI/BL data are not directly comparable to those from the RI/FS due to fewer samples collected in the nearshore areas during the PDI/BL for some COCs and the collection of mostly randomly placed samples compared to the targeted RI/FS risk assessment samples. Table 3 in Appendix G that summarizes the PDI EPCs is misleading because it only lists four chemical groups for the calculation of risks and hazards from the in-water sediment. As noted in Table 3-4 of the BHHRA, these are not the only chemicals that were included in the calculation of risks and hazards in the risk assessment. Although Section 2 of Appendix G explains this shorter list by indicating that an additional screening was conducted with comparison to regional screening levels (RSLs) to determine the COCs to be used in the PDI risk update, the screening is flawed. If a separate screening is to be conducted, then it should not arbitrarily exclude chemicals based on screenings conducted in previous reports without revisiting the reasoning in the previous screenings to verify that the assumptions are applicable to the current evaluation. The PDI risk update excluded total petroleum hydrocarbons-diesel range organics (TPH-DRO) although current PDI/BL data for TPH-DRO exceeded the RSLs. In addition, arsenic was excluded as a COC from the PDI risk update. Since the PDI risk update calculation for sediment direct contact was only calculated for sitewide risks, the PDI Report masks the variations in risk and hazard values that are evident in the BHHRA calculations completed by river mile for in-water sediment (Table 5-43 of the BHHRA). By only presenting the sitewide risks, the PDI Report presents an insufficient picture of the risks and hazards for the site. Figures 4a and 4b in Appendix G are intended to present comparisons of the 2013 BHHRA to the PDI risk update results for risks and hazards for direct contact with sitewide nearshore surface sediment. When compared to the risks and hazards presented in the 2013 BHHRA for the tribal fisher scenario, this graph appears to misrepresent the study areawide risks presented in Table 5-21 (EPA 2016a). According to Table 5-21, the tribal fisher scenario cancer risk is 3E-5 and the noncancer hazard is 0.4. It is unclear in the notes on Figures 4a and 4b if the 2013 BHHRA values presented were altered to only include the COCs identified in the PDI risk update. Table 5-21 indicates that the COCs that are the primary contributors to the total study areawide hazards from in-water sediment exposure for the tribal fisher are vanadium (54%), 2,3,7,8-TCDD eq (27%), and perchlorate (14%). By not including vanadium and perchlorate in the PDI risk update calculations, the hazards calculated by the PDI Report may not present an accurate representation of the hazards at the site.
7. The PDI risk update only evaluated fish consumption risk at a sitewide scale. The BHHRA determined that a 1-river mile exposure unit was necessary to evaluate fish consumption risk for small home range species such as sculpin and SMB. The PDI fish tracking data

confirm the validity of the 1-mile home range for SMB and by extension the 1-mile exposure unit. Therefore, this exposure unit is necessary to understand site risks at smaller spatial scales. See Specific Comments #13 and #16 and Appendix D.7 Comments for further discussion.

Appendix H Comments

1. The suitability of EPA implementation of the mechanistic FWM developed by the LWG to derive PRGs and subsequent CULs was called into question by the LWG and LSS in 2016 as documented in the final FS dispute (AR Doc # 100036161). EPA confirmed the validity of the FWM and provided the basis for this determination in LWG Dispute Issue 1l and LSS Dispute Issue 9. Additionally, the sediment CULs selected in the ROD for total PCBs and four of the five dioxins/furans were based on a background evaluation independent of the FWM. The information presented in Appendix H does not provide substantive justification or evaluations with new data that invalidate the FWM or EPA's 2016 dispute decision. Furthermore, EPA performed a multiple linear regression between fish tissue concentrations, fish tissue lipid content, surface sediment concentrations, and sediment total organic carbon. This was done to evaluate the relationship between the bioaccumulative COCs total PCBs, DDx, and dioxins/furans in fish tissue and proximal surface sediment samples. Averaging distances for sediment samples were evaluated at radii ranging from 100 feet to 1 mile from the fish capture location on the same river side to assess the statistical level of significance for the sediment-tissue relationship at different spatial scales. For total PCBs, DDx, and the three focused COC dioxins/furans (2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; 2,3,4,7,8-PeCDF) there were statistically significant positive associations between fish tissue concentrations and sediment concentrations. The strongest statistical relationships were observed when sediment averaging distances were less than 500 feet. This confirms that concentrations of bioaccumulative COCs in fish tissue are associated with sediment contamination at relatively small spatial scales approximating SMB home ranges (i.e., less than one mile).
2. The procedural flaws described in Section 3.2.1 are inaccurate. As stated in Section 2.11 of the ROD RS, the FWM was created by the LWG to develop a predictive relationship between chemical concentrations in sediment, water, and tissue that can be used to derive PRGs (EPA 2017a). EPA noted that the FWM was appropriately developed, parametrized, calibrated, and applied to develop risk-based PRGs and, by extrapolation, PTW thresholds. EPA approved the model approach developed by the LWG and determined that the model could be used for decision-making at the site. Documentation of the FWM and its calibration, adapted wholly from the report(s) submitted to EPA by the LWG, is provided in Appendix B of the 2016 FS (EPA 2016b). EPA noted in the final FS dispute, LWG Dispute Issue 1l (AR Doc # 100036161), that the LWG submittals repeatedly asserted that the FWM performs well for COCs other than PCBs, is suitable for calculating PRGs in sediment that are beyond the range of observed concentrations, and performs well at varying spatial scales. EPA also noted that absent claims that information previously submitted to EPA was either erroneous or deliberately misleading, no additional information was provided that the FWM, as developed by the LWG, was not suitable to derive PRGs.
3. The conceptual model flaws described in Section 3.2.2 are incorrect and inadequately characterize the supporting information for the FWM in FS Appendix B and the ROD RS

(EPA 2016b, 2017a). Extensive scientific evidence exists for the bioaccumulation of the organic chemicals included in the FWM. While quantifying chemical exposure from multiple sources in aquatic systems can result in a degree of uncertainty, mechanistic modeling reduces this uncertainty by using larger spatial scale calibration (i.e., sitewide), testing the FWM at smaller spatial scales with spatially explicit accounting for sediment exposure uncertainty, and verification through application to multiple chemicals with varying spatial distributions and physical properties that affect bioaccumulation potential (Windward Environmental LLC [Windward] 2015). Additionally, the LWG concluded that because the FWM is mechanistic, the model is appropriate for extrapolating beyond the empirically observed conditions in Portland Harbor. As stated in the final FS dispute, LWG Dispute Issue 1l (AR Doc # 100036161), the assumption of steady-state conditions is addressed in Appendix C of the bioaccumulation modeling report, which states, "Because of a lack of adequate time-dependent data for the Portland Harbor Study Area, the model has been simplified to assume steady-state conditions for the purposes of this application" (Windward 2015). Appendix H also misquotes FS Appendix B regarding the development of dioxin/furan PRGs. Congener-specific dioxin/furan PRGs were developed for fish tissue and sediment (using the FWM) because, while a relationship was not observed between the sediment toxic equivalent (TEQ) and fish tissue TEQ, the FWM could be appropriately calibrated for the individual congeners. Lastly, the absence of a fate and transport model coupled with the FWM does not invalidate the FWM or conclusions drawn from it. The FWM includes surface water uptake as an exposure route and therefore incorporates the contribution of upstream contaminant loading via surface water.

4. The technical flaws described in Section 3.2.3 are incorrect or otherwise do not represent a fatal flaw in the FWM and inadequately characterize the supporting information provided in FS Appendix B and the ROD RS (EPA 2016b, 2017a). As stated in the final FS dispute, LSS Dispute Issue 9 (AR Doc # 100036161), EPA disagrees that surface water concentrations alone can result in tissue concentrations that could pose unacceptable risk. Such a statement indicates a fundamental lack of understanding of the mechanistic FWM. The FWM is capable of estimating tissue concentrations resulting from exposure through a variety of processes such as gill uptake and dietary exposure and accounts for transformation and elimination of contaminants through metabolic pathways. The complex interplay between potential contaminant sources is such that in some cases (e.g., in species that prey on planktivorous fish), a combination of dietary and gill uptake alone may result in a tissue concentration exceeding risk-based concentrations. The potential for this type of scenario to occur was recognized by the LWG when they presented sediment PRGs as <0 for several contaminants, indicating that a sediment concentration of 0 in conjunction with the input water concentration resulted in an estimated tissue concentration exceeding the risk-based target (Windward et al. 2009). Lastly, the FWM inputs, such as the octanol-water partitioning coefficients, were not selected arbitrarily or "cherry-picked" as suggested; rather, they were selected following the process outlined in Windward (2015) and FS Appendix B (EPA 2016b).

Appendix I Comments

1. The RALs for the focused COCs presented in Appendix I completely omit the RI/FS data, which is inconsistent with Section 14.2 of the ROD, and are therefore invalid. Section 14.2 of the ROD states that post-ROD sampling will be conducted to support remedial design and to

refine the CSM such that the design and constructed remedy will reflect the newer information (EPA 2017a). The ROD however does not state that newly collected remedial design data will be used to develop new RALs; rather, the SMA footprints presented in the ROD will be refined based on higher-density sampling performed during remedial design. The PDI/BL data are not of sufficient spatial density for remedial design but will be incorporated into future remedial design datasets.

2. EPA performed an independent RAL curve analysis with the PDI/BL and RI/FS data following the methodology described in FS Appendix D Section D1 (EPA 2016b). This was done to assess whether the relationship between concentration and area remediated have substantially changed since the RI/FS data were collected. RAL curves were developed for the PDI/BL data only and for the PDI/BL data combined with the RI/FS data. The updated RAL curves show that the RAL concentrations for the focused COCs selected in the ROD are still appropriate. The RAL curves for total PAHs, DDx, and 2,3,4,7,8-PeCDF indicate little change while those for 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD show an increase in the area requiring active remediation. The increases in these areas are likely due to the sitewide dioxin/furan samples collected during the PDI/BL while the RI/FS samples were limited in spatial extent. The area requiring active remediation for total PCBs appears to have decreased, likely due to source control efforts and natural recovery. However, the total PCBs Alternative F RAL is still located in the knee of the curve and will result in substantial risk reduction without experiencing diminishing returns.

3. Section 5 states, "Site recovery is particularly notable in areas of the Site outside of the ROD SMA" and is used without scientific justification as supporting evidence for the need to develop new RALs. Areas outside of the ROD SMAs (i.e., areas selected for MNR in the ROD) represent approximately 84% of the total site area and were selected for MNR due to lower contaminant concentrations and multiple additional lines of evidence that are summarized in FS Appendix D Section D8 (EPA 2016b). EPA's evaluations with the PDI/BL data have determined that most of the ROD surface sediment SMAs still have RAL exceedances; this is corroborated by Section 2.2 of the PDI Report, which states that the highest concentrations of the focused COCs in the PDI/BL data were in the ROD SMAs, thus, "indicating stability of the high concentration areas." Therefore, decreases of contaminant concentrations in the ROD MNR areas and stability of high concentrations in the ROD SMAs support the CSM in the ROD and the ROD RALs.

Appendix J Comments

1. The refined SMAs presented in Appendix J are inconsistent with the ROD as they apply RALs other than those in the ROD, do not consider PTW, only include surface sediment data, replace older data using invalid assumptions, and rely on FS-level evaluations inadequate for remedial design. However, for comparative purposes, EPA estimated the lateral extent of surface sediment SMAs using the PDI/BL data alone and in combination with the RI/FS data following the methodology described in FS Appendix C (EPA 2016b). These evaluations show that the SMAs developed with the PDI/BL alone and in combination with the RI/FS data were 362 acres and 375 acres, respectively, compared to the ROD SMAs of 365 acres. This suggests that the contaminated areas greater than RALs and/or PTW thresholds have not changed appreciably since the RI/FS data were collected. The SMAs presented in the

ROD will need to be further delineated during remedial design with higher spatial density samples in both the surface and subsurface.

2. EPA disagrees with the 100-foot replacement methodology described in Section 2.2 as documented in EPA's comments on the Pre-RD Group's footprint report (EPA 2019). See General Comment #2 and Appendix D.2 comments for further discussion on EPA's assessment of the Pre-RD Group's comparison of RI/FS data to PDI/BL data.
3. The updated sediment volumes presented in Section 3.2 provide no supporting information for why 3 and 5 feet below mudline were chosen as appropriate depths of contamination. Additionally, these depths differ from the estimated depths of contamination presented in ROD Sections 6.6.1 and 10.1.1.3, which were used to develop volumes of contaminated sediment (EPA 2017a). The estimated volumes of contaminated sediment in Section 3.2 are arbitrary and cannot be directly compared to the estimates in the ROD or proposed ESD.
4. EPA has been reviewing the datasets collected after the FS database was finalized and approving them for unqualified use at the site. Geosyntec (2016) and Kleinfelder (2015) have not been approved for unqualified use by EPA, and any evaluations performed and conclusions drawn using these data cannot be corroborated by EPA. Furthermore, if an error is present in the Kleinfelder (2015) sediment sampling data report, as suggested in Table 2, a formal errata memorandum needs to be submitted to EPA for consideration, detailing the error with supporting documentation that substantiates any errors and corrective actions needed. Field photographs provided in Appendix D of Kleinfelder (2015) appear to show that samples were only collected from the top 10 cm of the grab sampler as discussed in Section 3.5 of the report.

Appendix K Comments

1. The ROD describes several types of caps (both passive and active) used in the FS evaluations and indicates that the use of reactive amendments may be needed for some caps, but the ROD does not require the use of amendments (EPA 2017a). The cap modeling discussed in Appendix D of the FS was intended to evaluate if contaminants at PTW levels can be contained using a reactive cap similar to the caps at other sediment Superfund sites over a range of seepage velocities observed at Portland Harbor (EPA 2016b). It was not meant to be an exhaustive modeling evaluation of all possible cap designs. Cap design details such as use of a reactive amendment, type of amendment, and isolation layer thickness are to be determined by performing parties during remedial design based on area-specific seepage velocities, initial concentrations, organic carbon content, and other design parameters.
2. Incorporating the impacts of groundwater source controls on seepage velocities is not prudent at this stage. Groundwater control systems can fail, require temporary shutdowns, and may not run in perpetuity. Caps are generally designed to contain contaminant flux under worst case scenarios, regardless of groundwater control systems. Using a reduced seepage velocity of 11 cm/year to account for hypothetical groundwater source controls is not conservative, and such a determination would have to be made based on area-specific measured seepage rates.
3. The seepage velocity of 24 cm/year used by the Pre-RD Group is at the lower end of velocities observed at the site. The median seepage velocity in the Portland Harbor RI is 0.5

cm/day (182.5 cm/year). While a seepage velocity of 24 cm/year may be observed in certain capping areas, it is not a good estimate for all Portland Harbor caps. EPA will require area-specific measured seepage rates to be used in cap design modeling by each performing party.

4. The organic carbon partitioning coefficient, Koc, used in the EPA modeling was selected for the tetrachlorobiphenyl homolog group as clearly stated in the FS Appendix D. The Pre-RD Group is using a total PCB Koc value provided in the Oregon DEQ chemical database, which is more consistent with pentachlorobiphenyls (DEQ 2015). EPA understands that a range of Koc values is available for different PCB homolog groups and different congeners; however, because tetrachlorobiphenyls such as PCB 77 have higher toxicity, it is more appropriate to use the lower Koc value as a conservative modeling input.

5. The initial porewater concentration, seepage velocities, and partitioning coefficients are a few of the most sensitive inputs for CapSim modeling (Lampert et al. 2018). Considering the issues with the seepage velocities and Koc value outlined above, EPA disagrees with the Pre-RD Group's conclusion that unamended caps will be sufficient for the entire site. While this may be the case for certain areas, EPA would require area-specific seepage velocities, initial concentrations, organic carbon content, and other design parameters to be used for cap design modeling to determine the required thicknesses and amendment requirements for caps. Each design party will have the ability to design caps that are most appropriate for their project area and that may or may not require the use of amendments.

6. It is unclear how the modeling results support the need for "maintaining the ROD remedial action level (RAL) of 1,000 μg/kg in the navigation channel without downward adjustment of the RAL to accommodate the ROD PTW criterion of 200 μg/kg." The PTW threshold for PCBs was based on a 10^{-3} cancer risk and does not hinge on the ability of caps to reliably contain higher contamination levels. The ROD requires removal of all sediment with concentrations exceeding PTW thresholds, and caps can only be used to contain PTW if it is present below the feasible depth limit of excavation technology.

Appendix L Comments

1. The ROD TADT (Appendix I Figure 28) is intended to be used along with the general design requirements provided in Section 14.2 of the ROD to determine what remedial technologies should be implemented based on the most recent design data. As stated in ROD RS Section 2.21.2, the TADT is purposely less prescriptive and is intended to provide a flexible framework to be followed during remedial design (EPA 2017a). Revisions to the TADT are not necessary because area-specific design level data collected by parties performing remedial design will govern final technology application decisions and the flexibility inherent in the TADT allows for area-specific decisions to be made during remedial design.

2. The purpose of enhanced natural recovery (ENR) is to accelerate the natural recovery process by adding a thin layer of clean sand. The Pre-RD Group is proposing the use of ENR in areas requiring active remediation, which contradicts the ROD TADT (Appendix I Figure 28). ENR is only meant to be used in areas greater than CULs but less than RALs to aid in natural recovery.

3. The ROD requires removal of sediment with concentrations exceeding RAL or PTW thresholds to the feasible depth limit of excavation technology, as approved by EPA, in the

Navigation Channel. The Pre-RD Group's recommendation for capping without dredging in this area is inconsistent with this ROD requirement. Additionally, the recommendation to require dredging only in areas with PTW NRC contradicts the ROD requirement to remove all non-aqueous phase liquid (NAPL) and PTW that cannot be reliably contained where technologically feasible.

4. Backfilling is required by the ROD in areas where impacts to the floodway and habitat considerations necessitate it. Area-specific flood rise modeling and habitat considerations will be accounted for during remedial design to determine the need for backfilling in dredged areas.
5. Placement of a residual layer is required in all dredging areas to isolate any dredge residuals and is common practice at sediment remediation sites. The Pre-RD Group recommends that a residual layer should only be required where needed but does not provide a decision process for determining what "where needed" means or any rationale or examples of areas where it may not be needed.
6. Appendix L consistently uses the phrase "mobile NAPL" to refer to PTW-NAPL. Note that the ROD does not distinguish mobile vs. non-mobile NAPL, and all PTW-NAPL and all NAPL are considered source material that require dredging and/or capping.

Appendix M Comments

1. LTM at sediment Superfund sites relies on trends in data collected at multiple time steps over extended periods. The ROD Table 17 COCs were selected based on extensive human health and ecological risk assessments. Excluding a subset of chemicals that are known to pose risks to human health and the environment based on a single sampling event is not reasonable, especially considering that surface sediment, surface water, and fish tissue data cannot account for future risks due to groundwater flux and other exposure pathways.

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