Responses to USEPA Comments on Baseline Ecological Risk Assessment for Operable Unit 4 - August 15, 2015
Anniston PCB Site, Anniston, Alabama

No	USEPA Comment	EPA Category	Comment Response
1	An explanation of why homolog analysis was done in some tissue types while Aroclors were measured in others (a lower detection limit [DL] can be obbtained analyzing homologs) needs to be added to the BERA.	Aroclors versus Homolog	The following text was added to Section 3.4, 3rd paragraph, 3rd sentence: "As described in the OU-4 Phase 2 FSP (Arcadis 2009), PCBs were analyzed for PCB homolog groups using modified (SIM) USEPA Method 8270C in many cases to achieve adequate detection limits in tissues where PCB concentrations may be low or the tissue mass may have been limited.,. PCBs in fish, molluscs and crayfish were analyzed as Aroclors and PCBs in the remaining tissue types were analyzed as homologs."
2	Ten percent (%) of the sediment and soil samples collected during the Phase 2 sampling were submitted for homolog analysis. Please add these data to the Appendix E spreadsheet.	Aroclors versus Homolog	Homolog data for the soil sediment and tissue samples are included in the appropriate tabs of the data appendix (E) to the BERA.
3	The total Aroclor polychlorinated biphenyl (PCB) concentrations tended to be lower than the total homologue PCB concentrations when the samples were analyzed by both methods. When total PCBs is different by the two methods, there is uncertainty in which value best represents the total PCB concentration for risk assessment purposes. The text of the uncertainty section should be modified so that it does not say that the samples of tissue that measured total PCB homologues overestimated the total PCB concentrations (see the Region 4 PCB Guidance [Wischkaemper et al. 2013]). Discussion of uncertainty regarding analysis of PCBs (weathered PCBs are not always picked up by Aroclor methods) and the uncertainty regarding total PCB Aroclors versus total PCB homologues should be added to Section 6.2.1.1. The table from the Sampling and Analysis Plan (SAP) that explains the analytical methods used for the different types of samples should be included. The methodology should be discussed. If possible, obtain a complete description of how the homolog analysis was conducted from the laboratory and add this to the BERA report.	Aroclors versus Homolog	The way in which total Aroclor concentrations relate to total homolog concentrations is addressed in the uncertainty analysis in Section 6.2.1.1. The following text was added to Section 6.2.1.1. "For PCBs, COPCs were measured as both Aroclors and homologs. Based on the ability of the Aroclor method to accurately characterize the quantity of PCBs, and the large amount of Aroclor data available for multiple media across OU-4 (and the Site), the data collection programs and the BERA generally used total PCBs as the sum of Aroclor results. As discussed in the response to Comment No. 1, PCBs were analyzed as homologs in some cases to achieve adequate detection limits with the available sample volumes. Specifically, surface water, fish tissue, crayfish tissue, and mollusc tissue were analyzed as homologs. Approximately 10% of soil and sediment samples were also analyzed as both homologs and Aroclors. To evaluate the uncertainty associated with the different measurement methods, paired data were evaluated for 106 soil samples, 16 sediment samples, nine surface water samples, and 25 tissue samples (including fish, crayfish and molluscs). A strong

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			correlation was found between the Aroclor and homolog results (Figure 6-21) with r^2 values ranging from 0.88 for soil to 0.99 for tissue. A regression was not run on the surface water samples because all Aroclor results were ND. The slopes of the regression line ranged from 0.77 for tissue to 0.83 for both soils and sediment indicating that the Aroclor method slightly overestimates PCB concentrations and that use of the Aroclor data may overestimate risks."
4	AVS-SEM data were collected on the sediments used in the toxicity testing. AVS-SEM results should be included in the interpretation of the toxicity testing. Results from these analyses would have aided in the interpretation of sediment chromium and lead concentrations and would have been supportive of the mercury (Hg) results. This comment is referring to Section 5.1.2 on sediment screening for metals and whether metals are contaminants of potential concern (COPCs).	Acid Volatile Sulfide/Simultaneously Extracted Metals (AVS/SEM)	The AVS-SEM data are included as Attachment 1 to the revised Appendix C of the BERA and are summarized in Table C-4 of Appendix C of the BERA. The AVS-SEM results are discussed in Section 3.1 of Appendix C, and in Sections 5.1, 6.1.2.1.1, and 6.2.3.3 of the BERA relative to metals bioavailability, toxicity and the COPC analysis. As noted in the Ingersoll et al. (2014) report, "Calculation of values of SEM minus AVS (SEM-AVS, molar basis), and SEM-AVS normalized to the fraction of sediment organic carbon (SEM-AVS/fOC; appendix 1, table 1-7) indicate low bioavailability and low probability of risk from these five metals (USEPA 2005)." The technical conclusion reached in the Ingersoll et al. report regarding the low probability of risk from these five metals is consistent with the evaluation and conclusions presented in the BERA.

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5	The concentrations of PCBs in sediment should be evaluated both on a dry-weight basis and on an organic-carbon normalized basis when comparing with the toxicity test effect concentrations from the Ingersoll et al. (2014) report. Normalizing concentrations to organic carbon will provide a better estimate of the bioavailability of PCBs to benthic invertebrates in finer-grained particles within each of the 10 reach sites, and will refine the exposure scenario and risk determination for benthic invertebrates.	Benthic Macroinvertebrates (BMI), Evaluation of Risk	Both organic carbon normalized and non-normalized PCB effects concentrations were calculated based on the toxicity endpoints and were provided in Appendix C of the draft BERA. They remain in the revised Appendix C (Table C-7) to the BERA. In addition, a point-by-point comparison based on both organic carbon normalized and non-normalized PCB concentrations is provided in the benthic invertebrate risk characterization in the revised BERA (Section 6.1.2; Table 6-4a). The following text was added to the end of the 2nd paragraph on p. 5-7 in Section 5.1.1. "The comparable EC values are also reported on an OC basis. The OC-normalized values introduce more uncertainty into the evaluation, as OC is generally low in most OU-4 sediments (and often not detected) with an average content of 1.1 % in the 268 surface sediment samples and with a standard deviation of 1.0. Given the low bias of the OC data, it is unlikely that OC is significantly influencing the bioavailability of PCBs in the aquatic system. While the OC-normalized results provide a datum, they are not useful for evaluating exposure-based risk due to the low OC concentrations and the high level of uncertainty in the results."
6	Risk to benthic invertebrates should be re-evaluated on a point-by-point basis, and the risk conclusions should be based on the number of samples within an Exposure Unit (EU) (Reach or Assessment Area) that exceed the toxicity threshold for the most sensitive endpoint, not all measured endpoints. Risk categories should be defined (low risk > EC0* and <EC10*, moderate risk > EC10* and < EC20*, and high risk > EC20*) and the number of samples within each EU that fall into these risk categories should be tabulated.	BMI, Evaluation of Risk	The benthic invertebrate risk characterization has been updated to include a point-by-point analysis by reach as requested (Section 6.1.2; Table 6-4a). The low, moderate and high risk categories are consistent with what was provided in the draft BERA and have been carried forward in the revised BERA.

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7	Another line of evidence (LOE) to evaluate potential risk to the benthic invertebrate community is to compare the measured tissue concentrations to tissue residue values reported in the literature. The no observed effect concentration (NOEC) tissue toxicity reference value (TRV) reported for PCBs by van Geest et al. (2011) was 0.59 milligrams per kilogram (mg/kg) wet weight, and the lowest observed effect concentration (LOEC) was 1.4 mg/kg wet weight. Measured concentrations in benthic invertebrates collected in the Upper Assessment Area (UAA), Middle Assessment Area (MAA) and Lower Assessment Area (LAA) and measured concentrations in *Lumbriculus* used in the site-specific bioaccumulation assays should be compared with the tissue-based TRVs. Concentrations of PCBs measured in mollusks collected on-site should also be compared to the tissue residue-based TRVs.	BMI, Evaluation of Risk	This reported "LOEC" is questionable and was not suggested by the authors. Van Geest et al. observed reduced survival (64% vs. >80%) of Hexagenia spp. (mayfly nymph) in sediment containing 4.8 ppm PCBs and where tissue accumulation was 1.4 mg/kg wet weight. These sediments contained elevated concentrations of other contaminants (PAHs, dioxins) that may have contributed to toxicity. In addition, the authors noted that the nymphs had difficulty burrowing into these particular sediments due to their "course physical structure which likely stressed the organisms". Alternate values based on tissue residue effects data have been employed as CTC values for benthic invertebrates. For benthic invertebrates, NOAEL and LOAEL CTC values of 28.4 and 76 mg/kg wet weight were selected (Borgmann et al. 1990, Nebeker and Puglisi 1974). Receptor-specific CTCs were also developed for oligochaetes, emergent insects, molluscs, and crayfish, as well as for terrestrial invertebrate receptors. Derivation of the CTCs is described in Appendix D. For *Lumbriculus*, site-specific worm bioaccumulation data were not compared to CTCs because the bioaccumulation tests were conducted with sediments that do not represent surface exposure in OU-4. The laboratory testing of *Lumbriculus* was conducted using sediments with a range of PCB and OC concentrations to evaluate bioaccumulation rates for aquatic invertebrates. These sediments were mostly mined from subsurface sediments and then were processed to remove all materials that were retained on a 2 millimeter sieve.

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8	For the benthic community line of evidence, the Hg tissue residues in benthic invertebrates should be compared with the tissue residue benchmarks for Hg from Mendez-Fernandez et al. (2015).	BMI, Evaluation of Risk	The suggested tissue residue benchmarks from Mendez-Fernandez et al. (2015) are not appropriate for mercury because the exposure sediments were from mine sites and contained high concentrations of many other metals that likely contributed to the observed toxicity. Similar to PCBs, receptor-specific CTCs were developed for mercury to assess risk to benthic invertebrates. For benthic invertebrates, NOAEL and LOAEL tissue effects benchmarks (10 and 16 mg/kg ww, respectively) were based on data from Borgmann et al. (1993). CTC values were also developed for emergent insects, molluscs, and crayfish, as described in Appendix D.
9	Section 6.1.2.4, Measurement Endpoint (ME) 2d: Evaluate the benthic community data on an assessment area basis. The hazard quotient (HQ) calculations suggest risk is elevated in the upper reaches and lower in the lower reaches.	Benthic Community Surveys	As requested, the benthic invertebrate community data have been evaluated on an assessment area basis in the revised BERA (section 5.6.2 and Tables 5-8 and 5-9). Both ANOVA and ANCOVA analyses have been provided to evaluate any differences relative to habitat types, seasonality and PCB concentrations in OU-4, relative to the reference areas.
10	The Shannon-Weaver diversity index equation on page 5-28 is incorrect. The logarithm should be replaced with the natural logarithm, and recalculated throughout the Operable Unit (OU)-4 BERA as needed.	Benthic Community Surveys	The equation was corrected in the text in Section 5.6. The calculations were performed correctly using the natural logarithm.
11	Table 5-5 needs to provide variance estimates for each of the mean values reported for each of the five subunits sampled within each of the 10 reach sites. Reach should be included as an additional factor in the analysis of variance (ANOVA).	Benthic Community Surveys	The benthic community data were evaluated on an assessment area basis as suggested in Comment #9 (see Section 5.6.2 and Tables 5-8 and 5-9). Both mean and standard deviation are provided in the revised table (Table 5-8).

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12	Section 6.1.2.4: The summary of benthic invertebrate community measures should not combine samples collected in different habitats and different seasons. One of the most significant metrics, abundance, is lower in OU-4 than reference, which suggests effects (mean abundance is 2% lower at OU-4 sample locations if all samples are combined. When adjusted for habitat and year, abundance is 39% lower at OU-4 sample locations than reference locations [footnote 1, Table 5-5]). The benthic invertebrate data are inconclusive. They do not definitively show an effect, but they also do not show the lack of an effect. The study lacks the power to conclude that there was or was not a difference attributable to the contamination – COPCs were not measured in the benthic community survey samples. Revise the text so that it does not state "there appear to be no adverse effects of PCBs or other COPCs on the benthic community." In the October 2008 EPA comments on the Phase 2 Field Sampling Plan (FSP), EPA noted that the samples for the benthic invertebrate community evaluation should be collected from the grab samples submitted for sediment chemistry and toxicity evaluation. The deficiency in the community data relative to the SAP should be acknowledged in the uncertainty section.	Benthic Community Surveys	An error was found in the analysis as presented in the draft BERA. The ANOVA results were misinterpreted and it should have been stated that mean abundance was higher in OU-4 compared to reference. The ANOVA analysis of the benthic community data was revised to include assessment area as a dependent variable as suggested in Comment #9 and to compare each assessment area to reference (Section 5.6, Tables 5-8 and 5-9). Differences related to habitat were appropriately controlled in the analysis as before by having habitat as a fixed effect in the model. The fall and spring data sets were evaluated separately. A power calculation is provided in the revised BERA indicating acceptable power (70%) to detect a 20% change in a benthic measure with 95% confidence. The benthic community sampling was not designed to evaluate associations with PCB concentrations since it was determined that the PCB concentrations did not vary enough or have a large enough range to evaluate such an association. Mean PCB concentrations in the three benthic sampling areas are not statistically significantly different. Therefore, we disagree that the data are deficient. However, in order to evaluate potential relationships between community measures and PCBs, each benthic community sampling location was paired with the mean PCB concentration of surface sediment samples within a half mile range of the location. An ANCOVA analysis was added to the BERA to determine if there is an association with PCB concentration while controlling for habitat and season (Section 5.6). The results of these analyses found no adverse effects of PCBs or other COPCs in OU-4 therefore the statement "there appear to be no adverse effects of PCBs or other COPCs on the benthic invertebrate community" has not been removed from the BERA.

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13	Section 6.2.4 needs to include a statement regarding how lack of replication associated with the invertebrate sampling contributes to uncertainty associated with spatial distributions of the community survey data within the five subunits sampled within each of the 10 reach sites.	Benthic Community Surveys	The following text has been added to Section 6.2.4, 5th paragraph: "In contrast, for the benthic invertebrates, sample size is high and adequate with 70% power to detect 20% or larger reductions in means (n = 158). While the sample size was adequate for statistical analysis, the benthic invertebrate sampling locations within each AA were limited in spatial extent which adds uncertainty to the interpretation. However, given that there is little variation in PCB concentrations throughout the AAs, it is unlikely that benthic communities are impacted elsewhere in the AAs outside the sampled areas."
14	Section 6.2.4 needs to include a statement regarding how use of different sampling methods in the different habitat types contributes to the uncertainty associated with the benthic community analysis and conclusions. For data to be valid and comparable between areas, they all must be sampled using the same methods and with the same amount of effort.	Benthic Community Surveys	The nature of benthic invertebrate habitats and organisms require different sampling methods be used and the appropriate methods were used in each habitat for this sampling program. The benthic community analysis included habitat as a fixed effect in the model to control for the variability within habitats and due to sampling methods. The following text has been added to the revised BERA in Section 6.2.4 5th paragraph: "In addition, different sampling methods were used in each benthic habitat type due to the nature of the invertebrates expected to be found within each habitat. The sampling methods were selected to be specific to the habitat types and were consistent between the assessment areas to minimize uncertainties associated with sample collection activities. The potential for these uncertainties were accounted for in the data evaluation process by adding habitat as a dependent variable."

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15	Section 6.3.3: Provide additional detail on the mean Ephemeroptera, Plecoptera, Trichoptera (EPT) index between reference and OU-4 (which differed by 18%). Were more sensitive species missing in the upper reaches (where contamination is higher), relative to lower reaches?	Benthic Community Surveys	To provide a more thorough understanding of EPT composition in the benthic sampling areas another benthic metric, percent EPT (of total abundance), was added to the benthic community analysis presented in section 5.6.2 and comparisons were made by AA. The following text has been added to Section 6.3.3 of the revised BERA: "The EPT index and percent EPT were very low in backwater and depositional locations in both OU-4 and reference areas. For other habitats, the EPT index was lower in UAA compared to reference in the fall and spring (riffle only) however the abundance of EPT taxa as a percent of total abundance was equal or greater in all AAs compared to reference. "
16	The data used for bioaccumulation factor development were not co-located. Use the data groupings agreed upon by the BAF Work Group; these are provided in Tables 1 and 2, and illustrated in the set of figures (File name “BERA Ecological Sample Groups PCBs.pdf”) to revise the Appendix B tables. Include a new section describing the data groupings and how they were developed. Refer to the data groupings instead of calling the data co-located. All references to co-located soil or sediment samples should be removed from the text.	Bioaccumulation Factors (BAFs) - Data to be utilized to calculate BAFs	The groupings shown on the figures are not completely consistent with the groupings presented in the Excel tables provided. As agreed in the workgroup, the 2007 and 2009 data that are proximal to the biological sampling areas are used as the soil or sediment data for the BAF recalculation, as presented in EPA Tables 1 and 2. Two samples (EUT-08-C3N-22 and EUT-08-C3N-23) are located across the creek from the EUT-08 BSA and were not included in the data grouping. Additionally, ELT-02-38 is located nearest to ELT-02 BSA and was included as part of C7 South, not C7 North, as noted by EPA. Text (Section 4.4.1.3) and Tables 4-18 and 4-19 were added to Section 4 and Appendix B to clarify how data were grouped.
17	The 2007 and 2009 sediment data should be utilized in the BAF calculations. Reference area data should be included in the regressions, but not in the median BAF calculations. It is noted that all reference sediment was non-detect (ND) for Aroclors.	BAFs - Data to be utilized to calculate BAFs	As described in Section 4.4.1.3 and Appendix B, the 2007 and 2009 data are used for the soil and sediment-based BAF calculations (soil, sediment, and biotic tissues). Reference data are included in the regressions when detected (i.e., for mercury but not for PCBs).

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18	The *Lumbriculus* data will only be used as benthic invertebrate tissue data. Mercury was not analyzed in the *Lumbriculus* samples, only PCBs. Tables 4-1 & 4-2 on exposure parameters should add a row for benthic invertebrate tissue for incorporation into the receptor diets.	BAFs – *Lumbriculus*	The exposure parameter tables (4-1 a and b and 4-2 a, b and c) were updated as requested and consistent with Tables 4 and 5 provided with the EPA comments.
19	Bioaccumulation factors for Hg bioaccumulation into *Lumbriculus* should be taken from those reported for the South River in Gilmour et al. (2013). The BAF for mercury bioaccumulation into *Lumbriculus* in dry weight for the South River was 1 in this study.	BAFs – *Lumbriculus*	Because there are no site-specific *Lumbriculus* data for mercury, the requested value of 1 was incorporated into the revised BERA, as presented in Section 4.4.1.4.
20	For the site-specific *Lumbriculus* PCB data, good regressions can be obtained several ways. The best regression is for wet weight and lipid normalized worm data versus Total Organic Carbon (TOC) normalized total PCBs as homologs in sediment; this regression model should be used to predict benthic invertebrate PCB concentrations. BAFs for *Lumbriculus* should not be calculated as worm tissue concentration divided by soil concentration (method currently in the BERA),	BAFs – *Lumbriculus*	All regressions for the laboratory bioaccumulation study were highly significant. The Aroclor sediment data were used rather than the homolog data to be more consistent with the other BAFs employed in the revised BERA. Additionally, the regression based on log-transformed worm wet weight and sediment Aroclors was selected to eliminate the need for an assumption regarding TOC content in sediment, which is highly variable in OU-4. The r^2 value for the log-transformed normalized homolog data was 0.93 and the r^2 value for the selected regression was 0.89. The selected regression was used in the Draft BERA and not a worm tissue divided by soil to as stated in Comment No. 20.
21	In Table 6-19, the laboratory BAFs should be considered to have low conservatism to overestimate risk (not moderate) and high confidence of risk prediction (not moderate). The *Lumbriculus* bioaccumulation data was analyzed by the high resolution gas chromatograph/mass spectroscopy (GC/MS) method versus the low-resolution GC/MS method that was used for the benthic invertebrate (Odonata) analysis. Additionally, the pairing of sediment to tissue is quite precise. It is of higher quality and was collected to serve as the standard against which all Phase 2 data was to be compared.	BAFs – *Lumbriculus*	This table is now Table 6-25. The ranking of each HQ line of evidence factors in all components of the HQ calculation (not just the BAF). As such, the level of conservatism for the sandpiper based on the *Lumbriculus* uptake model and the high sensitivity TRV was kept at high and confidence was changed to moderate because the high sensitivity TRV is likely not a good representation of a toxicity threshold for the sandpiper (or other water birds). The conservatism for the sandpiper based on the mid-sensitivity TRV and *Lumbriculus* BAF is moderate and the confidence is moderate because the HQ scenario

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			using measured tissue rather than modeled tissue was given higher weight overall.
22	To develop BAFs, EPA prefers using site-specific regressions over using the median BAF. There are significant regressions for PCBs to earthworms, small mammals and *Lumbriculus*. The method described in the next comment should be used when a significant regression cannot be developed.	Calculation of BAFs	Consistent with the approach used in the Draft BERA, significant positive regressions were employed. For PCBs, regressions are employed for *Lumbriculus*, earthworms, and small mammals. For mercury, regressions are employed for odonates (i.e., field benthic inverts), crayfish, snakes, frogs, and earthworms.
23	When a significant regression cannot be developed, calculate the BAFs as follows:	Calculation of BAFs	The requested approach was generally followed. The specific approach employed is outlined below, as described in Section 4.4.1 and Appendix B. Based on the general nature of Comment #22, it was assumed that this comment applies to both soil- and sediment-based BAFs.
	· A mean sediment concentration should be calculated for each biological sampling area (BSA). Non-detects should be included in the mean sediment concentration calculations. The mean sediment concentration should be calculated using ProUCL.		Sediment and soil nondetects were included when calculating the mean for BSAs. The Kaplan-Meier mean estimated by ProUCL was used. Nondetects were only excluded for soil and sediment when estimating regressions.
	· An individual BAF should be calculated for each tissue sample. Non-detected tissue samples should be excluded from the mean. Please include the non-detected tissue samples in the Appendix B tables, clearly marking them as non-detects; <u>but exclude non-detects when calculating the mean.</u>		The calculation of individual tissue ratios precludes the need for calculating a mean for tissue. BAF ratios were calculated for detected tissue concentrations only.

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	· Calculate a median BAF for each ~~(~~BSA~~)~~.		A median BAF for each BSA and each AA was calculated and each is presented in Table 4-21 and Appendix B Table B-26. However, because each reach (or EU) does not contain a BSA, it is not possible to use a BSA-specific BAF for HQ calculations in all EUs. Specifically, there are measured data for five of the 10 aquatic EUs and in seven of 19 terrestrial EUs. In addition, the number of tissue values for each BSA was three at most (except for fish), and when nondetect values are excluded for BAF calculation, this sample size is smaller in most cases. Thus, for consistency in approach for modeled and measured tissue estimates (as requested in comments 39 and 48), AA-specific BAF estimates are employed. For the modeled exposures, the soil or sediment EPCs are EU-specific and are used in the food web calculations with the AA-specific BAFs. For the AA-specific BAFs, the median of the individual BAF ratios for all BSAs within an AA was selected. HQs are also presented on an overall AA basis.
	· Calculate a median BAF for each assessment area (AA).		
	· Calculate summary statistics representing the variation around the mean.		Because median BAFs were requested, it is not clear what mean is being referred to here. To address this comment, variation around the median was estimated as presented in Table B-26.
	EPA is recommending this approach for the following reasons: · 1) Knowing the variation in your BAF term allows you to evaluate the uncertainty associated with your BAF estimate. 2) Knowing the variation in tissue term will also support an evaluation of whether you are seeing a broad or narrow accumulation range at a particular sediment concentration. 3) Knowing the variability in the calculated BAFs will help understand the uncertainty associated with tissue concentration estimates in the areas that lack measured tissue concentrations.		Comment noted. Uncertainty in the BAF term is also addressed by comparing the measured tissue-based HQs to those calculated using modeled tissue concentrations.

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24	To account for bioaccumulation in low-concentration areas, the percentage of NDs can be qualitatively evaluated. It can be noted that bioavailability cannot be numerically described accurately due to NDs, but that bioavailability is low.	Calculation of BAFs	These circumstances (i.e., for grasshoppers) are discussed in the uncertainty analysis in Section 6.2.2.1 2nd to last paragraph. The following text was added regarding grasshoppers. "For terrestrial insects (i.e., grasshoppers), PCBs were never detected in grasshopper tissue; therefore, a BAF of 0.17 derived from non-worm terrestrial insects along the Kalamazoo River (Blankenship et al. 2005) was selected for use in the modeled dietary HQ calculations. This literature-based BAF likely overestimates uptake to terrestrial insects such as grasshoppers and crickets, in which PCBs were not detected in OU-4 samples."
25	For PCBs the tissue data should be lipid-normalized and the sediments or soils should be TOC normalized. Mercury and other metals data should not be normalized. Please check the Lipid-normalized concentrations in Table B-27.	Calculation of BAFs	A detailed regression analysis was conducted using dry weight, wet weight, and lipid-normalized tissue as well as OC-normalized soil or sediment (Appendix B Tables B-9 through B-12 for sediment and Tables B-18 through B-21 for soil). In most cases, the lipid- and OC-normalized data resulted in a similar or worse regression fit than for non-normalized data (except for forage and predator fish for which a negative uptake relationship was found). Thus, the non-normalized data were used in BAF estimation to eliminate the need for a static assumption regarding TOC in sediment across an entire EU, which is quite variable. Section 4.4.1 summarizes the rationale for the selection of each model and BAF employed in the BERA. More detail is provided in Appendix B.

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26	For terrestrial biota, BSA-based BAFs should be calculated. If there are fewer than five BSAs with detects in tissues, and/or there is no spatial pattern to the detections, then the 95% upper confidence limit (UCL) for OU-4 overall should be calculated.	Calculation of BAFs	Rather than exclude BAFs based on these uncertainties, all receptor dietary doses are estimated based on modeled prey tissue estimates (i.e., using BAFs or regressions) as well as measured tissue concentrations (see also response to Comment 114). In a few cases, USEPA recommended a literature BAF due to low detections or lack of site-specific data (see responses to Comments #19, 32, and 33). For consistency with the HQs based on measured tissue (using AA-specific EPCs) and the aquatic modeled tissue HQs (using AA BAFs; see response to Comment No. 23), AA BAFs for the terrestrial prey tissues were used for PCBs and mercury, as described in Section 4.4.1.3
27	To calculate a BAF for toxic equivalents (TEQs), a BAF will be calculated for each congener, and a toxic equivalency factor (TEF) will be applied to the estimated concentration for that congener. Exposure estimated with the TEF-concentrations should then be summed to obtain the PCB TEQ and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) TEQ based exposure estimates.	Calculation of BAFs	The BAF calculation approach selected for other COPCs was used to calculate site-wide median BAFs for each congener, as data allowed. In the absence of sufficient data, site wide average ratio BAFs were calculated. If congeners were never detected in tissue, no BAF was selected. PCDD/PCDF and DL-PCB BAFs are presented in Appendix F Attachment 1. See also responses to Comments #23 and 165. The revised BERA includes the requested approach recognizing that while it's interesting to evaluate how each individual congener moves through the environment, congener-specific BAFs are not needed for risk assessment purposes and complicate an already complicated exposure model. Risks and hazards are not evaluated on an individual congener basis but rather on a TEQ basis. The important factor isn't how a particular congener moves from sediment to fish but how much dioxin equivalency is in the sediment and moves up the food web.

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28	Separate BAFs will be calculated for reptiles and for amphibians. Arcadis will go back to the dietary composition percentages in U.S. EPA (1993), and select the proportion of reptiles and amphibians for each receptor (this information is available for most of the receptor species).	Calculation of BAFs	BAFs for snakes and frogs are estimated individually based on the site-specific measured sediment and tissue, as described in Section 4.4.1.4 and Appendix B. The portion of each receptor diet comprising reptiles and amphibians are considered individually in the exposure models, as shown on Tables 4-1a through 4-2b. The diet composition is taken from Direct Comment Tables 4 and 5 provided by USEPA.
29	For emergent insects, a sediment concentration will be used to calculate the BAF for crane flies. A BAF for emergent insects will be calculated on an assessment area basis. The BAF will not be weighted by the percent of crane fly-only samples versus the total number of emergent insect samples collected.	Calculation of BAFs	The AA-specific BAFs (as outlined in response to Comment No. 23), result in crane-fly-specific BAFs for a portion of the UAA (two of the three sampling areas contained crane flies exclusively). These two samples resulted in tissue concentrations more than one order of magnitude greater than other emergent samples. Using the median BAF for the UAA results in a BAF value that is based on the crane fly tissue and sediment. The crane fly BAF is not consistent with other tissue/BAFs for emergent insects seen within OU-4 or at other sites. However, no crane fly-specific BAFs were located for direct comparison. The uncertainty associated with these two elevated tissue samples is discussed in Section 6.2.2.1.
30	Because terrestrial insects collected on-site did not include spiders or detritivorous insects, the OU-4 earthworm PCB and Hg data should be used as representative of all terrestrial invertebrate prey. Analyses provided to Arcadis on the BAF Work Group conference calls support the use of this approach.	Calculation of BAFs	The earthworm BAF regression was used for spiders (or other detritivores), as described in Section 4.4.1.5 and Appendix B. For the portion of the receptor diet that is specifically crickets or grasshoppers, site-specific data were employed. Using earthworms as a surrogate for grasshoppers is not warranted when OU-4 grasshopper data exist.

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31	Small mammal PCB concentrations should be predicted using the site-specific regression model. A comparison of OU-4 small mammal uptake with literature-reported uptake values should be provided. The degree of uncertainty in the mammal regression equation should not be deemed low just because a positive correlation was observed. Revise the text to remove characterization of the uncertainty in the BAF for small mammals as low.	Calculation of BAFs	The small mammal regression was used in the BERA to predict small mammal tissue concentrations in the food chain model. The following text and analysis was added to Section 6.2.2.1. "For small mammals, there is some uncertainty regarding the range of PCB concentrations included in the dataset. To evaluate this uncertainty, the OU-4 model was compared to a model based on data from the Kalamazoo River. While a regression could not be developed for the Kalamazoo River dataset (Blankenship et al., 2005), the static BAF based on average soil concentration and average tissue concentrations was estimated for small mammals. The static BAF ratio based on the same assumptions used in the OU-4 BERA (i.e., combined shrews and small mammals using an assumption of 32% solids) is 0.314. This BAF predicts mammal tissue concentrations approximately 2 to 4 times lower than those predicted by the OU-4 regression. As such, the mammal regression employed is considered adequately protective and to have relatively lower uncertainty compared to the ratio BAFs for which no relationship was found."
32	The OU-4 plant PCB data is too limited to develop a regression. All of the OU-4 plant data fall within the 95% prediction interval for the Ficko et al. (2013) plant data. The Ficko et al. (2013) model should be used to estimate plant PCB concentrations.	Calculation of BAFs	This BAF is used as requested for terrestrial plants in addition to the use of the measured data. Literature values were also requested for mercury for *Lumbriculus* (response to Comment 19) and plants (response to Comment #33). A description of selected BAFs for PCBs and mercury is provided in Section 4.4.1.4 and 4.4.1.5.
33	Mercury accumulation in terrestrial plants should be predicted using the BAF of 0.05 after Pant et al. (2011).	Calculation of BAFs	See response to Comment #32.

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34	To address EPA's concern regarding quantifying the variation in the BAF term with respect to the uncertainty in the BAF, add a confidence interval around the BAF. Formulas from Snedecor and Cochran (1980) were provided for the ratio of means BAF and a paper by Clarke and van Gorder (2013) was provided for the confidence intervals on the geometric mean regression BAF. The confidence intervals described above are for BAFs having significant regressions. The regression BAFs are developed from x, y pairs of ProUCL-derived means by location group. They differ from the average BAFs developed for situations where the regression is non-significant in that the latter uses itemized tissue data.	Calculation of BAFs	Confidence intervals around the regressions (when significant regressions were found) are provided in Appendix B using standard methods for ordinary least square regression (and not geometric mean regression). For ratio BAFs, confidence intervals around the selected median values are also provided.
35	The dietary exposure models currently use exposure point concentrations (EPCs) for abiotic media calculated using all available data. Run the risk characterizations on the 2007 and 2009 data separately from all the data to highlight recent conditions.	Dietary Exposure Model Calculations	See response to Comment #113.
36	For non-Hg metals, measured 95UCL tissue concentration forward risk calculations should be done, using a site-wide 95UCL tissue concentration estimate.	Dietary Exposure Model Calculations	Tissue EPCs for non-mercury metals are estimated based on the lower of the Site-wide 95% UCL and the Site-wide maximum, as described in Section 4.2.4 and presented in Tables 4-15 and 4-16.
37	For mercury, risk characterization should take into account the proportions of methylmercury in the prey. This can be done one of two ways: 1) evaluate estimated exposures using both methylmercury (MeHg) and inorganic Hg TRVs either based on the assumption that each comprises 100 % of the mercury present in a sample. Literature data for % MeHg and inorganic Hg in different prey species can be used to discuss implications of using this approach in the uncertainty section, or 2) by estimating % of each form present within a matrix using information provided by EPA (Table 3).	Dietary Exposure Model Calculations	This evaluation is not included in the main component of the BERA risk calculations. Section 6.2.3.9 describes the uncertainty associated with the assumption of 100% methyl mercury used in the BERA and provides an example calculation for the Carolina Wren to demonstrate the impact of this uncertainty. Tables 6-22 and 6-23 provide the inputs for the calculations.
38	For the forward risk calculations using measured tissue data, NDs should be included when calculating 95UCL tissue concentrations using ProUCL.	Dietary Exposure Model Calculations	NDs are included in all UCL calculations.

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39	For aquatic biota, because there are issues with both the BAF models (models tend to under-predict measured tissue concentrations) and with the available tissue data (several reaches lack measured tissue samples), the dietary exposure models will be calculated two ways:	Dietary Exposure Model Calculations	The calculation of modeled exposure (using BAFs) and measured exposure (using 95UCL tissue estimates) is included, as described in Section 4.4.1.
	1) Measured 95UCL tissue concentrations (calculated on an assessment area basis) forward risk calculations will be done for each reach using Reach-specific 95UCL abiotic EPCs.		Tissue UCLs are calculated on an AA basis and employed in conjunction with soil or sediment EPCs that are estimated on a reach-specific (i.e., aquatic EU) basis.
	2) Assessment area-specific BAFs will be used to estimate prey tissue concentrations for Reaches where biota were not collected.		AA-specific BAFs are estimated for PCBs and mercury for all tissue types. As described in response to Comment #23, HQs are calculated two ways in the BERA: 1) based on modeled exposure (using BAFs) and 2) based on measured prey tissue. Both the BAFs and the tissue EPCs are estimated on an AA-specific basis.
	3) When tissue concentrations are estimated for reaches that lack site-specific tissue data, the BAF should be applied to the 95%UCL of the abiotic media concentration in that reach.		As discussed in response to Comment #154, an SWAC-based UCL is employed as the sediment EPC, as described in Section 4.2.1.
	4) When presenting final conclusions in Section 8, distinguish the reaches with modeled predictions from the reaches with measured data.		As described above and in response to Comment #23, HQs are estimated based on both modeled and measured prey tissue estimates in the revised BERA.
40	Revise Table 4-13 so that it is clearer. Although there is no riparian area in Reaches 1 and 10, the risk calculations should be done on an assessment area basis. There is no need to adjust dietary percentages for riparian-feeding receptors in the UAA and the LAA. Scenario 1 should be added for sandpiper and mink.	Dietary Exposure Model Calculations	Risk estimated for Sandpiper 1 and mink 1 HQs were added (Tables 6-7a and 6-7b). The following text was added to the end of Section 4.3. "Note that for reaches C1 and C10, riparian habitat is not present. Risk to wildlife receptors exposed to riparian media in the food web models was evaluated using the UAA-wide riparian soil EPC for reach C1, and using the LAA-wide riparian soil EPC for reach C10. Additionally, metals were not analyzed in floodplain soil in reach C1. UAA-wide soil EPCs were used for metals to estimate risk associated with metals in reach C1. Uncertainties related to these assumptions

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			are discussed in Section 6.2.1.1." Table 4-13 was updated (now Table 4-17) to improve clarity.
41	The equation for deriving concentrations in eggs needs to be provided. From Office of Environmental Health Hazard Assessment (OEHHA) (2015) page 5-12, Equation 5.3.4.2: Cfa = (Inhalation + Water ingestion + Feed ingestion + Pasture/Grazing ingestion + Soil ingestion) * Tco . 1) Cfa = Average concentration in farm animals and their products (micrograms per kilogram [µg/kg]) ; 2) Inhalation, water ingestion, etc. = Dose through inhalation, water ingestion, etc. (micrograms per day [µg/d]) ; 3) Tco = Chemical-specific transfer coefficient of contaminant from diet to animal product (d/kg)	Bird Eggs	Equations were added to the document in Section 6 of Appendix B.
42	OEHHA (2015) also has TCos for polychlorinated dibenzo-p-dioxin/polychlorinated dibenzofurans (PCDDs/PCDFs) and 9 inorganics, including Hg and lead (Pb). Estimated egg concentrations of these COPCs should be evaluated.	Bird Eggs	Egg concentrations were estimated mercury and HQs based on egg residues were included in the Draft BERA using the OEHHA TCos. However, use of these TCos requires a dose estimate. The estimated dose is then multiplied by the TCo to estimate bird egg concentrations, in effect applying a second BAF (and associated uncertainty) to calculate the bird egg HQ. For PCDD/PCDFs, the congener-specific BAFs have a high degree of uncertainty due to the limited OU-4 PCDD/PCDF dataset. Due to this uncertainty in the BAFs and use of an additional uptake factor (i.e., TCo), the egg-based HQs have greater uncertainty than dietary HQs estimated using only the soil-to-biota BAFs. Egg-based HQs were not calculated in the TEQ assessment because they do not provide value in evaluating the potential TEQ risk to birds in OU-4. Thus, only the dietary analysis is included in the BERA for PCDD/PCDF. A lead TRV based on egg residues is not readily available in the literature; therefore, egg-based HQs for lead were not included in the revisions because this analysis would only serve to increase uncertainty compared to the LOEs already evaluated.

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43	Section 6.1.5.1, ME 5a and 5c: This should be 3 separate lines of evidence: 1) Estimated dietary exposure vs TRVs; 2) Estimated egg concentrations vs TRVs; 3) Estimated OU4 egg concentrations vs reference. Lines 2 and 3 are not discussed in the aquatic bird summaries pages 6-21 to 6-24, and are not shown on Table 6-12.	Bird Eggs	HQs for bird eggs are presented in Tables 6-9 and 6-10. In text summary tables referred to in this comment have been removed from the document. Results of the egg-based analysis are included in the text in Section 6.1.5.1 and egg-based analysis is included in the WOE summary table (Table 6-25). Egg concentrations vs reference for PCBs cannot be estimated because PCBs are ND in reference soil and sediment.
44	The quality and utility of the community survey data is highly dependent upon how they were collected. Unless field observations contain metrics that can be definitively associated with contaminant exposure, they should only have low weight in the weight of evidence; none of the community surveys included information on COPC concentrations at the survey sample locations. There is insufficient information presented in this BERA report regarding experimental design and statistical analysis methods used (e.g., type of experimental design, defining the experimental unit, randomness, sample unit independence, adequate spatial and/or temporal replication, power analysis) to evaluate whether conclusions based on the community surveys can be supported. The uncertainties associated with the statistical approach taken to evaluate the data, particularly in situations where the OU-4 site was considered as a single exposure unit, relative to reference, merit further discussion. Interpretation of differences (or lack of differences) between site and reference field survey data requires extensive evaluation and documentation of a wide range of potentially confounding factors that may affect the data (e.g., different site histories, underlying geology and soil types, land management practices, presence and potential influence of other stressors or other factors that may act to improve/impair quality of ecological measurements, etc.). Field data are highly variable spatially and temporally. For these data to be useful, they must be collected over an	Community Surveys	The uncertainty text in Section 6.2.4 address the following key uncertainties. "Key sources of uncertainty include (1) adequacy of sample size per habitat type which affects statistical power to detect differences, (2) similarity of reference locations to OU-4 locations, (3) seasonal or annual variability, and (4) detectability." Uncertainties associated with each of these are discussed in detail. The WOE confidence ranking for all community metrics except for benthic invertebrates has been changed to low. Benthic invertebrates are considered moderate and this is discussed in more detail in response to Comment #12 through 15.

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	adequately large area, with sufficient replication, and over a sufficient time frame. The description of the field data presented in Section 2.1 suggests that all of the field data are of a rather short duration - a couple of seasons over at most a couple of years - and have limited replication and sample sizes. As a consequence, these data are not definitive measures of possible effects but rather supportive at best. Comparison of biological community metrics between OU-4 and the reference habitats should not be considered a primary and robust line of evidence for any assessment endpoint, or be identified as the line of evidence with the highest degree of confidence.		
45	The BERA needs to specifically identify which community surveys were not evaluated relative to reference areas (plants, aquatic birds and mammals), and note that this is a significant limitation for these receptors.	Community Surveys	A new table (5-3) was added to Section 5 to clarify which analyses were conducted for each community data type.
46	All of the measurement endpoints related to COPC body burdens (ME1b, ME2b, ME3b, ME4b, ME6b) should state "comparison of **measured** concentrations of COPCs in organisms collected on-site with literature-derived tissue-based toxicity values", not comparison of estimated tissue concentrations with concentrations in organisms from the reference area. Use of the site-specific bioaccumulation factor BAF to estimate tissue concentrations underestimates risk, increases the uncertainty associated with the risk estimate, and decreases the level of confidence in the risk conclusions; HQs should have been calculated using the actual measured tissue concentrations. When comparing the measured concentrations in biota to tissue-residue TRV, use the 95% UCL on the tissue concentrations as the EPC. Alternatively, tissue concentrations can be compared on a point-by-point basis and reported as frequency of exceedance.	Critical Tissue Concentrations (CTCs)	Although previous versions of the MEs (i.e., in the BPF) included comparisons to critical tissue residues (CTRs [referred to in USEPA comments as critical tissue concentrations or CTCs]), the initial BERA MEs did not include comparison to CTRs for most receptors. This revision was based on discussions with USEPA and a desire to limit CTCs to those COPCs and tissues for which a consensus value was available from multiple studies in the literature. For most tissues, these toxicity data are lacking; therefore, the CTC ME text was removed. As requested in this comment, the revised BERA includes comparisons of measured tissue 95 UCLs to CTCs for PCBs and mercury (when values are available) and the ME text has been reinstated (Table 2-16). CTC comparisons are not included for non-mercury metals, except for lead and vanadium in fish, for which ample data were available. The AE and ME table (2-16) and text were updated to reflect this change. As described in response to Comment #23

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			and #39, dietary HQs are presented based on both modeled and measured prey tissue estimates.
47	Measurement endpoints ME1b, ME2b, ME3b, ME4b, and ME6b should be evaluated quantitatively. Figures 6-1 to 6-9 show the OU-4 data compared with the reference data (based on average concentrations). Present a statistical analysis (literature-based CTC versus LAA, MAA, UAA and reference).	CTCs	ME Xb was revised to include both a comparison to reference and a comparison to CTCs. Tissue-based HQs were calculated for all tissue for which sufficient toxicity data were identified. CTC comparisons are evaluated statistically by comparing the 95 UCL tissue EPC for an AA to the selected CTC value (this is the equivalent of a one-sample T-test). Results are presented in Table 6-11 and are discussed in the risk characterization. Figures 6-4 through 6-9 present box plots of reference and site data (by AA) for soil, sediment and biotic tissue for PCBs and mercury. Box plots for non-mercury metals in soil and sediment are provided in Figures 6-10 through 6-14. Statistical comparison is provided where adequate sample size was available (Tables 6-13 through 6-15). For non-mercury metals in tissue and PCDD/PCDFs in all media, sample size precluded statistical comparisons. Mean site and reference concentrations are compared qualitatively and show on Figures 6-15 through 6-20.

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48	The dietary exposure models should be re-run using measured tissue concentrations from reaches where data is available. For aquatic exposures, the risk calculations should be done on an assessment area basis.	Dietary Exposure Models	See responses to Comments #23 and #39. Both modeled and measured exposure estimates are provided on an EU and AA basis (using AA-specific tissue EPCs and BAFs in conjunction with EU-specific soil or sediment EPCs for each EU).
49	Results of dietary exposure models run with measured tissue concentrations should be compared with the models run using the estimated tissue concentrations; this will provide a measure of the uncertainty associated with the estimated tissue concentrations.	Dietary Exposure Models	See response to Comment #48.
50	Dietary exposure models for receptors that include *Lumbriculus* as a portion of the diet need to be clearly discussed in the text and presented on the appropriate Tables.	Dietary Exposure Models	The receptors that include *Lumbriculus* (i.e., benthic invertebrates) in diet are the sandpiper and raccoon. Table 4-17 and corresponding text in Section 4.5.2 were updated accordingly. In addition, the classification in Tables 4-1a, 4-1b, 4-2a, and 4-2b to distinguish between different types of invertebrates in receptor diet (see responses to Comments #108 and #110) provides clarity on this point.
51	Section 3.0: There needs to be a comprehensive data inventory for all site-related data. This should include a table (or series of tables) that clearly summarizes the number of samples from each reach for each medium (abiotic media and biota) and the analytes that were measured. In addition, there needs to be a table or tables that summarize the number and types of field community surveys that were conducted, and where and when they were performed.	Editorial Comments – Must be addressed	New tables 3-1 through 3-3 have been added to summarize the data available for each EU and AA for each COPC. The specific data employed in BAF calculation (which is different from the site-wide dataset) is presented in Appendix B. The available community data are summarized in Section 2 and the collection locations are presented on the Figure 2-1 series. Text was added to Section 5.6 to clarify which community data were used in the quantitative evaluation of this LOE. Table 5-3 was also added to summarize this information.
52	Tables 3-1 to 3-9, F-1: Summary statistics should be calculated by analyte, biota type, and area so that variation in concentrations among areas and biota can be readily evaluated. Graphical presentation as boxplots would also be helpful. Please revise.	Editorial Comments – Must be addressed	Summary statistics for each media type and COPC are provided in Tables 4-4 through 4-16. Box plots of PCB and mercury concentrations by AA and including reference are provided in Figures 6-4 through 6-9. Additional details for summary statistics for all media, COPCs, and exposure areas (site wide, AA, and EU)

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			are provided in Appendix E. See also response to comment No. 47.
53	Table 4-14: Need to include the regression for sediment to *Lumbriculus*.	Editorial Comments – Must be addressed	This table is now 4-20 and the selected regression was added.
54	Table 5-2b: Tissue TRVs are presented in dry weight but the fish tissue data in Tables 3-1 to 3-3 are in wet weight, so they are not comparable. Present the tissue TRVs on a wet weight basis.	Editorial Comments – Must be addressed	While most of the tissue CTCs are based on wet weight data reported in the literature, the dietary models are based on dry weight tissue EPCs and TRVs. As requested, both wet weight and dry weight tissue CTCs are presented in Table 5-2b.
55	Figures 5-2 and 5-3 should be presented as bar charts as opposed to line diagrams. Lines imply an association between measures - site and reference habitats are independent measurements.	Editorial Comments – Must be addressed	Figures 5-2 and 5-3 have been converted to a bar chart format.
56	Table 6-12: Add rows to show the egg TRV comparison results where LOEC HQs are > 1.	Editorial Comments – Must be addressed	Table 6-12 and similar in-text HQ summary tables for wildlife were replaced in the revised BERA to incorporate HQs for both modeled and measured tissues. Egg-based HQs are also included in the revised tables (Tables 6-9 and 6-10a for birds)
57	Table 6-12: There should be two rows for the spotted sandpiper, Sandpiper - 2 and Sandpiper (alternate BAF).	Editorial Comments – Must be addressed	Based on the requested revisions to receptor exposure scenarios provided in USEPA Directed Comment Tables 4 and 5, the alt BAF scenario is no longer included. Rather, sandpiper-1 includes a diet of 50% terrestrial worms and 50% aquatic invertebrates (estimated based on the field collected odonate tissue) and sandpiper-2 includes a diet of 50% aquatic invertebrates (based on odonate tissue) and 50% *Lumbriculus* (based on lab data). These dietary compositions are described in Tables 4-1a,b and in Section 4.5.2.1

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58	Table 6-18: Add a row with the site-specific sediment toxicity thresholds for survival, growth, and reproduction; these should all be classified as exhibiting a low level of conservatism to overestimate population risk and a high confidence in risk prediction.	Editorial Comments – Must be addressed	This table is now 6-24. Consistent with the analysis conducted in the BERA, the table contains rows for survival endpoints and reproduction endpoints. The EC values for the most conservative result in each category provide the basis of the conclusions. The survival endpoint is ranked as low conservatism and the reproduction endpoint is ranked as moderate conservatism because of the variability in the control response (i.e., the actual effect threshold could be much higher). Both LOE are given an overall confidence of moderate/high. Reproductive endpoints were ranked as higher conservatism than survival due to less certainty in population-level effects and high variability in the test results. No changes were made for reproduction ranks. Growth endpoints were removed.
59	Table 6-18: Add ME2b to this table (comparison of measured COCs in tissues to literature-based TRVs).	Editorial Comments – Must be addressed	See response to Comment #46.
60	Table 6-18: Based on the uncertainties associated with the community data, Table 6-18 should be revised to state that the community evaluation exhibits a high “level of conservatism to overestimate population risk” and low “confidence in risk prediction”.	Editorial Comments – Must be addressed	This table is now 6-24. For the benthic community, no rank of conservatism is applied because assumptions were not made. The WOE ranking is thus, based on the underlying uncertainty associated with the evaluation. For the benthic community, based on the 158 observations, this uncertainty is considered moderate and the overall confidence in the WOE conclusions is considered moderate.
61	Figures 6-1 through 6-9: These data would be much more interpretable if error bars and sample sizes were noted for each bar. Ideally, these data would be presented as a series of side-by-side boxplots so that the full distributions, including outliers, is displayed.	Editorial Comments – Must be addressed	For PCBs and mercury, these figures were revised and data are presented as box plots, now numbered 6-4 through 6-9. Box plots for non-mercury metals in soil and sediment are provided in Figures 6-10 through 6-14. For non-mercury metals in tissue and PCDD/PCDFs in all media, mean site and reference concentrations are compared qualitatively and show on Figures 6-15 through 6-20. Sample sizes are shown on all figures.

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62	Tables B-44 through B-46 should show all small mammal data and show the non-detected results in red ink.	Editorial Comments – Must be addressed	The complete summary of the data with NDs in red is provided in Tables B-15 and B-16 for PCBs and mercury, respectively. These are consistent with the tables provided for all other tissue types. Based on the requested changes to the BAF approaches, all of the BAF tables included in Appendix B were revised.
63	Tables B-36, B-37: It appears that B-flagged data was typically included in the tables as detected values. Confirm all B-flagged data are correctly indicated. Please discuss the uncertainty related to B-flagged values and how these were treated in the BAF calculations.	Editorial Comments – Must be addressed	B-flagged data are considered detected concentrations. All B-flagged data are for inorganic COPCs. The B flag in this case indicates an estimated value between the instrument detection limit and the reporting limit (RL). These values were included in the BAF calculations. Appendix E provides a sample qualifier key. Values indicated as ND in the referenced tables were re-verified.
64	Tables B-36, B 37: Some J-flagged small mammal data are shown in red ink. These results need to be checked to confirm they are the result, not reporting limit (RL)/2.	Editorial Comments – Must be addressed	Tables were updated appropriately to show ND values in red at the full RL.
65	On Table B-5, for the UAA, the sediment concentrations shown are for the LAA, while the fish tissue concentrations appear to be for the UAA. This comment applies to Table B-6 as well.	Editorial Comments – Must be addressed	This table was corrected. This issue did not affect the BAF calculations.
66	Table F-1: Present the dioxin-like (DL)-PCB TEQ along with the total and PCDD/PCDF TEQs.	Editorial Comments – Must be addressed	The DL-PCB TEQ was added.
67	An additional Table with summary data (mean, range, sample size, number of detects) for all PCDD PCDF and DL-PCBs needs to be provided. The number of samples cited on page F-20 do not match the number of samples in Appendix E. Also, the mean TEQ values reported in Table F-1 cannot be replicated using either the surface sediment or soil data, or the surface data from samples near the BSAs.	Editorial Comments – Must be addressed	Summary statistics for all constituents (including PCDD/DF and DL-PCB data) were added to Appendix F.

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68	The site-specific toxicity thresholds are reported to several significant digits in Table C-1, but are reported to no decimal places elsewhere (e.g., page 5-7, Table 5-2a). Toxicity thresholds need to be reported to the appropriate significant figures (e.g., 1 vs. 1.38 mg PCB/kg for the EC0* for *H. azteca* survival-normalized reproduction, 7 vs. 6.8 mg PCB/kg for the EC10* for *C. dilutus* emergence). Toxicity thresholds and HQs should be reported to two decimal places.	Editorial Comments – Must be addressed	Toxicity thresholds are shown to one decimal place when less than 10 and greater than 0.1. Values less than 0.1 are shown at one significant figure and values 10 or greater are shown as whole numbers. Results are presented as number of exceedances of individual samples rather than HQs.
69	Section 6.2: The uncertainty section goes to great lengths to state that the whole assessment is conservative and that almost all of the uncertainties serve to overestimate potential risks. The actual data and available information do not support this assertion. Text should be modified to more accurately reflect that the assessment is not necessarily conservative.	Editorial Comments – Must be addressed	While it is true that in many cases, uncertainties were mitigated by making conservative assumptions that would likely result in overestimation of exposure or risk, the uncertainty analysis was modified to clarify when the uncertainty could result in either over or underestimation of exposure or risk.
70	Page ES-7: States “Only one wild species with high sensitivity to PCBs, European starling (*Sturnis vulgaris*), is known to be present within OU-4.” There are three high sensitivity species present within OU-4, the starling, ruby-throated hummingbird (Table 2-4 and 2-11), and the gray catbird (Table 2-11) (Farmahin et al. 2013).	Editorial	The text in the executive summary has been modified as follows: "Three wild species with high sensitivity to PCBs, European starling (*Sturnis vulgaris*), ruby-throated hummingbird (*Archilochus colubris*), and the gray catbird (*Dumetella carolinensis*), have been observed within OU-4."
71	Global: Capitalize the 'l' in *Lumbriculus*.	Editorial	The requested change was made.
72	Section 2.3: For the dietary exposure models, HQs represent a modeled exposure estimate compared to a literature-based toxicity value. For all other HQ calculations, the HQ is a comparison of measured COPC concentrations to literature-derived benchmarks.	Editorial	Based on the modifications to the BERA that include dietary HQs for both measured and modeled tissues, the comment is no longer relevant and no changes to the BERA were necessary.
73	Table 2-3: Expand this table to provide details on fish species and tolerance.	Editorial	A column characterizing species as (I) intolerant, (M) moderately tolerant, or (T) tolerant was added to Table 2-3.
74	Table 2-5: The text describes bobcat tracks observed at the reference location. No bobcat on Table 2-5.	Editorial	Bobcat were observed in Reference Area 2 (Cheaha Creek) in spring 2007. Table 2-5 reports observations during the aquatic habitat surveys. The bobcat is noted on Table 2-12 which reports the terrestrial

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			habitat surveys and as a result, no changes were necessary.
75	Table 2-16: What is footnote 1 for?	Editorial	This footnote was removed.
76	Section 3.0: States "Data inputs needed in this OU-4 BERA to evaluate the assessment endpoints (AEs) and MEs identified in Section 2 include abiotic (i.e., surface sediment, water, and soil) and biotic (i.e., prey tissue) media COPC concentrations." Revise as: Data inputs needed in this OU-4 BERA to evaluate the AEs and MEs identified in Section 2 include abiotic (i.e., surface sediment, water, and soil), biotic (i.e., prey tissue) media COPC concentrations, and site-specific toxicity data. Secondary lines of evidence include community and population metrics data.	Editorial	Section 3 text was revised as follows: "Data inputs needed in this OU-4 BERA to evaluate the AEs and MEs identified in Section 2 include abiotic (i.e., surface sediment, water, and soil) and biotic (i.e., prey tissue) media COPC concentrations, and site-specific toxicity tests and community analysis. Secondary lines of evidence include community and population metrics data. "
77	Page 3-11: Indicate the plant tissues that were collected.	Editorial	There were 24 species collected in total across OU-4 and the reference areas. It is not feasible to list all 24 in text. Specific plants collected in each sample can be seen in Table 4 in Appendix E.
78	Section 4: Explain the methodology used to generate the OU-4 floodplain boundary and the rationale for not utilizing the Federal Emergency Management Agency's (FEMA's) 100-year floodplain, particularly because PCBs have been measured at elevated concentrations outside the boundaries of the study area used in the OU-4 BERA.	Editorial	The following text was added to Section 3.2 "The FEMA 100-year floodplain was not used exclusively for the project as it does not incorporate a significant amount of relevant data that have been collected for the Site. The hydraulic flow model that was used by FEMA to establish the 100-year floodplain was used as an initial starting point and was supplemented with site-specific data for Choccolocco Creek and the associated floodplain. These data included surveyed elevations for more than 100 creek-channel transects located along Choccolocco Creek and USGS Digital Elevation models for the floodplain. Other improvements to the FEMA model included a review of the aerial imagery to adjust roughness coefficients, and site photographs were used to improve information at the bridge cross sections. The development of the 100-year floodplain using the FEMA model is described in Appendix D of the Off-Site RCRA Facility Investigation (RFI) Report (BBL

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			2000). Subsequent to this effort, the floodplain was adjusted in 2010 using updated Digital Elevation models from the USGS and the surface water elevations from the HEC-RAS surface water flow model developed during the Off-Site RFI program. The overall process to update the floodplain footprint and the resulting minor adjustments are described in Attachment A of the Phase 3 Field Sampling Plan for Operable Unit 4 (Arcadis 2010b). The process of updating the 100-year floodplain in 2010 resulted in the minor expansion of several areas."
79	Section 4-4, Equation 2: The equation does not match the parameters in Tables 4-1 and 4-2. Soil ingestion in the tables is reported as a percent of diet whereas in the equation it is a normalized kg/kg/d rate (normalized ingestion $rate_{soil}$ [NIRs]). Need to explain the conversion and make the terms consistent. Either drop the dietary $fraction_{soil}$ (DFs) term because NIR_s already addresses soil ingestion, or NIR_s should be NIR_f to accurately represent scaling of food ingestion for the percent of soil ingested. It should be noted that food ingestion in this model is not a function of which food type but rather of all food. Scaling of food ingestion occurs as a function of both the COPC_k and FR_k terms. NIR should not really have a 'k' subscript, but rather an 'f' subscript to represent food. Define NIRw.	Editorial	The equation presented in Section 4.4 was updated to be consistent with the exposure parameters tables (Table 4-1a,b and 4-2a,b).
80	Please apply the comments on exposure parameters presented in Tables 4-1a, 4-1b, 4-2a, and 4-2b to the exposure parameters presented in these Appendix F Tables 2a, 2b, 3a and 3b.	Editorial	The exposure parameters evaluated in the BERA are consistent for all COPCs, including PCDD/PCDFs in Appendix F. Appendix F exposure parameter tables (F-4a through F-5b) are consistent with Section 4 exposure parameters tables (4-1a through 4-2b).
81	Section 4.5.2, Page 4-25: Drinking water ingestion rates (IRs) were actually calculated using Calder and Braun (1983).	Editorial	Section 4.5.2 discusses dietary composition. The drinking water ingestion methodology is discussed in Section 4.5.3 with the appropriate reference and thus no updates to the BERA were necessary.

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82	Tables 4-1a and 4-1b: All cited home ranges should be converted to the same unit (acres).	Editorial	The home ranges are presented differently at the request of USEPA to account for those receptors that have linear home ranges.
83	Tables 4-7, 4-8: Please add acreage of each subgroup be added to this table.	Editorial	The acreage was added to the text in Section 4.1.1 in the bullets that describe the reaches and EUs.
84	Figures 4-2a through 4-2k show the PCBs measured in soil and sediment in the 10 reaches. A similar set of figures for mercury measured in soil and sediment should be included for visualizing the distribution of mercury.	Editorial	Figures showing mercury concentrations were added to Section 4 (a new Figure 4-3 series presented in the same format as the Figure 4-2 series for PCBs).
85	Section 5.5.5: Cites "Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations (Thompson 1996)". This citation should be updated to the more recent version of this book (Shore et al. [2011] in Beyer and Meador 2011).	Editorial	This citation was updated.
86	Table 5-2b: Why were Eco-SSLs for lead not used?	Editorial	The Efroymson values were selected because they are more protective. The screening tables were revised to use the EcoSSL values for terrestrial plants and invertebrates as requested.
87	Page 6-9: States "For these metals, concentrations in the OU-4 Areas are generally similar to or lower than in the ERAs, suggesting that metals detected in plant tissue along Choccolocco Creek are not different from reference areas." For chromium, mean OU4 levels are four to almost seven times greater than reference. Vanadium is also greater in OU4 than in reference. The text should be revised.	Editorial	Sections 6.1.1.2, 6.1.2.2.2, 6.1.3.2.2, 6.1.4.2.2, 6.1.5.1.2 and 6.1.6.1.2 include a detailed description of how OU-4 and reference conditions compare for each COPC. While based on small datasets, these text sections discuss when OU-4 concentrations appear to be higher than reference conditions.
88	Pages 5-11, 6-82: The Hansen, Schimmel and Foster conference proceedings study of the sheepshead minnow should be Hansen et al. 1973 instead of 1974. Please correct the reference.	Editorial	This citation was updated.

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89	Page 6-43: States, "Overall, the [herptile] tissue PCB data collected from the OU-4 Areas indicate concentrations that are relatively higher than the ERAs. …This result is also similar to the relative PCB concentrations in small mammal tissue, and might be expected based on the relatively high mobility of these receptors." Tissue concentrations in small mammal tissue in the OU-4 areas were twice the tissue concentration in reference small mammals. Tissue concentrations in herptiles were 25 to 40 times higher in the OU-4 area than the reference area. The text should be amended accordingly.	Editorial	The text has been modified as requested to indicate the magnitude of the difference between site tissues and reference tissues. Section 6.1.6.1.2 reads "There are no statistically significant differences between mean small mammal tissue concentrations within the MAA and LAA (Table 6-15), but mean tissue PCB concentrations in the ERA were about one-half the tissue concentrations in the OU-4 AAs." Section 6.1.7.2 reads "Mean reptile and amphibian tissue concentrations in the ERA are significantly lower (30 to 170 times) than in the OU-4 AAs."
90	Page 6-47: Exposure of terrestrial wildlife receptors to mercury and other metals in Reach C-1 was not evaluated, as metals were not analyzed in soil samples from this reach. Tissue samples were not collected in Reach C-1, either (Table 4-3, BERA report). The uncertainty associated with the lack of site-specific data, and risk calculations for this area, should be highlighted.	Editorial	The following additional uncertainty text has been added to Section 6.2.1.1 to address this issue: "Data for mercury and other metals in soil are not available for the Snow Creek reach (C1). In this BERA, exposure to metals in this area is uncertain. For non-mercury metal COPCs in OU-4, concentrations are generally similar to reference areas and/or, there is no upstream to downstream trend, and the metals are not associated with historical Site use. Therefore, Site-related risk due to non-mercury metals is not anticipated in Snow Creek. The lack of non-mercury metals data in Snow Creek is not considered to significantly affect risk conclusions for this OU-4 BERA. For mercury, the lack of soil data in reach C1 represents an uncertainty. In sediment however, sediment mercury concentrations in C1 are similar to those for the Choccolocco Creek AAs (Figure 6-7), and therefore it is reasonable to assume that soil mercury concentrations in the UAA (and associated ecological exposures and risk) would also be similar to those in reach C1."

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91	Page 6-75, Equation 9: All input values used in Equation 9 need to be specified and supported.	Editorial	Definitions were added or clarified for all parameters in Equation 9 and the associated text of the BERA (Section 6.2.3.7) has been clarified.
92	Page 6-95: Remove the italicized part of the conclusion: "Results of the ecological community assessments indicate that, in general, OU-4 conditions are similar to reference*, and therefore, there appear to be no adverse effects of PCBs or other COPCs within OU-4"*. There are no associated media chemistry data to support any statements regarding effects of COPCs on community structure.	Editorial	This statement was revised as follows: "Based on this analysis, although there is variability in the benthic community for the two years evaluated, overall results suggest no adverse effect of PCBs or other COPCs on the benthic invertebrate community and overall health within OU-4 can be considered comparable to reference areas. While sediment chemistry is not specifically co-located with the community assessment locations, the fact that the sediment PCB concentrations are higher in OU-4 than in reference locations suggests that PCBs are not impacting the community. There is moderate level of confidence in this line of evidence because although it was based on 158 site-specific observations, the survey was spatially and temporally limited in scope."
93	Page 6-106, first bullet: Although high risk was predicted for carnivorous birds in Reach C1, the confidence in the prediction was moderate because cooper's hawks and barred owls were observed at the site. Text should be revised to remove statements linking moderate risks to observations of birds, because there is limited power in the observational data to detect whether there was potential impact to the bird community.	Editorial	All WOE discussions have been updated based on revisions to the overall approach in the BERA. The specific WOE for the carnivorous bird has been revised as follows in the 2nd bullet under the Carnivorous Birds sub-heading in Section 6.3.6.2. "Dietary HQs based on mid-sensitivity avian TRVs indicated that risk is negligible in all EUs within the LAA as well as C5 North. Predicted risk is low in C3 and C5 South, and moderate in all remaining EUs except C1 West where predicted risk is high. This evaluation is considered to be moderately conservative because the mid-sensitivity PCB TRVs are based on mourning dove studies, which are ecologically relevant and adequately representative of birds present onsite. Because the HQ results are based on doses estimated using predictive models for

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			prey tissue across the full range of PCBs measured in OU-4 soils, receptor specific parameters, and appropriate TRVs, and also because the red-tailed hawk and other carnivorous birds have been observed onsite, confidence in the risk prediction for terrestrial carnivorous birds based on this LOE is moderate to high."
94	Section 6.1.2.2, ME2b: This ME should be revised to state: "Body burdens of COPCs in OU-4 benthic invertebrate tissue as compared to tissue residue-based toxicity values and to concentrations measured in tissues collected in the reference area(s).	Editorial	This and other ME related to CTCs were updated in Table 2-16 in all cases where acceptable CTCs were identified. See also response to Comment Nos. 7, 46, 47, and 176.
95	Section 6.2.1.1: States “Changes in sediment deposition over time, such as increased sedimentation or erosion losses, may have occurred such that the surface soil/sediment dataset used in this OU-4 BERA is no longer representative of current conditions. Deposition of clean sediments or soils over historically contaminated materials may reduce exposure, or sediment removal in eroding areas may expose historically elevated concentrations of some COPCs.” Are there data to quantify this? If there are, this information should be included so that the implications of this can be quantified and used to improve the uncertainty analysis. What proportion of the site is depositional and what portion is erosional? How do these areas relate to risk estimates?	Editorial	The following information was added to the uncertainty analysis under a new sub-heading for exposure point concentrations (6.2.1.2) "For sediment EPCs, specifically, the EPC is weighted based on the estimated spatial extent of the three texture classes (i.e., fine, coarse, gravel) identified in OU-4 sediments. An evaluation of the visual classification of sediment cores (using the Unified Soil Classification System) collected from more than 180 transect locations in OU-4 indicated that much of Choccolocco Creek is composed of coarse-grained materials consistent with a high-energy non-depositional environment. For the 413 acres of creek between Snow Creek and the Highway 77 bridge, 174 acres can be characterized as consisting of gravel and having no recoverable material (42%) and 183 acres were classified as coarse grained (44%). This leaves only 56 acres (14%) defined as fine grained and likely depositional. Because the sampling for OU-4 was disproportionally focused on depositional areas, an approach to estimating EPCs based on the spatial area of each texture class was used. Thus, uncertainty associated with the class assignments may affect the estimated sediment EPCs." In addition, regarding potential changes in deposition or erosion

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			over time, Table R-1 (attached) shows that PCB concentrations in surface sediment are higher than subsurface concentrations in areas of Choccolocco Creek downstream of the backwater area, making the current exposure estimates protective. The PCB concentrations are higher in the Backwater area subsurface sediment than in the surface sediment, but sediments in this area are stable making the exposure of ecological receptors to deeper sediments unlikely. See response to Comment #137 regarding sediment stability and 155 regarding the proportion of the site that is depositional.
96	Section 6.2.1.1: States “Additionally for metals, only total metals were analyzed. The toxicity of many metals, notably including mercury, depends strongly on the oxidation state, presence of methylation, and other site-specific factors (such as presence of iron sulfide, or co-occurring metals) that influence uptake and effects. Thus, measurement of total metal concentration likely overestimates actual risk from most metals, because the specific form and toxicity of these metals in OU-4 is not accounted for.” For mercury, there is sufficient literature supporting the assumption that the majority of mercury present in fish tissue is MeHg, the most toxic form. As requested in Comment #37 (for the percentage of methylmercury in prey items to be factored into the risk equations) and in Comment #4 (consideration of the SEM-AVS data), which consider the form present, the text will need to be revised to remove statements that measurement of total metals likely overestimates the risk.	Editorial	The text regarding this uncertainty in the last paragraph in Section 6.2.1.1 under the Analytical Methods sub-heading is revised as follows: "Thus, measurement of total metal concentration may overestimate risks from some metals to specific receptors, because the form and toxicity of these metals in OU-4 is not accounted for. Additional information regarding the speciation of mercury and effects on risk estimates is presented in Section 6.2.3.9."

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97	Section 6.2.1.5: States "All selected exposure parameters are likely to overestimate potential exposure to actual wildlife present in OU-4." Statements like this need to be quantitatively supported. It is not unusual to need to approximate diets based on available tissue data. This is a real uncertainty, but it may result in either over- or under-estimation of risks.	Editorial	This statement in Section 6.2.1.5 (now 6.2.1.6) was revised to state: "The selected exposure parameters are likely to accurately represent or overestimate, but not underestimate potential exposure to actual wildlife present in OU-4, as the exposure parameters were selected to be more protective (i.e., high ingestion rates, low body weights, 100% site use/exposure to the upper bound of concentrations). For several of the selected representative receptors, significant uncertainty in the availability of preferred prey items in OU-4 was identified. Therefore, alternative diets were also considered. "
98	Section 6.2.2.1: States "It was also necessary to convert the estimated tissue to a dry weight value to be consistent with the dry weight IR used in the dose model. The average percent solids of 19% (84% moisture) was used for this purpose." If the dose model is using a dry weight ingestion rate, why not perform the regression using dry weight concentrations? Doing this post-regression wet-dry conversion using an average moisture content adds uncertainty to the estimated results. Sample-specific moisture values should have been used to reduce the uncertainty.	Editorial	The regression shown in Figure 6-23 was performed on a dry weight basis and the fit was not significant. The only fit that was significant was on a wet weight basis. The static assumption was necessary to compare possible results based on this regression to the results in the Draft BERA. No change was needed.
99	Section 6.2.2.1: States "To further evaluate the predicted sediment median BAFs based on the OU-4 data, BAFs available for two other PCB sites were considered." When comparing the BAFs developed for Anniston with those developed for other sites, please note the uncertainty in the OU4 BAFs on account of the limitations in the degree of co-location of tissue and sediment samples; comparisons to PCB BAFs from other sites should be caveated.	Editorial	The following text has been added to Section 6.2.2.1 to address this issue: "Differences in the OU-4 BAFs/BSAFs relative to those estimated for the Kalamazoo River and Housatonic River may be due to a variety of factors, including differences in the composition of PCBs at the sites, site-specific factors affecting bioavailability, and the degree to which abiotic and biotic samples were co-located (for OU-4, data were not precisely co-located)."
100	Table D-2: The acute and chronic values appear to be reversed.	Editorial	The acute and chronic values were corrected in Table D-2 and Table 5-2b.

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101	Table D-4, 5, 8, 9 and 10: More information should be included in this table - test species, test duration, number of doses, PCB form used, etc. Body weights and ingestions rates used to estimate doses should be as reported in the studies (i.e., Lillie et al. 1975 report different food ingestion rates for each dose and Aroclor tested). If values are not reported, food ingestion should be derived based on the chicken strain and test animal age using information from U.S. EPA (1988).	Editorial	Clarifying information, such as the number of doses, species tested, test duration, and PCB form was added to the tables in Appendix D. Note that these tables have been renumbered. Body weights and ingestion rates from the studies were used when available. If these parameters were not reported in the study, generic assumptions from USEPA (1993) and Nagy (2001) were used, consistent with the BERA.
102	Inconsistent terminology is used throughout the BERA to describe the site-specific toxicity thresholds (e.g., also referenced as benchmarks or criteria).	Editorial	Terminology has been made consistent throughout the BERA. Site-specific sediment toxicity thresholds are called out as such. Other literature derived soil or sediment-based values are referred to as benchmarks. Toxicity values for birds and mammals based on dietary dose are referred to as toxicity reference values (TRVs). Toxicity values for invertebrates, fish, bird eggs, and small mammals based on tissue residues are referred to as critical tissue concentrations (CTCs).
103	Terms including “similar to reference”, “impairment in some LOE” and “impairment in many LOE” are not clearly defined throughout the OU-4 BERA.	Editorial	The term impairment has been removed from the BERA. The term similar to reference is explained with each use. As an example, text in Section 5.6.7 for the mammal community metrics describes the term as follows: "Mean abundance and species richness were higher, although not significantly so, in OU-4 compared to reference. Diversity and dominance do not significantly differ between OU-4 and reference locations. Based on these results, the condition of the large mammal community within OU-4 appears to be similar to reference areas." In some cases, in order to improve readability of the text, these terms are used and supporting tables or figures are referenced that provide specific information clarifying the statements.

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104	Page 3-10, Appendix B: The description of the emergent insects should discuss how they were collected and whether adult or larval forms were collected. The Phase 2 FSP (Arcadis 2010a) indicated that benthic invertebrate sampling would include EPT species for larval and emergent life stages. There should be a discussion of what was collected relative to what was included in the FSP. Include a discussion of the number of samples collected versus the number of samples planned.	Emergent Insects	Additional text was added to Section 3.4 for each type of species collected that describes what was targeted and what was collected. In addition, two tables have been added that summarize this information. (Tables 3-13 and 3-14 for aquatic and terrestrial species respectively).
105	Section 3.4.1: Hymenoptera are not aquatic insects, but were included in the emergent insect samples. Halictid sweat bees are wholly terrestrial, nesting in the ground and foraging on plants. Why are sweat bees not mentioned in Appendix B?	Emergent Insects	The following footnote was added to the species list for emergent insects in Section 3.4" While Hymenoptera and Halictidae are not aquatic species, they were collected in the general area proximal to the creek where other emergent insects were collected in order to obtain adequate sample volumes for analysis."
106	The Biota-sediment accumulation factors (BSAFs) calculated for dragonflies and damselflies are significantly lower than those reported for bioaccumulation of PCBs into emergent insects at other sites, which ranged from 0.1 to 1.1 for total PCBs in the EPA BSAF database. To address uncertainty in the bioaccumulation factors for benthic invertebrates, literature values for bioaccumulation into emergent insects, or the crane fly data, should be used to predict PCB concentrations in infaunal emergent insects. Text on Page 6-57 should be revised to address the species collected as well as issues related to unknown degrees of exposure to contaminant concentrations in sediments including the sediment data pairing and sampling data concordance.	Emergent Insects	As described in responses to Comment Nos. 23 and 39, both modeled and measured tissue concentrations were evaluated in the revised BERA. In addition, the full range of BAFs based on crane fly and non-crane fly data were incorporated. The following text was added to the last subheading “Use of crane fly data for emergent insects” in Section 6.2.2.1 in the uncertainty analysis. “Literature values specific to crane flies were not available for comparison. However, as discussed below, data for emergent insects from the Kalamazoo river would suggest that the crane fly-based BAFs are likely to be overestimating uptake. While there is uncertainty associated with these elevated concentrations, they are based on validated data and are used in the BERA. The two crane fly samples affect exposure estimates for aquatic insectivorous birds and mammals within the UAA (including reaches C1 through C4) and may over predict exposure. The emergent insect BAFs employed for the MAA and UAA were based on samples primarily containing

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			damselflies and dragonflies (odonate species). These BAFs were compared to BAFs estimated using an available dataset from the Kalamazoo River (Kay et al., 2005). For all aquatic insects, a BAF of 0.54 was calculated based on the mean tissue (0.74 mg/kg ww converted to 2.39 mg/kg dw assuming the average of OU-4 emergent insect samples of 31 % solids) compared to the mean sediment concentration of 4.46 mg/kg dw. This BAF is very similar to the 0.56 and 0.53 BAFs calculated for OU-4 for the MAA and LAA respectively. A BAF value for odonates only (the species collected for the MAA and LAA), is estimated to be approximately 0.15. Thus, the selected BAF for the two downstream areas is considered adequately conservative."
107	The evaluation of exposure of higher trophic levels to emergent aquatic insects was based on a mean PCB concentration calculated across all emergent aquatic insect taxa sampled within each of the 10 reach sites (page 4-20). BAFs for emergent insects should be calculated on BSA and an assessment area basis. Crane flies should be evaluated separate from the other types of emergent insects. The dragonflies, damselflies and misc. winged insects can be combined and evaluated with separate grouping for BSAs and EDRs.	Emergent Insects	See responses to Comment Nos. 29 and 106.
108	Table 4-1a, 4-1b, 4-2a and 4-2b indicate that no receptors eat terrestrial invertebrates (worms). Worms should be included as a dietary component for the appropriate receptors.	Exposure Parameters	Tables 4-1a,b and 4-2a,b were updated as requested in USEPA Comment Table 4 (exposure parameters for birds) and Table 5 (exposure parameters for mammals).
109	Terrestrial invertebrates should have also included spiders/detritivores in addition to the herbivorous insects that were collected (Phase 2 FSP, Revision 2, page 50: Measured concentrations of COPCs in prey [e.g., edible plants, invertebrates [including spiders]]). Instead of using a literature-based BAF for spiders, the terrestrial worm data	Exposure Parameters	The portion of the receptor diet comprised of detritivores and spiders is estimated using the worm regression equation (or measured worm tissue). The portion that is comprised of crickets or grasshoppers is comprised of tissue estimates based on site-specific mercury data or estimated using literature PCB BAFs for terrestrial insects from Blankenship et

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	should be used as representative of all terrestrial invertebrate tissue concentrations.		al. (2005). These approaches are summarized in Sections 4.4.1.1 and 4.4.1.5 and described fully in Appendix B (Sections 4.4 and 4.5). See also response to Comment No. 30.
110	Table 4-1-a, 4-1b: Several receptors are indicated as feeding on aerial invertebrates that should be consuming terrestrial invertebrates (Carolina wren, spotted sandpiper, blue jay). See Tables 4 and 5 for recommended diet composition percentages.	Exposure Parameters	Tables indicate that all terrestrial invertebrates and insects should be modeled using the earthworm data in spite of the fact that portions of receptor diets are identified as crickets and grasshoppers. The earthworm model was used for detritivores and worms and the grasshopper data were used for that portion of the receptor diets.
111	Because they represent very different exposure pathways, reptiles and amphibians should not be combined as a dietary component. The exposure parameters need to be better refined so that the appropriate measured tissue concentrations/BAFs may be used independently. See Tables 4 and 5 for recommended diet composition percentages; the revised percentages of reptiles and amphibians were taken from U.S. EPA 1993.	Exposure Parameters	Reptile and amphibian portions of receptor diets were separated as outlined in USEPA Tables 4 and 5. See response to Comment No. 28.
112	Hazard quotients are presented for terrestrial receptors in Tables 6-4 & 6-5 that include receptors having a diet consisting of a mixture of terrestrial and aquatic prey items, including reptiles and amphibians for terrestrial receptors and small mammals for carnivorous wading birds. Clarify how the bioaccumulation factors for reptiles and amphibians exposed to sediment or small mammals exposed to soil were used and how this information was incorporated into dietary exposure calculations for these receptors.	Exposure Parameters	The BAFs for reptiles and amphibians are based on sediment and those for mammals are based on soil. The uncertainty associated with applying these values to another media to estimate tissue concentrations is discussed. In addition, the exposure estimates based on measured tissue also address this uncertainty.

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113	Use the abiotic data agreed upon by the BAF Workgroup for estimating the EPCs reflective of more recent conditions, i.e., 2007 – 2010 data. Risk calculations should also be run using the pre-2007 data for historical context and greater spatial coverage. Alternatively, risk calculations could be run using the most recent (2007-2009) data. The more recent data could be statistically compared with the older data; these relationships can be used to make conclusions concerning how risks may have changed.	Exposure Point Concentrations (EPCs) – Data used to calculate	BAFs and tissue EPCs were calculated using tissue and soil/sediment from Phase 1 and Phase 2 data only (i.e., 2007-2010). EPCs are not calculated using multiple iterations of the dataset. This would add an unwarranted level of complexity to the BERA (i.e., two complete sets of risk calculations). Moreover, the comparability of the datasets would be highly uncertain due to the differences in spatial areas sampled. Figures 6-4 and 6-7 provide box plots of the soil and sediment data for PCBs and mercury respectively based on the Phase 2 data only (synoptically collected with the biotic tissue) and the entire BERA dataset.
114	If the COPC was detected in environmental media or tissue in fewer than five location groups and there is no spatial pattern to the detections, use the measured OU4 site-wide 95% UCL (or maximum detection) as the EPC for all ecological exposure units.	EPCs – 95% UCLs versus Modeled EPCs	Terrestrial plants are the only tissue type with detections in fewer than five location groups. The measured site-wide UCL is used as the tissue EPC for terrestrial plants.
115	If the COPC was detected in greater than or equal to five location groups and displays a spatial pattern of concentrations, then use the measured 95% UCL by ecological EU or by AA depending on the amount of data available, for the ecological exposure units that have available data. Use predicted 95% UCL EPCs for ecological EUs lacking data using BAFs. For PCB and mercury bioaccumulation into terrestrial plants, literature based BAFs can be used.	EPCs – 95% UCLs versus Modeled EPCs	See responses to Comments Nos. 26, 23, and 39.
116	Use ProUCL to calculate the 95% UCLs and utilize the program's method of treating non-detects.	EPCs – 95% UCLs versus Modeled EPCs	See responses to Comments Nos. 132 and 154.
117	Compare measured fish tissue concentrations (forage, bottom fish and predator fish) with fish tissue TRVs. Use the 95UCL of measured fish tissue concentrations as the EPC.	Fish	As described in responses to Comments Nos. 23 and 39, both modeled and measured tissue concentrations were evaluated in the revised BERA.

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118	The conclusions in the BERA relate to fish in general and do not identify the higher degree of risk in predatory fish. Text should be revised to discuss risks to predatory fishes, including a discussion of the higher degree of bioaccumulation observed through trophic transfer, and the measured concentrations of PCBs and mercury in predator fish relative to tissue residue based TRVs. Tables 6-1 & 6-2 should include hazard quotients for predator fish.	Fish	Section 6.1.4.2 presents risks for each type of fish (i.e., forage, bottom and predator) based on tissue residues, with tissue HQs presented in Table 6-11 for the various fish tissue types.
119	Page F-18: The measured moisture content in fish from OU4 should be used to convert tissue concentrations to dry weight instead of using an assumed value.	Fish	The sample-specific moisture for fish was used to convert the wet weight tissue concentrations to dry weight.
120	Section 6.3.1: The Weight of Evidence (WOE) Approach boils all of the LOEs into two categories: HQ results and community survey results. This under-represents the lines of evidence that are available. There are 3 general types of LOE: a) Literature-based TRVs vs EPCs (either media concentrations or dietary exposure estimates) b) Site-specific bioassays, and c) Site-specific field surveys.	Lines of Evidence, Risk Conclusions	The recommendation in a and c are included. Bullet a is specifically expanded to include a range of values. However, the site-specific bioassays could not be incorporated because the studies were not designed to support evaluation of OU-4 spatially. Only three locations contained sediments that had a surface start depth. Of those, two included sediments mixed within the top 2 feet. Thus, these sediments cannot be used directly to evaluate conditions in surface sediments within OU-4. The results of the sediment toxicity testing are included in the context of site-specific sediment toxicity thresholds. This is consistent with the study design.
121	Each of these can be split further depending upon what data are available. Each LOE must be as independent as possible; different effect thresholds, different ways that exposure is evaluated. These are all then weighted to get at the WOE. Boiling everything down to two categories loses much useful information.	Lines of Evidence, Risk Conclusions	See response to Comment No. 120.
122	Literature-derived toxicity benchmarks and site-specific toxicity thresholds should be evaluated as separate LOEs as they are independent.	Lines of Evidence, Risk Conclusions	HQs based on literature benchmarks and site-specific bioassays were evaluated separately as noted in response to Comment No. 120 and in Section 6.3.3.

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123	Section 4.2.1: "Due to limitations with the dataset in a few of the reaches, in some cases reaches were grouped as shown below and surface weighted average concentrations (SWACs) (Table 4-5) for mercury were calculated for: • C1 - Oxford Lake Park; • C2 - Backwater Area; • C3 and C4 – Friendship Road to Coldwater Creek; and • C5 through C8 – Coldwater Creek to Jackson Shoals Dam". Pooling data among reaches decreases resolution, will increase uncertainty, and limit the ability to draw risk conclusions by reach. Table 4-5 should be revised to present summary statistics for mercury (minimum, maximum, mean, number of detects, 95%UCL, EPC and EPC basis) on an assessment area basis.	Mercury	Tables showing the summary statistics for mercury are included in Section 4. See response to Comment No 154 for specific changes to the sediment EPC approach. For mercury, data were pooled to estimate reach-specific EPCs due to small sample sizes. EPCs are also calculated on an AA basis.
124	The OU-4 BERA data set does not contain all of the available data compiled on the site to date. Although it is not explicitly stated in the Phase 2 FSP, the table on page 59 notes that the environmental media and prey tissue COPC concentrations used in the BERA will be the data collected during the Phase 2 sampling. Arcadis can respond to this in response to comments memo. Older tissue data available includes a lot of fillet data, with only limited site-specific data on whole body to fillet ratios (two locations, bass and catfish, section 4.3.1.2.3.2, Phase 2 FSP). Some fish locations do not have corresponding sediment data. The fish collection efforts for the human health risk assessment (HHRA) targeted different sized fish than the fish collected for the BERA. The current ecological fish data are of sufficient quantity and quality to support the risk analysis.	Missing Data	The tissue dataset for the BERA includes the data collected during the Phase 2 FSP. These data are considered to be the most accurate representation of current conditions in biotic tissues. It is noted that additional fish tissue data from historical collection efforts and ongoing monitoring are available. Historical data were not included because they do not represent current conditions and the 2007 dataset is sufficient for the BERA evaluations. Other data are primarily filet data. An extrapolation would need to be employed to use these data in the BERA. The uncertainty associated with extrapolation was considered unwarranted given the sufficiency of the 2007 dataset.
125	Soil and sediment data sets Arcadis did not have access to do not need to be included in the BERA.	Missing Data	Comment noted.

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126	Section 2: The measurement endpoints selected for evaluating the status of the selected assessment endpoints (specifically survival, growth and reproduction of benthic invertebrates) should add the survival and growth of freshwater mussels in laboratory toxicity tests. Although this report was not available when the BERA was done, the data are available now (Evaluation of the Toxicity of Sediments from the Anniston PCB Site to the Mussel *Lampsilis siliquoidea* [September 2015]; available at: http://www.fws.gov/daphne/Contaminants/index-AnnistonNRDA.html). Arcadis can include the mussel study results if they want to, but it is not required. EPA will consider data from the mussel toxicity study in the risk management decision.	Mussel Toxicity Tests	Perspectives on the USFWS (2015) mussel toxicity findings are provided in Sections 5.1.1 and 6.1.2.1.1 in terms of the relative sensitivity of these organisms compared to the midge and amphipod test species from which the invertebrate toxicity thresholds were derived. An evaluation of the mussel toxicity study results shows that they are less sensitive to OU-4 sediments than either the midge or amphipod. The effects concentrations provided in the BERA are, thus, protective for bivalves in the OU-4. Based on this finding, a formal concentration-response modeling exercise was not conducted, as it would not have produced effects concentrations that would replace those calculated for the infaunal test organisms. It is agreed that the mussel data are an important consideration for risk management in OU-4.
127	Estimates of PCB concentrations in pore water based on solid-phase microextraction (SPME) need to be discussed in the OU-4 BERA. Site-specific sediment toxicity thresholds based on concentrations of PCBs estimated for pore water need to be presented as a supporting line of evidence.	Pore Water	The SPME pore water PCB estimates are provided and discussed in the revised Appendix C to the BERA (Table C-5 and Attachment 1), including the raw data and a summary of the total PCBs by sample. The concentration-response relationships developed by Ingersoll et al. (2014) using these data showed an almost identical relationship and statistical significance for effects relative to those in sediments. As such, the sediment data are the most appropriate means for computing effects concentrations, as sediment is what risk management decisions will be based on. See also response to Comment No 146.

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128	EPA reviewed the approved 2006 Baseline Problem Formulation (BPF) for OU-4 to determine whether all agreed-to receptor groups, exposure pathways, AEs and MEs were evaluated in the BERA. For Table 2-16, the risk questions and measurement endpoints should be copied out of the approved Problem Formulation. Some of the agreed-to receptor groups were not evaluated as stated in Section 3.1.6 of the Problem Formulation (threatened and endangered [T&E] species), and some of the measurement endpoints presented in the BERA were not consistent with the measurement endpoints identified in the approved problem formulation (e.g., comparison of OU-4 tissue concentrations with reference tissue concentrations instead of critical tissue concentrations, comparison of estimated rather than measured tissue concentrations). The Problem Formulation document should be cited in the BERA.	Problem Formulation	Components are discussed in the responses to Comment Nos. 46 and 157. The Baseline Problem Formulation document is referenced in the problem formulation of the revised BERA as requested.
129	Section 2: The conceptual site model is incompletely described and does not include diagrams that establish the linkages between the various elements of the problem formulation.	Problem Formulation	A food web figure has been added (Figure 2.3) to clarify the linkages between Site media and the different trophic levels of the food web. The CSM text has been updated to more clearly describe fate and transport, trophic transfer and complete exposure pathways between the potential sources and the ecological resources at the Site.
130	Section 2.1: This Section should provide a description of: a general summary of the types of habitat and ecosystems present; the sources and releases of COPCs; COPC fate and transport; and potential receptors in OU-4. All discussion related to the community surveys except for the habitat description should be moved to Section 5.7, Effects on Ecological Communities, and discussed as an LOE.	Problem Formulation	Section 2.1 was updated as follows: 1) The CSM discussion has amended to include sections specific to sources, fate and transport, receptors. The summary information in Section 2.1 regarding community surveys has been revised to provide information on habitat and species present. Discussion of the results of these studies has been moved to Section 5.7.

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131	Section 2.5: There are some gaps in data developed for the BERA. Although methods are available to perform site-specific bioassays for a number of assessment endpoint groups, they were only performed for benthic invertebrates. Some discussion should be included to indicate that site-specific data could have been developed for plants, soil invertebrates, and amphibians, but were not. In EPA comments on the "Annotated Outline: Assessment Endpoints, Measures of Effect and Proposed Representative Ecological Receptors, Page 3" it states: "The identified MEs do not include laboratory toxicity tests with soil invertebrates. This should be considered in the BPF and a discussion should be provided as to why they were or were not included.	Problem Formulation	A footnote is added in Section 2.5 acknowledging that other bioassays could have been conducted. Footnote 4 includes the following: "Site-specific bioassays were conducted for benthic invertebrates only. While it is recognized that bioassays could have been conducted for other species such as terrestrial invertebrates, the evaluation of the range of other receptors included in the BERA were considered sufficient for characterizing and managing possible risks."
132	Use the most recent version of ProUCL, Version 5.0, which was released in 2013, for all EPC calculations.	ProUCL	All summary statistics were run using ProUCL 5.1.
133	If there are any data available to demonstrate that abiotic conditions associated with habitats in the reference sites were similar to abiotic conditions associated with habitats the 10 reach sites (e.g., temperature, depth, flow, grain size, total organic carbon, turbidity, lighting, physical-chemical properties), these data should be presented in the BERA.	Reference Locations	Available data regarding abiotic conditions was added to Section 5 in Tables 5-6 and 5-7. The following text was added to the end of the first paragraph in Section 5.6.2. "In general the ranges of abiotic conditions were similar at OU-4 and reference locations for each habitat type. Tables 5-6 summarizes water quality parameters during the 2006 and 2007 sampling events. Table 5-7 summarizes the composition of substrate at OU-4 and reference locations."
134	Global comment for all HQ tables (Tables 6-1 to 6-11) - the lowest threshold should be set at 1, not 1.5. Section 6, the risk summary section, would be easier to review if all HQs greater than 1 (not greater than 1.5) were shaded yellow.	Risk Characterization	The HQs are reliant on multiple variables with varying levels of precision and are comprised of values and calculations that in most cases over-estimate risk. An example of precision includes the BAF ratio that can be calculated to multiple decimal places but given what we know about the representativeness of the relationship, the precision in that estimate is low. Other inputs, such as food ingestion rates and toxicity reference values (TRVs), have similar levels of variability and hence uncertainty. The TRV and exposure concentrations are also examples where risks are overestimated in an effort to be protective.

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			Given these uncertainties, reporting HQs to anything more than a whole number implies a precision that is not supported in the inputs to the calculation. HQs were reported throughout the BERA to one significant digit (i.e. HQ = 1, not HQ = 1.0).
135	The risk summaries for each receptor need to be carefully reviewed and revised. The following risk categories should be used consistently throughout the summary: no observed adverse effect level (NOAEL) HQ < 1, risk is negligible or *de minimus*; NOAEL HQ > 1 and lowest observed adverse effect level (LOAEL) HQ ≤ 1, risk is low. LOAEL HQs > 1 should be evaluated based on the magnitude and the ecological significance of the response in the selected critical study. Most of the mistakes in the text are for HQs >1 and < 1.5.	Risk Characterization	The general categories outlined in this comment were employed. As described in response to Comment #134, an HQ of < 1.5 rounds to a whole number of 1 and is considered to be equal to 1 in the BERA.
136	Table 6-12, 6-13, 6-14, 6-15, 6-16, 6-17 under-represent risks. They only include COPCs with LOAEL HQs > 1.5. All HQs >= 1 should be included.	Risk Characterization	See responses to Comments Nos. 134 and 135.
137	The sediment biologically active zone of exposure evaluated was typically to a sediment depth of 5 centimeters (cm) (2 inches; Table 4-4, page 3-2, page 4-10). No data are provided illustrating the stability of sediments across the 10 reach sites (e.g., what is the potential for redistribution of sediments to a depth of more than 5 cm?). No conclusions can be made on the risks associated with exposure to deeper sediments (e.g., 5 to 15 cm) or on the risks associated with redistribution of contaminated sediments. The uncertainty associated with how the biologically active zone was defined and potential risk associated with exposure to deeper sediments should be discussed. Arcadis can respond to this comment in a response to comments memo. Reference the sediment stability study. The term “biologically active zone” was not used in the FSPs, although it was used in the BERA.	Sediment Sampling Depth	See response to Comment No. 138 regarding the depth profile sampled in sediment. The possibility of deeper sediment being exposed over time was evaluated to evaluate whether changes in deposition or erosion over time would result in changes to the surface concentrations such that the current risk assessment is underestimating exposure. As shown on Table R-1 (attached), in current surface sediment downstream of the backwater area, PCB concentrations are higher than subsurface concentrations, making the current exposure estimates protective. The PCB concentrations are higher in the Backwater area subsurface sediment, but sediments in this area are stable making the exposure of ecological receptors to deeper sediments unlikely. See also response to Comment No. 138.

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138	Add a quantitative analysis that demonstrates that the 0-2" results are equivalent to 0-6" results to the BERA. Include the figure cited at the 2-17 meeting showing PCB concentrations by sample event. Cite U.S. EPA 2015, "Determination of the Biologically Relevant Sampling Depth for Terrestrial and Aquatic ERAs".	Sediment Sampling Depth	The following text was added as the last paragraph in Section 6.2.1.1. "Depth Profile For soil and sediment sampling, different depth profiles were sampled during different sampling events. The soil was generally sampled from 0 to 1 foot, which is considered a reasonable estimate of the potential exposure zone for ecological receptors. The sediment was sampled at both 0 to 2 inches bgs and 0 to 6 inches bgs. While the most likely ecological exposures are likely in the most surficial sediments, the 0- to 6-inch interval is considered more comprehensive of the potential exposure depth and is what is recommended in the USEPA document entitled "Determination of the Biologically Relevant Sampling Depth for Terrestrial and Aquatic ERAs" (USEPA 2015). Thus, there is uncertainty associated with the use of the data from the 0- to 2-inch depth. To evaluate the comparability of data collected from the 0- to 2-inch interval versus the 0- to 6-inch interval, sediment cores collected during the RFI phase of the project and segmented into the 0- to 2-inch and 2- to 6-inch intervals were evaluated to identify any differences or bias that may be introduced by using data from the 0- to 2-inch interval as representative of the 0- to 6-inch intervals. A statistically significant (P-value <0.001) regression line for the 0 to 5 mg/kg dataset (which excludes one value greater than 5 mg/kg) has a slope of 0.962 indicating an almost 1:1 relationship for data from the two intervals. This correlation and slope indicate that the 0- to 2-inch interval are a suitable representative for the 0- to 6-inch interval. Surface sediment data (0-6 inches bgs) are shown by program on Figure 6-22." The correlation between the 0 to 2-inch and 0 to 6-inch horizons presented above in the text of the revised BERA are consistent with the mathematical

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			construct that was previously discussed with the USEPA. In the mathematical construct, the 0 to 2-inch interval would be representative of the biologically active zone (BAZ) even if the sediment in the 0 to 6-inch interval was the true BAZ. This is in contrast to the result of an approach that sampled the 0 to 6-inch interval when the true BAZ was less than 6 inches thick. The quantitative comparison results presented above and in the revised BERA confirm that results for the 0 to 2-inch and the 0 to 6-inch horizons are both representative of the BAZ, and are consistent with the mathematical construct that had been previously used so support using the results from both sample collections horizons.
139	The sediment toxicity tests, which are a site-specific measure of effect, should be considered a primary LOE for assessing risk to the benthic macroinvertebrate community	Sediment Toxicity Tests	The sediment toxicity tests are the primary line of evidence for the BERA. They form the basis for the calculation of all the site-specific toxicity thresholds used to conduct the point-by-point effects analysis by reach. Individual sediment samples used for toxicity testing were collected from various depths to capture a range of PCB concentrations and therefore are not representative of the exposure of a benthic organism in the biologically active zone (0-6"). Therefore, toxicity results from the testing program could not be used to evaluate risk at the sampling locations.

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140	An explanation of the differences in the Arcadis interpretation and the Ingersoll interpretation of the toxicity test study results should be added to the BERA report. A table should be developed which compares the various sediment quality benchmarks developed by Arcadis to those developed by Ingersoll, with any associated equations or a worked example provided so that the analysis is transparent. These comparisons should be done for multiple endpoints. This recommendation will provide context and added support for the values being used in the BERA. If Arcadis has issues with the toxicity thresholds derived or the analysis done in the Ingersoll et al. (2014) report, those issues should be clearly articulated in the BERA; an explanation of why the Ingersoll values were not used must be provided. EPA noted that including tables with both study interpretations, scatterplots, and overlays of the dose response curves produced by Arcadis and Ingersoll will support the discussion of the differences in the study interpretations.	Sediment Toxicity Tests	An explanation and comparison of the difference in the interpretation of the sediment toxicity test program results between Ingersoll et al. (2014) and the BERA is provided in the revised Appendix C (section 3.2 Table C-11) to the BERA. A table comparing the differences in the effects concentrations for some of the sensitive endpoints is included as part of this assessment (Table C-11). The USGS and USACE were contracted to conduct the field and laboratory studies and to provide the results of these benchtop studies for use in preparing the BERA. Ingersoll et al. (2014) conducted a separate evaluation of these benchtop data for NRDA purposes, and in some cases used inappropriate models and assumptions for their analysis. These are discussed in Appendix C (Section 3.2). The benthic invertebrate risk analysis in this BERA was conducted using the USGS/USACE benchtop testing data and technically appropriate data evaluation techniques.
141	The sub lethal endpoints from the Ingersoll toxicity tests were not used to develop site-specific toxicity thresholds. The statements regarding growth and reproduction as less certain measures of effects should be removed from the BERA. Additionally, the conclusion in the OU-4 BERA that "chironomid data provide a more appropriate test of potential toxicity because these receptors are known to be present at OU-4" is not correct and should be deleted. Chironomids are typically less sensitive, and *Hyalella* can be used as a surrogate for more sensitive species that may be present in the ecosystem. Test results from both surrogate species are equally relevant to evaluate impacts on benthic macroinvertebrates. Toxicity reference values generated for sensitive species and the most sensitive endpoints should be used preferentially to evaluate the potential for population level effects on benthic invertebrates and to develop risk conclusions for the BERA.	Sediment Toxicity Tests	All endpoints (except chironomid adult biomass - see response to Comment No. 142) were considered and evaluated in the Draft BERA. Revisions to the benthic community assessment in the BERA focus on the most sensitive reproductive and survival endpoints for the two species tested *(Table 6-4a)* Text regarding the relative applicability of chironomids was removed.

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142	The site-specific toxicity threshold for midges based on the most sensitive endpoint of adult biomass was not used in the OU-4 BERA to evaluate risks (page C-7, page 5-7). The larvae on Day 13 of the exposure would be at a stage where there is reduced feeding rate before pupation, so any density-dependent effects on weight of larvae, pupae, and resultant adults would be minimal. A strong correlation was observed between adult biomass and Day 13 average weight of midges (Figure A3-15L in Ingersoll et al. 2014), indicating that emergence of adults was not likely biased due to the possibility of a density-dependent influence of larvae surviving to the 4th instar subsequently dying before emerging as adult. Midge biomass should not be excluded as an endpoint.	Sediment Toxicity Tests	As described in the BERA, this endpoint is not included in the site-specific toxicity thresholds evaluated in the BERA because of the high level of uncertainty in the estimation method employed and based on a far-reaching and unsupportable assumption regarding linear growth in midges from day 13 through the end of the test. The reason for the strong correlation cited between adult biomass and Day 13 average weight is that the adult biomass was calculated by Ingersoll et al. (2014) based on the 13-day average weight. As a result, this correlation is meaningless. There are no data to support the assertions or to estimate adult biomass credibly from this testing program. The specifics of the uncertainty associated with this endpoint are discussed in Appendix C and Section 5.1.1
143	The bottom of the reference envelope was defined as the lowest reference response, whereas it is typically defined as the 5^{th} or 10^{th} percentile of reference responses (MacDonald and Landrum 2008; Hunt et al. 2001). This is the least conservative definition of the reference envelope. The BERA should calculate the 5th percentile of the reference envelope and present results both ways: compared to the lowest response percentage (currently in the BERA) and compared to the 5th percentile percentage.	Sediment Toxicity Tests	In most cases, the 5th percentile of reference response, when derived parametrically, is lower than the minimum observed reference response. Therefore, for most endpoints, and for all of the most sensitive endpoints, an EC0* based on the 5th percentile is higher than one based on the minimum. To illustrate this, a total PCB threshold value was also calculated for each endpoint at the 5th percentile of reference and presented in Table C-1 of Appendix C. These values were greater than EC0* for most endpoints and for all of the most sensitive endpoints.
144	Calculated lower prediction limits (LPLs) are not typically used in a risk assessment. The reference envelope is a defined lower limit; if survival, growth, or reproduction responses fall outside of the normal range of reference responses, a sample should be considered toxic. Lower limit statistics have to truncate at this level. It is also not clear how the LPL was calculated. The ProUCL user guide (U.S. EPA 2013, .pdf page 156) states: “In many environmental applications … one needs to compute lower	Sediment Toxicity Tests	The ProUCL technical guide and statistical output provide the equations and statistics needed to calculate the lower limit statistics using simple arithmetic operations. In the Draft BERA, HQs were presented and discussed based on 90%LPLs in addition to EC*0, EC10*, and EC20*. However, because the EPA has requested this change, the LPL was removed from the BERA.

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	limits including LPLs, lower tolerance limits (LTLs), or lower simultaneous limits (LSLs). At present, ProUCL does not directly compute a LPL, LTL or a LSL." Please remove the Lower Limit calculations from the BERA.		
145	Because of the high reliability of the Pb, Hg, and polycyclic aromatic hydrocarbon (PAH) thresholds from Ingersoll in correctly predicting toxicity, the EC0*, EC10*,EC20* and EC50* should be calculated for these analytes. The BERA currently only compares concentrations of metals and PAHs to the "consensus-based" probable effect concentrations (PECs) published by MacDonald et al. (2000) and concludes that metals and PAHs are not likely to have contributed significantly (relative to PCBs) to toxicity in OU-4 sediments. Reliability analysis in the Ingersoll report indicates that the Pb, Hg, and PAH thresholds correctly classify samples as toxic or nontoxic 74-96% of the time. Because these values are site-specific, these values should be included and compared with measured sediment Pb, Hg and PAH concentrations. Metals toxicity might be a factor in some of the tests based on AVS-SEM data. Some of the sediments with higher concentrations of PCBs also contained mercury concentrations within a range that was found to be toxic in other studies. Chibunda (2009) reported an effects concentration for 28-day emergence in *C. riparius* of 3.84 mg/kg of mercury in sediment and an effects concentration for 14-day growth of 2.42 mg/kg.	Sediment Toxicity Tests	Appendix C of the BERA has been revised to include an expanded discussion of the non-PCB data. Because the contaminant concentrations in tested sediments are correlated, it follows that concentration response models can be developed for lead, mercury, and PAHs (not a COPC for OU-4), and that the threshold values generated from these models (i.e., comparable to EC0* and EC10*) would show apparent predictability of toxicity. Ingersoll et al. (2014) recognized that this correlation may lead to false conclusions about causation among contaminants, and that the threshold values required further evaluation. Ingersoll et al. (2014) found the thresholds calculated for these compounds to be lower than either consensus-based SQGs (i.e. PECs) or LC50s generated in spiked toxicity tests, and therefore concluded that they are "likely not the main contributors to the observed toxicity." Lead toxicity thresholds ranged from 9 to 14 ppm, lower than both the TEC (35.8 ppm) and the PEC (128 ppm). This suggested to Ingersoll et al. that lead was not the driver of toxicity. The AVS-SEM data also supported that lead was not a primary contributor (see response to Comment No. 4). Mercury toxicity thresholds from Ingersoll et al. (2014) ranged from 0.3 to 1.3 ppm. These values are in fact within the range of the TEC (0.18 ppm) and the PEC (1.06 ppm). Site-specific values for mercury were calculated in a manner consistent with the PCB approach and are included in the revised BERA (Appendix C, Table C-6).

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146	A discussion of the SPME pore water COC concentrations and the toxicity test results should be added to the BERA as a supporting line of evidence.	Sediment Toxicity Tests	The SPME pore water PCB estimates are provided and discussed in the revised Appendix C to the BERA. See response to Comment No. 127.
147	Table 5-2a: The most sensitive EC0*, EC10* and EC20* values listed in Table 5-2a should be identified as the site-specific PCB toxicity values for benthic invertebrates. Presenting toxicity thresholds for all endpoints rather than just the most sensitive endpoints makes the benthic analysis difficult to follow.	Sediment Toxicity Tests	The new point-by-point effects assessment for the benthic invertebrate toxicity thresholds provides a more concise and easy to follow assessment, clearly focusing on the most sensitive endpoints (Table 6-4a).While risk estimates were based on these most sensitive endpoints, the survival endpoints have not been removed from the effects assessment as they provide important perspective on the risk characterization, particularly given the high degree of uncertainty associated with the most sensitive reproductive endpoints for amphipods.
148	Page 5-7, Tables C-1, 5-2a, Table E-1: The site-specific toxicity thresholds are reported to several significant digits in Table C-1, but are reported to no decimal places elsewhere (e.g., page 5-7, Table 5-2a). Toxicity thresholds need to be reported to the appropriate significant figures (e.g., 1 vs. 1.38 mg PCB/kg for the EC0* for H. azteca survival-normalized reproduction, 7 vs. 6.8 mg PCB/kg for the EC10* for C. dilutus emergence). Toxicity thresholds and HQs should be reported to two decimal places.	Sediment Toxicity Tests	See responses to Comment Nos. 68, 134, and 135.
149	Evaluate PCBs in surface water on an assessment area basis, not on an OU-4 overall basis.	Surface Water	For PCBs and metals, surface water was evaluated on an AA-basis (Table 6-12).
150	The first row in Table 8-1 indicates that the potential risk to aquatic invertebrates is low throughout OU4. Table 6-18 indicates that the potential risk to aquatic life was moderate due to the comparison of the surface water concentration to the Alabama Department of Environmental Management water quality standards (ADEM WQS); the chronic surface water standard for PCBs was exceeded at the site. Please correct Table 8-1 to show the potential risk to aquatic invertebrates as moderate.	Surface Water	Based on other revisions requested in the methods for evaluating surface water (see responses to Comment Nos. 149 and 153), this table was updated to reflect that ADEM WQS do not apply to benthic invertebrates. The WOE findings for the benthic invertebrate community are described in detail in Section 6.3.2 and summarized in Table 6-24.

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151	The use of water quality guidelines or the ADEM WQS to assess benthic species should be further supported. The data sets used to derive these guidelines likely do not contain data for benthic species. If the ADEM WQS values are considered relevant for benthic species, how is a 7-fold exceedance of the chronic guideline for PCBs considered to represent a low risk, as most benthic species are immobile and would experience chronic exposures?	Surface Water	The text in Section 6.1.2.1.2 was revised to clarify that WQS were compared to surface water concentrations to evaluate all aquatic life, recognizing that the WQS may not be directly applicable to benthic species.
152	National Ambient Water Quality Criteria (AWQC) should have been used as an additional source of surface water benchmarks. Describe the magnitude and frequency of exceedance of the AWQC and the ADEM WQS in the OU-4 BERA.	Surface Water	Values that are different between the two sources are evaluated in Table 6-12 and Section 6.1.1.1 (i.e., for mercury).
153	Table D-2, Footnote C: Hardness adjustment should have been done on a sample-by-sample basis, not using an assumed value of 200 milligrams per liter (mg/L). The AWQC default calculations use a hardness value of 100 mg/L. Use of a 200 mg/L value results in much higher hardness adjusted criteria (e.g., acute and chronic criteria for lead at 100 mg/L are 64.58 and 2.52 micrograms per liter [µg/L]. At 200 mg/L, criteria are 136.14 and 5.31 µg/L). Provide a table with site-specific hardness data (hardness data are not included in Appendix E).	Surface Water	The following text was added as the 2nd paragraph in Section 6.2.3.5. "ADEM WQS/NAWQC for some metals are estimated based on hardness assumptions. For the BERA, 200 mg/L hardness was assumed. Because hardness data are not available for OU-4, site-specific adjustments could not be made. This assumption could over or underestimate the toxicity benchmarks. If hardness is lower, the toxicity benchmarks employed would underestimate possible risk and if hardness is higher, they would overestimate possible risk."

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154	Section 4.1.1.2.1 of the Phase 2 FSP states that the 95% UCL will be the value used as the EPC in the risk calculations. Section 4.1.1.2.2 states that if the 95% UCL calculations result in HQs > 1, then a more "realistic" scenario will be evaluated using the SWAC as the EPC. However, the SWAC was used as the sediment EPC for all risk calculations. Tables 4-4 and 4-5 must be revised to present summary statistics similar to Table 4-6 (minimum, maximum, mean, number of detects, 95%UCL, EPC and EPC basis). Risk calculations need to be re-done using the 95%UCL as the sediment EPC for PCBs and Hg. The SWAC should not be used as the sediment EPC for PCBs and Hg. Table 4-4 should also present PCBs on an organic-carbon normalized basis.	Surface Weighted Average Concentration	As requested, the sediment EPCs for aquatic birds and mammals were calculated as an upper bound on the mean (i.e., a 95th percentile of the boot strap means) with areas having no recoverable sediment removed from the calculation process. The spatial weighting by textural class was included as the data were collected in a biased manner focused on fines (see response to Comment No. 95 for discussion of area for each class). Invertebrates and other prey items for fish, birds, and mammals may forage or reside in all texture classes (including fines, coarse, and the interstitial sediment in gravel areas). The areas with no sediment recovery were excluded in this revision because of the uncertainty associated with having to assume a concentration for these areas. This approach provides an upper bound on the exposure area while still accounting for the difference in area of the different sediment classes. Section 4.2.1 was updated to reflect the revised approach. Tables 4-4 and 4-5 a and b provide summary statistics by texture class for PCBs and mercury respectively. As described in response to Comment No. 25, PCBs were not evaluated on an organic carbon normalized basis.
155	It is not clear whether the sediment textural class used in the SWAC calculations was assigned based on field observations or laboratory grain size measurements, but it seems to have been based on the field observations. Discuss the uncertainty associated with doing this calculation based on visual field observations. For sediments that have grain size analysis, how well does the field observation correlate with the lab grain size analysis?	Surface Weighted Average Concentration	The following text was added to the uncertainty analysis as new subsection 6.2.1.2. "Exposure Point Concentrations" The primary uncertainty associated with any EPC is associated with how well it may actually represent the true exposure concentration. In all cases in this BERA, an upper bound on the mean was selected as the EPC to mitigate this uncertainty. For sediment EPCs, specifically, the EPC is weighted based on the estimated spatial extent of the three texture classes (i.e., fine, coarse, gravel) identified in OU-4 sediments. Because the texture class are factored into the sediment EPC estimation, the uncertainty associated with the class assignments

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			may affect the estimated values. The areas associated with the sediment EPC calculations were based on 769 attempted core locations from 187 transects in Choccolocco Creek. At each location where a core was collected it was visually classified using the Unified Soil Classification System. The photographs and field descriptions of each core were reviewed in the office and the sediments were classified into fine, coarse, and gravel categories. Organics (OL, PT), clays (CH, CL), silts (MH, ML), and fine sands (SC, SM) were called “fine grained” while coarse sands (SP, SW) were called “coarse grained). Gravels included classifications GM, GP, and GW. Samples from 28 core locations were submitted for laboratory analysis of grain size, and for these 28 cores, the grain size results were used along with the visual descriptions to classify the sediments in the fine, coarse, and gravel categories. For the surficial data used for the PCB contribution to the sediment EPC calculations, 57 of the 266 samples used a combination of grain size results and visual criteria to classify the sediments. The remaining samples were based on visual criteria only. The use of both laboratory and visual classification mitigates some of the uncertainty associated with the texture class assignments but for those samples where visual classification alone was employed, there is uncertainty. The primary way in which this would affect sediment EPCs is if fines were mis-classified as coarse or gravel, thereby underestimating the area of fines (which contain relatively higher COPC concentrations). The degree to which this uncertainty over or under predicts the EPC is unknown.

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156	To derive the SWAC, 187 transects along Choccolocco Creek (37 river miles; page 4-4) were sampled. What is the uncertainty associated with using a limited number of data points (approximately 5 transects per river mile) to define the surface area associated with each textural class? Did sampling density vary within the EUs? The presented SWAC is a calculation of the average PCB or mercury concentration across a designated area; it provides a false representation of precision, and has an associated measure of uncertainty or variability. U.S. EPA guidance for sediment sampling states that samples be collected from depositional areas, (U.S. EPA/ERT 1994). Is it appropriate to use a SWAC approach based on sediment textural classes that include textural classes that would not be the dominant grain size found in depositional areas? It is assumed that the few samples collected from each grain size class accurately represent the range of concentrations within a given class. This is not known. This uncertainty should be included with each SWAC, and the substantial uncertainties associated with this approach should be clearly discussed in the Uncertainty Section.	Surface Weighted Average Concentration	See responses to Comment Nos. 95, 155 and 138.
157	The following T&E species are known to exist within OU-4: Painted rock snail, *Leptoxis taeniata* (Threatened), Cylindrical lioplax, *Lioplax cyclostomaformis* (Endangered), Tulotoma snail, *Tulotoma magnifica* (Threatened). Other endangered, threatened, and proposed species that may also occur in OU-4. The 2006 Problem Formulation states “Risks to threatened and endangered species will not be explicitly evaluated. Rather, appropriate surrogate species will be identified as focal points for risk characterization for determining if risks to threatened and endangered species could be unacceptable. Surrogate species and appropriate risk assessment parameters will be developed in the BERA.” This was not done in the BERA and should be done.	Threatened and Endangered Species	The following revision has been made to the last paragraph of the problem formulation (Section 2): "Threatened and endangered species that may be found in OU-4 include the blue shiner (*Cyprinella caerulea*), painted rock snail (*Leptoxis taeniata*), Cylindrical lioplax (*Lioplax cyclostomaformis*), Tulotoma snail (*Tulotoma magnifica*). In addition, the United States Fish and Wildlife Service (USFWS; 2014) lists two mammals (Indiana bat [*Myotis sodalist*] and Gray bat [*Myotis grisescens*]), one bird (Red-cockaded woodpecker [*Picoides borealis*]), four flowering plants, eight clams, and two fishes (Pygmy Sculpin [*Cottus Paulus*] and Blue Shiner [*Cyprinella caerulea*]), as potentially occurring within Calhoun County, Alabama. In addition, one sensitive snail species (Wicker Ancylid [*Rhodacmea filose*] limpet), formerly thought to be extinct, has been observed in

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			lower Choccolocco Creek (O Foighil et al., 2011). Therefore, a range of risk estimates are provided in this OU-4 BERA to allow evaluation of the potential for adverse effects at both the individual and population levels."
158	The TEQ EPC is an OU4-wide mean concentration. The EPC should be the 95UCL, and should be evaluated on an assessment area basis. If data are limited, they should be evaluated on a point-by-point basis.	Toxic Equivalents (TEQ) - Congeners	See responses to Comment Nos. 165 and 163.
159	The mean TEQ values reported in Table F-1 cannot be reproduced using the values reported in Appendix E, 'Calculated TEQ values' table. Which samples were included in the total calculations? Only surface soil or sediment? Only the samples in Column N indicated as "within 50 feet of BSA"? Were duplicates removed? For Total TEQ, Non-detects (ND) Sub, the footnote needs to indicate the value used to substitute for NDs. The DL-PCB TEQ should be presented along with the total and PCDD/PCDF TEQs,	TEQ - Congeners	The approach for evaluating risk associated with TEQ has been revised. PCDD/PCDF and DL-PCB data collected within the BSAs were used to derive congener-specific BAFs, which were used to model prey tissue concentrations and calculate EPCs for dioxin/furan TEQ, DL-PCB TEQ, and total TEQ. Data included in the BAF datasets are presented in Appendix F Attachment 1 Tables F1-4 and F1-5. Modeled tissue calculations, including soil and sediment PCDD/PCDF and DL-PCB congener data included for the modeled tissue calculations (all soil and sediment data), are presented in Appendix F Attachment 2. The data employed are consistent with OU-4 samples presented in Appendix E.
160	Appendix F, page F-23: States "There is also some question whether the concentrations reported as PCB-126 are actually that congener. PCB-126 has been associated with Aroclor 1254 (Frame et al. 1996), which was only manufactured from 1974 to 1977 and was not produced in Anniston. Therefore, the possible source of any reported PCB-126 is unclear and not likely site-related." The detection limit for congeners in the Frame et al. (1996) study was 100 mg/kg. A more recent study determined the concentration of individual congeners in nine Aroclors to the sub-mg/kg level; PCB-126 was present in all nine Aroclors evaluated (Rushneck et al. 2004). This discussion should	TEQ - Congeners	The discussion related to the source of PCB 126 been removed, as PCB 126 has been found in nine Aroclors, including 1254 (Rushneck 2004). The Rushneck (2004) findings were cited in the uncertainty analysis of Appendix F and the BERA.

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	be included in the Uncertainty Section of the BERA, and the above statement should be deleted.		
161	A Table with summary data (mean, range, sample size, number of detects) for all PCDD PCDF and DL-PCBs needs to be provided.	TEQ - Congeners	Summary statistics for PCDD/DF and DL-PCB data were added to Appendix F.
162	The TEQs should be presented as toxic units. Evaluate the PCB signal versus the dioxin/furan signal.	TEQ - Congeners	Appendix F of the BERA has been revised and includes evaluation of DL-PCB TEQ in addition to PCDD/PCDF TEQ and total TEQ. Toxic units (TUs) are presented on Figures F-2 and F-3 to indicate the relative contribution of PCDD/PCDF and DL-PCBs to the total TEQ concentrations.
163	Report the number of samples that exceed the calculated site-specific risk-based concentrations (SSRBCs).	TEQ - Congeners	The TEQ evaluation is conducted as a forward assessment. HQs are presented based on AA-specific 95% UCL EPCs (soil) or maximum detected concentrations (sediment) The complete analysis is provided in Appendix F and key elements are summarized in Sections 6.1.4.1 for fish, 6.1.5.1.1 for birds, and 6.1.6.1.1 for mammals.
164	Sections 4.1 and 4.2, Page F-21: Use forward risk calculations based on 95UCL tissue concentrations to evaluate effects of TEQ exposure for mammals.	TEQ - Congeners	See response to Comment No. 163.
165	Section 4.4, Page F-22: Do not just use SSRBCs and average concentrations. Also use measured 95UCLs and forward risk calculations.	TEQ - Congeners	As described in response to Comment No. 163, DLCs were evaluated using forward risk assessment methods. Measured tissue UCLs were not included due to limited sample size. Tissue concentrations estimated in the calculations were modeled on a congener-specific basis as directed by USEPA (See response to Comment No. 27)
166	Although use of effect concentration (EC) values or dose-response curves would be the best approach, the current NOAEL and LOAEL value approach that is in the BERA will be followed. The percent effect associated with the LOAEL values will be incorporated in the risk characterization discussion.	Toxicity Reference Values (TRVs)	The percent effect associated with the LOAEL values selected as the basis for TRVs and CTCs has been included in the Appendix D text. When relevant, this information is also discussed in the risk characterization weight-of-evidence (6.3).

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167	Surface water benchmarks: 1) Table 5-2b should have the Tier II screening values for chronic and acute, not alternate values from Tables 1 and 2 in Suter and Tsao (1996). 2) Add AWQC as screening benchmarks.	TRVs – Specific TRVs	Table 5-2b includes Tier II values protective of aquatic life for barium, cobalt, vanadium, and PCBs (acute) because these lack ADEM WQ standards. Alternate screening levels specific to the individual receptor groups (aquatic plants, aquatic invertebrates, fish) from Tables 1 and 2 of Suter and Tsao (1996) are presented because the Tier II values are not specific to a particular receptor group. Surface water HQs for these receptor groups are presented using the values in Table 5-2b as well as ADEM WQS. NAWQC values for chromium, lead, and PCB are equivalent to ADEM values. For mercury, the NAWQC is higher for chronic and lower for acute and is based on inorganic mercury. Mercury ADEM values are based on methylmercury but apply to total mercury. Both the NAWQC and ADEM WQS for mercury were evaluated.
168	Soil Benchmarks: · The Oak Ridge National Laboratory (ORNL) values are either the lowest reported LOEC or the 10th percentile of observed LOEC values. These values are assumed to be protective, as many studies were carried out under high bioavailability conditions, or on unweathered spiked soils. The text should be revised to clarify the soil screen.	TRVs – Specific TRVs	Soil benchmarks are described in Section 2.3 of Appendix D as follows: "...These benchmarks are considered conservative screening levels equivalent to NOAELs. Although the ORNL values are either the lowest reported low observed effect concentration (LOEC) or the 10th percentile of observed LOEC values, these values are assumed to be protective, as studies were carried out under high bioavailability conditions, or on unweathered spiked soils."
	· The original soil benchmark for PCBs was the Region 4 screening value. Those values have been updated, and the equilibrium partitioning (EqP) approach should be used to develop the PCB soil screening numbers.		The USEPA Region 4 EqP-based approach was used to develop PCB soil screening values. NOAEL-equivalent and LOAEL-equivalent soil benchmarks were calculated using the EqP equation and toxicity threshold for aquatic invertebrates derived by Fuchsman (2006) using NAWQC methodology (Stephen et al. 1985). The NOAEL and LOAEL PCB soil benchmarks based on this approach are 3.6 and 36 mg/kg, respectively. Methods are described in detail in Appendix D Section 2.3.

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	· EPA provided several papers describing effects of PCBs in soil on soil invertebrates.		The USEPA-provided papers were reviewed as noted in response to Comment No. 168a, the soil PCB benchmark was revised based on the USEPA Region 4 screening value calculation methodology (2015) and a water quality benchmark from Fuchsman et al. (2006). The papers provided by USEPA related to this comment (Jensen et al. 2012, Paine et al. 1993, and Parmalee et al 1997) are discussed in the uncertainty section (Section 6.2.3.4).
	· Effects of mercury in soil on soil invertebrates have been reported by Son et al. (2007).		Son (2007) provides a Hg LOAEL-equivalent benchmark of 0.23 mg/kg based on reduced offspring production in springtail (collembola). This LOAEL value was evaluated in addition to the NOEC-equivalent SL for soil (0.1 mg/kg) from Efroymson et al. (1997) that was evaluated in the draft BERA. An alternate effect level of 2 mg/kg based on neutral population growth rate in soil invertebrates from Son et al. (2007) is also discussed in Section 6.2.3.4.
169	Amphibian/Reptile : 1) As per the Phase 2 FSP, concentrations of COCs measured in surface water will be compared with TRVs protective of herptiles, primarily those for amphibian eggs and larvae. The TRVs should be compared with the calculated 95UCL EPCs. This comparison should be included in the risk characterization section. 2) Tissue-based TRVs for amphibians and reptiles are scarce. Measured tissue concentrations in amphibians should be compared to available CTCs. This discussion can remain in the uncertainty section.	TRVs – Specific TRVs	Amphibian toxicity studies were provided by EPA for PCBs and Hg. Values for total PCBs (Aroclor 1254) include an acute survival LOEC of 10 mg/L. Lower effects values provided by EPA are not appropriate (for congeners or effects other than growth, reproduction, or survival). This value is orders of magnitude higher than the WQS value of 0.014 µg/L and is not appropriate for evaluation of chronic exposures. For mercury, aqueous toxicity thresholds are similar to or less sensitive than ADEM WQS. Available benchmarks and tissue CTCs, included data provided by EPA, have been evaluated in the Uncertainty Section (Section 6.2.3.8 and Section 6.2.3.13), but are not used as a LOE.
170	Fish: The benchmark value of 2.2 nanograms (ng) TCDD/gram (g) lipid, was selected as the TRV for fish. This value is based on the 95% UCL effects residue (based on the geometric mean of no observed effect residue [NOER]	TRVs – Specific TRVs	The UCL of the 90th percentile was selected because the Steevens (2005) dataset includes a large proportion of salmonid data. Salmonids are more sensitive to TCDD and are not present in OU-4.

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	and lowest observed effect residue [LOER] values) from the Steevens et al. (2005) study. Use of the UCL of the 90th percentile is not protective. The mean value (0.699 ng/g lipid) should be used as the TRV.		Discussion of uncertainty associated with species sensitivity and evaluation of the mean of the 90th percentile is included in the uncertainty analysis (Section 5.2) of Appendix F.
171	Bird Egg Critical Tissue Concentration: · The critical threshold for tPCBs in eggs of 6 micrograms per gram wet weight (ug/g ww) for intermediate sensitivity bird species cited in Harris and Elliott (2011) should be used as the egg tissue CTC for mid-sensitivity birds. This equals 24 ug/g dry weight (dw), assuming 75% moisture. * · If the Fernie et al. (2001a) value is retained, the following issues should be clearly addressed in the risk characterization section: 1) The kestrel is a low-sensitivity species, not a mid-sensitivity species. 2) At the egg concentration identified as a LOAEL from the Fernie et al. (2001a) study, 50% of nestlings produced by PCB-exposed adults died within 3 days of hatching, and 60% of PCB-exposed adult pairs with hatchlings failed to produce fledglings (Fernie et al. 2001b). In ovo exposure suppressed egg laying completely in 25% of PCB-exposed females (Fernie et al. 2001a).	TRVs – Specific TRVs	The recommended CTC of 6 µg/g ww is based on hatching success from Arenal et al. 2004 and Neigh et al. 2006. Arenal et al. (2004) is based on the European starling, which is a AhR Type I (high sensitivity) bird. As high sensitivity birds are evaluated in the BERA based on the domestic chicken TRVs, this value is not applicable. For Type 2 or 3 species, the egg concentrations reported in Neigh et al. 2006 are associated with reduced clutch size, but there were numerous confounding factors in the study and authors concluded that birds were not impacted. An alternate egg LOAEL TRV of 16.5 mg/kg ww based on mid-sensitivity birds was developed based on data from Peakall et al 1972, Peakall and Peakall 1973, as described in Section 5.5.1.1 and Appendix D (Section 3.1.2.2).

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172	Mid-sensitivity Bird PCB TRV: The ingestion rate and body weight values selected to convert dietary doses in mg COPC/kg diet to milligrams COPC per kilogram body weight per day (mg COPC/kg BW/day) have a large impact on the calculated TRV value. For the mid-sensitivity bird study, ERT used an ingestion rate (15 g/day) and body weight (160 g) for ring doves cited in Schwarzbach et al. (1991) for the conversion. Arcadis used a body weight of 150 g (source not specified—Table D-4) and the allometric equation from Nagy (2001) to calculate a food ingestion rate. Because the article (Peakall and Peakall 1973) noted that the diet concentration was wet weight, the fresh matter ingestion rate equation was used, resulting in a LOAEL TRV of 4.3 mg/kgBW/day. However, the article also noted that diet percent moisture was 10%, which is more of a dry weight diet than COCs incorporated into prey tissue (typically 60 to 85% moisture). Recalculating the TRV using the dry matter ingestion rate equation results in a LOAEL value for the Peakall study of 1.3 mg/kgBW/day, which is very close to the Koval et al (1987) study TRV of 1.4 mg/kgBW/day. The Nagy equations are a valid method for estimating food ingestion rate, the use of the Koval TRV as the mid-sensitivity bird dietary TRV is acceptable.	TRVs – Specific TRVs	Comment noted.
173	Mammal PCB TRV: Arcadis will use the TRV associated with the growth endpoint from the Restum et al. (1998) study. The alternate mink TRVs from the Aulerich and Ringer (1977) paper will be removed. This value is similar to an effect level reported in a more recent study by Bursian et al. (2013), which identified a dietary concentration estimated to result in 20% kit mortality at six weeks of age (0.34 µg PCBs/g feed; 0.033 mg/kg/day). It is also similar to an EC20 value that can be calculated using a dose response curve derived by Fuchsman et al. (2008), which describes the relationship between PCB exposure and production of surviving kits per female (EC20 = 0.059 mg/kg/day) [the paper reports an EC50 value of 0.17 mg/kg/day].	TRVs – Specific TRVs	Mink TRVs for growth (LOAEL = 0.05 mg/kg bw-d) and reproduction endpoints (NOAEL/LOAEL 0.05 / 0.1 mg/kg BW-d) from Restum et al. (1988) are both included in the revised BERA, and the TRV from Aulerich and Ringer (1977) was removed. The updated mink TRVs are described in Section 5.4.6 and Appendix D (Section 3.2.2).

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174	Field studies should be discussed as supporting information when discussing the HQ calculation results. Revise the text so it does not state that field study results were excluded from the risk evaluation phase.	TRVs – Use of field studies in TRV development	Revised text does not state that field studies were excluded. Field studies are discussed in specific examples as appropriate (e.g., Housatonic shrew study is discussed in the uncertainty analysis regarding small mammal analysis).
175	Include the Foekema et al. (2014) study of the effects of a mixture of 25 PCBs similar to Aroclor 1254 in the common sole (*Solea solea*). The study reported effects concentrations in lipid weight which can be converted to whole body concentrations using the lipids measured in sole in Foekema et al. (2012). Increased mortality, disrupted larval development, and edema between 6 and 40 days post fertilization was observed at a tissue-residue of 3.48 mg/kg. Disrupted or delayed development of larvae between 6 and 40 days post fertilization was observed at a tissue-residue of 6.82 mg/kg. Lethality to 50% of larvae, within 6 days post fertilization, occurred at a tissue-residue of 7.48 mg/kg. These results are of similar magnitude to the Hansen et al. (1973) study of the sheepshead minnow. The Foekema et al. (2014) study appears to be more conservative for establishing a fish tissue-based TRV for PCBs. Please incorporate the Foekema et al. (2014) study and lower the TRVs. EPA compared the lipids-normalized fish tissue concentrations were compared to the fish tissue benchmarks from Foekema et al. (2014) for disrupted development of common sole larvae. Based on calculated uptake curves for PCBs into forage fish, bottom fish, and predatory fish, the organic carbon normalized concentration in the fines to achieve the 341 mg/kg-lipid ww LOAEL would be 94 mg/kg-oc for the predatory fish, 255 mg/kg-oc for bottom fish, and 420 mg/kg-oc for forage fish. Apart from the lower reaches, the concentrations of PCBs in fine sediments are greater than the estimated protective values for the fish community.	TRVs – Specific updates requested using literature sources that have been released since TRVs were developed	The Foekema et al. (2014) study measured only 25 congeners associated with Aroclor 1254. The resulting TRVs are similar to TRVs from Hansen et al. (1973) used in the BERA in which Aroclor 1254 was measured. The discrepancy may be due to analytical methods, not toxicity. Therefore, the fish tissue TRVs based on Hansen et al. (1973) BERA were retained.
176	The benthic invertebrate tissue TRVs for PCBs and Hg noted in comments 7 (Van Geest et al. 2011) and 8 (Mendez-Fernandez et al. 2015) should be included as tissue TRVs.	TRVs – Specific updates requested	See responses to Comment Nos. 7 and 8.

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177	One commenter noted that tissue residue data could be matched up with closest Aroclor profile and then the appropriate Aroclor toxicity data could be used to estimate risk. This approach was used for Housatonic River BERA. See Moore et al. (2016).	TRVs - Specific updates requested	This approach does not account for differential uptake and metabolism of PCB congeners. Historical sources of PCBs are well understood within OU-4. This approach is not necessary to evaluate PCB risk in the revised BERA.
178	Review Moore et al. (2016) to determine if the small mammal tissue TRV should be revised.	TRVs - Specific updates requested	Moore et al. (2016) used a Aroclor 1254 dose-response curve for small mammals from the Spencer et al. (1982) gestational exposure study. The EC20 (assumed to be LOAEL-equivalent) appears to be between 1 and 10 mg/kg-d, which is greater than both the NOAEL and LOAEL values in the Draft BERA from the McCoy et al. (1995) multigenerational study with mice. No change to the small mammal TRV employed in the BERA based on this comment.
179	Dioxin/furan concentrations in fish tissues were not compared to a tissue-residue TRV possibly because one was not available. Dioxin/furan concentrations in fish tissues were in the hundreds of pico-grams per gram (pg/g) and did not exceed the tissue-residue TRVs for dioxin in fish of 3 – 16 nano-grams per gram (ng/g) (King-Heiden et al. 2012).	TRVs - Specific updates requested	King-Heiden (2012), a review article, cites the tissue CTC range for fish of 3-16 ng/g based on wasting/lethality in adult yellow perch, carp and bullhead, adult rainbow trout, bluegill, and largemouth bass. This comparison has been added to Appendix F.
180	Comment 37 recommends evaluating mercury as both MeHg and as inorganic Hg. Bird and mammals TRVs for dietary exposure to inorganic Hg are needed for the BERA. For birds, use the NOAEL of 0.45 mg/kgBW/day and the LOAEL of 0.9 mg/kgBW/day from Hill and Schaffner as cited in Sample et al. (1996). For mammals, use the LOAEL of 0.37 mg/kgBW/day reported by Atkinson et al. (2001).	TRVs - Specific updates requested	An analysis comparing methyl and inorganic mercury is provided in the uncertainty analysis (Section 6.2.3.9) for Carolina wren using the methylmercury TRVs from the BERA, inorganic TRVs from Hill and Schaffner (1976) for birds and from Aulerich et al. (1974) and Atkinson et al. (2001) for mammals, and the percentage methylmercury in prey items provided by USEPA (Table 6-22).

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181	There has been no consideration of the congener mixture at the site versus the congener mixtures in the various toxicity test results. Toxicity varies between Aroclor mixtures quite substantially for wildlife. Aroclors were only analyzed in fish, crayfish and mollusks; 1242, 1254, 1260 and 1268 were the only detected Aroclors. 1242 was detected in 63, 30, and 78% of fish, crayfish, mollusks respectively. 1254 was 100% detects. 1260 was detected in 100, 96, and 100% of fish, crayfish and mollusks. 1268 was detected in 2 of 135 fish samples (1%), and not detected in crayfish or mollusks. Measured concentrations of Aroclor 1254 were highest in all species groups, followed by 1260. Discuss the toxicity of site-specific Aroclors versus the Aroclors used to develop the TRVs.	TRVs – Additional Issues	Aroclor composition in fish, crayfish and mollusc tissue compared with Aroclors that provide the basis for CTCs are discussed in the uncertainty analysis in Section 6.2.3.12 and 6.2.3.14.1.
182	Page 6-75 identifies a NOAEL soil concentration of 21.1 mg/kg from a study conducted on shrews for the Housatonic. According to Moore et al. (2016, section 9 of supplementary material), this concentration is associated with about 15% additional mortality of shrews than occurs at 1-2 mg/kg dw tPCBs (Aroclor 1254 is dominant mixture at the site). The Housatonic River shrew study did not measure concentrations of PCBs in the diet or in the shrews. The degree of exposure to shrews is too uncertain or unknown to use the study to estimate a toxicity reference value. The power of the Housatonic River study to measure an effect was low due to low sample sizes. The study was confounded by the effect of the habitat differences among the sampling areas. During flooding it was suspected that the shrews moved among the sampling areas, so that shrews collected in one area might not have been exposed exclusively to the soils in that area. Due to the uncertainty in exposure, and the observed additional mortality, this concentration should be a LOAEL, not a NOAEL, concentration.	TRVs – Additional Issues	The Housatonic values were not included in the draft BERA as specific TRVs but were included as uncertainty discussion. The last sentence in the 3rd paragraph of Section 6.2.3.7 has been revised as follows: "While there is some uncertainty in this direct comparison because of possible limitations in the Housatonic River field study, different study design and differences in endpoints that were measured, this comparison does provide context for interpretation of modeled HQs primarily based on laboratory studies, relative to actual field observations of the robustness of a local community or population."

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183	Sensitivity analyses can be performed to see how much of an effect varying parameters has on results, and which parameters have the most influence in driving specific risk estimates.	Uncertainty Analysis	A quantitative sensitivity analysis was not performed. Additional examples of parameter sensitivity were added to the uncertainty analysis, such as evaluation sediment depth profile and the predictive nature of Aroclor vs homolog data (Section 6.2.1.1), implications of evaluation of mercury as 100% methyl mercury (Section 6.2.3.9) and evaluation of alternative BAFs for emergent insects and small mammals (Section 6.2.2.1)
184	Comments regarding definition of the riparian zone as the 100-foot corridor adjacent to the river: This was based on a historic agreement. Concentrations in the floodplain are higher near the river. If you used the whole riparian zone, it would dilute those samples. It was an operational decision--spatially restricted, but representative of highest exposure concentrations. This should be explained more clearly in the BERA.	100-foot Riparian Corridor	The following text replaces the text in the 3rd paragraph in Section 4.1. "The location, size, and shape of the foraging area may be influenced by the available food base and habitat due to the potential uncertainty in assigning EUs/AAs for a given receptor species. In addition, the PCB concentrations tend to be higher in the floodplain area closest to the creek with concentrations decreasing with distance away from the creek. Because, some receptors feed primarily on aquatic prey either within the creek itself or within the riparian corridor (the area of the floodplain closest to the creek) and other receptors feed more generally within the larger floodplain, the floodplain areas of each reach were divided into multiple EUs." The last sentence of the 2nd bullet below this paragraph was revised to "The riparian EUs and AAs are defined operationally as a 100-foot wide riparian corridor along both sides (200 feet in total) of the creek and are associated with creek miles. This 100-foot wide riparian zone is also consistent with much of the upland areas lining the creek with natural vegetation as opposed to the open field areas that are located beyond this treed buffer zone."

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