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River System
Baseline Ecological Risk Assessment (BERA)

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Acronyms and Abbreviations

AdH	adaptive hydraulics
amsl	above mean sea level
ANOVA	Analysis of Variance
AVS/SEM	acid volatile sulfides/simultaneously extracted metals
AWQC	ambient water quality criteria
BERA	Baseline Ecological Risk Assessment
BHC	hexachlorocyclohexane
BIC	benthic invertebrate community
$CaCO_3$	calcium carbonate
CBR	critical body residue
CF	correction factor
cfs	cubic feet per second
COEC	constituent of ecological concern
COPEC	constituent of potential ecological concern
CR_A	Clinch River Reach A
CR_B	Clinch River Reach B
CR_Ref	Clinch River Reference Reach
CRM	Clinch River mile
CSM	Conceptual Site Model
cy	cubic yards
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DMA	direct mercury analysis
DO	dissolved oxygen
DOC	dissolved organic carbon
DQA	Data Quality Assessment
EC	effect concentration
Eco-SSL	Ecological Soil Screening Levels
EDDMapS	Early Detection and Distribution Mapping System
EE/CA	Engineering Evaluation/Cost Analysis
EPC	exposure point concentration
EPSBTU	equilibrium partitioning sediment benchmark toxic unit
EPT	Ephemeroptera, Plecoptera, and Trichoptera
ER_A	Emory River Reach A
ER_B	Emory River Reach B
ER_C	Emory River Reach C
ER_Ref	Emory River Reference Reach

ERA	Ecological Risk Assessment
ERDCWES	U.S. Army Corps of Engineers Engineering Research and Development Center at Waterways Experiment Station
ERED	Environmental Residue-Effects Database
ERM	Emory River mile
ES	Environmental Standards, Incorporated
°F	degrees Fahrenheit
ft	feet
GSI	gonadsomatic index
ha	hectare
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
IC	inhibition concentration
ICP/MS	inductively coupled plasma/mass spectrometry
IDW	inverse distance weighted
Jacobs	Jacobs Engineering
KIF	Kingston Fossil Plant
LD	lethal dose
LERM	Little Emory River mile
LOAEL	low observed adverse effect level
LOE	line of evidence
LOEC	low observed effect concentration
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
MDL	method detection limit
mg	milligram(s)
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mL	milliliter(s)
mm	millimeter(s)
MPC	maximum permissible concentration
n	number
NN	nearest neighbor
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
NPDES	National Pollutant Discharge Elimination System
ORNL	Oak Ridge National Laboratory
ORP	oxidation-reduction potential
PAH	polycyclic aromatic hydrocarbon
PCA	principal component analysis

PCB	polychlorinated biphenyl
PLM	polarized light microscopy
QAPP	Quality Assurance Project Plan
QA/QC	quality assurance/quality control
r	correlation coefficient
RFAI	Reservoir Fish Assemblage Index
RL	reporting limit
SAP	Kingston Ash Recovery Project Non-Time-Critical Removal Action for the River System Sampling and Analysis Plan
SAS	SAS Institute Inc.
SC	specific conductivity
SedFlume	ERDCWES laboratory flume
Site	Tennessee Valley Authority Kingston Fossil Plant Release Site, in Roane County, Tennessee
SLERA	Screening-Level Ecological Risk Assessment
TAL	Target Analyte List
TDEC	Tennessee Department of Environment and Conservation
TDS	total dissolved solids
TIE	toxicity identification evaluation
TOC	total organic carbon
TR_A	Tennessee River Reach A
TR_B	Tennessee River Reach B
TR_Ref	Tennessee River Reference Reach
TRM	Tennessee River mile
TRV	toxicity reference value
TSS	total suspended solids
TVA	Tennessee Valley Authority
TWRA	Tennessee Wildlife Resources Agency
TWRC	Tennessee Wildlife Resources Commission
UCL	upper confidence limit
µg/L	micrograms per liter
USACE	U.S. Army Corps of Engineers
USDOE	U.S. Department of Energy
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
WOE	weight of evidence

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Executive Summary

This Baseline Ecological Risk Assessment (BERA) evaluates the potential effects on biota of ash residuals in the river system at the Tennessee Valley Authority (TVA) Kingston Fossil Plant (KIF) Release Site, in Roane County, Tennessee (the Site). This BERA was developed in support of the *Kingston Ash Recovery Project, Non-Time Critical Removal Action, River System Engineering Evaluation/Cost Analysis (EE/CA)* (TVA 2012), which evaluates alternatives for restoration of the river system impacted by the December 22, 2008 ash release. This BERA is consistent with the U.S. Environmental Protection Agency (USEPA) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA 1997a). As such, it consists of a Problem Formulation, exposure analysis, effects analysis, and risk characterization.

A comprehensive list of potentially exposed and sensitive ecological receptors was developed during the Problem Formulation. Assessment endpoints for the river system BERA include maintenance and reproduction of balanced communities or populations of the following receptor groups:

- benthic and pelagic fish
- benthic invertebrates
- aquatic and shoreline plants
- aquatic- or riparian-feeding birds and mammals
- aerial-feeding insectivorous birds and mammals
- aquatic- or riparian-feeding amphibians and reptiles.

General Problem Formulation – The TVA KIF is located at the confluence of the Emory and Clinch Rivers and near the confluence of the Clinch and Tennessee Rivers on Watts Bar Reservoir in Roane County, Tennessee. During the release event, approximately 5.4 million cubic yards (cy) of wet fly ash and bottom ash flowed into the river system adjoining the KIF. The initial ash release traveled upriver as far as Emory River mile (ERM) 5.75, and eventually was transported into the Clinch River, and as far downriver as Tennessee River mile 564 by high river flow events. Dredging in the Emory River began on March 20, 2009. Hydraulic dredging continued until May 29, 2010, with mechanical dredging for pockets of ash upstream of ERM 1.75 continuing until August 2010. Although dredging during the time-critical removal action removed approximately 3.5 million cy of released ash and sediment, because of legacy contaminants from former U.S. Department of Energy operations at Oak Ridge, no dredging was conducted downstream of ERM 1.75. Consequently, approximately 532,000 cy of ash are estimated to remain in the river system (Jacobs 2011a). The potential current and future ecological effects of this residual ash are the focus of this BERA.

An ecological Conceptual Site Model was developed to identify the potential mechanisms by which these different receptor groups might be exposed to ash and ash-related constituents of potential ecological concern (COPEC). Potential complete exposure pathways include: 1) direct contact with ambient media (i.e., seasonally-exposed and submerged sediments, sediment porewaters, and surface waters), 2) ingestion of COPECs in food items or surface water, and 3) incidental ingestion of seasonally-exposed and submerged sediments.

Multiple measures of exposure and effects (measurement endpoints) were evaluated for each of the assessment endpoints. Examples of exposure measures include concentrations of COPECs measured in environmental media, in prey items used for dietary exposure modeling, and in tissues of ecological receptors. Examples of effects measures include survival, growth (or biomass), and reproduction of ecological receptors. These measurement endpoints comprise the lines of evidence (LOE) used to characterize ecological risk.

An extensive suite of biological and environmental data was collected for this assessment, resulting in multiple LOEs for most receptors. A weight of evidence (WOE) paradigm was used to evaluate multiple LOEs in the risk characterization process. This was accomplished by first characterizing risk for each individual LOE, and then characterizing risk based on the entire body of evidence. At both levels of evaluation, the magnitudes and likelihoods of potential adverse ecological effects were estimated. The estimated potential risk to the assessment endpoint and the level of confidence in that estimate was summarized by receptor group. Potential risk was categorized as Negligible, Low, Moderate, or High. Confidence in the risk determination was categorized as being Low, Moderate, or High. These elements of the risk characterization for each receptor group are summarized in Exhibit ES-1.

The final characterization of risk also includes a determination as to whether risk management actions are recommended for an assessment endpoint. This recommendation is based on the WOE and provides a means for further characterizing the likelihood and magnitude of impairment of the assessment endpoint. Risk management recommendations are discussed further in the final section of the BERA and below. The receptor groups for which risk management is recommended are identified in Exhibit ES-1.

COPECs that might contribute to the risks were identified for each LOE and receptor for which a potential risk was indicated. A constituent was not considered to be a constituent of ecological concern (COEC) until the final determination of risks was made via the WOE process for each assessment endpoint. COECs are identified only for receptor groups for which risk management is recommended, as summarized in Exhibit ES-1.

Exhibit ES-1. Weight of Evidence Risk Characterization Summary by Receptor

Receptors	Potential Risk	Confidence in Risk Determination	Risk Management Recommended?	COECs
Fish	⊘	Moderate		
Benthic Invertebrates	◒ (ER) ○ (CR)	High	✓	As, Se, Ash
Aquatic Vegetation	○	Moderate		
Birds				
Piscivore - Heron	⊘	Moderate		
Piscivore - Osprey	⊘	Moderate		
Insectivore - Killdeer	○	Moderate	✓	As, Se
Omnivore - Mallard	○	Moderate		
Herbivore - Wood Duck	⊘	Moderate		
Aerial Insectivore - Tree Swallow	○	Moderate	✓	Se
Mammals				
Carnivore - Mink	○	Moderate		
Omnivore - Raccoon	○	Low		
Herbivore- Muskrat	○	High		
Aerial Insectivore - Gray Bat	○	Low		
Amphibians				
American Toad, Spring Peeper and Chorus Frogs	○	Moderate		
Reptiles				
Musk, Snapping, and Softshell Turtles	○	Moderate		

⊘ = risks are negligible; ○ = risks are low; ◒ = risks are moderate; ● = risks are high; ✓ = risk management is recommended.

COECs = Constituents of ecological concern.

ER = Emory River; CR = Clinch River.

Fish – The WOE suggests that ash and ash-related COPECs pose a negligible risk to the fish communities in the Emory, Clinch, and Tennessee Rivers. The available LOEs provide a moderate level of confidence in this risk determination, primarily due to the overall concordance of the LOEs.

The fish community surveys are given the greatest weight in the estimation of risk, because they are a direct measure of the assessment endpoint to be protected. These surveys provide robust evidence that the ash has not had measurable adverse effects on the fish communities at the Site.

The next-most-relevant LOEs are fish toxicity tests and fish reproduction studies. The results of each of these studies are consistent with the fish community survey results. This suggests there are no ecologically important impacts to individuals that might manifest themselves at the population or community level.

Fish health studies also suggest that no notable adverse effects are occurring at the organism or sub-organism level. Two types of data were compared with literature-derived effects values: COPEC concentrations in fish tissues and COPEC concentrations in surface waters. Several COPECs exceeded critical body residues (CBR), but CBRs are inherently conservative and uncertain effects values that provide only weak evidence of risk to the fish community. There were only a few limited exceedances of water quality effects values. Both of these types of literature-derived effects values are considered to have low relevance to the assessment endpoint. These LOEs are more appropriately used to help explain effects observed in community surveys, toxicity tests, and reproduction studies, rather than to predict effects. Comparisons with literature-derived effects values contribute little to the overall assessment of risks to fish, given the lack of effects observed for the community surveys, toxicity tests, and reproductive competence evaluations.

Benthic Invertebrates – The WOE suggests that ash and ash-related COPECs pose a moderate risk to Benthic Invertebrate Communities (BICs) in the Emory River and a low risk to BICs in the Clinch River. Statistical analysis indicates exposures to ash and arsenic appear to be most closely associated with the observed effects. Selenium also shows evidence of bioaccumulation in benthic invertebrates. The available LOEs provide a high level of confidence in these risk determinations and high confidence that risks are no greater than estimated.

The BIC surveys are given the greatest weight in the estimation of risk, because they are a direct measure of the assessment endpoint to be protected. Variability in the BIC appears to be explained primarily by the heterogeneity of habitats and hydrology inherent in the river system, rather than the presence of ash and ash-related COPECs. The confidence in these conclusions is high, as is the relative weight given to this LOE.

Three LOEs, sediment toxicity tests, invertebrate tissue concentrations, and sediment porewater concentrations, have moderate relevance to the assessment endpoint. The results for these LOEs are consistent, but contrary to the BIC survey results. All three LOEs indicate the potential for at least low risks to benthic invertebrates with some degree of correlation with arsenic and/or ash. This is particularly true for the toxicity tests for the Emory River, and confidence in these test results is high. Tissue concentrations of arsenic and selenium were higher in the Emory River than at reference locations and were measured at potentially toxic concentrations as compared to the few applicable studies reported in the literature. Although bioaccumulation provides direct evidence of bioavailability, confidence in the ability of this LOE to predict toxicity *in-situ* is relatively low. Porewater concentrations of some ash-related COPECs, including arsenic,

exceeded literature-reported adverse effects levels, again most notably in the Emory River, but the benthic community results reflect no significant adverse effects. Although the potential for risk appears to be low, these exceedances are consistent with the observed bioaccumulation of arsenic, and with the correlation of arsenic and ash in sediments with adverse effects in the sediment toxicity tests. Confidence in the porewater LOE is moderate.

The least-relevant LOE is the comparison of sediment concentrations with literature-derived effects values. These results suggest that ash-related COPECs pose a low risk to benthic invertebrates. There is high confidence that these conservative effects values do not underestimate risks; the observation that arsenic in the Emory River is among the COPECs exceeding benchmarks is consistent with the results of sediment toxicity tests, benthic invertebrate bioaccumulation measurements, and porewater measurements.

Aquatic Vegetation – The WOE suggests that ash-related COPECs pose (at most) a low risk to aquatic vegetation in the Emory, Clinch, and Tennessee Rivers. Arsenic appears to be the ash-related COPEC most consistently associated with potential effects. Selenium does not appear to be bioaccumulating in downstream periphyton or other aquatic vegetation. The available LOEs provide a moderate level of confidence in the risk determination, and there is relatively high confidence that risks are not greater than estimated.

Tissue concentrations of COPECs in periphyton and aquatic vegetation were given the most weight in the estimation of risks because they require the least extrapolation from measured concentrations to relevant exposures. However, they are still considered to have only a moderate weight due to their moderate relevance to the assessment endpoint. The potential for risks cannot be fully excluded based on these data, but risks are expected to be low, at most.

The least weight is given to the LOEs based on ambient media concentrations of COPECs. Of these, concentrations in sediments and porewaters are expected to be the most closely associated with potential ash exposures. The low potential for risks indicated by these data is consistent with the estimated low risks based on tissue concentrations.

Aquatic- and Riparian-Feeding Birds – The WOE suggests that ash-related COPECs pose a negligible risk to piscivorous (heron and osprey) and herbivorous (wood duck) birds that forage in and along the river system. Only the riparian-feeding insectivorous killdeer and omnivorous mallard, appears to be at some risk from exposure to ash-related COPECs. The killdeer dietary exposure model suggests that arsenic and selenium associated with the ash pose a low risk in the impacted reaches of the Emory and Clinch Rivers, and a negligible risk in the impacted reaches of the Tennessee River. Concentrations of these COPECs measured in benthic invertebrates from the Site are the primary drivers for the estimated risk to riparian-feeding insectivorous birds.

There is moderate confidence in the negligible risk determinations for heron, osprey, and wood duck, the three receptor species that do not feed predominantly on invertebrates. Risk to piscivores is based on multiple LOEs, including egg data that has moderate relevance to the assessment endpoints. Dietary exposure models are the only LOE for killdeer, mallard, and wood duck. There is moderate confidence in the negligible risk determination for species not feeding predominantly on invertebrates. There is moderate confidence in the low risk determination for insectivores; however, there is only a small probability (< 25 percent) of risk from arsenic and selenium.

Aquatic- and Riparian-Feeding Mammals – Based on the overall WOE for aquatic- and riparian-feeding mammals, there may be a low risk to mammalian species that feed on benthic invertebrates due to aluminum exposures in some reaches of the Emory, Clinch, and possibly Tennessee Rivers. The dietary exposure model suggests that aluminum poses, at most, a low risk to omnivorous and piscivorous species (as represented by mink and raccoon), but the risk could be negligible given that not all fish and invertebrate data upon which this conclusion was based were collected after dredging. Furthermore, aluminum concentrations in the prey items that are closely associated with sediments (i.e., mayfly nymphs and gizzard shad) are poorly correlated with the ash release and ash in sediments. Therefore, aluminum is not identified as a COEC based on the WOE for mammals. The WOE also suggests that ash-related COPECs pose a low risk to herbivorous mammals (muskrat) that forage in and along the river system.

Aerial-Feeding Insectivorous Birds and Mammals – The WOE suggests that ash-related COPECs pose, at most, low risks to aerial-feeding insectivorous birds and mammals that forage over the river system. Risk for tree swallows is based on site-specific reproductive measures that suggest a small, but uncertain, reduction in female fledglings produced per nesting female and indicate that the probability of selenium dietary hazard quotients exceeding 1 is low (10 to 30 percent).

No evidence exists for adverse effects of selenium on aerial-feeding mammal (bat) populations at the concentrations occurring in emergent insects at the Site using the dietary model (probability of exceeding 1 is near 0), though laboratory studies do show reduced growth in young at such concentrations. Literature-derived field studies of effects of selenium on mammal populations show no clear adverse effects, but are limited and uncertain.

There is moderate confidence in the low risk determination for tree swallows and low confidence in the low risk determination for gray bats. The moderate level of confidence for risks to tree swallows is based on the results for site-specific reproductive measures in tree swallow and the results of the probabilistic dietary risk model. Unlike for birds, the field studies of effects of selenium on mammal populations are limited and uncertain. The probabilistic dietary risk model based on those field studies indicate the probability of exceeding 1 is near 0, but other laboratory studies show reduced growth in young at such concentrations. When evaluated holistically, the WOE suggests, with moderate confidence, that overall risk

to aerial-feeding insectivores (birds and mammals) is, at most, low in the impacted reaches of the Emory, Clinch, and Tennessee Rivers.

Amphibians – The WOE suggests that ash and ash-related COPECs pose low risks to the amphibian populations at the Site (Exhibit 9-2 and Exhibit ES-1). Chemical concentrations in amphibian whole body tissues are substantially below levels associated with adverse effects. Surface water chemistry data indicate no toxicity to aquatic life, including an evaluation of amphibian-specific surface water benchmarks. Sediment chemistry data indicate no toxicity to amphibians, including an evaluation of amphibian-specific sediment toxicity benchmarks.

The risk evaluation for amphibians uses a combination of conservative and central tendency estimates for the exposure assessment, and conservative estimates for the effects assessment. This approach very likely results in a significant over-estimation of potential risks to amphibians. Even so, this evaluation suggests that COPECs at the Site do not pose substantial risks to amphibians.

Reptiles – The WOE suggests that ash and ash-related COPECs pose a low risk to the reptile populations at the Site. Turtle community surveys indicate that overall community structure is not altered and did not indicate any obvious anomalies or diseases. Chemical concentrations in turtle blood tissue indicate that concentrations of selenium and strontium were elevated compared to reference sites, documenting increased exposure for this receptor, but no adverse effects correlated with such concentrations are reported in the literature. Surface water chemistry data indicate no toxicity to aquatic life or amphibians, and less susceptible turtle indicator species are not likely to be affected. Sediment chemistry data indicate no toxicity to amphibians, and less susceptible turtle indicator species are not likely to be affected.

The risk evaluation for reptiles uses a combination of conservative and central tendency estimates for the exposure assessment and conservative estimates for the effects assessment. This approach very likely results in a significant overestimation of potential risks to reptiles. Even so, this evaluation demonstrates that COPECs at the Site pose only low risks to reptiles.

Exhibit ES-2. Receptor Risk Management Recommendations: Emory, Clinch, and Tennessee Rivers

Receptors	Emory River Reach C			Emory River Reach B			Emory River Reach A			Clinch River Reach B			Clinch River Reach A			Tennessee River Reach B			Tennessee River Reach A		
	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs
Fish	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Benthic Invertebrates	◒	✓	As, Se, Ash	◒	✓	As, Se, Ash	◒	✓	As, Se, Ash	○			○			○			○		
Aquatic Vegetation	○			○			○			○			○			⊘			⊘		
Birds																					
Piscivore - Heron	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Piscivore - Osprey	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Insectivore - Killdeer	⊘			○	✓	Se	○	✓	As, Se	○	✓	Se	○	✓	Se	⊘			⊘		
Omnivore - Mallard	⊘			⊘			○			⊘			⊘			⊘			⊘		
Herbivore - Wood Duck	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Aerial Insectivore - Tree Swallow	⊘			○	✓	Se	○	✓	Se	○	✓	Se	○	✓	Se	⊘			⊘		
Mammals																					
Carnivore - Mink	⊘			○			⊘			○			○			⊘			○		
Omnivore - Raccoon	⊘			○			⊘			○			○			⊘			○		
Herbivore - Muskrat	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Aerial Insectivore - Gray Bat	⊘			○			○			○			○			⊘			⊘		
Amphibians	○			○			○			○			○			○			○		
Reptiles	○			○			○			○			○			○			○		

⊘ = risks are negligible; ○ = risks are low; ◒ = risks are moderate; ● = risks are high; ✓ = risk management is recommended.
COECs = Constituents of ecological concern; Mgt = Management.

Risk Interpretations and Recommendations – The purpose of the risk assessment is to support the risk management decision-making process that occurs in Step 8 of the USEPA-approved EE/CA process (TVA 2012). Possible risk management response actions for the residual ash in the river system range from monitored natural recovery to capping and/or ash removal. A risk management recommendation in this BERA should not be interpreted as an endorsement of a particular response action. Rather, the recommendation for risk management is simply a means of conveying the assessors' best professional judgment as to the likelihood and magnitude of impairment of the assessment endpoint.

Risk management actions include an obvious spatial component for each receptor identified as warranting risk management. Areas and recommendations for action for one receptor may overlap with other receptors. Therefore, the determinations of risk and recommendations for risk management are summarized by river reach in Exhibit ES-2. This depiction also helps portray patterns of risks and exposure pathways.

Ecological risks from exposure to ash or ash-related COECs are, at most, moderate in the Emory River and low or negligible elsewhere in the river system. None of the assessment endpoints are estimated to be at high potential risk in any of the river reaches. Ecological risks related to residual ash and ash-related COECs are primarily associated with:

- Direct exposures to ash, arsenic, and selenium in surface sediment
- Dietary exposures to arsenic and selenium via consumption of invertebrates that inhabit ash-impacted surface sediments.

These risks are most pronounced for the BIC, largely because they are the receptors most exposed to the residual ash. They have direct and prolonged contact with ash and ash-related COECs in surface sediments and the associated sediment pore waters. The WOE for risks to the BIC based on bioaccumulation and sediment toxicity test results is tempered by the results of community surveys that show no clear association between BIC structure and ash exposure. These surveys provide direct evidence that potential effects on individuals are not being manifested at the community level; however, this interpretation is complicated by spatial variations in habitat characteristics that also affect community structure.

Risks are also indicated for wildlife that consume primarily invertebrates that inhabit surface sediments for at least one life stage. These wildlife receptors consume invertebrates that are in the surface sediments habitat, or insects that developed in the surface sediments and then emerged as flying adults.

- Selenium is most clearly associated with risks to wildlife via this exposure pathway. Selenium is known to bioaccumulate from aquatic environments to concentrations in higher trophic levels that can be toxic to wildlife, with the largest increases in concentration occurring at lower trophic levels (e.g., from water to algae and invertebrates, rather than from fish to piscivores)

- Arsenic in benthic invertebrates is associated with risks to insectivorous birds that consume aquatic life stages of invertebrates, such as the killdeer. Trophic transfer of arsenic is not typically considered a major exposure pathway, which is consistent with the limited risks to wildlife identified in this assessment.

The spatial distribution of risks is consistent with an exposure gradient for residual ash, with the most pronounced ecological risks occurring in the Emory River, fewer risks indicated in the Clinch River, and only low risks indicated in the Tennessee River. This interpretation is based on the magnitudes of risks and the distribution of receptors for which risk management is recommended.

- Risks to BICs are estimated to be moderate in the three impacted reaches of the Emory River and low in the Clinch River and Tennessee River. Risks to insectivorous birds (killdeer and tree swallow) are low in the Emory River (ER_B and ER_A) and Clinch River, but negligible in the Tennessee River
- Risk management is recommended for three receptors in the Emory River (i.e., BIC, killdeer, tree swallow), two receptors in the Clinch River (i.e., killdeer and tree swallow), and no receptors in the Tennessee River.

Taken together, these patterns for estimated risks are consistent with the interpretation that risks related to residual ash and ash-related COECs are:

- Primarily attributable to arsenic and selenium associated with ash in sediment
- Diminished with increasing distance from the release area.

This information is considered along with other factors in the EE/CA in order to further support the risk management decision-making process.

1. Introduction

This Baseline Ecological Risk Assessment (BERA) evaluates the potential effects on biota of ash residuals in the river system at the Tennessee Valley Authority (TVA) Kingston Fossil Plant (KIF) Release Site, in Roane County, Tennessee (the Site). The U.S. Environmental Protection Agency (USEPA) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA 1997a) describes an eight-step process. Figure 1-1 provides an overview of the USEPA eight-step Ecological Risk Assessment (ERA) process (USEPA 1997a). An expanded view of the components of each step is provided on Figure 1-2 (USEPA 1999a, 2000a). This process moves from conservatively biased screening analyses to more definitive, site-specific ecological risk characterization, as necessary. More specifically, an ERA is comprised of a Screening-Level ERA (SLERA) and, if necessary, a BERA.

A SLERA for the river system was conducted and presented in the *Kingston Fly Ash Recovery Project, Non-Time Critical Action Scope and Engineering Evaluation/Cost Analysis (EE/CA) Work Plan* (Jacobs Engineering [Jacobs] 2009). The SLERA evaluated the potential risks to ecological receptors exposed to detected constituents in sediments and surface waters in the vicinity of the Site. The SLERA results indicated it was not reasonable to exclude the possibility of adverse effects on biota that might be exposed to constituents of potential ecological concern (COPECS) in ash in the sediments or surface waters. Since the available data for the river system were not adequate to make a decision at that stage of the ERA process, the SLERA concluded that a BERA was warranted for this area of the Site.

A BERA Methodology was developed for evaluating the potential risks to ecological receptors in and along the river system at the Site. That methodology was used to help develop the *Kingston Ash Recovery Project, Non-Time-Critical Removal Action for the River System Sampling and Analysis Plan (SAP)* (Jacobs 2010a) which includes the BERA Methodology as Appendix B. Collection and analysis methods from the SAP were modified to some extent based on input from the agencies in spring 2010 and on a number of field change notices that were issued during sampling. A record of these changes can be found in the technical memoranda that were written for each media. These technical memoranda will also be included as part of the EE/CA Report, was submitted in June 2012.

The BERA and BERA Methodology for the river system are consistent with the following guidelines and guidance:

- *ECO-Update: Role of Screening-level Risk Assessments and Refining Contaminants of Concern in Baseline Ecological Risk Assessments* (USEPA 2001a)
- *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA 1997a)
- *Guidelines for Ecological Risk Assessment* (USEPA 1998).

This BERA is organized in a manner intended to facilitate review and interpretation of the numerous data/information sources and receptors evaluated at the Site. The selected report structure is different from typical ERA reports primarily in that the exposure analysis, effects analysis, and risk characterization are presented together for each line of evidence (LOE) for a given receptor. The individual LOEs are then integrated using a weight of evidence (WOE) process in the risk characterization for each receptor. Ultimately, the characterizations of risks for each receptor group are integrated to provide a unified risk characterization for the overall river system. More specifically, the remainder of this BERA is organized as follows:

- **Section 2. General Problem Formulation –** Presents the Problem Formulation for the overall site including: the Site and Environmental Setting, Assessment Endpoints, Conceptual Site Model (CSM), Measurement Endpoints, a description of Risk Characterization Techniques used for multiple receptors, and the Data Evaluation and Ecological Screening for the primary analytical chemistry datasets used in the BERA. Data used in the BERA are presented in Appendix A, and a summary of the data adequacy is presented in Appendix B. Some more recent data that were not included as part of the quantitative BERA analysis have been evaluated qualitatively and are presented in Appendix C. Finally, toxicological profiles developed to support the BERA are presented in Appendix D.

- **Section 3. Fish –** Summarizes the Problem Formulation elements specific to the fish community, identifies the LOEs used for fish, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for fish. Reports evaluating fish community, reproduction, health, and bioaccumulation data are presented in Appendices E through L.

- **Section 4. Benthic Invertebrates –** Summarizes the Problem Formulation elements specific to the benthic invertebrate community (BIC), identifies the LOEs used for benthic invertebrates, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for benthic invertebrates. Reports evaluating BIC, sediment toxicity tests, bioaccumulation, and sediment substrate data are presented in Appendices M through S.

- **Section 5. Aquatic Vegetation –** Summarizes the Problem Formulation elements specific to assemblages of aquatic vegetation, identifies the LOEs used for aquatic vegetation, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for aquatic vegetation. Reports evaluating aquatic vegetation data are presented in Appendix T.

- **Section 6. Aquatic- and Riparian-Feeding Birds –** Summarizes the Problem Formulation elements specific to populations of birds, identifies the LOEs used for birds, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk

Characterization for birds. Reports evaluating aquatic- and riparian-feeding bird data are presented in Appendices U through X.

- **Section 7. Aquatic- and Riparian-Feeding Mammals –** Summarizes the Problem Formulation elements specific to populations of mammals, identifies the LOEs used for mammals, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for mammals. Reports evaluating aquatic- and riparian-feeding mammal data are presented in Appendices W and Y.

- **Section 8. Aerial-Feeding Birds and Mammals –** Summarizes the Problem Formulation elements specific to populations of aerial-feeding insectivores, identifies the LOEs used for aerial-feeding insectivores, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for aerial-feeding insectivores. Reports evaluating aerial-feeding bird and mammal data are presented in Appendices W and Z.

- **Section 9. Amphibians –** Summarizes the Problem Formulation elements specific to populations of amphibians, identifies the LOEs used for amphibians, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for amphibians. Reports evaluating amphibian bioaccumulation data are presented in Appendix AA.

- **Section 10. Reptiles –** Summarizes the Problem Formulation elements specific to populations of reptiles, identifies the LOEs used for reptiles, presents for each LOE the Analysis of Exposure and Effects and the Characterization of Risks, and integrates the LOEs in the Risk Characterization for reptiles. Reports evaluating reptile bioaccumulation data are presented in Appendix AB.

- **Section 11. Risk Interpretation –** Presents a discussion and summary of the results of the risk characterization for each of the evaluated receptors and the river system as a whole.

In summary, the BERA contains the subject elements and associated technical material required by the USEPA (1997a) guidance, but the organizational structure has been tailored for conditions at, and the assessment of, the river system at the Kingston ash release area.

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2. General Problem Formulation

The Problem Formulation establishes the scope, goals, and direction of the BERA. It identifies assessment and measurement endpoints and important ecological resources that may be at risk. The Problem Formulation is Step 3 of the USEPA eight-step process (USEPA 1997a) and was presented in the approved BERA Methodology (Jacobs 2010a) for the Site.

This section presents the Problem Formulation for the Site overall, including a summary of the CSM and the assessment and measurement endpoints specified for the Site. These form the foundation for the receptor-specific sections that comprise the bulk of the BERA. The general Problem Formulation also includes a description of the risk characterization methodology used throughout the assessment. Finally, this section presents a summary of the analytical chemistry data for ambient media and biota, which are used for the assessment. Additional receptor-specific information is included in appendices and referenced in the BERA as appropriate. This section is organized as follows:

- Site and Environmental Setting, which includes site history and environmental conditions that are relevant to the assessment of ecological risks
- Assessment Endpoints and the CSM
- Measurement Endpoints for Assessment Endpoints
- Data Evaluation and BERA Screening Evaluation.

2.1 Site and Environmental Setting

Following is a summary of the site characteristics that may influence the interpretation of exposure and effects data for the Site.

2.1.1 Release History and Response

The TVA KIF is located at the confluence of the Emory and Clinch Rivers and near the confluence of the Clinch and Tennessee Rivers on Watts Bar Reservoir in Roane County, Tennessee (Figure 2-1). Ash, a by-product of a coal-fired power plant, is stored in unlined containment areas at KIF, including the former Dredge Cell that failed on December 22, 2008. While the released ash itself is primarily composed of fine silica particles similar to sand, it also contains trace amounts of arsenic, chromium, copper, lead, mercury, nickel, selenium, thallium, vanadium, zinc, and other elements that occur naturally in the coal (Jacobs 2010a).

Approximately 5.4 million cubic yards (cy) of wet fly ash and bottom ash were released during the ash release event and flowed into the surrounding area waters, including the Emory River, adjacent tributaries and sloughs, and adjoining shorelines (Figure 2-2). Evaluation of the spatial extent of ash deposits indicates

that ash initially traveled upriver as far as Emory River mile (ERM) 5.75, and eventually was transported into the Clinch River and as far downriver as Tennessee River mile (TRM) 564 by high river flow events.

Dredging in the Emory River began on March 20, 2009. Hydraulic dredging continued until May 29, 2010, with mechanical dredging in pockets of ash upstream of ERM 1.75 continuing until August 2010. The initial dredging pilot program was performed until July 20, 2009. Phase I production dredging began in August 2009 and focused on removing the greatest volume of ash in the quickest time frame to reduce the potential for upstream flooding and downriver migration. A second period of "precision" dredging (Phase II dredging) began in February 2010 to further minimize potential future ash migration downriver. This dredging focused on returning the river channel to its original (pre-release) depths while minimizing disturbance of legacy sediment. Engineering controls (silt curtains) and operational controls (i.e., reduced cutter head speed, reduced rate of advance, and reversed cutter head rotation) were implemented to minimize suspending solids during the dredging operations.

Although dredging during the time-critical removal action removed approximately 3.5 million cy of released ash and sediment, no dredging was conducted downstream of ERM 1.75 due to the presence of legacy contaminants. This includes ash in a portion of the Emory River (ERM 0.0 to 1.75) where cesium-137 was found at low levels in underlying sediment samples, as well as in the plant intake channel. Cesium-137 is not associated with the ash release, rather it is a result of historical releases from U.S. Department of Energy (USDOE) facilities on the Oak Ridge Reservation, as described in Section 2.1.5.2.

Approximately 532,000 cy of ash was estimated to remain in the river system, as described in the USEPA-approved *Kingston Ash Recovery Project, On-Scene Coordinator Report for the Time-Critical Removal Action at the TVA Kingston Fossil Fuel Plant Release Site, Roane County, Tennessee* (Jacobs 2011a). Residual ash estimates were based on interpretations of data from multiple sources, including post-dredging bathymetric data, pre-release river bathymetric data, dredging logs, visual surveys, and VibeCore™ data; these estimates therefore have inherent uncertainty.

Additional sediment samples have been collected to refine the estimates of remaining ash in the river system. Figure 2-3 depicts the most current estimates of the locations where this ash is likely present, based on interpretations of bathymetric survey information and results of VibeCore™ sampling data. Residual ash may be present in distinct pockets of ash, as well as being intermixed or interbedded with natural river sediments.

2.1.2 Historical Impacts

Military, mining, and industrial facilities have been in the area for decades and are likely sources of many of the legacy constituents in the river system. In addition, use of pesticides in mosquito control and agricultural pest control has been widespread since the 1940s. Chlordane, a legacy pollutant in this area, was

commonly used as a domestic and agricultural insecticide until 1978 and as a commercial insecticide until 1988 (Tennessee Department of Environmental and Conservation [TDEC] 2010). As a result, the Emory River arm of Watts Bar Reservoir has been included on the state 303(d) list of impaired waters as not fully supporting designated use classifications due in part to chlordane and polychlorinated biphenyls (PCB) contamination in fish (TDEC 2008). These legacy constituents are a baseline consideration in this BERA.

The section of the Emory River above the influence of the Watts Bar impoundment is listed because of mercury from atmospheric deposition. The Emory River may have elevated levels of some metals since several upstream tributaries are listed for manganese, iron, and pH from historic coal mining activities. Furthermore, a few of the upstream tributaries are also impacted by sediments or other pollutants from agriculture or development.

The Clinch River arm of Watts Bar Reservoir is also on the state 303(d) list due to PCBs, chlordane, and mercury contamination of sediments from industrial point-source discharges and atmospheric deposition. Tributaries to the Clinch River that are impaired by PCBs, chlordane, and mercury and one tributary that is listed for arsenic are potential sources of contamination to the Clinch River.

Some of this contamination is a result of former USDOE operations on the Oak Ridge Reservation that resulted in the contamination of sediments by radioactive (cesium-137) and other constituents, including PCBs (TDEC 2008). The primary concern is PCB contamination in fish. Other legacy contaminants include radioactive materials and metals. Information gathered during the implementation of an interagency agreement narrowed the contaminants of interest to cesium-137 and mercury (TVA 2009a). Because of cesium-137 contamination of sediments, the lower portion of the Emory River (ERM 0.0 to 1.8) was not dredged during the time-critical removal action.

2.1.3 Physical Setting

The ash was released into the Emory River, which flows from the Cumberland Plateau into the Watts Bar Reservoir. The Watts Bar Reservoir was formed when TVA constructed Watts Bar Dam in 1942 at TRM 529.9. Watts Bar Reservoir consists of three “arms:” 72 miles of navigable water on the Tennessee River (to Fort Loudoun Dam at TRM 602.3), 12 miles of navigable water on the Emory River, and 23 miles of navigable water on the Clinch River (to Melton Hill Dam at Clinch River mile [CRM] 23.1) (Figure 2-4).

Watts Bar Reservoir contains approximately 39,000 acres of surface water and 721 miles of shoreline at full pool (TVA 2009a). The reservoir is contained within portions of Meigs, Rhea, Roane, and Loudon counties in eastern Tennessee. The drainage basin encompasses approximately 17,310 square miles in Tennessee, North Carolina, and Virginia. Approximately 1,834 miles of streams drain directly to the reservoir; the majority of tributary inflow comes from five streams, one of which is the Emory River (TVA 2009a).

Exhibit 2-1. Tributary Information

Name	Drainage Area (square miles)	Confluence
Poplar Creek	136	CRM 12
Emory River	865	CRM 4
Whites Creek	138	TRM 545
Piney River	137	TRM 532
Little Tennessee River	2,630	TRM 601

Acronyms and Abbreviations:
CRM – Clinch River mile
TRM – Tennessee River mile

Source:
TVA 2009a

2.1.3.1 Climate

Climate in the region is warm during summer when average daily temperatures tend to be in the 70s degrees Fahrenheit (ºF) and cold during winter when temperatures tend to be in the 30s ºF. The warmest month of the year is July with an average maximum temperature of 87 ºF, while the coldest month of the year is January with an average minimum temperature of 25 ºF. Temperature variations between night and day tend to be moderate during summer with a difference that can reach 23 ºF, and moderate during winter with an average difference of 22 ºF (Jacobs 2010a).

The annual average precipitation at Kingston is 53.23 inches. Rainfall is fairly evenly distributed throughout the year. The wettest months of the year occur between November and April, with highest average monthly precipitation in March of 5.70 inches. The driest months of the year occur in August through October (Jacobs 2010a; National Weather Service 2006).

2.1.3.2 Hydrology

The Fort Loudoun Reservoir discharge comprises approximately 67 percent of the flow into Watts Bar Reservoir (mean annual flow of 18,200 cubic feet per second [cfs]). Melton Hill Reservoir contributes approximately 19 percent of the flow into Watts Bar Reservoir (mean annual flow of 5,000 cfs). Local streams and rivers contribute the remaining 14 percent of inflow to Watts Bar. Watts Bar Reservoir has an annual mean discharge of approximately 27,000 cfs. Flows from the Tennessee and Clinch Rivers are controlled by dams; the Emory River flow is unregulated. The seasonal reservoir drawdown decreases the water level in the main channel by approximately 5 feet (ft) from full summer pool (TVA 2009a). The normal summer and winter pool levels of Watts Bar Reservoir range from 741 and 735 ft above mean sea level (amsl).

The Site is located in an area of complex hydrology at the confluence of three rivers with distinctly different water quality signatures. Water flow and the mixing of these distinct water bodies is greatly influenced by dam operations, wind direction and speed, and by cooling needs at KIF.

The water intake for KIF is located at ERM 1.8. When the nine power production units are operating at KIF, the intake draws approximately 2,200 cfs. Because the Emory River often has flows of less than 1,000 cfs, the majority of water that the intake draws is from the Clinch River, which is drawn upstream from its confluence with the Emory River. TVA has facilitated this use of cold Clinch River water by constructing an underwater dam at CRM 3.9 that pools the colder (in summer) Clinch River water to be drawn up the Emory River to the KIF intake (TVA 2011). Cooling water from KIF is discharged to the Clinch River at CRM 2.5.

2.1.3.3 Geology

The Emory River watershed drains a portion of the Cumberland Plateau, which is dominated by sandstone, shale, and coal, and has numerous sandstone bluffs (Figure 2-5). The lower Clinch River watershed is predominately characterized by karst topography. Karst areas are formed when underlying soluble mineral deposits, such as limestone, dolomite, gypsum, and rock salt, are dissolved by surface water or groundwater. The dissolution of these deposits forms caves, disappearing streams, springs, and sinkholes; groundwater flows through underground microfractures, bedrock plains, and water-enlarged channels (TDEC 2010).

Alluvium and/or residuum generally cover bedrock. Alluvium is generally limited to the natural (pre-reservoir) floodplains of the Emory River and its tributaries. The alluvium consists of a thin (< 6 inch) layer of clayey silt, which overlies a layer of medium to very stiff silt and clay. The stiff silt and clay transitions to a very loose to medium dense sand and silt residuum (clayey soil derived from the weathering of the underlying bedrock). This geology is expected to cover the remaining upland areas within the region, but data regarding its thickness off site are currently unavailable (Jacobs 2010a).

2.1.3.4 Water Quality Parameters

Watts Bar Reservoir is eutrophic; its embayments range from 10 to 35 milligrams of chlorophyll per cubic meter, and the main channel remains around 15 milligrams of chlorophyll per cubic meter during the growing season (April through September). Strong thermal stratification occurs seasonally in the downstream (and deeper) lacustrine portion of the reservoir (TRM 530 to 545); embayments with low water exchange can also undergo strong thermal stratification. The transition zone (TRM 545 to 565) generally has higher chlorophyll concentrations than the downstream portion of the reservoir and exhibits weaker thermal stratification, except during very low-flow conditions. The more riverine portion of the reservoir (above TRM 565) has lower chlorophyll concentrations than the transition zone, and generally does not experience thermal stratification because of higher water velocities (TVA 2009a), except in embayments.

Thermal and oxygen stratifications were recorded during the summer months for areas monitored downstream prior to the release (TRM 530 to 545) or areas where low flows and a lack of mixing of the water column occurs (TVA 2009a). Also, the Clinch River has colder water temperatures than the Emory River during the summer months. The Emory River has lower hardness values than the Clinch River because it drains through the sandstone-dominated Cumberland Plateau.

Four of the 11 locations measured in the fall 2010 were also noted having thermal or oxygen stratification (ERM 1.0, CRM 3.5, and TRM 566, and 568.5). There is a potential for effects on aquatic biota within anoxic zones as they occur in the river; however, sampling to date offers low resolution as to where these areas exist.

2.1.3.5 Sediment Parameters

Sediments in the Emory River reflect the upstream riverine to downstream lacustrine conditions, which is more likely to have deposition. The river bottom at the upstream reference area (ERM 6.0) is comprised of bedrock and hard-packed sediments, silts, clays, sands, and detritus (leaves, twigs, and other natural organic materials) with no observable released ash. Further downstream (ERM 6.0 to 3.5), sediments increase in ash content and consists of mostly hard-packed clays and bedrock near overbank areas. Areas that are not scoured are comprised of gravels, fine silts, detritus, or sands with some coal particles. Continuing downstream (ERM 3.5 to 1.5), sediments are highly variable but the overall composition shifts to predominantly fine clays and silts. Near ERM 3.0, the substrate is a mix of ash, mostly fine silts and sands with detritus or hard-packed clay in some areas. Substrate in the lower Emory River (ERM 1.5 to 0.0) is predominantly fine silts and detritus mixed with ash (Appendix M).

The upstream Clinch River reference areas are bedrock or hard-packed sediments, gravel/sand mixes, and fine silts and detritus in depositional areas. The substrate between CRM 4.5 to 3.0 is comprised of mostly fine silts and detritus with some fly ash. The lower portion of the Clinch River, from CRM 3.0 to 1.5, has bedrock or fine silts and small amounts of ash in some areas. Sediments in the downstream-most portion of the Clinch River from CRM 1.5 to 0.0 consist of mostly silts and clays with detritus (Appendix M).

The upstream Tennessee River reference area near TRM 574 is in an area where no ash has been observed. Sediments at this location are comprised of mostly fine silts and clays with some sand, detritus, and gravel. The downstream Tennessee River, near TRM 566, is similar to the upstream in that the sediments are predominately fine silts and clays, with some sand, detritus, and gravel and trace amounts of observable ash (Appendix M).

2.1.4 Ecological Setting

Watts Bar Reservoir occurs within the Eastern Broadleaf Forest and Cumberland Plateau ecological provinces (Bailey et al. 1994). Vegetation includes limestone bedrock-affiliated species, such as oak (*Quercus*), pine (*Pinacaea*), hickory (*Carya*), black gum (*Nyssa sylvatica*), and sweetgum (*Liquidambar*) species (Bailey 1995). The tree cover is mosaic, recovering after southern pine bark beetle (*Dendroctonus frontalis*) outbreaks in the early 1990s that killed large stands of trees. Major plant communities in the area include Forested Bluff and Rocky Slopes; Mesic Deciduous Forest; Moist Woodlands; Forested Streamsides, Seeps, and Bogs; Forest Edges, Roadsides, and Fencerows; Prairies, Barrens, and Open Woodlands; Marshes, Wet Meadows, and Open Streamsides; and Gravel Bars and Boulders in Rivers and Large Streams (TVA 2009a). The mosaic pattern of tree growth recovery allows for the increased invasion of exotic species (e.g., tree of heaven [*Ailanthus altissima*], autumn olive [*Elaeagnus umbellate*], and Japanese honeysuckle [*Lonicera japonica*]) (Early Detection and Distribution Mapping System [EDDMapS] 2011).

Watts Bar Reservoir area is primarily rural and forested (Figure 2-6). There are approximately 771 shoreline miles of shallow water riparian habitat and approximately 29,000 of overbank habitat (TVA 2009a). Previous analysis indicated that 64 percent of vegetation within 25 ft of the shoreline and 59 percent of the vegetation between 25 and 100 ft of the shoreline was tree cover (TVA 1998).

A variety of native aquatic and terrestrial species occur within the Watts Bar Reservoir area. In addition, a number of invasive species have also established populations within the area, including five aquatic invertebrates, 15 fish, and seven aquatic plant species (Table 2-1). In addition, a number of threatened and endangered species have been identified in the Watts Bar Reservoir area, as described in the subsections below.

2.1.4.1 Aquatic Life

Aquatic habitat is largely affected by watershed land use and bathymetry, which in turn affects the types of aquatic life found in an area (TVA 2009a). Along residentially-developed areas adjacent to the reservoir seawalls and riprap dominate the shoreline. Along undeveloped areas, fallen trees provide shoreline habitat. Bedrock outcrops, shallow shoreline gravel beds, and rubble and cobble beds on steeper shorelines also provide varied aquatic habitats. Aquatic plant bed habitats are rare near Watts Bar Reservoir; high water flows and other factors likely eliminated the existing (invasive) Eurasian water milfoil (*Myriophyllum spicatum*) beds during the 1990s. Some beds of spiny-leaf naid (*Podocarpus spinulosus*), another exotic species, do exist (TVA 2009a). Aquatic species have different tolerances for temperature, water velocity, dissolved oxygen (DO) content, nutrients, and pH that limit where these species can live (e.g., main-stem river versus tributaries).

Fish

A total of 43 species of fish were caught during a TVA fish sampling effort in 2009; the predominant species included gizzard shad (*Dorosoma cepedianum*), bluegill sunfish (*Lepomis macrochirus*), freshwater drum (*Aplodinotus grunniens*), largemouth bass (*Micropterus salmoides*), and redear sunfish (*Lepomis microlophus*) (TVA 2009a). Creel data indicate that bluegill sunfish have been the most frequently caught species in the area, closely followed by largemouth bass (Tennessee Wildlife Resources Agency [TWRA] 2002). Other frequently-caught fish species include black crappie (*Pomoxis* spp.), white bass (*Morone chrysops*), white crappie (*Pomoxis* spp.), smallmouth bass (*Micropterus dolomieui*), and sauger (*Micropterus dolomieui*) (TVA 2009a).

Two federally-listed, threatened fish species, the snail darter (*Percina tanasi*) and the spotfin chub (*Cyprinella monacha*), may occur within Watts Bar Reservoir. The snail darter and the spotfin chub are also state-listed as threatened species. In addition, the blue sucker (*Cycleptus elongatus*) is state-listed as threatened, and the tangerine darter (*Percina aurantiaca*), the flame chub (*Hemitremia flammea*), and the Tennessee dace (*Phoxinus tennesseensis*) have been identified by the state of Tennessee as species in need of management (Table 2-2) (TVA 2009a).

Federally- and state-listed fish species observed in this area are known primarily from tributaries to Watts Bar Reservoir, although the flame chub, snail darter, spotfin chub, and Tennessee dace have been observed in embayments and margins of the reservoir. Snail darters spawn in tributaries, and their fertilized eggs are swept into the reservoir where the fry hatch before moving back upstream into the tributaries (TVA 2009a).

Mollusks

Native mussels in the Emory, Clinch, and Tennessee Rivers have been substantially impacted for decades. The impoundment of the Tennessee and subsequent creation of a transitional slack water habitat, strip mining activities upstream, and industrial and municipal wastewater discharges have altered mussel assemblages due to the changed current flows and DO levels (TVA 2009a).

Pre-release aquatic surveys indicated the presence of six mussel species (giant floater [*Anodonta grandis*], fragile papershell [*Leptodea fragilis*], pistolgrip [*Tritogonia verrucosa*], pimpleback [*Plethobasus cooperianus*], wartyback [*Plethobasus cicatricosus*], and threehorn wartyback [*Obliquaria reflexa*]) and a common aquatic snail (hornsnale [*Pleurocera foreman*]) in the Clinch and Emory River arms of Watts Bar Reservoir. All but one species (the pistolgrip) were likely to have occurred in the area impacted by the ash release, but likely in low numbers because of summer DO limitations in the Emory River slackwater (TVA 2009a).

Eight species of mussels and one species of aquatic snail have been federally-listed as endangered (Table 2-3). These species are also state-listed as endangered. In addition, the state identified the pyramid pigtoe (*Pleurobema plenum*) mussel as a species in need of management.

The pink mucket (*Lampsilis abrupta*), rough pigtoe (*Pleurobema plenum*), fanshell (*Cyprogenia stegaria*), and orange-foot pimpleback (*Plethobasus cooperianus*) mussels have been identified in the tailwaters of Melton Hill Dam (the lower Clinch River). The Alabama lampmussel (*Lampsilis virescens*), dromedary pearlymussel (*Dromus dromas*), fine-rayed pigtoe (*Fusconaia cuneolus*), purple bean (*Villosa perpurpurea*), and Anthony's river snail (*Athearnea anthonyi*) are considered to have likely been extirpated from the area after the river was impounded (TVA 2009a).

2.1.4.2 Riparian Life

The diverse habitats available around Watts Bar Reservoir, including upland forest, forested streams, early successional scrub-shrub, islands, and forested wetland habitats, support many species. Descriptions of plant, bird, mammal, amphibian, and reptile species observed in the Watts Bar Reservoir vicinity on TVA-owned lands are provided below (TVA 2009a).

Vegetation

A wide variety of aquatic and riparian vegetation grows in the Watts Bar Reservoir. Common species include aquatic grasses, sedges, hardwoods, pines, and more (Table 2-4). A number of aquatic and riparian vegetation species of concern are also present including three federally-listed threatened species, 13 state-listed threatened species, and seven state-listed endangered species (Table 2-5). However, none of the threatened or endangered species of vegetation have been observed along the study area shorelines (TVA 2009b).

Birds

Numerous bird species utilize the riparian and wetland habitats along the reservoir (Table 2-6). Common species include residential populations, wading shorebirds, aerial-feeding insectivores, and migratory species (Table 2-7). Some neotropical migrant species such as killdeer (*Charadrius vociforus*) and semipalmated plover (*C. semipalmatus*) are commonly found within the reservoir area, as well as waterfowl species including mallard (*Anas platyrhynchos*), American black duck (*Anas rubripes*), hooded merganser (*Lophodytes cucullatus*), resident Canada goose (*Branta canadensis*), and wood duck (*Aix sponsa*). There are also other water/wading birds such as pied-billed grebe (*Podilymbus podiceps*) and various tern and gull species (*Laridae* spp.) and aerial-feeding insectivores such as tree swallows (*Tachycineta bicolor*) and purple martins (*Progne subis*). Piscivorous birds, including double-crested cormorant (*Phalacrocorax auritus*), great blue heron (*Ardea herodias*), black-crowned night-heron (*Nycticorax nycticorax*), and osprey (*Pandion haliaetus*) are common and nest at the Site. Twenty-two heron colonies have been identified along the Watts Bar Reservoir.

Watts Bar Reservoir also supports bird species of concern. One federally- and state-listed protected species, the bald eagle (*Haliaeetus leucocephalus*), is occasionally observed foraging and resting in riparian

areas surrounding the Site. The closest known nest is located on the Tennessee River, approximately 2.5 miles from the mouth of the Clinch River (TVA 2009b). In addition, one state-listed endangered species, Bachman's sparrow (*Aimophila aestivalis*), is present within the reservoir but has not been observed along the study area shoreline. Three other state-listed species in need of management are also found, including barn owl (*Tyto alba*), least bittern (*Ixobrychus exilis*), and sharp-shinned hawk (*Accipiter striatus*) (Table 2-8) (TVA 2009a).

Mammals

The Watts Bar Reservoir area supports a number of mammal species in the riparian, wetland, and aquatic habitat types. Common mammals seeking food and cover in these habitats include muskrat (*Ondatra zibethicus*), mink (*Neovison vison*), white-tailed deer (*Odocoileus virginianus*), raccoon (*Procyon lotor*), eastern mole (*Scalopus aquaticus*), eastern cottontail rabbit (*Sylvilagus floridanus*), groundhog (*Marmota monax*), gray fox (*Urocyon cinereoargenteus*), and coyote (*Canis latrans*), along with others (Table 2-6).

Watts Bar Reservoir also supports several mammal species of concern. Only one federally- and state-listed endangered species, the gray bat (*Myotis grisescens*), is present within the reservoir area and likely forages along the Emory and Clinch River shorelines. The closest cave known to e occupied by gray bats is 16 miles from the Site (TVA 2009b). Additionally, the Eastern small-footed bat (*Myotis leibii*), southeastern shrew (*Sorex longirostris*), and southern bog lemming (*Synaptomys cooperi*) are state-listed as species in need of management (Table 2-9) (TVA 2009a).

Amphibians and Reptiles

The Watts Bar Reservoir area supports a number of amphibian and reptile species in the riparian, wetland, and aquatic habitat types. Common species in these habitats include bullfrog (*Rana catesbeiana*), green frog (*Rana clamitans*), Eastern narrow-mouth toad (*Gastrophryne carolinensis*), and Fowler's toad (*Bufo woodhousii fowleri*); while reptiles are represented by northern water snake (*Nerodia sipedon sipedon*), common snapping turtle (*Chelydra serpentina serpentine*), painted turtles (*Chrysemys picta* spp.), and red-eared slider (*Trachemys scripta elegans*) (Table 2-6).

While there are no federally-listed amphibian or reptile species of concern in this area, a number of state-listed species occur (Table 2-10). One state-listed threatened amphibian species, the Berry cave salamander (*Gyrinophilus palleucus*), has been found in the reservoir area, as well as one state-listed threatened reptile species, the Northern pine snake (*Pituophis melanoleucus melanoleucus*). In addition, the Eastern hellbender (*Cryptobranchus alleganiensis alleganiensis*), four-toed salamander (*Hemidactylium scutatum*), and Eastern slender glass lizard (*Ophisaurus altenuatus longicaudus*) are local species listed in need of management (TVA 2009a).

2.1.4.3 Wetlands

Wetlands occur throughout the Watts Bar Reservoir region. Forested wetlands are the most common wetland type in the reservoir area. Scrub-shrub and emergent wetlands are also found in association with larger areas of forested wetlands, as well as within embayments and along the aquatic to terrestrial shoreline gradient (TVA 2009a).

However, wetland areas along the Emory and Clinch Rivers are generally limited to narrow, fringe wetlands. This is due partly to the approximately 5-ft variation in water elevation between winter and summer pool levels. The relatively steep topography along much of the shoreline also limits the areas where soils remain saturated. Exceptions are typically located near shallow inlets fed by springs or small tributaries. The reservoir shorelines are sparsely populated with small beds of emergent vegetation located below the summer pool level. Wetland plants along the summer pool shoreline also are limited in distributions primarily along points and islands. These fringe wetlands appear to be comprised primarily of rushes or cattails. Riparian habitats along the Clinch and Tennessee Rivers are varied in nature and include mature deciduous (or mixed) forests, scrub/shrub, mixed herbaceous vegetation, rock or concrete retaining walls, and manicured lawns (Jacobs 2010a).

Wetland areas also existed in the Swan Pond Embayment prior to the ash release. However, the ash slide impacted the wetlands in the ash release area. These wetlands are addressed in the EE/CA for the Dredge Cell and Embayments (TVA 2009c); they are outside of the river system area being evaluated in this BERA.

2.1.5 Land Use

The majority of the reservoir area can be characterized as rural and forested. Larger developed communities near Watts Bar Reservoir include Spring City and Grandview in Rhea County; Rockwood, Kingston, and Harriman in Roane County; Lenoir City, Loudon, and Philadelphia in Loudon County; and Oak Ridge in Anderson County, Tennessee (TVA 2009a). The land use around the reservoir includes a combination of agriculture, fallow fields, pastures, industrial parks, barge terminals, residential shoreline developments and marinas, power production (TVA Kingston Steam Plant and TVA Watts Bar Nuclear Plant), wildlife conservation areas, and federal defense research (Oak Ridge Reservation).

TDEC has designated Watts Bar Reservoir for the following uses:

- Domestic water supply
- Industrial water supply
- Fish and aquatic life habitat
- Recreation

- Irrigation
- Livestock watering and wildlife.

The land around Watts Bar Reservoir includes private-, federal-, and state-owned tracts. TVA originally obtained approximately 55,000 acres of land around the reservoir; through land disposal actions, TVA has reduced its holdings to approximately 16,220 acres (TVA 2009a). Following the release, TVA purchased an additional 939.8 acres (as a direct result of the release or the subsequent removal actions) around the ash release area. The USDOE owns approximately 35,000 acres of land comprising the Oak Ridge Reservation site.

2.1.5.1 Tennessee Valley Authority Operations

TVA has two power-producing plants on Watts Bar Reservoir: Kingston Steam Plant and Watts Bar Dam, which produces hydroelectric power. TVA has several recreation and wildlife management areas on its reservoir properties and historically used pesticides on the reservoirs for malaria control. The TVA Clinch Breeder Reactor on the Clinch River may be used in the future for power generation or industrial purposes.

2.1.5.2 U.S. Department of Energy's Oak Ridge Reservation

The Oak Ridge Reservation is presently on the USEPA National Priorities List because a number of constituents on site require cleanup (TDEC 2010). Historically, tributaries to Watts Bar Reservoir were exposed to weapons-related constituents (e.g., uranium, metals, and PCBs) by stormwater runoff, unlined retention pond drainage, and buried container leakage (USDOE). White Oak Creek drains the Oak Ridge National Laboratory (ORNL) area. A dam was built on White Oak Creek, forming White Oak Lake. Poplar Creek and its tributaries, Bear Creek and East Fork Poplar Creek, drain the K-25 and Y-12 plant areas.

2.2 Assessment Endpoints

Assessment endpoints include important ecological resources that, if damaged, would significantly impact functioning of the ecosystem. Assessment endpoints are selected based on key ecosystems, communities, or ecological functions; COPECs present; the extent and magnitude of contamination; mechanisms of toxicity; and potential exposure pathways. The selection and definition of assessment endpoints are vital within a risk assessment because they focus the risk assessment design and analysis. Assessment endpoints provide a link between the measurement endpoints and the risk management process. The USEPA's definition of assessment endpoints (1997a) states that assessment endpoints should reflect "…explicit expressions of the actual environmental values (e.g. ecological resources) that are to be protected. Valuable ecological resources include those without which ecosystem function would be significantly impaired, those providing critical resources (e.g., habitat, fisheries), and those perceived as valuable by humans (e.g., endangered species and other issues addressed by litigation)."

Individual assessment endpoints typically revolve around key ecological trophic levels, communities, or specific ecological functions. Collectively, assessment endpoints are representative of the entire ecosystem. USEPA (1997) states that "useful assessment endpoints define both the valued ecological entity at the Site (e.g., a species, ecological resource, or habitat type) and a characteristic(s) of the entity to protect (e.g., reproductive success, production per unit area, and areal extent)." The associated risk question and the valued characteristic(s) to be protected for each of the assessment endpoints are provided in their specific chapters. Assessment endpoints for the river system BERA include maintenance and reproduction of balanced communities or populations of the following receptor groups:

- Pelagic fish communities
- Benthic fish communities
- Benthic invertebrate communities
- Aquatic plant communities
- Aquatic- or riparian-feeding herbivorous bird populations
- Aquatic- or riparian-feeding insectivorous (omnivorous) bird populations
- Aquatic- or riparian-feeding carnivorous (piscivorous) bird populations
- Aquatic- or riparian-feeding herbivorous mammal populations
- Aquatic- or riparian-feeding insectivorous (omnivorous) mammal populations
- Aquatic- or riparian-feeding carnivorous (piscivorous) mammal populations
- Aerial-feeding insectivorous (omnivorous) bird populations
- Aerial-feeding insectivorous (omnivorous) mammal populations
- Aquatic- or riparian-feeding amphibians
- Aquatic- or riparian-feeding reptiles.

2.2.1 Conceptual Site Model

The ecological CSM identifies the potential sources of ash contamination at the Site and the chemical, physical, and biological processes that occurred as a result of the ash release. Efforts to remove the ash from the Emory River occurred during the time-critical actions. However, because over-dredging did not occur, there may be small pockets of residual ash in the Emory River. Pockets of ash may also still remain in the Clinch and Tennessee Rivers. Some additional ash may have migrated into these river systems as a result of high flow events that occurred during the time-critical removal activities. In addition, ash may have become intermixed or interbedded with natural river sediments in all three rivers (Jacobs 2010a).

The primary transport pathways of contaminants from the source occur through river transport, groundwater transport, and biouptake. These pathways are discussed below. Surface erosion and transport in stormwater runoff was not considered a primary transport pathway of concern, since the non-time-critical removal action

for the Swan Pond Embayment and Dredge Cell will have removed the ash from the embayment and closed the Dredge Cell (Jacobs 2010a).

2.2.1.1 River Transport

Constituents within the ash may enter the water through suspension or dissolution and be transported in the river system with the river flow. At sufficiently high flow velocities, ash may be suspended in the water column. The small size and granularity of the ash particles mean that they may be easily detached and entrained in flowing water, so this material would continue moving downstream as suspended sediment during periods of high-flow rates. The fly ash deposits may also exhibit cohesive behavior, and may consolidate over time in the river channel. In addition, cenospheres (inert floating ash material) can move downstream on the water surface (Jacobs 2010a).

USEPA retained the U.S. Army Corps of Engineers Engineering Research and Development Center at Waterways Experiment Station (ERDCWES) to run a quantitative two-dimensional hydrodynamic and sediment fate and transport model, known as Adaptive Hydraulics (AdH). In general, this model uses system bathymetry, grain size distribution, and river flow data to predict the transport characteristics and deposition of fly ash in the system. Both cohesive and non-cohesive sediment transport can be simulated with the model, as well as non-alluvial sediments with unique characteristics such as coal fly ash. Additional information on AdH can be found at the following website: http://adh.usace.army.mil/ (Jacobs 2010a).

Modeling saturated coal fly ash in the system presents a unique problem. Fly ash has a lower particle density than native sediments, and a spherical shape as opposed to the irregular shapes found in quartz based sediments. In addition, the fly ash deposits can exhibit cohesive behavior, and may consolidate over time in the river channel. Because cohesive deposits are more resistant to erosion than non-cohesive deposits, the ERDCWES conducted tests in a laboratory flume (SedFlume) to characterize the resistance of fly ash to erosion. The SedFlume measures the erosion rate of a sample over time as flow is induced over the sample. This erosion rate is correlated to the bed shear stress induced by the flow. An empirical equation is developed from the tests and used in the AdH model to specifically predict erosion rates of the fly ash deposits. The natural bed sediment erosion, entrainment, transport, and fate are also simulated in the model along with the fly ash. Additional information on the SedFlume can be found at the following website: http://chl.erdc.usace.army.mil/chl.aspx?p=s&a=articles!630 (Jacobs 2010a).

The AdH model domain extends from 1 mile upstream of the KIF on the Emory River to the Watts Bar Dam. Both the Clinch and Tennessee Rivers are included in the domain up to Melton Hill and Fort Loudon Dams, respectively.

2.2.1.2 Groundwater Transport

Infiltration of precipitation through the ash in the closed Dredge Cell may result in leaching of ash-related constituents. Constituents in the leachate may then be transported to the water table and discharged to the Emory River with the groundwater flow. Because the sources are located adjacent to the river and in an area of upward groundwater flow, constituents entering shallow groundwater would be transported only a short distance prior to discharge. Once in the river, the groundwater would become intermixed with the surface water and transported with the river flow. The physical and chemical properties of constituents that influence their transport in groundwater include solubility in water and chemical affinity for solids. Metals tend to adsorb to ash and clay-rich soils, which retards their migration (Jacobs 2010a).

2.2.1.3 Biouptake

Constituents in the ash may enter into the food web through uptake by organisms. Two basic routes of exposure and subsequent uptake by organisms include: 1) direct exposure to dissolved constituents in surface water or porewater with subsequent transport across biological membrane surfaces (e.g., at the gill, gut, or root tip); and 2) ingestion of constituents in prey, surface water, and/or sediment particles with subsequent transport across the biological membrane surfaces. While uptake through direct exposure or ingestion generally describes the exposure for lower-trophic-level species, for upper-trophic-level species, ingestion of constituents in prey is the predominant route of exposure/uptake. Constituents in tissues of organisms in the food chain are likely to be ingested by the species that feed on them (i.e., those occupying higher trophic levels); the result of which may be the expression of toxicological effects in the higher trophic-level species (Jacobs 2010a).

The entire mass of a constituent in the environment is not necessarily available for biouptake. Bioaccessibility and bioavailability are the determining factors for biouptake. The portion that the organism can come into contact with is the bioaccessible fraction, which is a function of the behavior and physiology of the exposed organism. The portion that is potentially available for absorption or adsorption by an organism is the bioavailable fraction, which interacts with the exposed organism's biological membrane surfaces. A wide range of physical, chemical, and biological factors has the potential to influence the bioavailability of constituents in the environment. Bioavailability of constituents in sediment and surface water can vary with changing environmental conditions and is a function of chemical characteristics including chemical type and chemical speciation, as well as the physical and biological environmental conditions (Jacobs 2010a).

The processes of bioconcentration and bioaccumulation are important because they provide a basis of predicting potential constituent uptake by flora and fauna. Substances that enter an organism can accumulate, particularly in the lipid (fatty) tissue, which results in a higher concentration of the substance in the organism than in the surrounding environment or in its prey. Subsequent bioaccumulation of constituents can therefore magnify within the food web. Bioconcentration differs from bioaccumulation

because it refers only to the uptake of substances into the organism from water alone. Bioaccumlation is the more general term because it includes all means of uptake into the organism (Jacobs 2010a).

2.2.2 Complete Ecological Exposure Pathways

Ecological receptors (biota) at the Site could come into contact with COPECs through a variety of exposure pathways. Potential complete exposure pathways include:

- Direct contact with ambient media
  - Sediments (seasonally-exposed and submerged)
  - Porewaters
  - Surface waters (epi-benthic and water column)
- Ingestion of COPECs in food items
- Incidental ingestion of ambient media
  - Sediments (seasonally-exposed and submerged)
  - Surface waters
  - Porewaters.

Inhalation and dermal contact were evaluated as possible exposure routes during problem formulation, but were eliminated as pathways warranting explicit evaluation in the BERA (Jacobs 2010a). Under most exposure conditions, the inhalation and dermal contact pathways do not represent a significant contribution to receptor risk (USEPA 2005) and are not evaluated quantitatively under most circumstances. While fugitive dust is a potential release mechanism from seasonally-exposed sediments, inhalation of fugitive dust from seasonally-exposed sediments is not anticipated to present a significant contribution to ecological receptor risk associated with the river system. Feathers of birds, fur on mammals, and scales on reptiles are thought to reduce dermal exposure by limiting the contact of the skin surface with the COPECs in the river system (USEPA 2005). As a result, inhalation and dermal contact were not explicitly evaluated within the BERA.

The CSM represents the current understanding of potential sources in the river system based on the best available information, and depicts how COPECs from these sources may move through the ecosystem to receptors that make up the assessment endpoints. Primary components of the CSM (i.e., sources, release and transport mechanisms, exposure media, exposure pathways, and ecological receptors) are depicted on Figure 2-7. This CSM presents only viable exposure media and pathways for the river system. It does not include inhalation and dermal contact with fugitive dust (Jacobs 2010a).

2.3 Measurement Endpoints

A measurement endpoint is defined as "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint" (USEPA 1997a). Measurement endpoints can include both measures of exposure and measures of effect. Measures of exposure are measurements of the presence of stressors in the environment and their relationship or co-occurrence with the receptors being evaluated for each assessment endpoint. Examples of measures of exposure include concentrations of COPECs in environmental media, concentrations of COPECs measured in prey items used for dietary exposure modeling, and concentrations of COPECs measured in ecological receptor tissue. A summary of environmental data by media used for each assessment endpoint is presented in Table 2-11.

Measures of effect are the measurable changes in an aspect of an assessment endpoint (or its surrogate) in response to a stressor to which it is exposed. These are often expressed through the use of numerical observations (e.g., toxicity test results and community diversity measures) that can be compared statistically to a control or reference area to detect adverse responses to a site COPEC. Examples of measures of effect include survival, growth (or biomass), and reproduction of ecological receptors.

The measurement endpoints become the LOEs that are used to conduct the risk characterization. The linkage between the measurement endpoints to the assessment endpoints and the valued characteristics to be protected are presented in more detail in each of the assessment endpoint chapters. The measurement endpoints selected for this BERA (Jacobs 2010a) are presented, by receptor, below:

2.3.1 Fish

- Fish community metrics (e.g., species abundance and diversity)
- Early life-cycle fish toxicity tests (e.g., survival and growth)
- Fish reproductive studies (e.g., production of viable eggs)
- Fish health studies (e.g., ovary status and condition factors)
- Measurement of fish health metrics (e.g., liver enzymes)
- COPEC concentrations in fish tissues
- COPEC concentrations in surface waters (total and dissolved).

2.3.2 Benthic Invertebrates

- Quantitative surveys of site benthic invertebrate communities – relative abundance, diversity, and proportional abundance and diversity of key trophic guilds (e.g., shredders and filter feeders)
- Literature-derived toxicity data – no observed adverse effect levels (NOAELs), lowest observed adverse effect levels (LOAELs), lowest chronic values, screening values, and species sensitivity distributions

- Toxicity test results using laboratory-raised invertebrates and site sediments and surface waters in short-term to partial life cycle toxicity test (e.g., survival, reproduction, and biomass)
- COPEC concentrations in mayfly and snail tissues
- COPEC concentrations in surface waters (total and dissolved) and speciation of selenium and arsenic
- COPEC concentrations in sediments, speciation of selenium and arsenic, and acid volatile sulfides/simultaneously extracted metals (AVS/SEM)
- COPEC concentrations in sediment porewaters.

2.3.3 Aquatic Vegetation

- COPEC concentrations in emergent and shoreline vegetation and in periphyton
- Literature-derived toxicity data – NOAELs, LOAELs, lowest chronic values, screening values for sediment and aqueous exposures, and residue-effect levels
- COPEC concentrations in surface water (total and dissolved) and speciation of selenium and arsenic
- COPEC concentrations in sediments and speciation of selenium and arsenic
- COPEC concentrations in sediment porewater.

2.3.4 Aquatic- and Riparian-Feeding Birds

- Egg health metrics (e.g., weights, volumes, lengths, and widths) of great blue heron and osprey eggs
- COPEC concentrations in eggs of piscivorous birds
- Literature-derived toxicity data for dietary doses – NOAELs and LOAELs for bird growth or reproduction
- Refinement of literature-derived toxicity data for dietary doses – NOAELs and LOAELs for bird growth or reproduction
- Modeled COPEC dietary doses in receptors
- Toxicity reference values (TRVs) for effect concentrations (EC) – EC_{10}s and LOAELs for embryonic effects
- COPEC concentrations in surface waters
- COPEC concentrations in sediments
- Speciation of selenium and arsenic in sediment and fish tissue
- COPEC concentrations in tissues of dietary items.

2.3.5 Aquatic- and Riparian-Feeding Mammals

- TRVs for raccoon (*Procyon lotor*) concentrations representing benchmarks for toxic effects to mammals (EC_{10} or LOAEL)
- COPEC concentrations in raccoon tissues

- Health metrics of exposure in raccoons (e.g., blood count and plasma biochemistry panel)
- Screening-level literature-derived toxicity dietary doses
- Refined screening-level literature-derived toxicity dietary doses
- Modeled COPEC dietary doses
- COPEC concentrations (total) in surface waters
- COPEC concentrations in sediments
- Speciation of arsenic and selenium in sediment and fish tissue
- COPEC concentrations in tissues of dietary items
- COPEC concentrations in tissues of raccoons.

2.3.6 Aerial-Feeding Birds and Mammals

- Productivity metrics for tree swallow (e.g., clutch size, hatching success, nestling survival, and female fledglings produced per nesting female)
- Tree swallow (*Tachycineta bicolor*) egg health metrics (e.g., weights, lengths, widths, and volumes)
- Tree swallow nestling health metrics (e.g., nestling weights, tarsus lengths, and feather lengths)
- TRVs for eggs, eggshells, and nestlings concentrations
- Screening-level literature-derived toxicity dietary doses for individual growth or reproduction for each assessment endpoint
- Refined screening-level literature-derived toxicity dietary doses for individual growth or reproduction for each assessment endpoint
- Modeled COPEC dietary doses
- COPEC concentrations (total) in surface waters
- COPEC concentrations in tissues of dietary items
- COPEC concentrations in tree swallow eggs, eggshells, and nestlings.

2.3.7 Amphibians

- Literature-derived toxicity data
- COPEC concentrations in surface waters (total and dissolved)
- COPEC concentrations in sediments
- Health metrics of exposure – measured whole body tissues COPEC concentrations of selenium (and other bioaccumulators) in amphibians.

2.3.8 Reptiles

- Turtle community survey data

- Literature-derived toxicity data
- Health metrics of exposure – measured blood COPEC concentrations of selenium (and other bioaccumulators) in reptiles
- COPEC concentrations in surface waters (total and dissolved)
- COPEC concentrations in sediments.

Because each receptor group is evaluated using multiple endpoints along with primary and secondary LOEs, a WOE approach is used to evaluate the strength of each measurement endpoint and enable further interpretation of the potential for site-related risks. Additional information regarding the WOE approach is presented in the next section.

2.4 Risk Characterization Techniques

This section summarizes techniques and methodologies that were employed for the receptor groups in the BERA risk characterizations. Multiple LOEs may be needed in order to reasonably demonstrate that COPECs from the ash release are likely to cause adverse effects on an assessment endpoint. Therefore, an extensive suite of biological and environmental data was collected for this assessment, resulting in multiple LOEs for most receptors. The WOE paradigm was used to evaluate multiple LOEs in the risk characterization process. This paradigm is the standard approach used in ecological risk assessment to:

- Compare and contrast results for multiple measurement endpoints
- Provide a framework for developing an overall estimation of risks to an assessment endpoint
- Identify and describe the level of confidence in the risk estimate.

The BERA Methodology (Jacobs 2010a, Appendix B) described the LOEs being collected for this assessment as falling into two general categories: primary LOEs and secondary LOEs. The principal distinction between these groups is the degree to which they relate to the community and population attributes of the assessment endpoints. Primary and secondary LOEs are briefly discussed below, followed by a description of the process used to weight multiple LOEs in the BERA.

2.4.1 Primary Lines of Evidence

Primary LOEs provide direct evidence of risk and are commonly used in the decision-making process. The primary LOEs for the river system include evaluating literature-derived effects values, conducting toxicity tests, and performing site-specific biological surveys (biosurveys) as described below.

2.4.1.1 Literature-Derived Effects Values

Toxicity data obtained from the literature were compared with concentrations of COPECs measured at the Site. The literature-derived toxicity data can be for ambient media or for biological tissues. The literature-derived toxicity data selected for use in this BERA correspond as closely as possible to the receptor-specific assessment endpoints in terms of taxonomy, life stage, response, exposure duration, and exposure conditions. In general, this LOE includes comparison of:

- Measured COPEC concentrations in ambient media with NOAELs, LOAELs, and chronic values
- Measured COPEC concentrations in biota (whole organisms or specific tissues of birds, mammals, fish, benthic invertebrates, reptiles, and amphibians) with literature-derived toxicity data
- Measured or estimated COPEC concentrations in prey and ambient media ingested by wildlife with literature-derived dietary TRVs.

2.4.1.2 Toxicity Tests

Toxicity tests were conducted for multiple aquatic test species and included as LOEs for fish and benthic invertebrates. Such tests provide a measure of the bioavailability and toxicity of COPECs in site-specific media. Toxicity test species and endpoints were selected based on relevance to the Site, availability of standard test methods, and relevance to the assessment endpoints.

- Sediments: whole sediment toxicity testing was conducted using amphipods (*Hyalella azteca*) and insect larvae (*Chironomus dilutus*) are standard test organisms for sediments or surface waters. In addition, cladocerans (*Ceriodaphnia dubia*) and fathead minnows (*Pimephales promelas*) are commonly used test organisms for surface water toxicity tests, and to a lesser extent for sediment toxicity (USEPA 1994a).
- Surface waters: toxicity testing was conducted using *C. dubia* and *P. promelas*, both of which are standard test organisms for surface waters.

2.4.1.3 Community Surveys

Community surveys are direct measurements of the quality of the biological community being assessed at the Site. They are generally the most ecologically relevant LOE and are recommended by USEPA for inclusion in ERAs (USEPA 1990).

Various community surveys were conducted at the Site and are used in this BERA. These include reproduction and morphometric assessments of birds, population surveys of reptiles, and community surveys of fish and benthic invertebrates.

2.4.2 Secondary Lines of Evidence

In contrast with primary LOEs, secondary LOEs provide only indirect evidence of risk to the assessment endpoints. These measures of exposure or effects may help explain or refine observations made with primary LOEs, but are insufficient by themselves as a reliable estimate of potential risk. Health metrics are general categories used to describe the secondary LOEs used in this assessment.

Health metrics are measures of exposure to various environmental stressors (such as COPECs).They typically indicate whether the organism has been exposed to one or more types of stressors, but generally lack the dose-response relationship to establish causality for an individual COPEC. Furthermore, they are primarily evaluated at the suborganismal level by assessing molecular, biochemical, and physiological endpoints. Quantitative relationships between such measures and adverse effects on the population or community have not been established in the scientific literature. Example health metrics measured for the river system include liver enzyme levels in fish and mammals. Other health metrics in the context of this assessment refer to measures of effects at the suborganismal level in receptors potentially exposed to COPECs. They often include changes in biomolecular, biochemical, or physiological parameters that are measures of stress on an individual organism. Example health metrics measured for the river system include organ dysfunction and increased frequency of histopathological lesions in fish.

2.4.3 Weight-of-Evidence Framework

LOEs were interpreted and integrated using a WOE approach. As noted above, this was accomplished by first characterizing risk based on each individual LOE, and then characterizing risk based on the entire body of evidence. At both levels of evaluation, ecological risks were characterized by estimating the magnitude and likelihood of potential adverse effects. This included, either implicitly or explicitly, consideration of at least four factors:

- An estimate of severity (magnitude) of potential adverse effects
- Confidence (or uncertainty) in the estimated magnitude of potential effects
- Evidence of causality
- Relevance of the data to the assessment endpoints.

These factors were synthesized together as an estimated potential risk to the assessment endpoint and a level of confidence in that estimate. This integration of information was performed for each individual LOE and then for the entire body of evidence. The WOE process includes assigning a weight to each LOE. For this BERA an intrinsic relative weight was explicitly defined for each LOE.

Numerous WOE methodologies are presented in the literature. These range from purely qualitative to highly quantitative (Linkov et al. 2009). The WOE approach presented below can be categorized as being

near the middle of qualitative-quantitative continuum. It is primarily logic-based but specifies a "qualitative" weight for each LOE. It is not a scoring or indexing approach, in that it does not apply a numerical weight to each LOE and then mathematically derives a numerical WOE decision. Such methods are consistent and explicit in the weights applied, but they often imply a degree of precision and certainty that the science does not support.

The WOE approach used for this BERA considers the unique characteristics of the river system, the stressors, and available data used to characterize ecological risks. The result is a holistic and comprehensive characterization of risks for each receptor group evaluated. The lack of standard methods for evaluating many of these LOEs makes professional judgment an essential component of this evaluation. Several aspects of ecological significance may be considered, including the nature and magnitude of the effects, the spatial and temporal patterns of the effects, and the potential for recovery once the stressor, in this case ash, has been removed. Different aspects of each assessment endpoint may drive the assessment conclusions, depending on the relevance and confidence for each LOE.

Key elements of the risk characterization for each receptor group are presented in two risk tables. The first presents the evaluation of four principal attributes of each LOE; the second summarizes the characterization of risk for each LOE and the overall WOE. These tables are populated and presented in the risk characterization section for each receptor group. Templates of the Risk Characterization Attributes table and the Summary of Risks table for fish are provided below as examples (Exhibits 2-2 and 2-3).

Exhibit 2-2. Example Risk Characterization Attributes Table--Fish

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Community Surveys	Negligible	High	Moderate	High
Toxicity Tests	Negligible	Moderate	High	Moderate
Reproductive Studies	Negligible	High	High	Moderate
Tissue Concentrations	Moderate	Low	Moderate	Low
Health Metrics	Low	Low	Moderate	Low
Surface Water Chemistry	Low	Low	Moderate	Low
Weight of Evidence	**Moderate**	**Moderate**	**Moderate**	**--**

Exhibit 2-3. Example Summary of Risks Table--Fish

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Community Surveys	High	Negligible	High	--
Toxicity Tests	Moderate	Negligible	High	--
Reproductive Studies	Moderate	Negligible	High	--
Tissue Concentrations	Moderate	Moderate	Low	Aluminum, Arsenic, Cadmium, Copper, Lead, Selenium, Zinc
Health Metrics	Low	Low	Low	--
Surface Water Chemistry	Low	Negligible	Low	--
Weight of Evidence	**--**	**Negligible**	**Moderate**	**--**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

2.4.3.1 Attributes Evaluated

Following is a brief description of the four attributes of an LOE noted above, along with an explanation of how these attributes were evaluated in this BERA. An example of how these attributes align for multiple LOEs is then provided in tabular form.

Estimate of Magnitude of Adverse Effects (Negligible, Low, Moderate, or High)

This factor is an estimate of the "severity" of adverse effects. In this risk characterization strategy, the magnitude of potential effects is considered separately from the associated uncertainty. Two general measures of magnitude were considered for each assessment endpoint and are briefly described below:

- **Exceedance of Effects Values** – Exposure concentrations may be greater than effects values (e.g., TRVs) by varying degrees. These exceedances may range from slight to moderate to large. The definitions of these ratings are based on best professional judgment. The magnitude of the exceedance (e.g., value of hazard quotient [HQ]) was considered along with the nature of the exposure value (e.g., maximum concentrations) and effects values (e.g., NOAELs vs. LOAELs) used to calculate the HQ. It is recognized that HQ magnitude is not uniformly or quantitatively proportional to the severity of effects; however, there is a qualitative difference between HQs of 2, 20, and 200.

- **Difference from Reference** – Measures of effects at potentially affected reaches of the river system may diverge from those of reference locations to varying degrees. For example, benthic communities may have fewer taxa than reference locations, or sediment toxicity tests may result in lower growth or reproduction than reference sediments. Statistical significance is one measure of this divergence, but the absolute difference from reference is also important. The interpretation of these differences are based on best professional judgment and, to the extent practicable, explicitly and quantitatively described.

The estimated magnitude of effects generally fell into four qualitative categories: Negligible, Low, Moderate, and High. These determinations were based on best professional judgment and are supported by explicit documentation in the risk characterization section for each receptor group. The magnitude of effects was generally considered to be:

- "Negligible" if conservative effects values were not exceeded or site metrics were not different from those of reference (background) locations.
- "Low" for small exceedances of conservative effects values (e.g., NOAELs), non-exceedances of less conservative effects values (e.g., LOAELs), and small differences in site metrics relative to reference (background) locations.
- "Moderate" for large exceedances of conservative effects values (e.g., NOAELs), small exceedances of less conservative effects values (e.g., LOAELs), and moderate differences from reference conditions.
- "High" for large exceedances of non-conservative effects values (e.g., LOAELs) or large differences from reference (e.g., > 30 percent reduction).

The definitions of negligible, low, moderate, and high are based on professional judgment and typically vary by type of evidence. For example, low and high exceedances may differ by orders of magnitude.

Confidence in Magnitude of Adverse Effects (Low, Moderate, or High)

This attribute is an estimate of the likelihood that the reported magnitude of adverse effects is representative of that experienced by the receptors at the Site. This measure of uncertainty was based on the extent to which the available data and scientific understanding support the magnitude estimate. Some of the factors considered in the interpretation of the estimated magnitude of effects are briefly described below.

- **Concordance of Evidence** – Separate pieces of evidence within a single LOE may be relatively consistent or inconsistent with each other. For example, multiple toxicity tests at similar exposure levels may result in similar or different responses. This could be an issue for any LOE, including biological surveys, tests of site media, or laboratory tests reported in the literature. The relative concordance of multiple LOEs is a principal consideration in the WOE process.

- **Study Robustness** – The methods used to generate and analyze the available data may vary based on the study design or execution.
- **Organism Sensitivity** – Test organisms may be more or less sensitive to stressors of interest than the species or populations at the Site. This factor was considered throughout the evaluation of the literature-derived effects data as well as for cultured laboratory specimens used in toxicity tests with site sediments or waters.
- **Divergence of Bioavailability** – COPECs in toxicity tests or other case studies were evaluated to determine whether they were more or less bioavailable than in site media or under site-specific conditions.

Similar to the rating used for the magnitude of effects, the confidence rating is based on best professional judgment backed by explicit and detailed analyses and descriptions. The confidence level was generally considered to be:

- "Low" if the estimated magnitude of adverse effects is as likely to be "incorrect" as it is to be "correct." This suggests there are substantial uncertainties associated with the estimation process.
- "Moderate" if it is more likely than not that the estimated magnitude of adverse effects is correct. (i.e., it is possible that refining the uncertainties would result in a substantial change in the estimate of magnitude of effects, but the probability of that type of change is low.)
- "High" if there is very little chance that refining the uncertainties will result in a meaningful change in the estimated magnitude of effects. This implies that the available exposure and/or effects data are sufficiently robust such that any additional data or information is unlikely to substantially alter the magnitude estimate.

In some cases there may be additional data or information that could be reasonably gathered to reduce uncertainties or better describe inherent variability. The value and importance of collecting more data was considered in this BERA.

Evidence of Causality (Low, Moderate, or High)

This attribute is the degree to which the data indicates that observed adverse effects are due to the ash or ash-related COPECs. This is considered for each LOE and for the overall WOE. The criteria for evaluating causality considered during the risk characterization process for each receptor group (as applicable) are provided below:

- **Spatial Correlation** – Effects occur at the same place as exposure; effects do not occur where there is no exposure.
- **Temporal Correlation** – Effects occur with or after an exposure.
- **Biological Gradient or Strength** – Effects decline as exposure declines over space or time.

- **Plausibility (Mechanism)** – Evaluation of the strength of the evidence for a mechanism (direct or indirect) by which the stressor (ash or specific ash-related COPECs) causes the observed effects.
- **Plausibility (Stressor-Response)** – The magnitude of observed effects is consistent with the level of exposure to the stressor.
- **Experimental Verification** – Effects observed under controlled experimental conditions appear to be reflected in field data.
- **Consistency of Stressor-Effects Association** – Similar effects are observed in other studies or locations with the same or similar stressors.
- **Specificity of Cause** – The effect tends to be associated with a particular stressor.

Based on the evaluation of applicable criteria, the evidence of causality generally fell into three qualitative categories: Low, Moderate, and High. These determinations were based on best professional judgment and supported by explicit documentation in the risk characterization sections. The evidence of causality was generally considered to be:

- "Low" for the non-positive associations, be they strongly negative or too weak to reasonably classify as positive.
- "Moderate" if it is more likely than not that the observed effect is caused by the stressor of interest (ash or ash-related COPECs). It is possible, though not probable, that one or more other factors are responsible for the observed effects.
- "High" if the observed adverse effects are probably caused by the stressor of interest. (i.e., it is unlikely that one or more other factors are responsible for the observed effects.)

Relevance to the Assessment Endpoint (Low, Moderate, or High)

This attribute is the degree of association between the measure of potential adverse effects and the actual assessment endpoints to be protected at the Site. Standard ecological assessment endpoints are intended to be protective of populations and communities, rather than individuals. Where direct measurement of assessment endpoints is not feasible, population-related attributes (e.g., survival, growth, or biomass, and reproduction) are used as measurement endpoints. The uncertainties inherent in the extrapolation from measurement endpoints to assessment endpoints are the basis for this attribute of an LOE. This is sometimes referred to as biological or ecological relevance.

Therefore, relevance to the assessment endpoint was generally considered to be:

- "Low" for LOEs that rely on literature-derived effects values, especially those that include assumptions of bioavailability (e.g., direct exposure to ambient media or dietary exposures).
- "Moderate" for LOEs that use site-specific receptor data that are direct measures of population-related attributes (e.g., reproductive condition in target fish species collected at the Site) or use measures of

population-related attributes in laboratory organisms exposed to site media (e.g., survival and biomass of invertebrates in toxicity tests site sediments).

- "High" for LOEs that include direct measurement of the assessment endpoint being evaluated (e.g., abundance and species richness of the fish community at the Site).

To help ensure consistency among receptors, relevance to the assessment endpoint for a given LOE was not adjusted based on the number or type of other LOEs available for a receptor.

2.4.3.2 Summary and Integration of Evidence

Following is a brief description of the primary elements of the risk characterization for each LOE and the overall WOE.

Relative Weight (Low, Moderate, or High)

This is a measure of how much influence an LOE has when weighing multiple LOEs and making a risk determination for the receptor. In general, LOEs with greater relevance to the assessment endpoint receive greater weight. For example, the fish community data are highly relevant because they are a direct measure of the assessment endpoint (the fish community) and, thus, have a "High" relative weight rating.

Potential Risk (Negligible, Low, Moderate, High)

This is the determination as to whether the LOE is consistent with the assessment endpoint being at risk and the probable magnitude of potential adverse effects on the assessment endpoint. The risk determinations are based on best professional judgment backed by explicit and detailed analyses and descriptions.

Confidence in Risk Determination (Low, Moderate, or High)

This is an estimate of the overall uncertainty associated with the risk determination for a given LOE and for the WOE. As with the risk determination, the confidence rating (i.e., Low, Moderate, or High) is based on best professional judgment backed by explicit and detailed analyses and descriptions. The two risk characterization attributes described above that have the most bearing on the confidence in the risk determination are the Confidence in Magnitude of Adverse Effects and the Evidence of Causality.

- A "Low" confidence rating suggests that the risk determination is as likely to be "incorrect" as it is to be "correct." There are various reasons one might have low confidence in a risk determination, some of which are COPEC-specific and/or receptor-specific. The basis for "Low" confidence ratings are specified in the risk characterization and uncertainties for each LOE.

- A “Moderate” confidence rating suggests that it is more likely than not that the determination of risk is correct. It is possible, but not probable, that refining the uncertainties will change the determination of risks. There are various reasons for a “Moderate” rating, including that there is moderate evidence of causality or the magnitude of adverse effects is marginal.
- A “High” confidence rating suggests that there is little chance that refining the uncertainties will change the determination of risks. The magnitude of effects may be so great or the available dataset sufficiently robust that any additional data or information is unlikely to significantly alter the determination of risk.

For each of these ratings there may or may not be additional data or information that could reasonably be gathered to reduce uncertainties or better describe inherent variability, depending on the nature of the available data. The value and importance of collecting more data is addressed in the uncertainties for the receptors.

COPECs/COECs

These are the COPECs most likely to be associated with observed or predicted adverse effects. As the name implies, COPECs do not necessarily pose a risk to the assessment endpoint; rather, they have the “potential” to pose a risk and must be further evaluated to determine whether the available evidence is sufficient to make a risk determination. COPECs can be eliminated from further evaluation at various points in the risk characterization process; however, they are not considered to be constituents of ecological concern (COECs) until the final determination of risks is established. This risk determination is made via the WOE process for each assessment endpoint. Therefore, COPECs are identified for each LOE, whereas COECs are identified only for the final WOE. For example, in Exhibit 2-3, seven COPECs were identified for the surface water chemistry LOE (i.e., aluminum, arsenic, cadmium, copper, lead, selenium, and zinc), but none were identified as COECs based on the WOE. The list of COPECs identified for the BERA are presented in Exhibit 2-4, below.

2.5 Data Evaluation and Ecological Screening

The following subsections describe the data collected to support the assessment of risks to biota. Spatial and temporal considerations are defined, as well as the data usability. A brief description of each medium is provided, along with initial screening results for abiotic media based on the full dataset used in this BERA. This is a refinement of the preliminary screening presented in the SLERA, for which only limited ambient water quality data were available.

The sample types and quantities discussed in the following sections are the actual number of environmental samples and do not include quality assurance/quality control (QA/QC) samples that were collected per the requirements of the *Quality Assurance Project Plan for the Tennessee Valley Authority Kingston Ash Recovery Project, Revision 1* (QAPP; TVA 2010), which was supported by the USEPA. QA/QC samples

planned in support of this plan include field duplicates, field blanks, equipment rinsate blanks, and matrix spike/matrix spike duplicates. The QA/QC samples are defined in Section 14 of the QAPP and were collected per the frequencies detailed in Table 14.1 of the QAPP.

Analysis of 26 metals/metalloids, naturally-occurring radionuclides, and legacy constituents, and other legacy constituents associated with the Emory and Clinch Rivers selected as COPECs in the BERA are presented in Exhibit 2-4.

Exhibit 2-4. Constituents of Potential Ecological Concern (COPECs)

Metals / Metalloids	Radionuclides / Legacy Constituents	
Aluminum	Cesium-137	Total PCBs
Antimony	Cobalt-60	Total PAHs
Arsenic	Potassium-40	Chlordane
Barium	Radium-226	Mercury
Beryllium	Radium-228	
Boron	Thorium-228	
Cadmium	Thorium-232	
Calcium	Uranium-234	
Chromium	Uranium-235	
Cobalt	Uranium-238	
Copper		
Iron		
Lead		
Magnesium		
Manganese		
Mercury		
Molybdenum		
Nickel		
Potassium		
Selenium		
Silver		
Sodium		
Strontium		
Thallium		
Vanadium		
Zinc		

Notes:
This list includes only the specifically identified legacy constituents for the Emory and Clinch Rivers. It does not include the full suite of radionuclides, PCBs, or pesticides analyzed.

Acronyms and Abbreviations:
PAHs - polycyclic aromatic hydrocarbons
PCBs – polychlorinated biphenyls

2.5.1 Spatial Considerations

Spatial considerations were evaluated for each of the assessment endpoints established within this BERA, as well as the various types of media needed for each endpoint. The spatial boundaries of the river system are those areas impacted by the ash release, upstream to downstream within the Emory, Clinch, and Tennessee Rivers. Each river was divided into reaches to more adequately characterize potential effects to the selected assessment endpoints. The following reaches were established for each river (Figure 2-8):

- **Emory River** – Reach A (ERM 0.0 to ERM 1.5); Reach B (ERM 1.5 to ERM 3.5); Reach C (ERM 3.5 to ERM 6.0)
- **Clinch River** – Reach A (CRM 0.0 to CRM 3.0); Reach B (CRM 3.0 to CRM 4.5)
- **Tennessee River** – Reach A (TRM 550.0 to TRM 566.0; Reach B (TRM 566.0 to TRM 568.0).

The following reference locations were also established for each river to provide an accurate comparison to pre-release conditions:

- **Emory River** – Reference (above ERM 6.0)
- **Clinch River** – Reference (above CRM 4.5)
- **Tennessee River** – Reference (above TRM 568).

2.5.2 Temporal Considerations

Temporal considerations were also evaluated for each assessment endpoint established for this BERA, as well as for the various types of media needed for each endpoint. The temporal boundaries were primarily defined by the following events: the December 22, 2008 fly ash release; completion of the dredging associated with the time-critical removal action; and completion of the EE/CA (and associated risk assessments) for the river system. These events define the following time periods for which various types of data were available for use in the BERA:

- **Pre-Release Period** – prior to December 22, 2008
- **Time-Critical Removal Period** – from May 11, 2009 to completion of dredging associated with the time-critical removal action on May 29, 2010
- **Post-Time-Critical Removal Period** – from completion of dredging associated with the time-critical removal action (May 29, 2010) to completion of the river system BERA and EE/CA (anticipated June 2012).

While comparison of results collected a year or more later to data from samples collected immediately after the ash release should provide an indication of trends in bioaccumulation, concentrations of bioaccumulative

ash-related COPECs may increase in biota over time. Consequently, collection of data over more than one sampling period may be necessary for quantifying bioaccumulation trends and for long-term management. Two to three years of sampling often is necessary to characterize potential bioaccumulation to higher trophic-level receptors. A portion of the data was collected during and shortly after dredging (Table 2-12). Dredging occurred in Emory reaches B and C (ER_B and ER_C), and engineering controls were used to minimize migration of disturbed sediment and ash. Therefore, the potential influence of dredging is likely to be limited. More weight was given to samples collected in the post-time-critical removal period, assuming that these data better represent the current condition of the river system.

Most data collected in the spring and fall of 2011 were not available for use in the quantitative BERA evaluation. When possible, these data were used to quantitatively supplement the estimated risks to ecological receptors, as presented in the uncertainty sections at the end of each receptor section.

2.5.3 Data Usability Assessment

This section focuses on the evaluation of data quality and usability for risk assessment purposes. The following issues were considered in the data usability assessment: data quality, data qualifications, data preparation, detection limits, and data adequacy.

2.5.3.1 Data Quality

The USEPA's *Guidance for Data Quality Assessment: Practical Methods for Data Analysis* (USEPA 2000c) provides general guidance on assessing data quality criteria and performance specifications for decision making. As described by USEPA, "Data Quality Assessment (DQA) is the scientific and statistical evaluation of data to determine if data obtained from environmental data operations are of the right type, quality, and quantity to support their intended use." The guidance specifies that DQA is built on the fundamental premise that data quality, as a concept, is meaningful only when it relates to the intended use of the data. As such, one of the questions the DQA is designed to address is: "Can the decision (or estimate) be made with the desired confidence, given the quality of the dataset?"

Data from various investigation activities have been collected at the Site, beginning immediately after the ash release and continuing through the present. Percent recoveries of spiked samples and certified reference materials were within the QAPP-specified or laboratory-derived acceptance limits in most cases. Data associated with recoveries outside these limits were qualified as estimated (J flagged). Specific data quality issues for individual samples or sampling events are discussed in the site-specific risk assessment sections.

2.5.3.2 Evaluation of Data Qualifiers

The following criteria were employed to determine whether a particular sample result was usable for risk assessment purposes. To determine whether a particular sample was usable for risk assessment purposes, the following data validation qualifiers were assigned to the result:

- U* – This result should be considered "not detected" because it was detected in an associated field or laboratory blank at a similar level.
- R – Unreliable positive result; analyte may or may not be present in sample; sample result was rejected.
- UR – Unreliable reporting or detection limit; analyte may or may not be present in sample; sample result was rejected.
- J – Quantitation is approximate due to limitations identified during data validation.
- UJ – This analyte was not detected, but the reporting or detection limit may or may not be higher due to a bias identified during data validation.

Sample results rejected during data validation were not considered usable for risk assessment. These data, indicated by an "R" or "UR" data qualifier, were eliminated from the risk assessment dataset. The laboratory performed a percent moisture analysis on each sediment sample in accordance with USEPA SW-846 procedures for determining dry sample weight, as well as for the biota samples. In some instances, sample masses were too small or samples were held too long to accurately measure percent moisture in several types of biotic tissue (Environmental Standards, Incorporated [ES] 2011); accordingly, percent moisture results in biotic tissue were qualified as unusable and were not used to adjust sample results to a dry-weight basis. When possible, valid percent moisture estimates for samples were averaged to create a site-specific percent moisture average for each tissue type. When site-specific percent moisture values were not available, default literature-derived values were used.

A correction factor (CF) was established for some mercury data. Results for split samples generated by multiple organizations revealed an apparent low bias for mercury data when analyzed by inductively coupled plasma/mass spectrometry (ICP/MS) (SW-846 Method 6020) as compared to cold-vapor atomic absorption (SW-846 Method 7471) spectroscopy and direct mercury analysis (DMA) (SW-846 Method 7473). This bias was confirmed by TVA's contracted laboratory through the analysis of multiple standard reference materials by each of the three methodologies. Based on the observed inconsistencies, studies are underway to evaluate ICP/MS bias compared to DMA data on a species-specific and tissue-specific basis. These studies are intended to identify appropriate CFs for application to historical mercury data generated by ICP/MS analysis.

For fish fillet samples, a statistical analysis of previously reported mercury data was performed to determine the number of samples required to evaluate bias with 90 percent certainty. Based on this analysis, 28 samples were selected from archive and analyzed for mercury by ICP/MS and DMA. The results for

mercury by the two methods were compared for each fillet sample, and it was determined that the DMA mercury results were consistently 1.6 times greater than the ICP/MS mercury results (r^2 = 0.93). Additional studies are underway to evaluate bias in various fish tissue aliquots, macroinvertebrates, and avian eggs. Data generated during late 2011 and early 2012 for macroinvertebrates are currently under analysis to determine whether a sufficient dataset has been generated to determine the CF for these species with an appropriate certainty (e.g., 90 percent).

2.5.3.3 Data Preparation

The data were organized into different datasets for the different data end uses. For example, the dietary exposure modeling used mid-column surface water data while the invertebrate assessment used sediment porewater data. Prior to usage, the datasets were prepared as described below.

Treatment of Duplicates

The following rules were applied to duplicates whether they were field duplicates or analytical duplicates. Analytical results for the same COPEC that report the same type of data (e.g., total metal) in one sample obtained by two or three different analytical methods or by different laboratories were combined. (Note: This does not apply for metals analyzed in specialty analyses such as AVS/SEM, sequentially extracted metals, or metals speciation).

- Values for non-detects were included at the reporting limit (RL) reported by the laboratory.
- The greater result of the parent and duplicate sample was used if a COPEC was detected in both. The lower of the RL was used if the COPEC was not detected in either the parent sample or the duplicate, but the detected value was used when the COPEC was detected in one of the parent/duplicate samples.

Constituents of Potential Ecological Concern Classes

Some COPECs behave similarly, especially those in the same class of compounds; thus, they were grouped together for evaluation in this BERA. The sum of the detected concentrations plus the method detection limits (MDLs) for non-detected results were reported for use in the BERA. Chemicals included in the summed groups were as follows:

- Total dichlorodiphenyltrichloroethane (DDT) (sum of three constituents):
  - 4,4′-Dichlorodiphenyldichloroethane (DDD)
  - 4,4′-Dichlorodiphenyldichloroethylene (DDE)
  - 4,4′-DDT

- Total low-molecular-weight PAHs (LPAHs) (sum of six constituents: molecular weight ≤ 192):
  - Acenaphthene
  - Acenaphthylene
  - Anthracene
  - Fluorene
  - Naphthalene
  - Phenanthrene

- Total high-molecular-weight PAHs (HPAHs) (sum of 10 constituents: molecular weight ≥ 202):
  - Benzo(a)anthracene
  - Benzo(a)pyrene
  - Benzo(b)fluoranthene
  - Benzo(g,h,i)perylene
  - Benzo(k)fluoranthene
  - Chrysene
  - Dibenzo(a,h)anthracene
  - Fluoranthene
  - Indeno(1,2,3-cd)pyrene
  - Pyrene

- Total Aroclor PCBs (sum of seven constituents):
  - Aroclor 1016
  - Aroclor 1221
  - Aroclor 1232
  - Aroclor 1242
  - Aroclor 1248
  - Aroclor 1254
  - Aroclor 1260

2.5.3.4 Evaluation of Detection Limits

Detection limits were evaluated to determine suitability of data for BERA purposes. As a conservative measure, RLs were used for non-detected COPEC concentrations. Individual sample RLs may have varied from the laboratory's routinely reported limits; this variance typically was the result of dilution factors, sample weight or volume available to perform the analysis, dry weight adjustment for solid samples, the presence of analytical background contaminants, or other sample-related or analysis-related conditions. In some instances, results may be reported between the RL and MDL. When results were reported between the RL and MDL, those results were reported as estimated values (TVA 2010).

Target reporting limits were compared to ecological screening values for abiotic media and were considered adequate for use in risk assessment screening (Jacobs 2010a). Screening values for all biotic media were generally not available for comparison to reporting limits; however, the lowest possible reporting limits were requested from the laboratory and were generally below literature-based adverse effects levels. Exceptions to this are discussed in the specific tissue evaluations for each receptor.

2.5.3.5 Data Adequacy Summary

Analytical data generated for the Kingston Ash Recovery Project underwent a critical QA review as specified in the QAPP. In general, the data met the data quality objectives defined for the BERA and are acceptable for use. A summary of the description of the laboratory deliverables reviewed, the level of review (verification and validation), the QC measures included in the review, and summary tables representing the overall data quality are presented in Appendix B.

2.5.4 Abiotic Media

Abiotic media collected in support of the BERA included ash deposits, seasonally-exposed sediments, submerged sediments, residual sediments, sediment porewaters, surface waters, and groundwaters. A description of each dataset, including distribution of sampling, applicable screening values, and results of the ecological screening is provided in the subsections below.

2.5.4.1 Ash Deposits

The volume and distribution of any residual ash deposits (e.g., depth, thickness, areal extent, and degree of mixing or layering) were estimated to distinguish ash from non-ash sediment. These methods included:

1. Visual observations, where the gray ash can be readily differentiated from the natural brown river sediments.
2. Polarized light microscopy (PLM), which provides a quantifiable estimate of the proportion of ash in a sample of sediment.

The PLM method views mineral specimens under polarized light. In this method, a set number of random fields is examined by polarized microscopy and the proportion of spherical particles to total particles counted is used to quantify the proportion of ash in the sample.

Sampling of ash deposits was done using Vibecore™ sampling techniques. In general, samples were taken on a regular grid layout, with samples at any given grid transect taken at mid-channel and both to the left and right of the channel (Figures 2-9 through 2-11). Results of the U.S Army Corps of Engineers--Engineer Research and Development Center (USACE) adaptive hydraulics sediment transport modeling were used to

help guide sampling, focusing on locations where ash deposits were likely to occur. Sampling frequency was greater in reaches where ash deposition from the release was greatest and decreased outside those reaches. Samples were visually described, particularly visual presence of ash, thickness, and any depositional layering. Ash thicknesses were measured to within 0.1 ft. Quality control included confirmatory laboratory PLM estimates on at lease 10 percent of the total samples and also on all samples field-estimated as being greater than 50 percent ash. A total of 268 field PLM samples were collected, and 113 samples were sent for confirmatory laboratory PLM analysis.

2.5.4.2 Seasonally-Exposed Sediments

Seasonally-exposed sediments are sediments that become exposed on the river banks during the times that the reservoir levels are low (i.e., winter months--December to April). These sediments are submerged in shallow water the remainder of the year, usually near the river banks. Aquatic plants and aquatic- or riparian-feeding birds and mammals may be exposed to these shallow water sediments. Benthic fish and invertebrates may be exposed to these sediments whenever the reservoir is at the summer pool level (Jacobs 2010a).

Samples of seasonally-exposed sediment were collected using hand-auger methods at winter pool elevations. Samples were collected randomly along each shoreline (right and left bank), regardless of particle make-up (ash/native sediment proportion). The study boundaries were areas impacted by the ash release, upstream to downstream. Exposures would occur only in the upper 6 inches of seasonally-exposed or shallow sediments; therefore, the depth of sampling was limited to the upper 6 inches. A total of 70 samples were collected (Figures 2-12 and 2-13).

Samples were visually described, with focus on visual presence of ash and ash thickness or any depositional layering. Field PLM estimates of percentage of ash were made on site, based on a scale of greater than or less than 50 percent ash, and were recorded in the VibeCore™ sampling summary log. Confirmatory laboratory PLM estimates were made at an off-site laboratory on 10 percent of the total samples (eight total) as QC. Samples (70) were analyzed in an off-site analytical laboratory for total metals and metalloids.

Exposures to naturally-occurring radionuclides at near-background levels of radioactivity may pose risks to ecological receptors as a result of direct exposure to the exposed sediments. Therefore, 25 percent of the grab samples (18 total) were also analyzed for naturally-occurring radionuclides (potassium-40, radium-series, thorium-series, uranium-series), plus the USDOE legacy radionuclides (cesium-137 and cobalt-60).

In order to estimate potential effects of legacy constituents, 25 percent (18 samples) of the grab samples of seasonally-exposed sediments also were analyzed for river-specific legacy constituents. Emory River legacy constituents are polycyclic aromatic hydrocarbon (PAHs), PCBs, and pesticides. Clinch and Tennessee Rivers

legacy constituents are mercury, PAHs, PCBs, cesium-137, and cobalt-60. Because mercury, cesium-137, and cobalt-60 are already included in the analysis above, the 25 percent Clinch River legacy samples were analyzed for PAHs and PCBs only. Analyses for PAHs were to determine concentrations of the 34 parent and alkylated PAHs specified in the USEPA procedure for evaluating PAH mixtures in sediment (USEPA 2009). Because of the seasonally-exposed sediments' potential impacts on COPEC migration and/or toxicity, the same 25 percent of the samples also were analyzed for chemical speciation for arsenic, mercury, selenium, and chromium.

Concentrations of ash-related COPECs in the seasonally-exposed sediments were compared with literature-derived effects values from USEPA *Region 4 Waste Management Division Sediment Screening Values for Hazardous Waste Sites* (USEPA 1994b), so the potential exposures for ecological receptors could be estimated. Tables 2-13 through 2-18 present summaries of maximum seasonally-exposed sediments concentrations compared to the available sediment criteria. Seasonally-exposed sediments sample data are presented in Appendix A. As shown in Exhibit 2-5, the maximum detected values of some COPECs exceeded screening values. For 16 COPECs, screening values were not available.

Exhibit 2-5. Seasonally-Exposed Sediments – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available	
Arsenic	Aluminum	Strontium
Chromium	Barium	Thallium
Copper	Beryllium	Vanadium
Lead	Boron	Arsenic (speciation)[2]
Mercury	Cobalt	Mercury (speciation)[3]
Nickel	Hexavalent Chromium	PAHs[4]
Zinc	Manganese	Pesticides[5]
Chlordane (total, alpha, and gamma)	Selenium[1]	Radionuclides[6]
Total HPAH		
Total PCBs		

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Selenium includes selenium (total), selenite, inorganic selenium, and organic selenium.
[2] Arsenic speciation includes arsenate, arsenite, inorganic arsenic, and organic arsenic.
[3] Mercury speciation includes methylmercury and inorganic mercury.
[4] PAHs include: total alkylated PAHs and total un-substituted PAHs; individual PAHs without screening values are listed in Tables 2-1 through 2-18.
[5] Pesticides include: alpha-BHC, beta-BHC, heptachlor, and total pesticides.
[6] Radionuclides include: actinium-228, bismuth-214, cesium-137, lead-212, lead-214, potassium-40, radium-226, radium-228, thallium-208, thorium-228, thorium-230, thorium-232, thorium-234, uranium-234, uranium-235, and uranium-238.

Acronyms and Abbreviations:
BHC – hexachlorocyclohexane
COPEC – constituent of potential ecological concern
HPAH – high-molecular-weight PAH
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

2.5.4.3 Submerged Sediments

Submerged sediments are sediments that are below water year-round. Aquatic biota are directly exposed to submerged sediments, and wildlife that inhabit or forage in the river system may be exposed to submerged sediments in their diet.

The study boundaries were areas impacted by the ash release, upstream to downstream. Ecological exposures would occur only in the upper 6 inches of submerged sediments; therefore, the depth of sampling was limited to the upper 6 inches. A total of 82 submerged sediments samples were selected from the ash deposit sampling (Figures 2-14 through 2-16). All grab samples were analyzed in an off-site analytical laboratory for total metals and metalloids. Submerged sediments samples were collected concurrently with ash deposit samples. In general, samples were selected randomly from among the ash deposit samples regardless of particle make-up. Results from the submerged sediments sampling provides an estimate of the spatial heterogeneity in the sediments from the Emory, Clinch, and Tennessee Rivers.

Samples were visually described, particularly visual presence of ash and ash thickness or any depositional layering. Field PLM estimates of percentage of ash were made on site, based on a scale of greater than or less than 50 percent ash, and were recorded in the VibeCore™ sampling summary log. Confirmatory laboratory PLM estimates were made at an off-site laboratory on 10 percent of the total samples and also on the samples estimated as being greater than 50 percent ash (35 total) as QC. Samples were analyzed in an off-site analytical laboratory for total metals. Grain size analyses were also performed.

Exposures to naturally-occurring radionuclides at near-background levels of radioactivity may pose risks to ecological receptors as a result of direct exposure to the submerged sediments. To confirm this low risk, 25 percent of the samples (22 total) were analyzed for naturally-occurring radionuclides (K-40, radium-series, thorium-series, uranium-series) plus USDOE legacy radionuclides (cesium-137 and cobalt-60).

To estimate potential effects of legacy constituents, 25 percent of the samples (22 total) also were analyzed for river-specific legacy constituents. Emory River legacy constituents are PAHs, PCBs, and pesticides. Clinch and Tennessee Rivers legacy constituents are mercury, PAHs, PCBs, cesium-137, and cobalt-60. Because mercury, cesium-137, and cobalt-60 are already included in the analysis above, the 25 percent Clinch River legacy samples were analyzed for PAHs and PCBs only. Analyses for PAHs determined concentrations of the 34 parent and alkylated PAHs specified in the USEPA procedure for evaluating PAH mixtures in sediment (USEPA 2009). Samples analyzed for PAHs, PCBs, or pesticides also were analyzed for total organic carbon (TOC) so that equilibrium partitioning methods could be used to estimate ecological exposures. To evaluate potential impacts of submerged sediments on COPEC migration, bioavailability, and/or toxicity, 25 percent of the samples (22 total) also were analyzed for chemical speciation for arsenic, mercury, selenium, and chromium, and for AVS/SEM.

Concentrations of ash-related COPECs in the submerged sediments were compared with literature-derived effects values (USEPA 1994b), so the potential exposures for ecological receptors could be estimated. Tables 2-19 through 2-28 present summaries of maximum submerged sediments concentrations compared to the available sediment criteria. Submerged sediments sample data are presented in Appendix A.
As shown in Exhibit 2-6, the maximum detection of some COPECs exceeded the screening value. For 16 COPECs, screening values were not available.

Exhibit 2-6. Submerged Sediment (VibeCore™) – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available	
Arsenic	Aluminum	Strontium
Chromium	Barium	Thallium
Copper	Beryllium	Vanadium
Lead	Boron	Arsenic (speciation)[2]
Mercury	Cobalt	Mercury (speciation) [3]
Nickel	Hexavalent Chromium	PAHs[4]
Zinc	Manganese	Pesticides[5]
Total HPAHs	Selenium[1]	Radionuclides[6]
Total LPAHs		
Total PCBs		
Aroclor 1260		

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Selenium includes selenium (total), selenite, inorganic selenium, and organic selenium.
[2] Arsenic speciation includes arsenate, arsenite, inorganic arsenic, and organic arsenic.
[3] Mercury speciation includes methylmercury and inorganic mercury.
[4] PAHs include: total alkylated PAHs and total un-substituted PAHs; individual PAHs without screening values are listed in Tables 2-19 through 2-28.
[5] Pesticides include: alpha-BHC, beta-BHC, delta-BHC, heptachlor, and total pesticides.
[6] Radionuclides include: actinium-228, bismuth-214, cesium-137, lead-212, lead-214, potassium-40, radium-226, radium-228, thallium-208, thorium-228, thorium-230, thorium-232, thorium-234, uranium-234, uranium-235, and uranium-238.

Acronyms and Abbreviations:
BHC – hexachlorocyclohexane
COPEC – constituent of potential ecological concern
HPAH – high-molecular-weight PAH
LPAH – low-molecular-weight PAH
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

Based on the results from the submerged sediments samples collected in the Emory and Clinch Rivers, a subset of locations were chosen for laboratory sediment toxicity testing, hereafter referred to as “bulk sediment.” These included ten locations on the Emory River (including two reference locations), and ten locations on the Clinch River (including two reference locations) (Figure 2-17). After collections were

complete, the sediments from the two reference locations for each river were composited to create one reference sample for each river. These samples (18 total) were analyzed for the same suite of 26 metal and metalloid COPECs; chemical speciation (arsenic, chromium, mercury, and selenium); legacy PAHs, PCBs, and pesticides; grain size and TOC; percent ash; and sequentially extracted metals. Samples also were analyzed for AVS/SEM. Note: reference samples were not collected for use in statistical comparisons with impacted locations. Rather, these samples were collected in order to compare exposures at upstream locations and were also used to dilute impacted site samples in laboratory sediment dilution series toxicity tests.

Concentrations of ash-related COPECs in the bulk sediments also were compared with literature-derived effects values (USEPA 1994b), so the potential exposures for ecological receptors could be estimated. Tables 2-29 through 2-35 present summaries of maximum bulk sediment concentrations compared to the available sediment criteria. Bulk sediment sample data are presented in Appendix A. As shown in Exhibit 2-7, the maximum detection of some COPECs exceeded the screening value or a screening value was not available.

Exhibit 2-7. Bulk Sediment for Laboratory Toxicity Tests – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available	
Arsenic	Aluminum	Strontium
Copper	Barium	Thallium
Mercury	Beryllium	Vanadium
Nickel	Boron	Arsenic (speciation)[2]
Zinc	Cobalt	Mercury (speciation) [3]
Total HPAH	Hexavalent Chromium	PAHs[4]
Total LPAH	Manganese	Pesticides[5]
Total PCBs	Selenium[1]	Radionuclides[6]
Aroclor 1254		

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Selenium includes selenium (total), selenite, inorganic selenium, and organic selenium.
[2] Arsenic speciation includes arsenate, arsenite, inorganic arsenic, and organic arsenic.
[3] Mercury speciation includes methylmercury and inorganic mercury.
[4] PAHs include: total alkylated PAHs and total un-substituted PAHs; individual PAHs without screening values are listed in Tables 2-29 through 2-35.
[5] Pesticides include: aldrin, alpha-BHC, beta-BHC, endosulfan II, heptachlor, and total pesticides.
[6] Radionuclides include: actinium-228, bismuth-214, cesium-137, lead-212, lead-214, potassium-40, radium-226, radium-228, thallium-208, thorium-228, thorium-230, thorium-232, thorium-234, uranium-234, uranium-235, and uranium-238.

Acronyms and Abbreviations:
BHC – hexachlorocyclohexane
HPAH – high-molecular-weight PAH
PAH – polycyclic aromatic hydrocarbon
COPEC – constituent of potential ecological concern
LPAH – low-molecular-weight PAH
PCB – polychlorinated biphenyl

2.5.4.4 Sediment Porewater

Sediment porewater is the interstitial water present between grains of sediment that is the primary source of exposure to rooted aquatic plants and benthic invertebrates (particularly burrowing organisms) near the base of the food web. The purpose of evaluating sediment porewaters separately from whole sediments is to understand the factors involved in desorption of COPECs from the ash/sediments and toxicity of the COPECs apart from any physical effects of ash on benthic growth (Jacobs 2010a). The sediment remaining after porewater has been extracted (residual sediment) also was evaluated.

When possible, porewater samples were collected from the same locations as the bulk sediment samples used in laboratory sediment toxicity testing (Figure 2-18). Porewater (and residual sediment) samples were collected with a boxcore sampler and lexan tubes, sealed, and shipped to an off-site analytical laboratory where they were centrifuged. Porewater was extracted and residual sediment was also retained for sampling. Deviations from SOP specified sampling procedures occurred at ERM 3.5, ERM 5.5, CRM 6.5, and CRM 7.5, when the boxcore sampler met refusal when sampling from the boat. ERM 3.5 could not be sampled for porewater due to hard river and shoreline substrates; however, the remaining three locations were sampled by hand near the shoreline using lexan tubes. A total of 19 samples were collected. Reference porewater samples were combined post-porewater extraction at a 1:1 ratio to create one reference sample for each river. Samples were filtered and analyzed for dissolved COPECs only, including metals. Samples were also tested for chemical speciation (arsenic and selenium), major ions (sulfides, chlorides, and hydroxides), hardness as calcium carbonate ($CaCO_3$), and dissolved organic carbon (DOC). DO, pH, oxidation-reduction potential (ORP), and specific conductivity (SC) were measured in the laboratory after extraction of porewaters.

Porewater samples were not tested for radionuclides, PAHs, PCBs, or pesticides due to practical limitations of collecting sufficient sample volumes and because bulk sediment and/or surface water analyses and organic carbon content can be used to estimate porewater concentrations of these COPECs.

Concentrations of ash-related COPECs in the porewaters were compared with literature-derived effects values for aquatic invertebrates (USEPA 1994b), so that the potential exposures for ecological receptors could be estimated. Hardness-dependent screening values were adjusted based on a conservative hardness value of 50 milligrams per liter (mg/L) for the reaches. Summaries of sediment porewater concentrations compared to the available aquatic life criteria are presented by river reach in Tables 2-36 through 2-42. Sediment porewater sample data are presented in Appendix A. As shown in Exhibit 2-8, the maximum detection of some COPECs exceeded the USEPA Region 4 chronic surface water screening value or a screening value was not available.

Exhibit 2-8. Porewater – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available
Aluminum	Barium
Arsenic	Cobalt
Boron	Manganese
Copper	Molybdenum
Iron	Strontium
Lead	Vanadium
Arsenite	Arsenic (speciation)[1]
	Selenium (speciation)[2]

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Arsenic speciation includes arsenate, inorganic arsenic, and organic arsenic.
[2] Selenium speciation includes selenite, inorganic selenium, and organic selenium.

Acronyms and Abbreviations:
COPEC – constituent of potential ecological concern

Residual sediments, the remaining sediments after porewater extraction, were evaluated for comparative purposes with other submerged sediments and were available in the event that porewater concentrations indicated a potential risk to ecological receptors. These samples were analyzed for metals, metals speciation (arsenic and selenium), and ash content, post-extraction. All 19 residual samples were analyzed. Similar to porewaters, residual sediment samples were not tested for radionuclides, PAHs, PCBs, or pesticides due to practical limitations of collecting sufficient sample mass.

Concentrations of ash-related COPECs in the residual sediments were compared with literature-derived effects values (USEPA 1994b), so that the potential exposures for ecological receptors could be estimated. Summaries of maximum residual sediment concentrations compared to the available sediment criteria are presented in Tables 2-43 through 2-49. The residual sediments sample data are presented in Appendix A. As shown in Exhibit 2-9, the maximum detection of some COPECs exceeded the screening value or a screening value was not available.

Exhibit 2-9. Residual Sediments – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values		Screening Values Not Available	
Arsenic	Mercury	Aluminum	Manganese
Copper	Nickel	Barium	Selenium[1]
Lead	Zinc	Beryllium	Strontium
		Boron	Vanadium
		Cobalt	Arsenic (speciation)[2]

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Selenium includes selenium (total), selenite, inorganic selenium, and organic selenium.
[2] Arsenic speciation includes arsenate, arsenite, inorganic arsenic, and organic arsenic.

Footnotes:
COPEC – constituent of potential ecological concern

2.5.4.5 Surface Waters

Surface water samples were collected to characterize water quality in the Clinch and Emory Rivers following completion of dredging. Aquatic biota are directly exposed to surface waters, and birds and mammals that inhabit or forage in the river system likely are exposed to surface waters in their diet.

The surface water data used in the assessment are the results from the 2010 surface water sampling that were collected from 11 fixed-station monitoring locations in the Emory, Clinch, and Tennessee Rivers (Figure 2-19). Surface water samples were collected at two discrete depth intervals using a peristaltic pump. One sample was collected at each location at mid-depth for use in evaluating pelagic fish and wildlife exposures, and a second sample was collected approximately 1.5 ft above the bottom for use in evaluating epi-benthic water for bottom-dwelling organisms. The 11 fixed-station monitoring locations were sampled once each week for eight weeks (8/31, 9/8, 9/14, 9/21, 9/29, 10/5, 10/12, and 10/19) to obtain sufficient quantity of data to demonstrate variability.

Water quality parameters (temperature, turbidity, DO, pH, ORP, and SC) were measured in the field using a multi-analyte programmable data logger. If temperature and DO measurements indicated thermal stratification of the water column, samples were collected at mid-depth in the epilimnion, mid-depth in the hypolimnion, and the epi-benthic water. For this assessment, if an epilimnion and hypolimnion sample were collected, the samples were treated essentially as duplicates, and only one result per location was reported (Section 2.5.3.3). Samples were analyzed in an off-site analytical laboratory for metals (total and dissolved). In addition, samples were analyzed for total suspended solids (TSS), total dissolved solids (TDS), DOC, and hardness as $CaCO_3$. A total of 176 samples were collected.

Based on the risk assessments conducted for the Swan Pond Embayment/Dredge Cell EE/CA (Jacobs 2010a), external exposure to gamma radiation was not a significant pathway of human exposure

for surface water, particularly for the naturally-occurring radionuclides present at the Site at near-background levels of radioactivity. To confirm that is an insignificant risk ecological pathway, 25 percent of the samples (44 total) were analyzed for naturally-occurring radionuclides (K-40, radium-series, thorium-series, and uranium-series), and USDOE legacy radionuclides cesium-137 and cobalt-60.

The analysis of surface water for legacy constituents was unlikely to provide useful information because PAHs, PCBs, and pesticides have low water solubility and partition strongly to sediment particles, making detection in surface water unlikely. Submerged and seasonally-exposed sediments were analyzed for these legacy constituents (Jacobs 2010a). Because of their potential importance to COPEC migration and/or toxicity, 25 percent of the surface water samples (44 total) were also analyzed for chemical speciation for arsenic and selenium (dissolved metals analysis only).

Concentrations of ash-related COPECs in the surface water were compared with literature-derived effects values for aquatic invertebrates (USEPA 1994b), so the potential exposures for ecological receptors could be estimated. Hardness-dependent screening values were adjusted based on a conservative hardness value of 50 mg/L for the reaches. Summaries of surface water concentrations compared to the available aquatic life criteria are presented by river reach in Tables 2-50 through 2-58 (mid-column surface water) and Tables 2-59 through 2-67 (epi-benthic surface water). Surface water summaries were separated into mid-column and epi-benthic water. Surface water sample data are presented in Appendix A. As shown in Exhibit 2-10, the maximum detection of some COPECs exceeded the USEPA Region 4 Chronic Surface Water Screening Value or a screening value was not available.

Exhibit 2-10. Surface Water (Mid-Column and Epi-Benthic) – Detected Constituent of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available	
Aluminum	Barium	Vanadium
Mercury	Cobalt	Arsenic (speciation)[1]
Lead (epi-benthic only)	Manganese	Selenium (speciation)[2]
	Molybdenum	Radionuclides[3]
	Strontium	

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] Arsenic speciation includes arsenate, inorganic arsenic, and organic arsenic.
[2] Selenium speciation includes selenate, inorganic selenium, and organic selenium.
[3] Radionuclides include: bismuth-214, cesium-137, lead-214, radium-226, radium-228, thorium-232, uranium-234, and uranium-238.

Footnotes:
COPEC – constituent of potential ecological concern

In addition to the routine surface water samples collected for the BERA, surface water samples were also collected for use in the sediment toxicity tests, hereafter referred to as "bulk water." Bulk water samples were collected from one location on the Emory River and one location on the Clinch River (Figure 2-17). Samples were collected and shipped on an as-needed basis to the toxicity testing laboratories (30 samples total). Bulk water was collected with a peristaltic pump and analyzed for metals, DOC, and TSS analysis.

Concentrations of ash-related COPECs in the bulk water were also compared with literature-derived effects values for aquatic invertebrates (USEPA 1994b), so the potential exposures for ecological receptors (particularly in the sediment toxicity tests) could be estimated. Exceedances of screening values and differences in hardness were considered part of the exposures to laboratory toxicity test organisms, and results were used for the evaluation of the toxicity test results. Hardness-dependent screening values were adjusted based on a conservative hardness value of 50 mg/L for the reaches. Summaries of bulk water concentrations compared to the available aquatic life criteria are presented by river in Tables 2-68 and 2-69. Bulk water sample data are presented in Appendix A. As shown in Exhibit 2-11, the maximum detection of some COPECs exceeded the USEPA Region 4 Chronic Surface Water Screening Value or a screening value was not available.

Exhibit 2-11. Bulk Water – Detected Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values	Screening Values Not Available
Aluminum	Barium
Silver	Cobalt
	Manganese
	Molybdenum
	Strontium
	Vanadium

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
COPEC – constituent of potential ecological concern

2.5.4.6 Groundwater

Groundwater exposure evaluations in this BERA are based on conditions in the shallow alluvial and residual soils where groundwater discharges to the river sediments predicted by groundwater flow and quality modeling results. Because current conditions do not reflect long-term COPECs flux to the river following closure of the Dredge Cell and Ash Pond, the concentrations of selected ash-related COPECs in groundwater were predicted based on modeled fate and transport simulations. The goal of the groundwater transport analysis was to quantify ash-related COPEC concentrations and mass loadings entering the

reservoir via groundwater seepage from source areas. These predictions were used in evaluating long-term risks to aquatic receptors.

The COPECs selected for modeling purposes were arsenic, mercury, chromium, selenium, radium-226, and thorium-228. These COPECs are naturally-occurring metals, metalloids, and radionuclides that are concentrated in the ash through the coal combustion process. These COPECs are representative of the primary constituents in ash that have been shown to contribute to potential risk to human or ecological receptors exposed to environmental media within the river system. These COPECs were modeled to predict future mass loadings of these COPECs following dredge cell closure and to represent the fate and transport behavior of other ash-related COPECs. A complete description of the groundwater transport modeling methodology can be found in Groundwater Transport Modeling Methodology (Jacobs 2010a).

In addition to the modeling, two permanent monitor wells were also used to evaluate groundwater inputs to the river system. Groundwater monitoring of the Dredge Cell was accomplished through sampling a number of temporary and permanent wells. Two permanent wells located nearest to the Dredge Cell wall, 6AR and 22, were selected to evaluate the potential for groundwater migration to the Emory River (Figure 2-20). Data from these two wells are used to provide a conservative estimate of groundwater exposure to benthic invertebrates living along the wall. All other organisms would only potentially be exposed to this medium after the groundwater dilutes with surface water. Samples from these two groundwater wells were analyzed for the same suite of 26 metals and metalloids, radionuclides, major anions, TSS/TDS, and ammonia.

Concentrations of ash-related COECs in the groundwater were conservatively compared with literature-derived effects values for aquatic invertebrates (USEPA 1994b), so the potential exposures for ecological receptors could be estimated. Hardness-dependent screening values were adjusted based on a conservative hardness value of 50 mg/L for the reaches. Summaries of groundwater concentrations compared to the available aquatic life criteria are presented in Tables 2-70 through 2-74. Groundwater summaries were separated into modeled concentrations and permanent wells. Groundwater sample data are presented in Appendix A. As shown in Exhibit 2-12, the maximum detection of some COECs exceeded the USEPA Region 4 chronic surface water screening value or a screening value was not available.

Exhibit 2-12. Modeled Groundwater and Permanent Well (6AR and 22) Constituents of Potential Ecological Concern

Maximum Exceeded Screening Values[1]	Screening Values Not Available
Aluminum	Barium
Beryllium	Cobalt
Boron	Manganese
Cadmium	Strontium
Copper	Sulfate
Iron	Radionuclides[2]
Nickel	
Thallium	
Zinc	

Notes:
Tables include only those COPECs that were detected at least once.

Footnotes:
[1] No exceedances occurred in the modeled groundwater results.
[2] Radionuclides include: bismuth-214, lead-214, and radium-228.

Footnotes:
COPEC – constituent of potential ecological concern

2.5.5 Biotic Media

Biotic media collected in support of the BERA included fish, benthic invertebrates, aquatic vegetation, piscivorous bird eggs, insectivorous bird eggs, raccoon tissues, amphibian tissues, and turtle tissues. These various types of wildlife were collected to estimate bioavailability and/or bioaccumulation of COPECs from their diet or other exposure pathways, as well as to estimate exposure and risk to their respective predators. Descriptions of each dataset, including the distribution of sampling, species and tissue selections, and chemical analyses are described in the subsections below.

2.5.5.1 Fish

Fish communities of interest include both pelagic (open-water or “top-feeding”) and benthic (bottom-dwelling or “bottom-feeding”) communities. Concentrations of ash-related COPECs may bioaccumulate in fish over time due to exposure to water, ash, and dietary items. In addition, fish are a food source for aquatic- or riparian-feeding piscivorous wildlife (animals that feed primarily on fish), providing a potential pathway for ash-related COPECs to enter the broader food web.

Fish data used quantitatively in this BERA are from sampling that began in 2009, immediately following the ash release, and continued through 2010 and early 2011. (Note: Fish samples collected in January 2011 were included as part of the “Fall 2010” fish collections.) Throughout the course of the sampling, a total of

11 fish species were collected, with various tissue types being analyzed. Largemouth bass and bluegill sunfish were selected as indicator species of pelagic fish communities, and channel catfish (*Ictalurus punctatus*) were selected as indicator species of benthic fish communities, each representing different behaviors and feeding ecologies.

In addition, gizzard shad were selected to evaluate the role and importance of a main forage fish (gizzard shad) as a mechanism by which COPECs are incorporated into lower levels of the aquatic-riparian food web. Gizzard shad are the primary forage fish of the larger piscivorous fish (i.e., bass) and piscivorous birds (i.e., great blue heron and osprey). Since gizzard shad feed on primary producers (algae), they represent an important step in the trophic transfer of ash-related COPECs.

Samples were collected from four river reaches (including one upstream reference location) in the Emory River and from three river reaches (including one upstream reference location) in the Clinch River (Figure 2-21). Samples were collected using boat electrofishing and gill netting in order to obtain sufficient sample masses for analysis.

Both filet and non-filet portions of individual fish samples (largemouth bass, bluegill sunfish, and channel catfish) were analyzed for metals and metalloids, and for arsenic speciation. Approximately 25 percent of the samples also were analyzed for PCBs and pesticides. Samples were not analyzed for PAHs because PAHs are readily metabolized and excreted. Approximately 25 percent of filet portions of largemouth bass, bluegill sunfish, and channel catfish were analyzed for radionuclides.

Gizzard shad samples were collected from the same sample reaches; however, because of their small size, up to ten individual fish (or more as necessary) were composited into a single sample, and up to three composite samples were collected at each location. Gizzard shad samples were analyzed for metals in whole fish, whole fish with gut contents removed, and gut contents only. In addition, approximately 25 percent of the gizzard shad samples were analyzed for arsenic speciation, PCBs, pesticides, and radium-226 and radium-228.

Summaries of fish concentrations are presented by river reach in Tables 2-75 through 2-116. Fish summaries were separated by tissue types. Fish sample data are presented in Appendix A.

Fish tissue collections continued in the spring and fall of 2011, following similar methodology from previous years. These data were not available for inclusion in the main BERA interpretation of fish; however, a brief evaluation of these data is presented in the Uncertainty Analysis in Section 3.3.2.

2.5.5.2 Benthic Invertebrates

Benthic invertebrates are organisms without backbones that live in the sediments on the bottoms of streams and rivers. These include crustaceans (e.g., crayfish and amphipods), mussels, clams, snails, aquatic worms, and immature forms of aquatic insects such as mayfly nymphs and non-biting midges. Because they inhabit the sediments and are relatively sedentary, benthic invertebrates are highly exposed to COPECs in the sediments and porewaters and are, therefore, indicators of environmental quality. This is particularly relevant for this area because the released ash was deposited on the river bottom. Concentrations of ash-related COPECs may bioaccumulate in benthic invertebrates over time. In addition, because they are at the base of the food web, benthic invertebrates and their emergent adult life stages are prey for other ecological receptors. The birds, mammals, and fish that feed on benthic invertebrates or emergent insects are ecological receptor populations of particular interest for bioaccumulative COPECs (Jacobs 2010a).

Benthic invertebrate sampling began in 2009, immediately after the release, and continued in 2010. One species of mayfly, *Hexagenia bilineata*, and one species of aquatic snail, *Pleurocera canaliculatum*, were selected as indicator species. In 2009, samples of larval mayflies and snails were collected at a total of six locations: one reference location upstream in each of the Emory, Little Emory, and Clinch Rivers (ERM 6.0, Little Emory River mile [LERM] 1.0, and CRM 6.0); two locations in the impacted Emory River reaches (ERM 1.0 and ERM 2.5); and one location in the impacted Clinch River reaches (CRM 1.5). In 2010, another reference location was added on the Tennessee River (TRM 572.5), and four additional potentially impacted locations were added at ERM 4.0, CRM 3.5, TRM 566.3, and TRM 560.8 (Figure 2-22). Because emergence of adult *Hexagenia* is unpredictable and sporadic, collections were made opportunistically at the same approximate locations. Larval mayflies were collected by taking multiple Ponar/Peterson grabs of sediment and selectively removing the organisms. Adult mayflies were collected using a combination of methods such as direct removal with forceps from vegetation along the shoreline and sweep nets. Snails were collected by hand from shallow rocky or stable wooden structures near the shoreline. Individual samples were composited by species. Three composite samples each of snails and larval mayflies were collected at each location, one of which was depurated (i.e., evacuated of digestive system contents) before analysis. Emergent mayflies do not feed and, therefore, were not depurated. When possible, depending on availability of organisms, three composite samples were collected within each reach to evaluate variability at that location.

Whole body snail samples and both life stages of mayfly samples were freeze dried and then analyzed for metals and metalloids. These tissues were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation due to the large quantity of material required for radionuclides and chemical speciation. Summaries of tissue concentrations in larval mayfly, adult mayfly, and snail sample data are presented by river reach in Tables 2-117 through 2-181, respectively. Samples are divided into depurated and non-depurated, as appropriate. Results are presented in Appendix A.

Benthic invertebrate tissue collections continued in the spring and summer of 2011, following similar methodology from previous years. Analysis of the 2011 results is used in the uncertainties analysis.

2.5.5.3 Aquatic Vegetation

Aquatic vegetation refers to plants that live in direct contact with sediments and/or surface waters at least part of the year. These include macrophytes, algae, and periphyton. Macrophytes are vascular aquatic plants that may be emergent (e.g., cattails, rushes, and sedges), submergent (e.g., elodea and milfoil), or floating (e.g., duckweed). Algae are non-vascular plants that may be single celled, multi-cellular, or filamentous. Periphyton refers to the complex of algae, bacteria, and detritus attached to submerged surfaces. Periphyton and algae account for a significant portion of primary productivity in many freshwater systems and are an important food source for various species of benthic invertebrates, amphibians, fish, birds, and mammals. Emergent and floating macrophytes can comprise a large portion of the diets of waterfowl and aquatic-feeding herbivorous mammals. Rooted macrophytes are in direct contact with sediments, porewaters, and surface waters. Epi-benthic algae and periphyton may be in contact with sediments and porewaters, in addition to surface waters. As such, aquatic plants are a potentially important pathway for trophic transfer of COPECs in water and sediment to aquatic biota and aquatic-feeding wildlife (Jacobs 2010a).

COPEC concentrations in aquatic plants were measured to evaluate the magnitude and likelihood of risk to ecological receptors through food web exposures. Birds and mammals that feed on macrophytes (e.g., ducks and muskrats) often forage in shallow waters and along shorelines. Therefore, macrophyte sampling focused on near-shore emergent vegetation. Isolated beds of emergent macrophytes were observed in some areas targeted for seasonally-exposed sediments sampling. Isolated patches of rushes, sedges, and cattails also were observed at the summer pool waterline in some parts of the river system. Measured COPEC concentrations in these plants were used as direct inputs for dietary exposure models for herbivorous and omnivorous wildlife.

COPEC concentrations in periphyton also were measured for evaluating bioavailability and the potential for trophic transfer of metals. Periphyton occur most densely in areas where ample sunlight is able reach the substrate. This also is where the aquatic snails being sampled for this project tend to be located because periphyton are a primary food item for these and other grazers. Therefore, locations selected for collection of aquatic snails were also sampled for periphyton once during the growing season.

Samples of emergent vegetation from along the summer pool shoreline were collected for chemical analysis at three locations in each reach of the Emory, Clinch, and Tennessee Rivers. Samples of emergent vegetation from below the summer pool shoreline also were collected for chemical analysis at three locations in each reach of the Emory, Clinch, and Tennessee Rivers. When possible, these were co-located with samples of seasonally-exposed sediments (Figure 2-23). Leaves and stems of emergent plants were

collected from above the ground or water surface. Samples of periphyton grown on artificial substrates were collected for chemical analysis at each of the benthic invertebrate (snails) sampling locations. Three periphyton samples were collected at each location and composited into one sample.

Aquatic vegetation samples were analyzed for metals, metalloids, and percent moisture. Aquatic plants were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation, both because hydrophobic COPECs are not expected to accumulate in plant materials, and due to the large quantity of material required for radionuclide and chemical speciation analyses. Summaries of constituent concentrations in emergent and shoreline samples, as well as periphyton, are presented in Tables 2-182 through 2-203, respectively. Aquatic vegetation sample data are presented in Appendix A.

2.5.5.4 Piscivorous Birds

Various types of aquatic- and riparian-feeding birds inhabit or forage along the river system. Concentrations of ash-related COPECs may bioaccumulate in these birds over time and may adversely affect their populations. Great blue heron and osprey were selected for evaluation because both are present throughout the Site, feeding on fish and other aquatic organisms that may be in direct contact with ash in the sediment or water column.

Heron and osprey egg sampling efforts began in 2009, approximately 4 months after the ash release, to provide baseline concentrations for future location comparisons. These sampling efforts were continued in 2010. When possible, nests located around the ash-impacted area near the KIF, downstream of the ash-impacted area, and upstream of the ash-impacted area (reference location) were sampled (Figure 2-24). One egg was collected from each occupied, accessible nest, with a target number of ten eggs per area. Egg samples were analyzed for metals, metalloids, and percent moisture, but were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation due to the large quantity of material required. Summaries of constituent concentrations in great blue heron and osprey egg sample data are presented in Tables 2-204 through 2-213, respectively. Results are presented in Appendix A.

Great blue heron egg collections continued in the spring of 2011, following similar methodology from previous years. These data were not available for inclusion in the main BERA interpretation of piscivorous birds; however, a brief evaluation of these data is presented in the Uncertainty Analysis in Section 6.3.2.

In addition, dietary exposure models were developed for piscivorous birds as well as for herbivorous, omnivorous, and insectivorous avians. These models were performed to assess wildlife exposure to the ash COPECs by calculating an exposure dose from each COPEC from the food web and abiotic media. Dietary exposure models contained species-specific parameters (e.g., body weights, ingestion rates, and diet compositions) and site-specific concentrations from abiotic media and prey items to estimate risk to birds.

2.5.5.5 Mammals

Various species of aquatic- and riparian-feeding mammals inhabit or forage along the river system. Concentrations of ash-related COPECs may bioaccumulate in these populations of mammals over time and adversely affect these wildlife populations. Raccoons were selected for evaluation because they are opportunistic omnivores present throughout the Site, consuming a wide variety of lower trophic-level prey items.

Raccoon sampling began in the fall of 2009 and continued in the fall of 2010. Traps were set in areas surrounding the KIF, as well as in off-site reference areas (Figure 2-25). In both years, adult raccoons were captured in live traps, blood samples were collected from live animals, and necropsies were performed following euthanasia of specimens. In addition to blood, hair, brain, gonad, kidney, liver, and muscle tissue samples were collected during necropsies. A total of ten raccoons from around the ash release area and five raccoons from a reference location were collected in each year.

Raccoon tissue samples were analyzed for metals, metalloids, and percent moisture. These tissues were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation. Summaries of the concentrations in each tissue type are presented in Tables 2-214 through 2-244, respectively. Results are presented in Appendix A.

Raccoon tissue collections continued in the fall of 2011, following similar methodology from previous years. These data were not available for the BERA interpretation; however, a brief evaluation of these data is presented in the Uncertainty Analysis in Section 7.3.2.

In addition, dietary exposure models were developed for herbivorous, omnivorous, and piscivorous mammals. These models were performed to assess wildlife exposure to the ash COPECs by calculating an exposure dose from each COPEC from the food web and abiotic media. Dietary exposure models contained species-specific parameters (e.g., body weights, ingestion rates, and diet compositions) and site-specific concentrations from abiotic media and prey items to estimate risk to mammals.

2.5.5.6 Tree Swallow

Aerial-feeding insectivorous birds, such as the tree swallow, commonly inhabit or forage along the river system. The tree swallow is widely used as an indicator species of local sediment contamination (Bishop et al. 1995; Nichols et al. 1995; Secord et al. 1999). They readily use nest boxes, which can be established at specific areas of interest. They feed near their nest box, primarily on emergent aquatic insects. As a result, residues in their tissues reflect location-specific sediment contamination for those chemicals that transfer into the insect biota. Concentrations of ash-related COPECs may bioaccumulate in these birds over time and may adversely affect their populations.

Tree swallow egg and nestling sampling began in 2009, approximately 5 months after the release, and continued and expanded in 2010. When possible, nests were established and sampled in locations around the ash-impacted area near the KIF, downstream of the ash-impacted area, and upstream of the ash-impacted area (reference location) (Figure 2-26). One egg was collected from each occupied, accessible nest, with a target number in each area of ten eggs in 2009 and 15 eggs in 2010. In addition, one 15-day nestling was collected from each occupied, accessible nest, with a target number in each area of ten nestlings in 2009 and 15 nestlings in 2010. When possible, egg and nestling collections were co-located from the same nest. In addition, a subset of 20 egg shells was selected and analyzed for metals in 2010. This included eggshells from potentially-impacted areas (13 eggs) and reference areas (seven eggs).

Tree swallow egg, eggshell, and nestling samples were analyzed for metals, metalloids, and percent moisture (when sufficient sample mass was available). Tree swallow tissues were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation due to the large quantity of material required for radionuclides and chemical speciation analyses. Summaries of tree swallow egg, eggshell, and nestling data are presented in Tables 2-245 through 2-274, respectively. Results are presented in Appendix A.

Tree swallow tissue collections continued in the spring of 2011 following similar methodology from previous years. In addition to data collected in previous years, a comprehensive reproductive study was also begun. These data were not available for the BERA interpretation; however, a brief evaluation of the reproductive data is presented in the Uncertainty Analysis in Section 8.3.2.

2.5.5.7 Amphibians

Amphibians have been considered a “sensitive sentinel to environmental change” due to their unique life cycle, which exposes them to aquatic dissolved metals as well as metals sequestered to sediments (Hopkins and Roe 2009, as cited in Sparling et al. 2010). Concentrations of ash-related COPECs may bioaccumulate in amphibians over time and adversely affect these populations. In addition, amphibians can be a food source for higher trophic-level predators.

Amphibian samples were collected in 2009, approximately 3 months after the release, and again in the spring of 2010. Three species were collected, including American toad (*Bufo americanus*), spring peeper (*Pseudacris crucifer*), and upland chorus frog (*Pseudacris feriarum*), and analyzed as whole body specimens. Amphibians were collected from impacted locations (West and North Embayments), locations to the north of the ash release area that were not impacted (Dawson and Rocky Top Farms), and an additional reference location from a pond in a Knox County subdivision (Figure 2-27). A target of ten adults per species was collected from each location.

Whole body amphibian samples were analyzed for metals, metalloids, and percent moisture (when sufficient sample mass was available). Amphibians were not tested for PAHs, PCBs, pesticides, radionuclides, or

chemical speciation due to the large quantity of material required for radionuclides and chemical speciation. Summaries of whole body spring peeper, upland chorus frog, and American toad data are presented in Tables 2-275 through 2-290, respectively. Results are presented in Appendix A.

Amphibian tissue collections continued in the spring of 2011, following similar methodology from previous years. These data were not available for the BERA interpretation; however, a brief evaluation of these data is presented in the Uncertainty Analysis in Section 9.3.2.

2.5.5.8 Reptiles

Reptiles can inhabit both aquatic and riparian habitats, consuming a variety of plants, invertebrates, other reptiles, amphibians, fish, small birds, and small mammals in both ecosystems. Throughout their lifespan, reptiles can act as both predators to smaller species, as well as being prey items for higher trophic-level predators. Concentrations of some ash-related COPECs may bioaccumulate in reptiles over time and adversely affect their populations. In addition, reptiles can be a food source for higher trophic-level predators.

Turtle sampling efforts began in 2009, approximately 8 months after the release, and continued in 2010. Three species of turtles, including common musk/mud turtle (*Chelydra serpentine*), Eastern spiny softshell turtle (*Apalone spinifera*) (2010 only), and common snapping turtle (*Chelydra serpentina serpentine*), were selected to represent various trophic levels of reptiles. Blood and carapace (softshell only) samples were collected, and then each turtle was released unharmed. Sampling locations (Figure 2-28) included locations in the Emory River, the Clinch River to its confluence with the Tennessee River, the Tennessee River below the confluence with the Clinch River, and the Tennessee River above the confluence with the Clinch River (the 2010 Reference Area). Specimens were also collected from a reference location in nearby Knox County (the 2009 Reference Area).

Turtle blood and carapace samples were analyzed for metals, metalloids, and percent moisture (when sufficient sample mass was available). Turtle samples were not tested for PAHs, PCBs, pesticides, radionuclides, or chemical speciation due to the large quantity of material required for radionuclides and chemical speciation. Summaries of common musk/mud turtle, Eastern spiny softshell turtle, and common snapping turtle blood and carapace data are presented in Tables 2-291 through 2-313, respectively. Results are presented in Appendix A.

Turtle tissue collections continued in the spring and summer of 2011, following similar methodology from previous years. In addition, reproductive data was also collected in order to evaluate long-term and/or latent effects on the early growth, development, physiology, and survival of hatchlings. These data were not available for the BERA interpretation; however, a brief evaluation of these data is presented in the Uncertainty Analysis in Section 10.3.2.

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3. Fish

Fish were selected as a focus for this BERA because they are an important part of the aquatic food web at the Site, have a relatively high likelihood for potential exposure, and can be sensitive to the effects of some COPECs that may be present in the ash material. Historically, fish have been studied throughout the Emory, Clinch, and Tennessee Rivers by TVA and others. Specifically, TVA has conducted fish surveys as one of its natural resource stewardship functions and in areas near its power plants as part of their National Pollutant Discharge Elimination System (NPDES) permit renewal requirements. These studies provide a baseline of historical population and condition data that are useful in helping to evaluate potential risks from ash-related COPECs.

3.1 Problem Formulation

The Clinch, Emory, and Tennessee Rivers support an abundant and diverse fishery typical of large river systems in the ecoregion. For example, 43 fish species were observed during recent TVA fish sampling efforts; the predominant species were gizzard shad, bluegill sunfish, freshwater drum, largemouth bass, and redear sunfish (TVA 2009a). The fishery is also utilized by recreational anglers. Historic creel data indicate that bluegill sunfish were the most frequently caught species, closely followed by largemouth bass (TWRA 2002). Other fish species frequently targeted by anglers include black crappie, white bass, white crappie, smallmouth bass, and sauger (TVA 2009a).

Several threatened or endangered fish species occur in the area, including the flame chub, snail darter, spotfin chub, and Tennessee dace. Their occurrence is known primarily from tributaries to Watts Bar Reservoir, although some species have also been observed in embayments and margins of the reservoir (TVA 2009a).

Fish in the Clinch, Emory, and Tennessee Rivers downstream of the release may be directly exposed to inorganic compounds and other COPECs in sediments and surface waters. Exposure pathways may include direct contact through skin and gills, ingestion of surface waters and sediments, uptake through the food web, and maternal transfer of constituents to eggs. Concentrations of ash-related COPECs may bioaccumulate in fish over time. At high enough concentrations, toxicity from these COPECs can affect the abundance and diversity of fish. In addition, fish are a food source for aquatic- or riparian-feeding piscivorous wildlife.

Fish communities of interest for this BERA include both pelagic (open-water or “top-feeding”) and benthic (bottom-dwelling or “bottom-feeding”) communities. The evaluations used to characterize risks to fish include studies that are generic to all fish, as well as studies conducted specifically on representative species of benthic fish (e.g., channel catfish) and open-water fish (e.g., largemouth bass).

3.1.1 Selection of Assessment Endpoints

As described in Section 2.2, assessment endpoints are used in the ERA process to identify focus areas for the risk assessment and to develop a method for evaluating ecological risks. The initial selection of the assessment endpoints was presented in BERA Methodology of the SAP (Jacobs 2010a). As discussed in that methodology, the assessment endpoint for fish used to develop this BERA is the survival and maintenance of pelagic and benthic fish populations in the Emory, Clinch, and Tennessee Rivers.

Based on this assessment endpoint, the ecological risk questions established for evaluating fish at the Site are as follows:

- Do concentrations of ash-related COPECs in surface waters or food items pose unacceptable risks to pelagic fish communities?
- Do concentrations of ash-related COPECs in surface waters, submerged sediments, seasonally-exposed sediments, or food items pose unacceptable risks to benthic fish communities?

These risk questions are used to help focus subsequent steps of the fish assessment, including the selection of measurement endpoints. If unacceptable risks are determined to exist, the spatial extents and magnitudes of these risks to the fish community must be evaluated in the WOE assessment.

3.1.2 Selection of Measurement Endpoints

The measurement endpoints for fish selected for the BERA are as follows.

Measures of exposure:
- COPEC concentrations in surface waters (total and dissolved)
- COPEC concentrations in submerged and seasonally-exposed sediments
- COPEC concentrations in fish tissues
- Measurement of fish health metrics (e.g., liver enzymes).

Measures of effects:
- Results of fish health studies (ovary status, condition factors, etc.)
- Results of fish reproductive studies
- Results of embryo-larval fish toxicity tests
- Fish community surveys.

3.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to fish. The LOEs used to evaluate fish include:

- Fish community surveys
- Surface water and sediment toxicity tests
- Fish reproductive studies
- Fish tissue bioaccumulation
- Fish health metrics
- Surface water and sediment chemistry data evaluation.

Each LOE is summarized below.

3.2.1 Community Surveys

Biological surveys provide a useful LOE for evaluating population-level risks because they rely on direct measurement of fish abundance and diversity. Two programs for fish community surveys are used in this BERA to characterize fish abundance and diversity in the vicinity of the Site. The first is TVA's annual evaluation of the spring sport fishery that has been conducted since 2002 for the Watts Bar Reservoir. The second is TVA's Reservoir Fish Assemblage Index (RFAI) methodology, which has been used since 2001 to assess the condition of the fish community in the general vicinity of the KIF. Each of these programs is evaluated separately as part of this LOE.

3.2.1.1 Analysis of Exposure and Effects – Spring Sport Fish Survey

The results of the spring sport fish survey are summarized in the *Evaluation of Spring Sport Fish Survey Results for Watts Bar Reservoir, 2002 – 2011* (Appendix E). The evaluation focuses on results for black bass (*Micropterus*) surveys conducted annually from 2002 through 2011. The surveys were conducted in the following five areas (and years):

- CRM 2.5 (2002 to 2005; 2009 to 2011)
- Caney Creek (2002 to 2011)
- Blue Springs (2002 to 2011)
- Watts Bar Forebay (2002 to 2011)
- ERM 2.5 (2009 to 2011).

The Emory River and Clinch River sampling areas are the closest in proximity to the KIF plant and ash release area and are considered near-field in the context of this study report.

The survey includes 12 sampling stations at each of the five areas. Electrofishing surveys are conducted during the April to May timeframe. At each sampling station, a 30-minute continuous electrofishing run is conducted within the littoral zone and across various habitats. Fish are identified by species, weighed, measured, enumerated, and noted for general health prior to releasing.

The spring sport fish data were used to develop key population metrics for black bass populations, including measures of catch rates, general fish condition (e.g., parasites and anomalies), and length frequency and relative weights. Largemouth bass length frequency histograms were developed for each area, and the average pre-release length-frequency (2002 to 2005) for the Clinch River was compared to the post-release length frequencies found in the Clinch River and Emory River areas. Relative weights for three size classes of largemouth bass – stock (8 to 11 inches), quality (12 to 14 inches), and preferred (15 to 19 inches) – were evaluated per year across the sampling areas.

The catch rates and number of parasites and anomalies per sample area by collection year are presented in Table 3-1. A summary of the data is provided as follows.

- **Catch Rates** – Catch rates for near-field areas ranged from 42 to 51.8 fish per hour (pre-release at Clinch River) to 27.8 to 62.7 fish per hour (post-release at Clinch and Emory Rivers). Catch rates for far-field areas ranged from 42.5 to 78 fish per hour (pre-release) to 46 to 96.5 fish per hour (post-release). The highest catch rates for near-field areas occurred in 2009 and 2010, and the lowest catch rates were observed in 2011. Catch rates were also above the long-term average at far-field locations from 2008 through 2010, and included the highest catch rate observed at each location. Similar to near-field locations, catch rates declined at these locations in 2011, indicating area-wide declines in catch rates (e.g., at the Watts Bar Forebay and Caney Creek). Catch rates indicated that average pre-release conditions (54.6 fish per hour) were slightly lower reservoir-wide than those observed in the post-release (57.5 fish per hour) years (as shown in Table 3-2). A significant rainfall event preceded the April 2011 sampling effort; increased turbidity and flow rates likely reduced the catch rate for that sampling effort. The influence of these conditions is most evident for the upstream Clinch and Emory River locations, where the effects of rainfall and local runoff have greater impacts on the more riverine environments in those reaches (Appendix E).

- **Fish Condition** – The percent parasites observed indicated a notable difference between pre- and post-release conditions, as reservoir averages showed an increase from 1.4 percent incidence of parasites to 8.3 percent. However, this increasing trend in percent parasites began in 2007. Fish condition, respective to anomalies observed, indicated a slight difference between pre- and post-release conditions, as reservoir averages showed an increase from 3.1 to 4.2 percent. Environmental variables

(e.g., meteorological conditions) likely had a greater influence on the overall fish condition than any localized impacts due to the fly ash release (Appendix E).

- **Fish Length** – Largemouth bass length frequency histograms for the Clinch and Emory River sample stations are represented by collection year in Figures 3-1 through 3-3. Within the near-field areas there was greater representation of age-one fish in 2002 (CRM 2.5) and 2009 (CRM 2.5 and ERM 2.5) as compared to other years, which likely indicates better recruitment of young-of-year fish from the preceding year. Table 3-3 provides a summary of recruitment information for the Clinch and Emory River locations. The histograms also show that the lower catch rates at the locations (near field and far field) in 2011 resulted largely from collecting fewer 10- to 13-inch or 10- to 14-inch fish (predominately fish age 2 to 4) than in most previous years (Figures 3-1 through 3-3) (Appendix E).

- **Fish Weight** – Relative weights are a reflection of fish condition and are derived from the ratio of the actual weight of a fish to a standard weight for a fish of similar length. For this evaluation, standard weights were derived from historical data from Tennessee Valley reservoirs. Relative weights for largemouth bass were evaluated for three different size classes: stock (8 to 11 inches), quality (12 to 14 inches), and preferred (15 to 19 inches). Results for these three size classes indicated a range of 80 to 107 percent; as presented in Figure 3-4. Relative weights for largemouth bass within the Clinch River have typically been 2 to 6 percent lower as compared to the three far-field locations. Similarly, relative weights for largemouth bass in the Emory River are typically 5 to 10 percent lower as compared to far-field locations. Additionally, relative weights for stock and quality size classes of largemouth bass across the locations indicated similar temporal trends (pre- and post-release) including a consistent decrease in 2003 and 2010. Relative weight differences may be partially due to the sampling locations being on three river systems with differing characteristics (e.g., watershed geology, hydrological regimes, and water quality) and to the longitudinal gradients in physical, chemical, and biological properties (e.g., primary productivity) that exist within the reservoir because of a downstream transition from more riverine environments to more lacustrine environments. This reservoir-wide temporal pattern may be influenced by factors common among locations such as, perhaps, annual differences in meteorology and hydrology, which in turn affect overall conditions in the reservoir (e.g., water temperature and primary productivity) and, thereby, affect fish growth, spawning, and/or recruitment (Appendix E).

Fluctuations in several measures used to evaluate bass populations provided no clear evidence of ash-related effects. Similar spatial and temporal patterns in catch rates, length-frequency distributions, relative weights, and instances of external parasites among locations (near field and far field) both before and after the ash release, suggest strong evidence that the patterns observed were associated with factors that extended reservoir-wide (e.g., meteorology) and/or location differences in limnological properties (e.g., velocity and primary productivity) (Appendix E).

The lower catch rates for black bass at near-field locations in 2011 would be the most concerning, except they were preceded by two post-release surveys with above average catch rates and the fact that the 2011 surveys coincided with rather extreme fluctuations in reservoir conditions (i.e., pool level, flow, turbidity, and water temperature) that likely affected sampling efficiencies. Additionally, the likelihood that an adverse impact of the release would be realized concurrently over an area of approximately 10 river miles, to the same degree, 2.5 years after the release, is low. Likewise, the general similarities with results at far-field locations also indicate reservoir-wide influences during the 2011 survey (Appendix E).

3.2.1.2 Analysis of Exposure and Effects – Annual Fish Community Survey

The results from the annual fish survey are summarized in the *Evaluation of the Fish Community in the Vicinity of Kingston Fossil Plant, 2001 – 2010* (Appendix F). As part of the studies, fish communities near the KIF plant have been evaluated using TVA's RFAI methodology, as part of their NPDES permit renewal requirements since 2001. The RFAI uses 12 fish community metrics from four general categories: species richness and composition, trophic composition, abundance, and fish health. Together, these metrics provide a balanced evaluation of fish community integrity (Appendix F).

Fish community samples were taken once every 2 years from 2001 through 2007 at locations upstream (CRM 4.4) and downstream (CRM 1.5) of the KIF heated discharge. Following the ash release in December 2008, fish community sampling has been conducted annually at three locations, including the two locations sampled prior to the ash release and at an additional location established at ERM 2.5 to evaluate immediate near-field effects through 2010. Figures 3-5 through 3-7 detail the sampling locations established for ERM 2.5, CRM 1.5, and CRM 4.4.

Fish sampling methods included both boat electrofishing and gill netting. At each location, electrofishing consisted of 15 electrofishing boat runs near the shoreline, each 300 meters long, with duration of approximately 10 minutes. At each location, approximately 15,000 ft was sampled. Ten overnight experimental gill net sets were used at each area. Experimental gill nets were used as an additional gear type to collect fish from deeper habitats not effectively sampled by electrofishing.

Fish collected using these sampling methods were identified by species, counted, and examined for anomalies (such as disease, deformations, or hybridization) (Appendix F). Community results were evaluated using the multi-metric scoring RFAI method.

Possible RFAI scores range from 12 to 60, and ecological health ratings are ranked as follows:

- 12 to 21 "Very Poor"
- 22 to 31 "Poor"
- 32 to 40 "Fair"

- 41 to 50 “Good”
- 51 to 60 “Excellent”.

Variation among RFAI scores from year to year per location is expected, with an acceptable average variation for RFAI scores in TVA reservoirs established at up to 6 (± 3) for similarity (Appendix F). In other words, a RFAI differential score above 6 for a location compared to a historical average RFAI score would indicate potential real (vs. normal variability) differences.

The RFAI results were used to determine if ash-related effects were evident in the fish community by comparison with historical scores for the existing locations on the Clinch River, both of which may have had some impact from the release (Appendix F). The Emory River location was also compared to the historical data from the Clinch River locations, based on the 2 years of post-release data, to provide information about unique characteristics at that location. An inspection of individual RFAI metric results and species of fish used in each metric provides more insight into any observed changes and differences between monitoring locations.

RFAI results including individual metric scores, contributing species, and overall RFAI scores for each sampling location for each sample year are listed in Tables 3-4 through 3-6. Metric scoring criteria for the transition areas are shown in Table 3-7. The total number of each species collected in electrofishing and gill netting samples combined is listed for each sampling location for each sample year in Table 3-8. The species collected, including trophic level, indigenous, and tolerance classifications, as well as catch per effort during electrofishing and gill netting at each sampling location for each sample year, are listed in Appendix A (Baker 2011a) of the *Evaluation of the Fish Community in the Vicinity of Kingston Fossil Plan, 2001-2010* (Appendix F).

In general, RFAI scores across the locations were relatively consistent, as scores rated “Good” to “Fair,” with the lowest scores occurring at the Clinch River locations in 2007 and 2009, respectively. These scores were lower based on a lesser abundance in the number of indigenous species collected. In 2010, RFAI scores for the locations rated “Good” and individual metric scores were very similar among locations.

The number of indigenous species collected at CRM 4.4 and CRM 1.5 in 2009 and 2010 was similar to that observed prior to the ash release, as the number of species ranged from 28 to 35. In 2009, the immediate near-field ERM 2.5 location indicated 32 indigenous species as compared to the 28 species observed at CRM 4.4 and CRM 1.5. In 2010, these three locations indicated a number of indigenous species equal to or the highest observed during either the pre-release or post-release studies.

Species richness metrics typically received the maximum number of points, indicating a “good’ number of species were represented in samples. Exceptions were the moderate “number of indigenous species” collected at CRM 1.5 and CRM 4.4 in 2007 and 2009, and the moderate to low “number of benthic

invertivores" collected at each location each sample year. Because the number of indigenous species declined consecutively at Clinch River locations from 2005 to 2007, with 2007 yielding the lowest number of indigenous species to date, the moderate richness observed in 2009 is not believed to be related to any adverse effects of the ash release. Additionally, because results for benthic invertivores were similar in pre-release and post-release samples, there is no clear evidence of ash-related effects (Appendix F).

"Percent tolerant individuals" and "percent dominance by one species" usually scored in the moderate to low range at the locations, with no apparent difference between pre- and post-release samples attributable to ash-related effects. The low scores for "dominance" were largely due to the high percentage of bluegill sunfish and/or gizzard shad in most samples. Likewise, these species are considered tolerant of degraded water quality and contribute to the composition of tolerant individuals (Appendix F).

"Percent non-indigenous species" have shown a trend of increasing percentages in electrofishing samples mainly due to an increase in the frequency and numbers of inland silverside (*Menidia beryllina*) collected. This species was first collected in Kentucky and Pickwick reservoirs in 1993 and has continued to spread throughout the Tennessee River system (Appendix F).

Scores for trophic composition ("percent top carnivores" and "percent omnivores") have fluctuated between high, moderate, and low. The principal difference between pre- and post-release results was the improved score (lower composition) for "percent omnivores" in electrofishing samples at CRM 1.5 and CRM 4.4 in 2009 and 2010; a result of the higher overall number of fish collected those years (Baker 2011b).

Fish abundance was low in electrofishing and gill netting samples at the locations each sample year. Fewer fish were collect at both Clinch River locations in 2007 than in other years. The number of fish in electrofishing samples was higher during post-release surveys than during pre-release surveys, but fish abundance continued to score low or at the low end of the moderate range. Additionally, the higher-than-average number of fish collected at both locations in 2009 and 2010 was largely the result of increases in tolerant (e.g., bluegill sunfish, gizzard shad, largemouth bass, and/or spotfin shiner) species and inland silversides, which are not indigenous (Appendix F).

The principal difference in the percentage of fish with anomalies among years was the higher percentages (2.8 to 4.3 percent) in electrofishing samples in 2009. The majority of the increases were attributable to bluegill sunfish infected with common parasites, which also coincided with considerable increases in numbers of bluegill sunfish collected when compared to most previous years (Appendix F).

Comparison of fish community results for pre-release to post-release conditions in the historic sampling areas indicates RFAI scores were generally consistent and the historical fish assemblage data were representative of transition zones found in upper main stem Tennessee River reservoirs. This comparison of results from multiple years of sampling provides strong evidence that the area near KIF continues to

support species of fish in numbers and conditions typically observed in the area before the release (Appendix F).

Some variability in the characteristics used to assess the fish assemblage was observed, but is also expected as a result of natural population cycles and movements of the species being measured (Tennessee Wildlife Resources Commission [TWRC] 2006), as well as the variability that arises from the fact that nearly any practical measurement, lethal or non-lethal, of a biological community is a sample rather than a census of the entire population (Appendix F).

3.2.1.3 Risk Estimation

The results from the various fish community surveys indicate that although variability in the fish assemblage was observed, the level of variability is not unexpected. The results from the spring sport fish survey (Appendix E) indicate that similar spatial and temporal patterns in catch rates, length-frequency distributions, relative weights among locations (near-field and far-field) both before and after the ash release, suggest that the patterns observed appear to be associated with factors that extended reservoir-wide. Similarly, the annual fish community survey results for pre-release to post-release conditions indicates RFAI scores are generally consistent, and the fish assemblage data are representative of the habitat. Though there is some indication that individual fish health may have been impacted immediately after the ash spill, as reflected by increased parasite loads, increased parasite loads have been higher since 2009 compared to prior 2009 reservoir-wide, suggesting other factors may be contributing to the increase.

Risk Conclusion for Fish Community Surveys – To date, no substantive impairment to the fish community is observed in the surveys' results.

3.2.2 Surface Water Toxicity Tests

Toxicity tests provide a direct measure of potential toxicity and are a useful LOE for evaluating ecological risks. Toxicity tests using fish as the test species were conducted as part of the surface water toxicity testing program. Fish toxicity tests were also conducted using ash-impacted sediments. Summaries of these tests are provided in the following sections.

3.2.2.1 Analysis of Exposure and Effects – Surface Water Toxicity Testing

In fall 2010, Emory River surface water samples were collected for aquatic static-renewal toxicity tests with fathead minnows. The objective of the testing was to determine if the surface water from the Emory River in the vicinity of the ash release was toxic to aquatic organisms.

Surface water samples were collected from five locations within the Emory River, including one upstream reference location along the Emory River (ERM 8.0), and four site-related water locations (ERM 1.0, ERM 2.0, ERM 3.0, and ERM 4.0). Toxicity tests were conducted by Environmental Enterprises USA, Inc., in accordance with USEPA toxicity testing protocols for 7-day chronic tests (USEPA Method 1002; USEPA 2002a). Test organisms were less than 24-hours old fathead minnows. The test was run with and without ultraviolet treatment of the water, and controls consisted of moderately hard synthetic freshwater and Emory River water from a reference location. The tests included serial dilutions (10, 20, 40, 60, 80, and 100 percent site water) for each sample. The surface water from the reference location was used as the dilution water. The results from the dilution series were used to calculate for each sample a no observed effect concentration (NOEC), a lowest observed effect concentration (LOEC), and an IC_{25} (concentration that inhibits 25 percent of organisms) for growth and survival.

A summary of the surface water toxicity analysis results is presented in Table 3-9. There were no effects on fathead minnow growth or survival. As a result, the LOEC and IC_{25} were determined to be greater than or equal to 100 percent for all treatments.

3.2.2.2 Analysis of Exposure and Effects – Embryo-Larval Tests

Fish embryo-larval studies were conducted by ORNL as part of a larger biological monitoring program administered by TVA. The tests measured the responses of fathead minnow embryos to exposure to fly ash in laboratory toxicity tests, and test methods were adapted from a standard 7-day embryo-larval survival and teratogenicity test (method 1001.0 in USEPA 2002a). Mortality, hatching success, and the incidences of developmental abnormalities were used as measured endpoints (Appendix G).

Bulk sediment samples containing ash from the Kingston ash release were collected for TVA from four locations in the lower Emory River within the release zone (Figure 3-8 and Table 3-10). Reference sediment was collected from a location in the Emory River upstream of the release's influence. Percentages of ash in the lower Emory River samples varied from 46 to 78 percent (Table 3-11), and these samples had elevated concentrations of mercury, selenium, and arsenic as compared with the reference sediment (Table 3-12). Prior to testing, sediment samples were remixed with reference sediment to obtain a concentration series for embryo-larval toxicity testing (100 percent [only sediment samples A, B, C, or D], 75 percent, 50 percent, 25 percent, and 0 percent [only reference sediment]).

Test and control water were obtained the day prior to the initiation of tests from the Emory River at ERM 8.0. Test water chemistry, including pH, conductivity, alkalinity, hardness, and DO, was analyzed prior to daily use, and DO and final pH were measured in daily discard water.

The day prior to the initiation of a test, approximately 35 milliliters (mL) of a sediment sample was layered on the bottom of each of four replicate 185-mL polystyrene chambers and overlaid with 100 mL of test water

(control dishes were provided with 135 mL of test water only). Embryos for tests were obtained from in-house fathead minnow cultures at ORNL. The morning of test initiation, fertilized eggs were collected from spawning tiles placed in stock aquaria the prior afternoon, ensuring that embryos were ≤ 24 hours old at the time of collection. Eggs from at least five different aquaria were pooled and allowed to water-harden in aquarium water for approximately 1 hour after collection, then viable embryos < 6 hours old as indicated by developmental stage were transferred to dishes containing filtered Emory River test water prior to randomization and final allocation to test dishes.

Following randomization of the fertilized eggs, 15 embryos were transferred to each test chamber and carefully deposited directly to the surface of the settled sediment layers. Placing the embryos directly on sediment provided both a more stable platform for pre-hatch development to proceed and a worst-case scenario for contact exposures to ash in the sediments during embryonic and early larval development.

For each test, embryos and larvae were incubated for 7 days through the absorption of the larval yolk sac without feeding. Embryos were scored daily for survival, hatching success, and developmental abnormalities, and dead embryos or larvae were removed and discarded. At test termination, fish were euthanized, and embryos and/or larvae were removed from the test chambers with a transfer pipette.

Endpoints of the fathead minnow embryo-larval toxicity test were based on mortality, incidences of developmental abnormalities, and total number of dead and deformed embryos and/or larvae (USEPA 2002a). For the purposes of data analysis and presentation, any developmental abnormality scored at the time of test termination was considered lethal, and those embryos and larvae were added to the final totals for mortality.

Summary results for the fathead minnow embryo-larval tests conducted on Emory River sediment are shown in Tables 3-13 through 3-16. Mean survival in controls, consisting of filtered Emory River water without added sediment, was 80 percent or greater for each of the four tests (Figure 3-9). Although embryo-larval survival generally tended to be slightly less in sediment exposures than in control water exposures, only in the 100 percent reference sediment group of the sample D test was there a statistically significant reduction from control survival. In none of the tests did total mortality, defined as the sum of dead and deformed embryos and/or larvae at test termination, differ significantly within a sediment dilution series (Tables 3-13 through 3-16). Therefore, the NOEC was 100 percent for each of the tested sediment samples. The EC_{25} (concentration at which a 25 percent reduction in survival is expected) could not be statistically calculated because total mortality did not differ significantly among groups; therefore, the EC_{25} was > 100 percent for each sample.

Hatching success was also unaffected by exposure to sediments containing ash (Tables 3-13 through 3-16). Variation in hatching success among replicates and between treatment groups (concentrations) simply reflected normal variation between replicates and treatment groups in embryo mortality.

Incidences of developmental abnormalities were relatively low across the tests. Deformities ranged from sometimes transient edema and mild spinal curvatures to severely stunted embryos with numerous and varied deformities that typically caused death prior to test termination. The overall percentage of deformed embryos or larvae at test termination across the four tests was < 2 percent of the total embryos used in the tests (28 of 1440).

3.2.2.3 Risk Estimation

Direct contact exposures of fathead minnow eggs to sediments from the Emory River containing up to 78 percent ash had no significant effects on embryonic or early larval development in these relatively short-term laboratory tests, although chemical COPECs with known toxic effects on fish early larval stages, particularly selenium, were significantly elevated in the sediment samples relative to reference locations (Table 3-12). Developmental endpoints including embryo-larval survival, hatching success, and the incidences of developmental abnormalities did not differ statistically across concentration series ranging from 0 to 100 percent of each ash-containing sediment sample. These results, thus, do not support a hypothesis that ash in areas of the Emory and Clinch Rivers affected by the release presents a significant risk to developing fish eggs, either from contact with ash or exposure to COPECs potentially leaching from the sediment into overlying river water. That many of the developing embryos in these tests were relatively coated with sediments from inadvertent disturbances during handling of test chambers and water changes provided further evidence of a lack of a substantial direct toxic effect in these experimental exposures.

Risk Conclusion for Fish Toxicity Tests – To date, no significant health effects (i.e., no direct observations of reduced survival, hatching success, or developmental abnormalities) have been observed from the fish toxicity test studies.

3.2.3 Fish Reproductive Studies

Fish reproductive studies were conducted by ORNL to evaluate the bioaccumulation of metals of reproductive concern in fish from the Emory and Clinch Rivers downstream of the ash release (Appendix H). The focus of the studies was organs such as the liver and ovary that would provide a route of maternal transfer of reproductive toxicants (such as selenium) to the developing eggs. The studies also evaluated the reproductive health of the fish to assess potential relationships between any observed reproductive effects and the ash release. These objectives were addressed through a comparison of bioaccumulation and selected reproductive criteria in key fish species sampled from locations either affected by the ash release or locations located upstream of the release as reference locations.

3.2.3.1 Analysis of Exposure and Effects

Fish were collected by TVA personnel by boat electrofishing from five locations in the Emory and Clinch Rivers (Figure 3-10) at the beginning of the 2009 and 2010 reproductive seasons. Fish were collected from upstream reference locations (ERM 8.0 and CRM 8.0) and locations at or downstream of the ash release (ERM 3.0, ERM 0.9, and CRM 1.5). An additional reference location in the Little Emory River (LERM 2.0) directly upstream of the release location was added to the study locations in 2010.

Largemouth bass, bluegill sunfish, and white crappie were the initial target species for the study. However, sufficient white crappie in appropriate reproductive condition could not be collected from some of the study locations in 2009 due to lack of preferred habitat. Redear sunfish were collected beginning in 2010. At each location and for each species, the goal was to collect at least eight adult female fish. Fish were returned live to ORNL, euthanized, and processed as described in Appendix H. For the purposes of assessing reproductive condition, ovaries were removed from the body cavity and weighed to the nearest 10 milligrams (mg). Representative pieces of tissue were carefully cut from the midpoint of the largest ovarian lobe, weighed, and placed in vials containing either a half-strength solution of Karnovsky's Fixative for later analysis of ovary stage, oocyte (immature developing eggs) condition, and the estimation of fecundity. Additional slices of ovary tissue were placed in vials with 10 percent buffered formalin for subsequent histopathology analysis.

The gonadsomatic index (GSI), or relative size of the gonad to body size determined as a percentage, was calculated for each fish. The GSI is a commonly used and easily determined indicator of fish reproductive status (Nikolsky 1963), but by itself provides little definitive information about actual gonadal condition. Alterations in the GSI (generally downward) have been demonstrated with contaminant exposure in a number of field studies, including exposures to pulp and paper mill effluent (Janz et al. 1997; Munkittrick et al. 1994; van den Heuvel et al. 2002), and to PAHs, PCBs, and other chlorinated compounds (Johnson et al. 1999). Because of the rapidity and ease with which this parameter is measured, the GSI remains one of the most widely used reproductive indicators of environmental stress. However, the GSI is best suited for use as part of a larger suite of reproductive indicators because it provides only limited data on reproductive condition (Greeley 2002).

The reproductive condition of ovaries was quantitatively evaluated by sizing, staging, and enumerating the oocytes above a certain threshold size contained within a weighed piece of ovary by procedures adapted from Greeley et al. (1987), Hsiao et al. (1994), and Lin et al. (1989). Ovarian pieces fixed and stored in Karnovsksy's Fixative were soaked and washed with several changes of distilled water to remove potentially toxic residues. Each pre-weighed ovary portion was gently separated. Both largemouth bass and redear sunfish ovaries contained oocytes ranging in size from less than 0.1 millimeter (mm) in diameter up to 2.0 mm in diameter. Individual oocytes of a species-specific diameter (generally 0.3 to 0.5 mm) and larger were measured to the nearest 0.1 mm. The developmental stage of each oocyte was recorded, as was the

presence of any atretic (dead or nonviable) oocytes and post-ovulatory follicles (tissue that surrounds the developing eggs prior to ovulation and expulsion).

Oocyte staging was adapted from Greeley et al. (1988) and Wallace and Selman (1981). Because oocyte development is essentially uniform throughout the length and width of both largemouth bass and redear sunfish ovaries, results of analyses on these representative ovarian pieces were extrapolated to give the total number of oocytes of a certain size class or stage in each ovary. Measures of reproductive condition derived from this exercise include one or more estimates of fecundity, total counts of vitellogenic ("yolk-laden") oocytes within an ovary, and the prevalence of parasitic lesions and oocyte atresia (oocyte death and/or reabsorption). Similar procedures were used previously to evaluate the reproductive condition of both largemouth and bluegill sunfish in Watts Bar Reservoir (Adams et al. 1999) for remedial investigation feasibility studies at that respective location.

The production of viable eggs is among the most sensitive indicators of contaminant exposure and effects in experimental life-cycle tests of potential toxicants (Suter et al. 1987). Fecundity has been shown to be very sensitive to environmental contaminants in a number of laboratory studies (Carlsson et al. 2000; Kovacs et al. 1995; Tam and Payson 1986). In fish species with large single clutches of eggs, fecundity is often determined with a volumetric approach by enumerating the number of eggs in a small subset of the entire egg mass. However, for fractional or multiple spawners, such as largemouth bass and redear sunfish that are capable of spawning more than once a season, eggs are generally ovulated and spawned very quickly, so that even if fish ovaries have some eggs when collected these may represent only a fraction of the actual eggs produced during a single spawn. Therefore, the morphometric method employed to estimate fecundity by enumerating either preovulatory oocytes before a spawn or postovulatory follicles immediately after a spawn is the most accurate method to provide a fecundity estimate for such fractional spawners.

In the study, no attempt was made to estimate annual fecundity for sunfish, because this requires careful monitoring of egg development patterns throughout the breeding season. Therefore, to best compare the reproductive potential of sunfish populations at the various study locations, batch or clutch fecundity estimates were calculated for fish collected at or just prior to the beginning of the breeding season. For female largemouth bass and redear sunfish that spawned prior to collection, freshly shed post-ovulatory follicles were counted to determine the number of eggs released in the spawn. For redear sunfish collected before their first seasonal spawn (the majority of fish), batch fecundity estimates were based on the number of synchronously developing oocytes in the leading clutch of developing oocytes destined to be released during the first spawn. Clutches are more difficult to distinguish from oocyte size-distributions in pre-spawn largemouth bass ovaries, in bass lacking post-ovulatory follicles, the oocytes = 1.3 mm in diameter (the size of oocytes typically absent from ovaries with freshly shed follicles in post-spawn fish) were enumerated to provide an estimate of batch fecundity. For largemouth bass, an estimate of annual fecundity was also calculated by enumerating the oocytes = 0.7 mm (as suggested by Kelley 1961) plus any post-ovulatory

follicles present in the ovaries, although this may be an underestimation of the true annual fecundity because smaller vitellogenic oocytes appear to continue to grow into this size range during the breeding season.

The presence of atretic oocytes, characterized by the degeneration and necrosis of developing oocytes and the subsequent infiltration by macrophages in fish ovaries, is representative of gamete quality rather than quantity and, thus, may be one of the better reproductive indicators of an actual pathological condition (Greeley 2002). Elevated incidences of oocyte atresia have been reported in fish populations affected by environmental stressors, including contaminants. However, atresia has also been observed to occur at specific times during or immediately following the breeding season of unstressed fish populations (Hsiao et al. 1994; Hunter and Macewicz 1985), and it is clear that atresia is probably not a significant factor in actually reducing fecundity except in acute stress-related cases.

Histopathological analysis was done by Dr. Swee J. Teh, Toxicology Consulting, Davis, California. Lesions measured in ovary slices included:

- Pre-vitellogenic oocyte necrosis
- Follicular vitellogenic oocyte atresia
- Macrophage aggregates
- Parasitic infections of follicular and previtellogenic oocytes.

The severity of lesions in ovaries were scored based on a qualitative scale of 0 = no lesions present, 1 = mild lesions, 2 = moderate lesions, and 3 = severe lesions, with a score of 2 or 3 being considered a significant lesion. A composite lesion score for each species at each location was calculated as the average sum of lesion scores for each organ for that species and location. These results are presented along with other histopathological analyses in Appendix I.

The following sections describe the results of the fish reproductive studies for each species. Additional details are found in the primary report.

Bluegill Sunfish

Results of reproductive analyses conducted on female bluegill sunfish sampled at the beginning of the 2009 and 2010 breeding seasons are shown in Tables 3-17 and 3-18 and Figures 3-11 through 3-13. In the spring of 2009, mean GSIs were statistically significantly lower in female bluegill sunfish from the release area (ERM 3.0) than in fish collected from the upstream reference location at ERM 8.0 (Figure 3-11), although GSIs at locations farther downstream at ERM 0.9 and CRM 1.5 remained similar to those at the reference location. The abundance of vitellogenic oocytes and the batch fecundity, normalized for fish size, also tended to be lower at ERM 3.0 than at either the upstream reference location or locations farther downstream of the release (Figure 3-12), although reductions were not statistically significant. Furthermore,

the incidences of histopathological lesions in bluegill sunfish ovaries from the Site were also greater than at the reference location or farther downstream locations (Figure 3-13). Oocyte atresia was relatively uncommon at the locations sampled in 2009, including ERM 3.0 (Figure 3-11); therefore, direct toxicity to the developing oocytes was not a factor in either the lower GSIs or decreased abundance of vitellogenic oocytes and batch fecundity at this location. The larger vitellogenic oocytes in ovaries from ERM 3.0 were relatively smaller and less developed than at the other study locations, suggesting a delay in oocyte development in the ovaries of fish at this location.

By the spring of 2010, bluegill sunfish GSIs were similar between the release area and the Emory River reference location, while GSIs at locations farther downstream were actually significantly higher than at the reference location (Figure 3-11). The abundance of vitellogenic oocytes and the batch fecundity of bluegill sunfish did not differ significantly among the study locations in 2010 (Figure 3-12). Oocyte atresia was noticeably more common in bluegill sunfish ovaries at the locations sampled in 2010, and tended to be somewhat more prevalent at both the downstream locations and the other reference locations (LERM 2.0 and CRM 8.0) than at the Emory River reference location (Figure 3-11). Incidences of histopathological lesions in sampled ovaries were relatively fewer at both the release area and study locations downstream of the release than at any of the three upstream reference locations (Figure 3-13).

Considered together, these results suggest that the reproductive condition of female bluegill sunfish may have been temporarily affected in the immediate vicinity of the release location during the 2009 breeding season. Specifically, the observed delay in ovarian development may have been due to habitat alterations and food web disruptions or direct toxic effects. In contrast, in 2010 the reproductive condition of female bluegill sunfish at this location and other study locations downstream of the release did not differ in any ecologically significant manner from fish at the other study locations.

Largemouth Bass

Results of reproductive analyses conducted on female largemouth bass sampled at the beginning of the 2009 and 2010 breeding seasons are shown in Tables 3-19 and 3-20 and Figures 3-14 through 3-17. In the spring of 2009, largemouth bass GSIs were significantly lower at CRM 1.5, and also tended to be lower at the release area than at either ERM 0.9 or the upstream reference locations (Table 3-19 and Figure 3-14). A similar pattern was observed in the site-to-site variation in batch fecundity (Figure 3-15), although the observed differences between locations were not statistically significant.

The total number of vitellogenic oocytes (Figure 3-15) and the estimated annual fecundity (Figure 3-16) were both markedly lower at the other locations as compared to ERM 8.0, with annual fecundity statistically lower at both Clinch River locations (including the upstream reference location at CRM 8.0). Oocyte atresia was observed at the locations, but there were no significant differences between locations due to the

relatively high variability. Incidences of histopathological lesions were not significantly greater at ash-affected locations than at the ERM 8.0 reference location (Figure 3-17).

In the spring of 2010, largemouth bass GSIs were similar among the study locations (Table 3-20 and Figure 3-14). The abundance of vitellogenic oocytes, the batch fecundity (Figure 3-15), and the estimated annual fecundity (Figure 3-16), normalized for fish size, were actually higher at other study locations than at the ERM 8.0 reference location, in certain cases with statistically significant differences. Oocyte atresia was again observed at the locations in the spring of 2010 including the ERM 8.0 reference location (Figure 3-14), but with no statistically significant differences noted due to relatively high variability within each location. As in 2009, the incidences of histopathological lesions were not significantly greater at ash-affected locations than at the ERM 8.0 reference location (Figure 3-17).

Considered together, these results do not provide definitive evidence that the ash release had any ecologically significant adverse impact on the reproductive condition of female largemouth bass during either the 2009 or 2010 breeding seasons. There were some downward trends and even some statistically significant reductions among several reproductive parameters measured in fish from the upstream reference location at ERM 8.0 during the spring of 2009 and the downstream locations at ERM 3.0 and CRM 1.5 in particular. However, values of these reproductive parameters at the downstream locations were similar to those measured in fish from the Clinch River reference location in both 2009 and 2010 and also to values measured at these same locations in 2010. In fact, most of the observed differences among locations in both years appeared to be due primarily to the relatively high variability from year to year in the values of the reproductive parameters measured in bass from the ERM 8.0 reference location rather than from the other locations.

Redear Sunfish

Results of reproductive analyses conducted on female redear sunfish sampled at the beginning of the 2010 breeding season are shown in Table 3-21 and Figures 3-18 through 3-20. In the spring of 2010, none of the reproductive parameters measured in redear sunfish GSIs differed significantly between the upstream reference location at ERM 8.0 and the other study locations (Table 3-21 and Figures 3-18 through 3-20). Considered together, these results do not provide definitive evidence that the ash release had any ecologically significant adverse impacts on the reproductive condition of female redear sunfish at the beginning of the 2010 breeding season.

3.2.3.2 Risk Estimation

The fish reproduction study evaluated the reproductive condition of three species: bluegill sunfish, largemouth bass, and redear sunfish. The results from the fish reproductive study indicate that bluegill sunfish reproduction was delayed at the ash release site in the Emory River (ERM 3.0) during the 2009

breeding season immediately following the ash release. However, an observed delay in ovarian development may have been due to habitat alterations and food web disruptions or direct toxic effects. Delayed development alone is not conclusive evidence of an adverse effect on spawning success or population maintenance. There also was no conclusive evidence of adverse reproductive effects in female bluegill sunfish sampled from the other study sites in 2009 or from any of the locations sampled in 2010. Thus, any delays in development were localized, transient, and not representative of post-dredging conditions. Results of the studies with largemouth bass also did not provide definitive evidence that the ash release had any ecologically significant adverse impacts on the reproductive condition of female largemouth bass during either the 2009 or 2010 breeding seasons, although there was a downward trend in female reproduction condition in largemouth bass. Similarly, results of reproductive analyses on redear sunfish collected prior to the 2010 breeding season did not provide conclusive evidence that the ash release had ecologically significant adverse impacts on the reproductive condition of females of this species either.

Risk Conclusion for Fish Reproductive Studies – To date, no conclusive evidence of ecologically significant effects has been observed from the fish reproductive studies. No adverse effects on reproductive condition of the three tested species were observed in 2009 or 2010, although an apparently localized delay in reproductive development was noted at one location (ERM 3.0) and only in the spring of 2009. This delay may be attributed to habitat alterations and food web disruptions or direct toxic effects, and is not representative of current conditions at the Site.

3.2.4 Fish Tissue Concentrations of COPECs

Another LOE to evaluate potential risks to fish is the comparison of fish tissue data to critical body residues (CBRs) and the generation of HQs from this comparison. A number of fish species, tissue types, locations, and sampling events were included in this analysis, and the analysis focuses on fish that were collected in 2010, while dredging was still being conducted. An evaluation of fish collected immediately after the release was conducted in order to establish baseline concentrations (Appendix J). However, the preliminary assessment of risks to fish was based on comparison of study locations results with reference locations that were collected while dredging was still in process. Initial evaluation of fish tissue concentrations was conducted by ORNL (Appendix K), with additional analyses conducted by ARCADIS (Appendix L).

The evaluation of tissue concentrations in representative fish species provides a measure of the potential bioavailability of ash-related COPECs. In addition, the concentrations are compared to CBRs to determine whether uptake is significant enough to be adversely affecting the health of the organisms. There is substantial uncertainty associated with the development of CBRs, and so comparisons should be cautiously interpreted. Concentrations below these values can be interpreted as not causing an effect, but concentrations above indicate only that the potential for impact exists and do not indicate a direct cause and effect.

3.2.4.1 Analysis of Exposure and Effects

Fish tissue samples were collected from various sample reaches within both the Clinch and Emory Rivers. Samples were collected from a Clinch River reference reach (CR_Ref) located at CRM 8.0, and two sample reaches including Clinch reach B (CR_B) located between CRM 6.0 and CRM 3.0 and Clinch reach A (CR_A) located between CRM 3.0 and CRM 0.0. The Emory River reference reach (ER_Ref) includes both LERM 2.0 and ERM 8.0. Sample reaches include ER_C located between ERM 6.0 and ERM 3.5, ER_B located between ERM 3.5 to ERM 1.5, and Emory reach A (ER_A) located between ERM 1.5 to ERM 0.0. Sample reaches are presented in Figure 2-8.

Fish tissue samples collected during both spring and fall 2010 sampling events were used to calculate HQs. Fish were collected between April and July for the Spring 2010 sampling event through the collaboration of TVA, TDEC, TWRA, and ORNL. Samples were collected from reaches ER_Ref (ERM 8.0, LERM 2.0), ER_C (ERM 4.5), ER_B (ERM 3.0, ERM 2.0), ER_A (ERM 0.9), CR_Ref (CRM 8.0), CR_B (CRM 3.5), and CR_A (CRM 1.5). Species collected during spring 2010 include bluegill sunfish, channel catfish, gizzard shad, largemouth bass, redear sunfish, and white crappie. The fall 2010 TVA sampling event was conducted in September, October, and December 2010, as well as January 2011. TVA, TDEC, TWRA, and ORNL collaborated to collect the samples. Fish were collected from reaches ER_Ref (ERM 8.0, LERM 2.0), ER_C (ERM 4.5), ER_B (ERM 3.0, ERM 2.5), ER_A (ERM 0.9), CR_Ref (CRM 8.0), CR_B (CRM 3.5), and CR_A (CRM 1.5). Species sampled include bluegill sunfish, channel catfish, gizzard shad, and largemouth bass. Sample counts by tissue type and specific locations are presented in Table 3-22. These sampling events are presented in more detail in the *Kingston Ash Recovery Project, Non-Time-Critical Removal Action, River System Sampling and Analysis Plan Task Completion Technical Memorandum for Fish Sampling* (Jacobs 2011b).

For the purpose of comparison to CBRs, the most robust datasets (by species and tissue type) were chosen. Datasets were chosen based on the spatial distribution of samples collected (reference and sample reaches included), the trophic position of the species sampled (pelagic, benthic, and forage fish), and the overall number of samples collected for each tissue type. Tissue samples were analyzed for trace metals. Largemouth bass and channel catfish fillet samples were also analyzed for legacy constituents (pesticides and PCBs). With the exception of calcium, potassium, and sodium, which are considered nutrients, metals and legacy constituents (fillet samples only) are presented in this analysis.

The following datasets were included in the comparison to CBRs:

- Fillet
  - Bluegill sunfish
  - Channel catfish
  - Largemouth bass

- Whole body
  - Bluegill sunfish
  - Gizzard shad
- Whole body without the gut or its contents
  - Gizzard shad
- Liver
  - Bluegill sunfish
  - Largemouth bass
- Ovary
  - Bluegill sunfish
  - Largemouth bass

Tissue data were compared to CBRs, including NOAELs and LOAELs. CBRs were selected for each metal and legacy constituent for each different tissue type. Toxicity values were selected from the U.S. Army Corps of Engineers (USACE)/USEPA Environmental Residue-Effects Database (ERED) (http://el.erdc.usace.army.mil/ered/). After querying the database for each constituent, the CBRs were chosen based on the following rules:

- Select the highest NOAEL, if available
- Select the lowest LOAEL, if available
- If a NOAEL is not available, divide the chosen LOAEL by 10
- If no LOAEL is available, multiply the chosen NOAEL by 10
- If the highest NOAEL is greater than the lowest LOAEL, the LOAEL divided by 10 will be used for the NOAEL.

This is a conservative approach to selecting CBRs and estimating potential risks for a BERA. The comparison of tissue data to CBRs is considered a conservative and highly uncertain approach for evaluating ecological risks to fish. Several of the CBRs are based on limited datasets and often include unbounded NOAELs or LOAELs, which increases the uncertainty regarding the actual level of COPECs in tissue associated with an overall effect. Also, most metals undergo transformation in fish tissue to less toxic metal complexes, based on the capacity of organisms to sequester metals in forms that are not biologically reactive. For example, arsenic is routinely converted to relatively non-toxic organic arsenic complexes (arseno-betaine), which are relatively harmless and, thereby, increases the uncertainty and conservatism with applying a tissue-based CBR.

CBRs for fish tissues are presented in Table 3-23. The full dataset from the ERED database is presented in Appendix D. HQs are calculated by dividing the tissue concentration by the CBR at the NOAEL and LOAEL

for each individual sample. With the exception of calcium, potassium, and sodium, which are considered nutrients, metals and legacy constituents (fillet samples only) were included in this analysis.

Results of statistical analyses were also considered when interpreting HQs calculated for fish tissues. One-way analysis of variance (ANOVA) tests were conducted to look for spatial differences in concentrations of COPECs across reaches for fillet, whole body, whole body minus gut and its content, liver, and ovary samples. Separate analyses were performed for each species and tissue type. Further detail on statistical analyses is presented in Appendix L. HQs were calculated for both results that were significantly different from a reference reach, and results that were not. Results of the statistical analyses are summarized in Table 3-24.

The comparison of fish tissues to CBRs indicated exceedances of CBRs at both the NOAEL and LOAEL level in various fish tissues. HQs calculated from this comparison are discussed below for each tissue type.

Fillet Samples

CBRs for nine COPECs were available from the ERED database for fillet samples. These COPECs include aluminum, arsenic, cadmium, copper, lead, mercury, nickel, selenium, and total PCBs. It should be noted that COPECs not discussed below do not have either HQs > 1 or a CBR. Individual HQs for each sample are provided in Appendix L.

NOAEL HQs calculated for bluegill sunfish fillet samples range from < 1 to 15, with the highest HQs from selenium. The only LOAEL HQs > 1 in bluegill sunfish fillet samples were from selenium, and the highest HQ was 1.5. In channel catfish fillet samples, NOAEL HQs from detections range from < 1 to 5, with the highest HQs from selenium. No LOAEL HQs exceeded 1 for channel catfish fillet samples. Largemouth bass fillet samples showed NOAEL HQs from < 1 to 10, with the highest HQs again from selenium. No LOAEL HQs exceeded 1 for largemouth bass fillet samples from 2010.

In fillet samples, HQs were > 1 for aluminum, arsenic, copper, mercury, and selenium. A summary of fillet HQs by species, reach, and COPEC is presented in Table 3-25. Individual HQs for each sample are provided in Appendix L. As shown on Table 3-25, NOAEL HQs > 1 for aluminum were only noted in a few bluegill sunfish fillet samples from one sample reach, and no LOAEL HQs exceeded 1. Aluminum does not appear to be a risk driver in fillet samples. Similarly, NOAEL HQs > 1 for copper were only seen from a few reaches in largemouth bass and channel catfish fillets (Table 3-25). No LOAEL HQs > 1 were seen in any fillet samples. Although some of the sample reaches with NOAEL HQs for copper > 1 were from sample reaches with significantly higher concentrations than reference reaches, the majority of samples did not have HQs > 1 (Table 3-25).

Arsenic was detected with HQs > 1 in most largemouth bass fillet samples. Concentrations in the Emory River were also shown to be significantly higher than reference reach (ER_Ref). In bluegill sunfish fillet samples, arsenic HQs > 1 were more sporadic, with approximately half of samples showing HQs > 1. Of these bluegill sunfish samples, significantly higher concentrations were noted from some reaches in both the Clinch and Emory Rivers (Table 3-25). HQs > 1 for arsenic were only noted from a few channel catfish samples, with none of the results being from reaches with significantly higher concentrations than a reference reach.

Concentrations of mercury resulting in HQs > 1 were seen in approximately half of the fillet samples for largemouth bass and channel catfish and were spread across the sample reaches and reference reaches. Fewer HQs > 1 were seen in bluegill sunfish fillet samples. No LOAEL HQs > 1 for mercury were noted from any fillet samples. It should be noted that mercury concentrations from the three species were not significantly higher than reference reaches for fillet samples.

Concentrations of selenium above the NOAEL CBR are widespread for fillet samples (Table 3-25). Selenium NOAEL HQs were greater than 1 in nearly all fillet samples collected from both sample reaches and reference reaches in the Clinch and Emory Rivers. LOAEL HQs > 1 were noted in some bluegill sunfish fillet samples from sample reaches in both the Clinch and Emory Rivers. No LOAEL HQs > 1 were seen in reference reaches for fillet samples. Significantly higher concentrations of selenium than a reference reach were noted for bluegill sunfish, largemouth bass, and channel catfish fillet samples at most locations in the Clinch and Emory Rivers.

Whole Body Samples

CBRs for 15 trace metals were available from the ERED database for whole body samples. These COPECs include aluminum, arsenic, beryllium, cadmium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. It should be noted that COPECs not discussed below do not have either HQs > 1 or a CBR. Individual HQs for each sample are provided in Appendix L.

NOAEL HQs for bluegill sunfish whole body samples range from < 1 to 581, while LOAEL HQs for bluegill sunfish range from < 1 to 58. The highest HQs were due to aluminum, iron, manganese, and zinc. NOAEL HQs for gizzard shad whole body samples ranged from < 1 to 847; the maximum LOAEL HQ was 85. Again, the highest HQs were from aluminum, iron, manganese, and zinc. In gizzard shad whole body samples without the gut or its contents, NOAEL HQs ranged from < 1 to 236, and the maximum LOAEL HQ was 24. Again the highest HQs were the result of aluminum, iron, manganese, and zinc.

In bluegill sunfish and gizzard shad whole body samples and also gizzard shad whole body samples without the gut or its contents, NOAEL HQs > 1 were seen for all COPECs with an available CBR with the exception of beryllium, silver, and thallium. HQs > 1 were seen in the sample reaches and reference reaches. LOAEL

HQs > 1 were also noted in both bluegill sunfish and gizzard shad (with and without the gut and its content) whole body samples for aluminum, iron, manganese, selenium, and zinc in most samples. A few LOAEL HQs > 1 were noted for concentrations of arsenic, copper, and cadmium from various river reaches. Whole body HQs are summarized in Table 3-26.

Due to the conservatism used when selecting CBRs, the whole body CBRs are quite low. For example, in cases where a fillet and whole body CBR were available in the ERED database (seven instances), whole body CBRs were approximately 20 times lower than CBRs for fillet samples. As previously discussed, HQs > 1 were seen in both sample and reference reaches (Table 3-26). Comparison of whole body samples to these CBRs is an overly conservative approach for estimating overall risk to fish from exposure to COPECs in ash.

Liver

CBRs for liver tissue are available from the ERED database for 12 trace metals including aluminum, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, vanadium, and zinc. Individual HQs for each liver sample are provided in Appendix L. NOAEL HQs for bluegill sunfish liver samples range from < 1 to 14,598, and LOAEL HQs reach 1,459. The highest HQs were from cadmium; the next highest NOAEL HQ (excluding cadmium) is only 7. The CBR for cadmium is obviously extremely low. Similarly for largemouth bass liver samples, the highest NOAEL and LOAEL HQs (8,029 and 803, respectively) were from cadmium. Excluding cadmium, the next highest NOAEL HQs for largemouth bass liver samples was 7.

In bluegill sunfish liver samples, NOAEL HQs > 1 were seen in the majority of samples for aluminum, cadmium, chromium, selenium, vanadium, and zinc. These COPECs have NOAEL CBRs > 1 from the sample reaches and reference reaches. Cadmium LOAEL HQs were > 1 for bluegill sunfish liver samples. A few LOAEL HQs > 1 were also noted for vanadium. Some significantly higher concentrations of these COPECs were noted in sample reaches when compared to reference reaches (Table 3-27). NOAEL HQs > 1 for lead were seen in only two bluegill sunfish liver samples, with no LOAEL HQs > 1. Individual HQs for each sample are provided in Appendix L.

Due to the elevated HQs in both the sample reaches and reference reaches for aluminum, cadmium, chromium, selenium, vanadium, and zinc, liver HQs provide limited information in estimating overall risks to fish from exposure to trace metals in ash.

Ovary

CBRs for ovary tissues were only available from the ERED database for arsenic and selenium. No HQs for arsenic calculated for bluegill sunfish or largemouth bass ovary samples were > 1. Conversely, the bluegill sunfish and largemouth bass ovary samples had NOAEL HQs > 1 for selenium, with the maximum being 5. These include samples from both sample reaches and reference reaches. No LOAEL HQs > 1 were noted

(Table 3-28). Individual HQs for each sample are provided in Appendix L. Due to the lack of CBRs for ovary tissues, and the presence of HQs > 1 for the selenium samples (including reference locations), ovary HQs provide little information useful in estimating overall risks to fish from exposure to trace metals in ash.

3.2.4.2 Risk Estimation

Fish tissue samples collected in 2010 were compared to CBRs as a conservative assessment of potential risk to fish due to exposure to legacy compounds and COPECs related to the ash release. CBRs for both NOAELs and LOAELs were selected from the ERED database. Fish tissue samples included several species (bluegill sunfish, largemouth bass, and gizzard shad) and tissue types (fillet, whole body with and without the gut and its contents, liver, and ovary). Exceedances of CBRs were noted for the tissue types and across both sample reaches and reference locations, especially in whole body samples, drawing attention to the conservatism in the CBR selection process.

Although the CBRs were exceeded for many COPECs in each tissue type, a similar number of exceedances were noted in the reference reaches from both the Clinch and Emory Rivers. Statistical analyses did show significantly higher concentrations of some COPECs in sample reaches when compared to reference areas. These differences are presented in detail in Appendix L. For COPECs that had significantly higher concentrations in sample reaches compared to reference areas, arsenic and selenium concentrations exceeded CBRs most regularly in the various tissue types. In fillet samples, NOAEL HQs were as high as 15 for any species, with the highest values being from selenium in bluegill sunfish, channel catfish, and largemouth bass. For whole body samples, the highest NOAEL HQ was 847. The highest whole body HQs were from aluminum, iron, manganese, and zinc in both bluegill sunfish and gizzard shad (with and without the gut and its contents). In liver samples, NOAEL HQs went as high 14,598, but these values were for cadmium, which has an extremely low CBR. Excluding cadmium, whole body NOAEL HQs are much lower (maximum of 7). The only CBRs available for ovary samples were for arsenic and selenium. No HQs were > 1 for arsenic and the maximum HQ from selenium was 5.

Risk Conclusion for Fish Tissue Chemistry – The LOE for the comparison of fish tissue data to CBRs indicates several exceedances of conservative CBRs from the literature, many of which are questionable in terms of their relevance to the Site or population impacts to fish. In addition, similar exceedances are frequently observed in fish collected from reference areas. Overall, the chemistry results indicate limited uptake/bioaccumulation of ash-related metals and no substantial risks to fish.

3.2.5 Fish Health Metrics

The third LOE for fish is based on the ORNL fish health studies for the Clinch, Emory, and Tennessee Rivers following the ash release. The results of these studies were reported in the *Fish Health Studies Associated with the Kingston Fly Ash Spill* (Appendix I). The fish health studies were initiated in early spring

2009 and were conducted to determine if exposure to ash-related metals causes health effects on sentinel fish species.

3.2.5.1 Analysis of Exposure and Effects

The fish health studies included five seasonal studies: spring 2009, fall 2009, spring 2010, fall 2010, and spring 2011 (although the spring 2011 data are not yet reported and are therefore not discussed below). The studies included three to four sentinel fish species that represent different feeding habits, behaviors, and home ranges.

The spring field sampling was conducted during April and May when fish were reproductively mature and developing their gonads. Only female fish were collected in the spring because emphasis was on reproductive integrity, while in the fall a mix of both males and females were collected. A summary of the spring and fall collections is provided below:

- In spring 2009, largemouth bass, bluegill sunfish, and white crappie were collected at ERM 8.0 (upstream reference), ERM 3.0, ERM 0.9, CRM 1.5, and CRM 8.0
- In spring 2010, largemouth bass, bluegill sunfish, white crappie, and redear sunfish were collected at ERM 8.0, LERM 2.0, ERM 3.0, ERM 0.9, CRM 1.5, and CRM 8.0
- In fall 2009, bluegill sunfish, largemouth bass, and channel catfish were collected from ERM 8.0, ERM 3.0, ERM 0.9, CRM 1.5, and CRM 8.0
- In fall 2010, bluegill sunfish, largemouth bass, and channel catfish were collected from ERM 8.0, LERM 2.0, ERM 3.0, ERM 0.9, CRM 1.5, and CRM 8.0.

At each location and for each species and sampling period, the goal was to collect eight adult fish, of which fish health analyses were conducted on the eight individuals, and metals were analyzed for in six of these eight fish. Fish were collected via electrofishing. Immediately upon collection, a blood sample was taken from the caudal veins of each fish using vacutainers (lithium as anticoagulant), and fish were transported alive to the laboratory for processing. At the lab, fish were processed for a suite of health indicators, the liver and ovary harvested for metal analysis, and each individual filleted for metals analysis of muscle tissue.

Fish health indicators representing different functional response groups and levels of biological organization were measured. Specific health indicators were as follows:

- Condition indices
- Bioenergetic and hematological responses
- Histopathological indicators
- Indicators of carbohydrate-protein metabolism

- Organ dysfunction responses
- Measures of electrolyte homeostasis
- Indices of feeding and nutrition including those parameters associated with digestive physiology.

A full description of the fish handling techniques is presented in Appendix I. Briefly, fish were processed in the laboratory for a variety of fish health measurements, including length and weight and any external signs of injury including presence of disease, infection, or parasites. Following euthanization, fish were dissected and major organs including liver, gills, kidneys, and ovaries were examined for anomalies including parasites, necrosis, or discoloration. Ovarian tissue was further processed for various measurements of reproductive integrity (reproduction status report). Sections of gill, liver, and kidney tissue were preserved in formalin for histopathological analysis.

Measures of overall health and condition included the liver, visceral, and spleno-somatic indices, which were calculated as the mass of these respective organs divided by total body mass. The condition factor for each fish was also calculated. A Health Assessment Index (Adams et al. 1993) was also determined for each fish based on observation, scoring, and evaluation of a variety of internal and external lesions and anomalies. Bioenergetic status was assessed by estimating the relative amount of lipid stored in the mesenteries of the visceral (scored between 0 and 4), and the nutritional/feeding level of each fish was estimated by the fullness of the stomach and intestine (scored between 0 and 4).

For histopathological analysis, internal organs such as the liver, gill, and ovary were dissected, fixed in formalin, and shipped to the University of California-Davis toxicopathology laboratory for analysis. At the lab, stained tissue sections were screened for lesions and subjected to histopathologic analysis. The severity of lesions in the liver, gill, kidney, and ovarian lesions were scored based on a qualitative scale of 0 = no lesions present, 1 = mild lesions, 2 = moderate lesions, and 3 = severe lesions, with a score of 2 or 3 being considered a significant lesion. A composite lesion score for each species at each location was calculated as the average sum of lesion scores for each organ for that species and location.

The fish health results were reported by Adams and Fortner (2011b) as both individual responses and integrated (holistic) responses. Individual health responses are presented by functional health metric groups for each species including bluegill sunfish and largemouth bass for each of four seasons (spring 2009 and 2010; fall 2009 and 2010), channel catfish (fall 2009 and 2010), white crappie (spring 2009 and 2010), and redear sunfish (spring 2010). The integrated fish health analysis incorporated the measured fish health parameters together within a multivariate context to assess the holistic response of fish to environmental conditions at each sample location. Statistical tests (i.e., canonical discriminant analysis) were used to assess the integrated health status of bluegill sunfish, largemouth bass, channel catfish, white crappie, and redear sunfish at each sample location.

Individual Health Responses

The individual health responses are presented in this section as a function of fish species and season. For each species, comparisons of individual responses are made on the basis of spring 2009 versus spring 2010 and fall 2009 versus fall 2010.

A suite of health metric were used in this study to investigate the possible relationship between exposure of fish to ash-associated COPECs and health response of these fish in the Clinch and Emory River systems. Seven functional groups of fish health responses were measured that reflect different physiological functions and represent varying degrees of sensitivity and specificities to environmental stressors, and also reflect different levels of ecological relevance. Results presented in figures from Appendix I provide examples of two representative health metrics for each of these seven functional groups including condition indices, bioenergetics and hematology, histopathology, carbohydrate-protein metabolism, organ dysfunction, electrolyte homeostasis, and feeding and nutrition. Comparisons of results between the spring 2009 and spring 2010 and between fall 2009 and fall 2010 for four species (bluegill sunfish, largemouth bass, channel catfish, and white crappie) show that there is a wide variation in species, seasonal, and location responses for the various measured health metrics.

For temporal comparisons of individual health responses, the number of spatial gradients observed in 2009 versus those seen in 2010 was determined considering both bluegill sunfish and largemouth bass for the 14 representative health metrics. For these two species in spring and fall 2009, there were 15 obvious spatial gradients and seven instances where a moderate or slight spatial gradient was detected (Table 3-29). In spring and fall 2010, 17 obvious spatial gradients were observed with six moderate level gradients (Table 3-29). Therefore, because approximately the same number of spatial patterns was observed in 2009 compared to 2010, it appears that no definitive conclusions can be reached regarding changes in temporal patterns in individual health responses from 2009 through 2010.

The reasoning behind this conclusion is that if a spatial pattern is observed in some individual health metric along the gradient of the Emory and Clinch Rivers (where a response change along this gradient is based on differences from the upstream reference location), such a change is assumed to be caused by exposure to ash (Adams and Fortner 2011b). The fish bioaccumulation report submitted to TVA in mid-June (Adams et al. 2011a) clearly demonstrated that there were spatial patterns in selenium and, to a lesser degree, arsenic in the Emory and Clinch Rivers as a function of the location of the ash release. If similar spatial patterns are observed in individual fish health responses along the same downstream gradients of these systems, the inference is that there could be a causal relationship between ash-associated COPECs and health responses. Therefore, changes in the spatial patterns of individual health responses over time would suggest the following:

- If the number and/or frequency of spatial trends increases over time after the release, fish may be incurring increased damage or injury
- If the frequency or number of spatial trends decreases over time, some level of recovery or improvement could be occurring
- If the number or frequency of spatial trends does not change over time, no conclusion on the health of the sentinel fish species can be made

The results of this individual fish health analysis, based on the assessment of individual health metric responses alone, is inconclusive for these sentinel species because the number or frequency of spatial trends did not change appreciably from 2009-2010. However, this analysis is based on the behavior of individual health responses over time, which can be highly variable as a function of location, season, and species.

To address the issue of multiple sources of variability, an integrated assessment approach was also taken using the individual health metrics together in a holistic context, which collapses the multiple sources of variability into one variance structure and provides an overall view of spatial (differences among sample locations) and temporal (differences among seasons and years) changes in fish health. Therefore, this BERA also considered the integrated fish health response as described below.

Integrated Health Responses

The integrated health status of fish was evaluated as five main groups of temporal or seasonal comparisons (Adams and Fortner 2011b) including:

- Bluegill sunfish in spring of 2009 versus spring 2010
- Largemouth bass in spring 2009 versus spring 2010
- Bluegill sunfish in fall 2009 versus fall 2010
- Largemouth bass in fall 2009 versus fall 2010
- Channel catfish in fall 2009 versus fall 2010.

In addition, the health status of two other species (white crappie and redear sunfish) were evaluated on a spatial basis for one season (spring 2010), because in spring 2009 white crappie could not be collected at two locations due to scarcity of preferred habitat for this species, and sampling for redear sunfish was not initiated until spring 2010.

- **Bluegill Sunfish (spring 2009 versus spring 2010)** – In spring 2009, the health status of bluegill sunfish from CRM 1.5 was more similar to the reference locations than were bluegill sunfish from the remaining locations including ERM 3.0, ERM 0.9, and CRM 8.0 (the positive control). In this multivariate analysis, results for ERM 3.0, ERM 0.9, and CRM 8.0 were grouped at about the same multi-

dimensional distance from the reference location results, and therefore were somewhat similar in their health status compared to the reference locations (Figure 3-21, top panel). One year later in spring 2010, the health status of bluegill sunfish from ERM 0.9 was more similar to the reference locations than in 2009 (ERM 0.9 moved closer to the reference locations) (Figure 3-21, bottom panel). In this case, the statistical linear distance between the reference locations and ERM 0.9 was 14 units in spring 2009 and decreased to 10 units in spring 2010. Over this period, however, bluegill sunfish from CRM 1.5 became less similar in their health status to the fish from the reference locations, while the overall health of fish from ERM 3.0 and CRM 8.0 did not change appreciably over this period.

- **Largemouth Bass (spring 2009 versus spring 2010)** – The health of largemouth bass from ERM 3.0 was the most similar to the reference locations compared to largemouth bass from the other three locations (linear distance of 10 in Figure 3-22), and relatively longer distances separated the other three locations from the reference locations. These other three locations were approximately equidistant (or equally dissimilar) to the reference locations, with CRM 1.5 being the most dissimilar because of its linear distance of 32 units from the reference locations (Figure 3-22). In spring 2010, there was a shift in the positions of the study locations results relative to the reference locations. Results for both ERM 0.9 and CRM 1.5 moved closer to the reference locations, while largemouth bass at ERM 3.0 moved further away. As was also observed for bluegill sunfish between spring 2009 and 2010, health status of largemouth bass at CRM 8.0, the positive control location, maintained the same health status over the year because it remained approximately equidistant from the reference locations both years. As was also seen for bluegill sunfish during the spring, the two reference locations were similar to each other.

- **Bluegill Sunfish (fall 2009 versus fall 2010)** – In fall 2009, health status of bluegill sunfish from CRM 1.5 were very similar to the reference locations, with bluegill sunfish from the three remaining study locations demonstrating a large difference in health status compared to the reference locations (Figure 3-23). Over the year, health status of bluegill sunfish from the two locations immediately below the release (ERM 3.0 and 0.9) moved quite close to the reference locations, while health status of bluegill sunfish from CRM 1.5 moved further away, and those from CRM 8.0 maintained approximately the same distance from the reference locations as that in spring 2009. In the spring of both years, the two reference locations maintained close proximity to each other.

- **Largemouth Bass (fall 2009 versus fall 2010)** – As was also observed in fall 2009 for bluegill sunfish, largemouth bass from CRM 1.5 had a very similar health status compared to the reference locations (Figure 3-24). Another similarity to bluegill sunfish in fall 2009 is that largemouth bass from ERM 0.9 are most dissimilar to the reference locations, while largemouth bass from CRM 8.0 were intermediate in their similarity (i.e., distances from reference locations) to the reference locations being located between ERM 3.0 and ERM 0.9 in Figure 3-24. Interestingly, bass from ERM 3.0 were also very similar to the fish from the reference locations. In fall 2010, two shifts in health status occurred compared to the previous fall. Even though largemouth bass from CRM 1.5 maintained a consistent health status over the year,

largemouth bass from ERM 0.9 moved closer to the reference locations, while those from ERM 3.0 moved further away. As has been observed in the previous comparisons between seasons for bluegill sunfish and largemouth bass, the positive control location (CRM 8.0) shifted very little or at the relative to the reference locations during the year.

- **Channel Catfish (fall 2009 versus fall 2010)** – A very similar spatial pattern in fish health status appears to exist between largemouth bass and channel catfish during fall 2009. In both cases, fish from CRM 1.5 and ERM 3.0 were very similar to the reference locations, while fish sampled from ERM 0.9 are the most dissimilar to the reference locations, and those from CRM 8.0 were intermediate between ERM 0.9 and ERM 3.0 (Figure 3-25). Over the period from fall 2009 to fall 2010, the health status of channel catfish collected from the locations did not change appreciably, with the Emory and Clinch River locations being approximately the same distance from the reference locations during both fall periods. It is interesting to note that compared to bluegill sunfish and largemouth bass, the health of channel catfish did not appear to change over a year including both spring and fall periods.

- **White Crappie (spring 2010)** – Compared to bluegill sunfish, largemouth bass, and channel catfish, only a few white crappie could be collected at some locations in spring 2009 due to poor availability of preferred habitat. Even though some white crappie were collected in spring 2009 at the locations, at two locations only one to two individuals were captured despite repeated sampling efforts and, therefore, the discriminant analysis could not be performed because of a low sample size. In spring 2010, however, an adequate number of white crappie was collected at each location to perform the discriminant analysis procedure. On a spatial basis, the health status of the two reference locations was very similar to each other (Figure 3-26). Compared to CRM 1.5, CRM 8.0, and ERM 0.9, white crappie from ERM 3.0 were the most similar to the fish from the reference locations. White crappie from ERM 0.9 were most dissimilar, while individuals from CRM 8.0 and CRM 1.5 were approximately equidistant or similar in their health status compared to the reference locations.

- **Redear Sunfish (spring 2010)** – Sampling for redear sunfish was initiated in spring 2010; therefore, no data are available for the spring, 2009, for temporal comparisons of health status. As opposed to all other species for both sample seasons, the health status of redear sunfish at the two reference locations was not similar (Figure 3-26). Because of the dissimilarity of the two reference locations, there are no obvious spatial patterns or differences among locations.

The results for the discriminate analysis provided both a spatial and temporal assessment of the integrated fish health responses among seasons and sample locations (Adams and Fortner 2011b). In the spatial and temporal comparisons of fish species among locations and seasons (Figures 3-21 through 3-26), the positive control location maintained its same relative position to the reference locations (approximate same linear statistical distances between mid-points of circles) by not moving either closer or farther away from the reference locations over time (Table 3-30). In almost all cases, the sites downstream of the ash release

either moved closer or farther away from the reference locations as a function of time (Table 3-30). The positive control site, CRM 8.0, is several miles upstream in the Clinch River above the confluence of the Emory River and, therefore, not influenced by the ash release. Therefore, fish sampled from this location would not be expected to exhibit behavior relative to increasing or decreasing health status over time as a result of the ash release.

For the integrated fish health responses shown in Figures 3-21 through 3-26, several temporal patterns in fish health status can be observed. Table 3-30 summarizes these temporal patterns for bluegill sunfish, largemouth bass, and channel catfish at each sample location. At the release location (ERM 3.0), the integrated health status of bluegill sunfish improved between fall 2009 and fall 2010 and between spring 2009 and fall 2010. There was no change in overall health for bluegill sunfish from spring 2009 to spring 2010. Over the 18-month study period (spring 2009 to fall 2010), there was an indication in improved health of bluegill sunfish at ERM 3.0. At ERM 0.9, 2 miles below the release, there appeared to be improvement in the health of bluegill sunfish over the seasonal comparisons and over the entire 18-month study period (Table 3-30). At CRM 1.5, approximately 6 miles below the release, the overall health status of bluegill sunfish declined (compared to the reference locations) over the seasonal comparisons and over the entire 18-month study period. At CRM 8.0, the positive control location, there were no temporal changes in health of bluegill sunfish compared to the reference locations. In the seasons when two reference locations were sampled, the reference locations maintained very close similarity to each other. However, it is uncertain whether these sublethal effects (e.g., changes in bioenergetic status) are predictive of ecologically significant health effects (i.e., obvious damage or injury).

For largemouth bass, the integrated health status declined at ERM 3.0 between spring 2009 to spring 2010, between fall 2009 and fall 2010, and over the entire study period of spring 2009 to fall 2010 (Table 3-30). However, at ERM 0.9, the overall health status improved over the seasonal comparisons and over the 18-month study period. At CRM 1.5, overall health improved from spring 2009 to spring 2010 and over the entire study period, but little improvement was seen over the fall period. As was observed for bluegill sunfish, there was no change in the health status of largemouth bass over time at CRM 8.0 compared to the reference locations, and the reference locations maintained very close similarity to each other.

For channel catfish from the locations over the fall 2009 to fall 2010 period, there was no change in health status compared to the reference locations. The finding of no temporal trends or changes in health of channel catfish is most likely related to its large home range, which suggests that the health status of channel catfish is influenced by a wide variety of environmental conditions that occur in the various areas inhabited by this species that are integrated over the total time period of its large range movements.

3.2.5.2 Risk Estimation

The conclusions from the individual and integrated fish health studies are as follows:

Individual Fish Health

- Several individual health metrics deviate from reference conditions downstream of the ash release.
- Even though the health metrics of fish immediately below the ash release appears to deviate from reference areas to some degree, the effects seem to be localized to areas closest to the release because the deviations were not widespread, and were less pronounced in downstream areas.
- Assessment of fish health using a number and variety of individual health metrics show that there is a wide variation in species, seasonal, and location responses for the various measured health metrics.
- Because approximately the same number of spatial patterns in individual health metrics were observed in 2009 compared to 2010 (i.e., individual health metrics either increased or decreased downstream of the reference locations), it appears that no definitive conclusions can be reached regarding changes in temporal patterns in individual health responses.

Integrated Fish Health

- The multivariate discriminant analysis procedure assesses the integrated health status of fish on both spatial and temporal scales, and indicates the following:
  - The reference locations were very similar in their integrated responses over species and seasons.
  - There were no temporal changes in the integrated responses of any species at the positive control location relative to their behavior compared to the reference locations.

- From 2009 to 2010, the health status in bluegill sunfish and largemouth bass at two of the three study locations (either ERM 3.0, ERM 0.9, or CRM 1.5) downstream of the ash release approached reference conditions. The health status of channel catfish did not change over time.

Risk Conclusion for Fish Health Studies – To date, although some individual health metrics appear to deviate from reference conditions in the vicinity of the ash release, no significant health effects (i.e., obvious damage or injury) have been observed, suggesting that:

- The various observed changes in individual fish health metrics did not appear to be of sufficient magnitude or duration to result in significant adverse impacts.
- In these species, although the rate of biological repair or recovery was not directly assessed, it appears to offset any impact potentially caused by exposure to COPECs.

3.2.6 Surface Water Chemistry

Another LOE used to evaluate potential risks to fish is the comparison of surface water data to acute and chronic water quality criteria. This method utilizes the surface water data from the near-location and reference reaches, and compares the data to regulatory benchmarks that were developed to protect fish and other aquatic resources. The ratios of the measured concentration to the regulatory benchmarks are used to calculate HQs for individual chemicals and individual reaches.

3.2.6.1 Analysis of Exposure and Effects

The data used in the assessment are the results from the 2010 surface water sampling and include inorganic compound results from samples collected at mid-water column and near the water-sediment interface (i.e., epi-benthic samples). Surface water samples were collected in 2010 during eight events (dates were 8/31, 9/8, 9/14, 9/21, 9/29, 10/5, 10/12, and 10/19). The data include a total of 11 stations, with one or more stations located in each of nine reaches. Locations (and reach designations) were as follows:

- Three stations along the Clinch River (CRM 2.0 [CR_A]; CRM 3.5 [CR_B]; and CRM 6.0 [CR_Ref]).
- Six stations along the Emory River (ERM 0.3 [ER_A]; ERM 1.0 [ER_A]; ERM 2.0 [ER_B]; ERM 3.0 [ER_B]; ERM 4.0 [ER_C]; ERM 8.0 [ER_Ref]).
- Two stations along the Tennessee River (TRM 566.0 [Tennessee River reach A (TR_A)]; TRM 568.5 [Tennessee River reference (TR_Ref)]).

The surface water sample locations are shown in Figure 2-19.

At each of the 11 locations, two types of surface water samples were collected: one from mid-water depth and one from near the bottom. Samples included both filtered and unfiltered samples, and were analyzed for Target Analyte List (TAL) metals. The data used in the comparison include both the mid-water depth and the epi-benthic samples, and include both total (unfiltered) results and the dissolved (filtered) results. Use of the unfiltered surface water data is conservative, because for metals it is the dissolved fraction that is bioavailable and potentially toxic. For example, according to USEPA (1992a), toxicity testing has shown that dissolved measurements are better predictors of toxicity than total recoverable measurements.

The benchmarks used to evaluate risks for fish are the USEPA (2009) numeric water quality standards for the protection of aquatic life (ambient water quality criteria [AWQC] criterion continuous concentration values). These benchmarks were derived based on data for many species of aquatic organisms and are considered protective of fish. Because the criteria must be protective of a wide range of species exposed under a variety of conditions, they are conservative. As a result, their application to a BERA likely result in risk estimates that are generally higher than would occur for individual species in the field.

Several AWQC are hardness dependent (i.e., cadmium, chromium, copper, lead, nickel, and zinc); however, as a conservative approach, the evaluation uses a default hardness value of 50 mg/L. The assumption of a hardness value of 50 mg/L is conservative given that measured hardness values range from 66.2 mg/L to 138 mg/L. Both filtered and unfiltered surface water data are compared to criteria, and criteria are adjusted (based on USEPA guidance; USEPA 2009) by a conversion factor to obtain dissolved criteria.

A summary of the surface water benchmarks and comparison to surface water data is presented in Table 3-31 (mid-water) and Table 3-32 (epi-benthic water). HQs (calculated as the ratio of the detected concentration to the benchmark) are presented in Table 3-33 (mid-water) and Table 3-34 (epi-benthic water).

The results from the surface water criteria comparison are summarized as follows.

Mid-Water Data

- For total metals, aluminum exceeded the chronic benchmark in each of the nine reaches except for the ER_Ref location which was non-detect (Table 3-31). Concentrations were below the acute benchmark. Mercury exceeded the chronic value in four reaches: one reach along the Clinch River (CR_A) and three locations along the Emory River (ER_A, ER_B, and ER_Ref). Mercury concentrations were below the acute benchmark. The chronic HQ values for total metals range from less than 1 to 4 for aluminum, and less than 1 to 19 for mercury (Table 3-33). Acute HQ values for total metals were less than 1.
- For dissolved metals, aluminum exceeded the chronic benchmark in one Clinch River reach (CR_B) (Table 3-31). Concentrations were below the acute benchmark. Mercury exceeded the chronic value in only the CR_Ref. Mercury concentrations were below the acute benchmark. The chronic HQ values for dissolved metals were less than 1 except a single aluminum value (2) and a single mercury value (17) (Table 3-33). Acute HQ values for dissolved metals were less than 1.

Epi-Benthic Water Data

- For total metals, aluminum exceeded the chronic benchmark in each of the nine reaches (including the reference locations) (Table 3-32). Aluminum concentrations for two Emory River reaches (ER_B and ER_C) also exceed the acute benchmark. Lead exceeded the chronic value in one reach along the Emory River (ER_B) and one reach along the Clinch River (CR_A). Lead concentrations were below the acute benchmark. Only total lead measured in two reaches exceeded the chronic benchmark; however, these samples were below the benchmark when sample-specific hardness values were used in place of the conservative 50 mg/L value. Mercury exceeded the chronic value in two reaches along the Emory River (ER_A and ER_Ref), and mercury concentrations were below the acute benchmark. The chronic HQ values for total metals ranged from less than 1 to 14 for aluminum, less than or equal to 1 for lead,

and less than 1 to 19 for mercury (Table 3-34). Acute HQ values for total metals were less than 1 except two values for aluminum (1 and 2).

- For dissolved metals, concentrations were below both the chronic and acute benchmarks (Table 3-32). Acute HQ values for dissolved metals were less than 1 (Table 3-34).

3.2.6.2 Risk Estimation

Collectively, the surface water data for the Clinch, Emory, and Tennessee Rivers indicated that total (unfiltered) concentrations for several metals (primarily aluminum, and to a lesser extent lead and mercury) exceeded the chronic values, and in rare instances exceed the acute values. However, exceedances of aluminum and mercury also occurred for upstream (reference) reaches. Exceedances were much less frequent for dissolved metals than for total metals. These data indicate that during the time of sampling, the ash release did not have a significant impact on surface water quality within the Clinch, Emory, or Tennessee Rivers. Although there are a few exceedances of conservative water quality benchmarks, exceedances were relatively infrequent for dissolved concentrations. Because dissolved concentrations provide a more reliable predictor of bioavailability (and risk) to fish, the LOE represented by the surface water data do not indicate significant risks to fish.

Risk Conclusion for Surface Water Chemistry – The LOE for the comparison of surface water chemistry data to water quality benchmarks indicates few exceedances of water quality benchmarks. These results indicate limited potential for exposure to ash-related COPECs in surface water, and significant risks to fish are not expected.

3.3 Risk Characterization

Risks to fish populations were evaluated based on multiple LOEs, including the assessment of biological survey data, toxicity tests, reproductive studies, tissue concentrations, health metrics, and surface water chemistry. The ecological relevance, potential risk, and confidence in the risk determination are summarized in Exhibit 3-1.

Exhibit 3-1. Fish Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Community Surveys	Negligible	High	Moderate	High
Toxicity Tests	Negligible	Moderate	High	Moderate
Reproductive Studies	Negligible	High	High	Moderate
Tissue Concentrations	Moderate	Low	Moderate	Low
Health Metrics	Low	Low	Moderate	Low
Surface Water Chemistry	Low	Low	Moderate	Low
Weight of Evidence	**Moderate**	**Moderate**	**Moderate**	**--**

Based on concordance of the LOEs, potential risks to fish are considered negligible with moderate confidence (Exhibit 3-2).

Exhibit 3-2. Fish Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Community Surveys	High	Negligible	High	--
Toxicity Tests	Moderate	Negligible	High	--
Reproductive Studies	Moderate	Negligible	High	--
Tissue Concentrations	Moderate	Moderate	Low	Aluminum, Arsenic, Cadmium, Copper, Lead, Selenium, Zinc
Health Metrics	Low	Low	Low	--
Surface Water Chemistry	Low	Negligible	Low	--
Weight of Evidence	**--**	**Negligible**	**Moderate**	**--**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

3.3.1 Weight of Evidence

A WOE approach is used to evaluate overall risk to fish. The risk for each LOE is evaluated, along with the confidence in the risk estimate. The ecological relevance of each LOE is considered when assessing overall risk.

3.3.1.1 Fish Community Surveys

The fish community survey results provide a direct measure of possible population-level effects on fish. The results from the Spring Sport Fish Survey (Appendix E) indicate similar spatial and temporal patterns in catch rates, length frequency distributions, relative weights, and instances of external parasites among locations (near field and far field) both before and after the release, and provides strong evidence that any patterns observed were associated with factors that extended reservoir-wide. Similarly, the annual fish community survey results for pre- release to post- release conditions indicates RFAI scores are generally consistent, and the fish assemblage data are representative of the habitat. To date, no significant impairment to the fish community is observed in the surveys' results.

3.3.1.2 Surface Water Toxicity Tests

The fish toxicity tests include the surface water toxicity testing data and the fathead minnow larval fish studies. The tests provide a direct measure of potential toxicity of ash-related materials to fish. To date, no significant health effects (i.e., no direct observations of reduced survival or biomass) have been observed from the fish toxicity test studies. These observations are consistent with the fish community surveys, because overt toxic effects were not observed in the population data.

3.3.1.3 Fish Reproductive Studies

The fish reproduction study evaluated the reproductive condition of three species (bluegill sunfish, largemouth bass, and redear sunfish) and included direct observations of fish reproduction health. The results from the fish reproductive study indicate that bluegill sunfish reproduction was delayed at the ash release area in the Emory River (ERM 3.0) during the 2009 breeding season immediately following the ash release. However, an observed delay in ovarian development was most likely due to habitat alterations and food web disruptions. Delayed development alone is not conclusive evidence of an adverse effect on spawning success or population maintenance. There also was no conclusive evidence of adverse reproductive effects in female bluegill sunfish sampled from the other study locations in 2009 or from any of the locations sampled in 2010. Thus, any delays in development were localized, transient, and not representative of current conditions. Results of the studies with largemouth bass and redear sunfish did not provide conclusive evidence that the ash release had ecologically significant adverse impacts on the reproductive condition of females of these species. To date, no significant health effects (i.e., obvious damage or injury) have been observed from the fish reproductive studies.

3.3.1.4 Fish Tissue Chemistry

The LOE for the comparison of fish tissue data to CBRs indicates several exceedances of non-site-specific conservative literature-derived benchmarks. However, similar exceedances are frequently observed in fish

collected from reference areas. These results indicate that although there has been limited uptake of ash-related metals, and levels often exceed values associated with adverse effects, the conservative nature of the screening is not indicative that effects to the fish population have actually incurred. Conversely, the site-specific fish population surveys and toxicity testing results indicate that there are no risks to fish populations at the Site.

3.3.1.5 Fish Health Metrics

The fish health studies included five seasonal studies: spring 2009, fall 2009, spring 2010, fall 2010, and spring 2011(although the spring 2011 data are not yet reported and are therefore not discussed) and three to four sentinel fish species that represent different feeding habits, behaviors, and home ranges. The results from the studies indicate that although some individual health metrics for fish in the vicinity of the ash release differ from reference locations, no significant health effects (i.e., obvious damage or injury) have been observed, suggesting that:

- COPEC exposure is not of sufficient magnitude or duration to elicit observed changes in biological responses
- In these species, the rate of biological repair exceeds or equals the rate of damage caused by exposure to contaminants.

3.3.1.6 Surface Water Chemistry

The LOE for the comparison of surface water chemistry data to water quality benchmarks indicates few exceedances of water quality benchmarks. These results indicate limited potential for exposure to ash-related metals in surface water, and limited or no risks to fish.

3.3.2 Uncertainties

There are many sources of uncertainty inherent in BERAs, and the objective of an uncertainty analysis is to present key information regarding assumptions and uncertainties in the risk assessment process and place the quantitative risk estimates into proper perspective. As described in the following sections, this BERA uses a conservative approach that is more likely to overestimate risks to fish than underestimate risks. Sources of uncertainty related to the characterization of potential risks to fish are described below and include uncertainties associated with:

- The risk assessment approach (e.g., selection of measurement endpoints)
- Data used to characterize the Site
- Toxicity benchmarks
- Fish toxicity testing

- Fish health and reproduction studies
- Fish community survey data
- The final interpretation (or risk characterization) of the evaluation.

3.3.2.1 Risk Assessment Approach

The BERA follows standard ERA guidelines developed by USEPA (1992b, 1997a). The guidelines reference a technical approach that includes the identification of assessment endpoints, measurement endpoints, and risk questions. The measurement endpoints selected for fish include multiple LOEs, including comparison of fish tissue data to CBRs, comparison of surface water data to criteria, fish toxicity studies, fish health and reproduction studies, and direct observations of fish community abundance and diversity. Given the selection of multiple and diverse LOEs for fish, the overall level of uncertainty associated with the general risk assessment approach for fish is relatively low.

3.3.2.2 Site Data

The data used to evaluate risks to fish are from multiple site investigations that were conducted after the ash release (and in some cases prior to the release). The investigations were developed in consultation with state and federal agencies and included sampling and analysis of surface waters and fish tissues, as well as fish collections for health and reproductive studies and fish community studies. Collectively, these data are sufficient to adequately characterize risks to fish associated with release. However, there is a degree of uncertainty regarding site data and the ability of the data to represent site conditions. For example, an overall higher level of uncertainty (in regards to the data) is associated with the surface water data, because these data were collected during a relatively brief period in 2010 (dates were 8/31, 9/8, 9/14, 9/21, 9/29, 10/5, 10/12, and 10/19) and may not reflect current conditions.

Recent Site data (Appendix C) was preliminarily evaluated. A comparison of fish tissue concentrations of selected metals in samples collected in 2010 and 2011 was conducted. Metals included in this analysis were aluminum, arsenic, cadmium, copper, lead, selenium and zinc. Samples from both the Emory and Clinch Rivers were included from all river reaches (ER_R, ER_C, ER_B, ER_A, CR_R, CR_B, and CR_A). Fillet samples of largemouth bass, bluegill, and channel catfish were included as well as whole body samples of gizzard shad and whole body samples of gizzard shad without the gut or its contents. Percent change in concentrations from 2010 to 2011 was calculated using the average tissue concentration of each metal at each sample reach. Average concentrations were calculated using the full detection limit for non-detect samples. This analysis did not include any weighting of the averages to consider sample sizes from each reach and species, adding some uncertainty to the assessment. Considering all of the species, the seven metals previously mentioned, and all the sample reaches collectively, the average tissue concentrations went up slightly from 2010 to 2011 (4 percent). However, this preliminary evaluation does not change the overall interpretation of concentrations of metals in fish tissue.

3.3.2.3 Toxicological Benchmarks

Toxicological benchmarks that were used in the assessment of risk to fish include the AWQC to evaluate surface water data and CBRs to evaluate fish tissue data. These benchmarks are not site-specific, and there is a relatively high level of uncertainty in their application for evaluating risks to fish. The level of uncertainty associated with the fish tissue CBRs is of particular importance. Although CBRs are used to tie fish tissue residues to toxicological outcomes, CBRs are not an accurate predictor of actual site-related population risks, but instead are literature-derived benchmarks for specific endpoints/effects to individual fish under experimental conditions. The inaccuracy of CBRs to predict site-specific risks is evidenced by the fact that background fish tissue concentrations frequently exceed CBRs.

When expressing CBRs on total internal concentrations, the capacity of organisms to sequester metals in forms that are not biologically reactive is neglected. In other words, the biological magnitude of accumulated metal concentrations will be dependent on the way organisms cope with the metal (Martinag et al. 2004), and CBRs may not accurately incorporate these coping methods. Also, ecologically meaningful CBRs can only be derived if several conditions are met, including whether the toxicological mechanism has been identified, if the parent study provides adequate time to estimate steady state, and there is evidence of a concentration-response relationship. Not meeting these conditions adds considerable uncertainty to the assessment. For example, specific tissues involved in the sites of toxic action may be unknown and, therefore, it may not be possible to select a tissue that is closest to the sites of toxic action. Practical constraints on the quantity and quality of CBRs may also require that other tissues (such as whole body CBRs) be considered. CBRs are also uncertain because contaminant interactions cannot be reliably predicted.

3.3.2.4 Fish Toxicity Testing

Toxicity tests provide information on the potential for adverse biological effects from contaminants and are recognized as a key component of the ERA process (USEPA 1998). Toxicity tests used in this BERA to evaluate risks to fish include the surface water 7-day chronic tests and the ash laboratory toxicity tests. The test procedures used to evaluate risks to fish were based on standard testing protocols, using a laboratory organism (i.e., fathead minnows). However, there is a level of uncertainty associated with the toxicity tests. For example, the toxicity testing is done using laboratory-raised organisms, which have historically shown to be more sensitive to contaminants than field organisms. Also, there may be increased uncertainty because the test durations were relatively short. However, the tests used sensitive life stages and, therefore, should reflect the potential for adverse effects.

In 2011 a fish developmental study was conducted. Specifically, redear sunfish and bluegill sunfish were collected during the spring 2011 breeding season from up to five sites in the Emory and Clinch Rivers. Mature eggs and milt were collected from the fish, and developing embryos were subsequently raised in

replicate incubation chambers through hatching and yolk absorption in water from both the respective sites of origin and reference water (from ERM 8.0) in a two-way (crossover) ANOVA experimental design. Test endpoints included fertilization success, survival, hatching success, and incidences of developmental abnormalities through the yolk-absorption stage of development. Although the data are currently being evaluated, embryo larval survival for both species was relatively high greater than 70 percent for all study sites with both sunfish species, an unexpectedly high rate of survival considering the developmental nature of these in vitro spawning procedures. This preliminary evaluation does not change the overall interpretation of risk to fish (Appendix AC).

3.3.2.5 Fish Health and Reproduction Studies

The fish health and reproduction studies were carefully designed to evaluate the possibility for exposure and subsequent effects on fish health and reproduction. Given that the fish were collected from the field (including areas in the immediate vicinity of the ash release), there is little uncertainty that the subjects represent actual conditions. However, the largest source of uncertainty with the study results is the actual implications of measures towards population-level effects to fish. For example, the fish health studies indicated evidence of exposure to metals (based on increases in tissue metals concentrations compared to background) and some levels of effect on individuals. However, individual health responses can be highly variable as a function of location, season, and species, and it is uncertain whether these sub-lethal effects (e.g., changes in bioenergetic status) are predictive of ecologically significant health effects (i.e., obvious damage or injury).

3.3.2.6 Fish Community Survey Data

Community survey data are used in this BERA as an LOE to evaluate risks to fish. These types of data are useful for characterizing ecological risks because they are based on direct observations of the local populations. These types of community data are often highly variable, and a substantial level of effort is required to adequately characterize the environmental community to the extent where measured response may be interpreted. Regardless, the fish community data represent a robust dataset with observations recorded for several locations and time periods, and there is a relatively high degree of certainty that they are representative of current conditions and appropriately reflect localized fish populations.

3.3.2.7 Risk Interpretation

This BERA uses multiple measurement endpoints to evaluate potential ecological effects, and a qualitative WOE approach is used to help interpret the information. The WOE approach takes into account the strengths and weaknesses of the different measurement methods to determine whether the results show that a stressor has caused (or could cause) a harmful environmental effect. The WOE approach used in this BERA is qualitative, and identifies a general weighting scheme of low, medium, or high based on a list of

10 relevant attributes. This information is used to help interpret the risk assessment data, to determine how well the measurement endpoint represents the assessment endpoint, and to determine how well the measurement endpoint estimates or predicts the effect. The information can also assist in the understanding of which LOEs have a stronger relative technical basis and can help reconcile conflicting LOEs.

3.3.3 Summary and Conclusions

Based on the combined overall WOE, risks to fish within the Emory, Clinch, and Tennessee Rivers are expected to be negligible. The LOEs that are most supportive of the low estimates of risk are the lack of population-level impairment from the fish community studies, and the lack of effects from the fish toxicity testing.

4. Benthic Invertebrates

Benthic invertebrates are an important receptor group for this BERA. They live within and on top of sediments, are typically sedentary, and play a key role in cycling nutrients and processing organic matter. In addition, they provide a food base for various fish and wildlife species. Because their lives are spent in localized benthic habitats they are directly exposed to chemical constituents in sediments and surface waters and can bioaccumulate constituents into their tissues. This provides a route for trophic transfer of constituents to predators. The composition and structure of the benthic community is often used as an indicator of ecosystem health. They are often the most sensitive receptor group to metals and other stressors in sediments and related porewaters.

4.1 Problem Formulation

Benthic invertebrates live in or attached to the sediments on the river bottom or near the sediment-water interface. Typical species mixtures in the site-related waterways include crustaceans (e.g., crayfish and amphipods), bivalves (e.g., mussels and clams), snails, aquatic worms, and the immature forms of a variety of aquatic insects (such as mayfly nymphs and midges). Historical surveys in the site area have identified several protected or sensitive species of invertebrates. Five species of protected mollusks have historically been observed in Watts Bar Reservoir area and its tributaries, but have not been found in 30 years: Alabama lamp mussel *(Lampsilis virescens)*, dromedary pearly mussel (*Dromus dromas*), purple bean (*Villosa perpurpurea)*, fine-rayed pigtoe (*Fusconaia cuneolus*), and Anthony's river snail *Athearnia anthonyi* (TVA 2009a). These species were most likely extirpated when the dam was constructed (TVA 2009a). The pink mucket *(Lampsilis abrupta)*, rough pigtoe *(Pleurobema plenum)*, fanshell *(Cyprogenia stegaria)*, and orange-foot pimpleback (*Plethobasus cooperianus*) have been identified in the tail waters of Melton Hill Dam (the lower Clinch River) (TVA 2009a).

Because they are so closely associated with sediments and are relatively sedentary, benthic invertebrates are highly exposed to constituents in the sediments and porewaters, and can, therefore, be used as indicators of sediment/habitat quality. In sufficient concentrations many of the ash-related metals can cause reduced growth and/or survival of some invertebrates; some may bioaccumulate over time to high enough concentrations to pose risks to invertebrate populations. As the base of the food web, benthic invertebrates and the emergent adult life stages of many species are prey for other ecological receptors and may transfer contaminants to their consumers.

4.1.1 Assessment Endpoints

As described in Section 2.2, assessment endpoints are used in the ERA process to identify focus areas for the risk assessment and develop a method for evaluating ecological risks. The initial selection of the assessment was presented in the BERA Methodology of the SAP (Jacobs 2010a). Because of their

importance in the aquatic food web, the viability of benthic invertebrates in the Emory, Clinch, and Tennessee Rivers was included as an assessment endpoint.

Based on this assessment endpoint, the ecological risk question established for the evaluation of benthic invertebrates at the Site is as follows:

- Do concentrations of ash-related COPECs in surface water, submerged sediment, sediment porewater, or shallow seasonally-exposed sediment pose unacceptable risks to benthic invertebrate communities?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective must be evaluated in the WOE assessment. This risk question helps focus subsequent steps of the benthic invertebrate assessment, including the selection of measurement endpoints.

4.1.2 Measurement Endpoints

The measurement endpoints for benthic invertebrates selected for the BERA are as follows:

Measures of exposure:
- COPEC concentrations in surface waters (total and dissolved)
- COPEC concentrations in sediments
- COPEC concentrations in porewaters and groundwaters
- COPEC concentrations in invertebrate tissues.

Measures of effects:
- Literature-derived toxicity data
- Laboratory toxicity tests with invertebrates and site sediments
- Biological surveys of site benthic invertebrates – relative abundance, diversity, and taxa richness.

4.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to benthic invertebrates. The LOEs used to evaluate benthic invertebrates include:

- Abiotic media evaluation (i.e., surface waters, porewaters/groundwaters, and sediment chemistry data)
- Invertebrate tissue data
- Sediment toxicity tests
- BIC surveys.

The information obtained from each individual LOE was qualitatively combined in an overall WOE approach. Data on water, sediments, and porewater chemistry and other (physical) characteristics were collected to assess chemical composition and to document other factors that could influence the distribution and abundance of benthic species (e.g., TOC, grain size). The results of toxicity tests and evaluation of tissue data provide information to evaluate the potential bioavailability and toxicity of the COPECs present. Measures of *in situ* biological effects, such as benthic community structure provide information on actual alterations of resident communities that may be related to sediment chemistry and/or general habitat conditions. Integration of these three components provides comprehensive information which may be used to evaluate and rank the priority of the areas that have been surveyed (Burton et al. 2002; Chapman 1996; Chapman and Anderson 2005; Chapman et al. 2002). Collection locations are shown in Figures 4-1 through 4-9. Each LOE is summarized below.

4.2.1 Community Surveys

Assessments of the composition and structure of benthic invertebrate communities can provide direct evidence of the effects of sediment constituents/ general habitat conditions on naturally-occurring communities. Deviations from expected community characteristics (as may be demonstrated by statistical comparisons with reference area conditions) may be due to the presence of chemical constituents. However, they may also be due to other factors (e.g., sediment grain size and organic content) unrelated to observed levels of COPECs. Therefore, it is essential to make comparisons with the benthic invertebrate communities in reference areas with similar sediment characteristics except for the presence of site-related COPECs.

Prior to the ash release at KIF, the benthic invertebrate community was evaluated as part of TVA's Valley-Wide Vital Signs Monitoring Program. This monitoring occurs in 31 reservoirs; however, only the Tennessee River was included in pre-ash release monitoring. Therefore, there are no historical invertebrate baseline data for the Emory and Clinch Rivers near the Site prior to the ash release. All invertebrate data in the Emory and Clinch Rivers were collected post-release in 2009 and continued each year, along with the already monitored Tennessee River. Following the release in 2008, benthic invertebrates were collected in 2009 and 2010 to evaluate potential impacts from ash and ash-related COPECs in the Emory, Clinch, and Tennessee Rivers. Because the 2009 data were collected before the initial dredging was completed, they are more representative of the immediate impacts of the release, rather than the long-term, residual effects. Therefore, the BIC data collected in 2010 were the primary data evaluated to assess potential site-related effects from the 2008 TVA Kingston ash release. The benthic community and corresponding substrate data (including ash) and sediment chemistry data collected in 2010 are the most recent and complete information available to evaluate the recovery of the benthic community from the Kingston ash release. Appendix M includes a detailed temporal comparison between the 2009 and 2010 results.

4.2.1.1 Analysis of Exposure and Effects

Methods

The sampling and data assessment methods used for this study are described below. Greater detail on the sampling methodology can be found in Appendix M.

Sampling Methods

A standard ponar or Peterson grab sampler was used to collect the top 6 inches of bulk sediments along transects in the Emory, Clinch, and Tennessee Rivers (Figure 4-10). Each sample was washed through screens to separate benthic organisms from sediments and debris, and the BIC samples collected were used to evaluate potential site-related effects from the TVA Kingston ash release. In total, 187 samples were collected from 16 transects. Each transect was positioned perpendicular to the shore and ran across the entire wetted width of the channel (Figure 4-10). In most cases, a total of 10 samples were collected at equally-spaced intervals along each transect. In a few cases, more or fewer samples were collected. (Table 4-1) Some transects were considered representative of "site" conditions (adjacent to or downstream of site influences); others were representative of "reference" conditions uninfluenced by the release (Table 4-1).

Upon collection, the bulk sediment samples were washed through screens to remove finer materials; the remaining material was transferred to jars, fixed with 10 percent buffered formalin solution, and submitted for laboratory processing and identification of organisms to the lowest practical taxonomic level (typically genus or species). At each sample location, water depth was measured, and a physical description of the sediment in the sample was characterized by a minimum of two team members that reached agreement. This included estimates of percent ash presence, as well as quantification of substrate (i.e., habitat) composition (i.e., grain size, woody debris, detritus, relict shells, and coal). These data are co-located with the BIC data and were collected along each transect shown in Table 4-1 and Figure 4-10. As such, these data should be useful in understanding the potential effects of ash to the BIC versus the potential effects of different habitat variables.

In 2010 sediment chemistry, grain size, TOC, and ash samples were collected during separate sampling programs at nearby locations and submitted for laboratory analyses as described in Section 2.5.4 Abiotic Media and Appendix M. Sediment chemistry, grain size, TOC, and ash samples were collected as part of the VibeCore™ sampling of submerged sediments and as part of the bulk sediments collected for laboratory toxicity tests. Sediment chemistry samples were analyzed for metals and metalloids (BIC compared to total metals/metalloids; see Section 2.5.4 for additional metal analyses), PCBs, PAHs, and pesticides. Figures 4-11 through 4-14 show, in relation to the BIC transects, the submerged sediment sampling locations, the laboratory toxicity test sample locations, and the number of locations where sediments were analyzed for percent ash, respectively. Collectively, these non-co-located data provide "area-wide"

information to aid in the interpretation of the benthic data and potential factors that might influence the BIC. Additional benthic community and sediment chemistry data were collected in January 2012 to further aid in the interpretation of potential factors that might influence the BIC. These data were collected from the same sampling locations (i.e., co-located), and may provide a more direct measure of potential COPEC-related effects to the BIC. Preliminary findings of the 2012 benthic community data are presented in Section 4.3.2.1 of the uncertainty section.

Assessment Methods

BIC data were described using summary statistics and analyzed using standard multi-metric procedures. Metrics are measures used to quantify aspects of community structure and function that change in predictable ways with increased human influence and/or perturbation (Barbour et al. 1995). Metrics also provide a consistent theoretical framework for analyzing complex assemblage data. Because multi-species BIC data may be complex, several metrics are typically used to characterize communities (USEPA 1999c; 2006). USEPA (1999c; 2006) supports the use of metrics that are best suited to evaluate anticipated effects, including measures of benthic abundance, diversity, tolerance, and life history adaptations.

BIC metrics were compared between site and upstream reference transects to evaluate potential site-related effects from the TVA Kingston ash release. Metrics were selected based on the following. (Note that the analysis was not designed for metric calibration, index development, or threshold establishment.)

- Availability in regulatory protocols. Many states have well-developed stream protocols, although fewer have BIC assessment protocols for large rivers (USEPA 2006) that would be applicable to the Emory, Clinch, and Tennessee Rivers. This appears to be true for the state of Tennessee, which has stream protocols (TDEC 2011), but does not have a companion manual for large rivers. However, five of the six semi-quantitative metrics used in Tennessee's stream protocol were coincidentally determined to be useful in this assessment.
- Use of metrics by other organizations or in the scientific literature. TVA has conducted numerous water quality studies that also include sampling and analysis of the BIC. Metrics have been selected by TVA for the Emory, Clinch, and Tennessee Rivers. Other studies are cited as applicable.
- Applicability to site conditions. Metrics appropriate for use should reflect the type of BIC present and its associated habitat. The metrics selected for use should also be sensitive to potential site-related COPECs. In this case, metals and metalloids appear to be important ash-related COPECs.
- Balanced representation of community structure. USEPA (1999c) supports the use of metrics that are best suited to evaluate anticipated effects, including measures of benthic invertebrate abundance, diversity, tolerance, composition, and life history adaptations (feeding and habit guilds).

The metrics selected for this evaluation, based on the parameters just discussed, are provided below with a brief explanation regarding their interpretation. In general, these are commonly-used standard measures found in the scientific literature. Table 4-2 provides examples of selected citations regarding the use of these metrics, including support for those that are sensitive to metals concentrations. Table 4-2 also lists the expected responses of the metrics to increased disturbance and/or perturbation.

- **Number of Organisms** – Count of the individuals in the sample. Used as the denominator for several other metrics and also useful for observing differences in the number of organisms between different locations.
- **Number of Taxa** – Total number of distinct taxa per sample. Measures the overall variety of the species assemblage. Taxa richness is an important component in most evaluations of invertebrate integrity. The underlying reason is the basic ecological principle that healthy, stable biological communities have high species richness.
- **Number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) Taxa** – Total number of distinct Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa per sample. Many EPT species are sensitive to changes in water quality and will disperse under environmental disturbance or human perturbation. This metric is most applicable in higher gradient streams, where many EPT reside (coarse substrate), but is useful here due to the presence of the burrowing mayfly (*Hexagenia*) whose larval stage lives in soft substrates.
- **Shannon Diversity** – Measures the diversity of the sample in terms of number of taxa and relative species abundance. Scores for this metric typically range from 0 to 4 with larger numbers indicating greater diversity.
- **Tolerance Index** – Calculated measure to evaluate effects of organic pollution based on abundance and tolerance values ascribed originally by Hilsenhoff (1987) but adapted by TDEC (2011) based on NCDENR (2011). The tolerance index has also been found to be sensitive to the effects of impoundments, heated discharges, and chemical pollution (Hilsenhoff 1998). Scores for this index typically range from 0 (intolerant taxa) to 10 (tolerant taxa).
- **Percent 2 Dominant Taxa** – Measures the dominance of the two most abundant taxa, with often greater representation and relative abundance of tolerant invertebrate taxa in disturbed conditions.
- **Percent Oligochaetes** – The relative percent contribution of aquatic worms. A composition metric that provides information on the makeup of the assemblage and relative contribution to the total BIC.
- **Percent Chironomids** – The relative percent contribution of midge larva. A composition metric that provides information on the makeup of the assemblage and relative contribution to the total BIC.
- **Feeding Guild** – Functional feeding group measure that identifies feeding strategies based on food type and availability. Feeding groups can become skewed and unbalanced based on changes to environmental conditions (USEPA 1999c). Guilds will be presented in the evaluation; however, emphasis is placed on those that are more sensitive to change (shredders).

- **Organism Habit** – Mode of existence based on differing adaptations for movement and maintaining position. Similar to feeding guild in that this metric can become skewed and unbalanced based on changes to environmental condition (USEPA 1999c). Habits will be presented in the evaluation: however, emphasis is placed on those that are more sensitive to change (clingers).

Metric results were compared between site and reference transects using summary statistics and hypothesis tests to evaluate potential site-related effects from the TVA Kingston ash release. Differences in BIC composition between rivers (Figures 4-15 and 4-16) precluded pooling of data, and as such, analyses were performed on individual rivers. Differences in the community composition between rivers may be due to differences in benthic habitat (especially smaller substrate sizes and increased effects of thermal stratification on dissolved oxygen as habitat becomes more lacustrine) and fluvial geomorphology.

Hypothesis tests were performed using the Kruskal-Wallis test (data were predominantly non- normal), a non-parametric analogue to one-way ANOVA, and the Wilcoxon Rank Sum pairwise comparison test (one-sided) to evaluate differences between each site transect and its corresponding reference transect. (Note: One upstream reference transect was sampled in each of the Emory and Tennessee Rivers; two upstream reference transects were sampled in the Clinch River. These data [CRM 6.0 and CRM 8.0] were combined to aid in the statistical comparisons.)

Other tests were performed to support the multi-metric assessment. This included comparing the BIC data to co-located physical data (water depth, substrate, and ash), and non-co-located physical and chemistry data (chemistry, substrate, TOC, and ash), as applicable, to evaluate the effects of these factors on the BIC. This was done by using similarity analysis to assess the correlation between BIC data and physical/chemical data (Clarke and Gorley 2006), and by linear regression analysis to evaluate the relationship between the BIC metric data and co-located ash data alone. Geospatial statistics were used to assign non-co-located physical and chemistry data to individual BIC sampling locations.

Results

The initial BIC sampling in January 2009 was limited due to the ash that filled the Emory River channel nearest the Site, as well as to the limited access to locations between ERM 4.0 and ERM 2.0 that were blocked during the dredging period. In December 2009, two sites were added at ERM 2.2 and 4.1, and in December 2010 both ERM 2.6 and 3.0 were also added. Substrate before and after the spill consisted of mostly fine sized material with transects farther from the site consisting of heterogeneous substrate types, as well as varying channel morphologies.

Results of the 2009 BIC sampling indicate that the only discernable effect of the ash spill on the BIC was the initial release and subsequent covering of the BIC that occurred when ash filled the Emory River channel nearest the Site. Results of the 2010 BIC sampling indicate that not only are these organisms present in

areas of the Emory, Clinch, and Tennessee Rivers affected by the ash release, but the BIC composition and structure in these areas is generally similar to those found at upstream reference locations, which were un-impacted by the release of ash. Given the magnitude of the ash release, these results over time appear to be promising, and suggest that the BIC is in the process of recovering. However, some potential for limited site impairment does appear to exist. Appendix M includes a detailed temporal comparison between 2009 and 2010 results. The details provided below focus on spatially comparing 2010 BIC survey results and drawing any relationships to sediment chemistry and physical results.

Benthic Habitat

Characterization of the benthic habitat in the Emory, Clinch, and Tennessee Rivers at each of the sampling transects is presented in Table 4-3. This includes measures of water depth and visual estimations of percent ash, grain size, and organic material in the grab samples, or as encountered during sampling (bedrock). Grain sizes were categorized as bedrock, gravel, sand, fines, and hard pack clay. Organic material consisted of woody debris, detritus, shells, or coal-gravel.

The Emory, Clinch, and Tennessee Rivers are relatively large rivers with deep, slow-moving waters. Mean transect water depths in each river ranged from 10 to 30 ft in the Emory River, 13 to 31 ft in the Clinch River, and 26 to 40 ft in the Tennessee River (Table 4-3). Light penetration to bottom substrates appears to be minimal, as no aquatic vegetation was observed in the grab samples. In the three rivers, the predominant substrate type and benthic habitat was fines, which are unconsolidated silts and clays (Table 4-3). Sand and detritus were the next most prevalent, natural substrate types. The relative abundance of other substrate types varied in lesser amounts, with some differences between rivers (e.g., more clay and coal-gravel in the Emory River; more bedrock and mollusk shells in the Clinch River; and less detritus in the Tennessee River).

Benthic habitats between site and upstream reference transects within each river were generally similar (Table 4-3). This allows for stronger inference and better interpretation of the data when comparing site and reference communities. However, it is recognized that due to the location of the Kingston site, which is just upstream of the confluence of the Emory, Clinch, and Tennessee Rivers, and also the Watts Bar Dam, there could be some influence to the BIC due to other habitat variables. Specifically, the upstream reference transects are typically located in more channelized portions of the rivers, with smaller cross-sectional areas, while the downstream site transects are often located in more impounded, lake-like areas, with larger cross-sectional areas (Figure 4-1) (Appendix M). As such, substrates tend to be somewhat more coarse upstream and less coarse downstream, particularly in the Emory and Clinch Rivers (Table 4-3). These differences could not be controlled for given the physical nature of the Site.

The presence of ash varied between transects, but overall, the amount of ash observed in the grab samples generally decreased from upstream to downstream (Table 4-3). Mean percent ash in the site transects ranged from 0 to 31 percent (18 percent median) in the Emory River, 5 to 32 percent (9.5 percent median) in

the Clinch River, and 0 to 16 percent (8 percent median) in the Tennessee River. No ash was observed in the upstream reference transects.

Benthic Invertebrate Community

BIC data (taxa and counts) are presented in Table 4-4. Mean percent abundance results of higher order taxa are presented in Table 4-5, and are shown graphically on Figure 4-17. Results show a BIC that is typical of finer-grained large river sediments. Of the 15 major taxonomic groups represented in Table 4-5, four groups represent the majority of the overall taxa in the Emory, Clinch, and Tennessee Rivers. These are the dipterans (fly larva, including chironomids), oligochaetes (aquatic worms), ephemeropterans (mayflies), and bivalves (mussels/clams). Dipterans and oligochaetes are often the dominant taxa in finer-grained sediments because they are adapted to live in this type of environment (Merritt et al. 2008; Thorp and Covich 2001). Certain taxa of ephemeroptera and bivalves are also adapted to burrow or reside in finer-grained sediments, such as burrowing mayfly and *Musculium* (fingernail clam). These are the two most abundant genera of mayflies and bivalves in each of the three rivers.

Although the four major taxonomic groups that represent the majority of the overall benthic invertebrate taxa in the Emory, Clinch, and Tennessee Rivers is the same, the relative contribution of each of these groups is different for each river as shown below.

- **Emory River** – 99 percent total contribution of: dipterans (66 percent), oligochaetes (28 percent), ephemeroptera (3 percent), and bivalves (2 percent).
- **Clinch River** – 96 percent total contribution of: dipterans (44 percent), oligochaetes (30 percent), ephemeroptera (13 percent), and bivalves (9 percent).
- **Tennessee River** – 97 percent total contribution of: dipterans (41 percent), oligochaetes (9 percent), ephemeroptera (15 percent), and bivalves (33 percent).

Dipterans were comprised primarily of phantom midge (*Chaoborus*), biting midges (*Ceratopogonidae*), and several species of non-biting midges (*Chironomidae*). Oligochaetes were comprised primarily of aquatic worms and other tubificids (*Branchiura sowerbyi*). Ephemeropterans were comprised primarily of burrowing mayfly. Bivalves were comprised primarily of Asiatic clam (*Corbicula fluminea*), zebra mussel (*Dreissena polymorpha*), and fingernail clam (*Musculium transversum*) (Table 4-4).

As seen from above, the relative contributions of fly larvae and aquatic worms generally tends to decrease, and burrowing mayfly and bivalve contribution generally tends to increase in the downstream direction. The different relative contributions of these four major taxonomic groups explains, in part, the differences observed between BICs between rivers (Table 4-5 and Figures 4-15 and 4-16), and supports the idea that data from each river should be evaluated separately. The reason for the different BIC composition between rivers may be due to differences in benthic habitat and fluvial geomorphology, as mentioned previously.

Other taxonomic groups present in the Emory, Clinch, and Tennessee Rivers occur in much lesser amounts (typically a fraction of 1 percent), and include the following: dragonfly/damselfly (*Odonata*), stonefly (*Plecoptera*), alderfly (*Megaloptera*), caddisfly (*Trichoptera*), beetle (*Coleoptera*), mite (*Acariform*), scud (*Amphipoda*), leech (*Hirudinea*), roundworm (*Nematoda*), and snail (*Gastropoda*) (Table 4-5). Although the relative abundance of organisms in these groups is comparatively small within each river, they each contribute to the organization of BIC structure and function.

Benthic Invertebrate Community Metrics

BIC metric results are presented in Table 4-6 and are shown graphically on Figures 4-17 through 4-22). Multi-metric analyses on the BIC data confirm the fact that benthic organisms have recolonized areas in the vicinity of the ash release. Analysis also shows that most metric results were generally within the range of upstream reference conditions, and that results between rivers were generally similar (i.e., no large disparity was observed in metric scores between rivers). Although the potential for site-related impairment may exist, the data suggest that impacts may be limited.

Emory River

Of the ten metrics evaluated, almost all results for the site transects were similar to the upstream reference transect results. In total, results for nine of the ten metrics at each of the six site transects were similar to or fared better than the upstream reference transect. These were number of organisms, number of taxa, number of EPT taxa, Shannon diversity, tolerance index, percent two dominant taxa, percent chironomids, percent shredders, and percent clingers. Only percent oligochaetes at transects ERM 1.0 and 3.0 (but not the other four site transects) were statistically different from upstream reference ($p < 0.10$ one-sided ANOVA; Table 4-7). This may suggest the potential for some site-related impairment, but as indicated below, it may also indicate other potential factors as no relationship between percent oligochaetes and ash was observed.

Although the majority of metric results were similar between site transects and the upstream reference transect, some metrics showed a decreasing relationship with increasing ash. Specifically, the number of organisms and the number of taxa in the Emory River site transects showed a statistically significant decrease with increasing ash ($p < 0.10$ linear regression; Table 4-8, Figure 4-23). These results suggest the potential for some site-related impairment. However, it should be noted that the r-square values for these were very low ($r^2 = 0.058$ and 0.051, respectively), which indicates that ash explains only a small amount of the variance. Furthermore, the results of these two metrics, although showing a decline with increasing ash, were within the range of the upstream reference transect results and were not statistically different as indicated in the hypothesis tests above. It should be noted that none of the other eight BIC metrics showed a consistent relationship with ash. This indicates that site-related impacts, if present, are limited in the Emory River.

Clinch River

Results of the multi-metric evaluation for the Clinch River were more variable. Four of the ten metrics evaluated for the site transects were statistically different from the upstream reference transect, while six of the ten metrics at the Site were similar to the upstream reference transect ($p < 0.10$ one-sided ANOVA; Table 4-9). The four metrics that were statistically different from reference were number of organisms, number of taxa, percent shredder, and percent clingers. The six metrics that were comparable between location and reference were number of EPT taxa, Shannon diversity, tolerance index, percent two dominant taxa, percent chironomids, and percent oligochaetes.

These results suggest that some potential site impairment may be occurring. However, it is not clear if the results are due to the presence of ash (and associated COPECs) or due to differences in habitat. For example, the upstream reference in the Clinch River had the most gravel and least amount of fines of any transect, which could influence the BIC (Table 4-3). Additionally, results of the similarity analysis, which are presented in the next section, suggest that the BIC assemblage in the Clinch River is correlated more to substrate type and water depth than percent ash composition for co-located data. In fact, no trends were apparent between the metrics and ash data for the Clinch River, and as such, linear regression analysis was not performed.

Tennessee River

Results of the multi-metric evaluation for the Tennessee River were also mixed. Four of the ten metrics calculated for transect TR 560.8 (number of organisms, number of taxa, tolerance index, and percent shredder), and two of the ten metrics calculated for transect TR 566.3 (number of EPT taxa and tolerance index) were statistically different from the upstream reference transect ($p < 0.10$ one-sided ANOVA; Table 4-10). The remaining metrics at these transects were similar to the upstream reference transect.

These results suggest that some potential site impairment may be occurring. However, it is not clear if the results are due to the presence of ash (and associated COPECs) or due to differences in habitat for transect TR 566.3. For transect TR 560.8, the results may be due solely to differences in habitat as no ash was observed at this transect. Results of the similarity analysis, which are presented in the next section, suggest that the BIC assemblage in the Tennessee River is correlated more to substrate type and water depth than percent ash composition for co-located data. No trends were apparent between the metrics and ash data for the Tennessee River, and as such, linear regression analysis was not performed.

Feeding and Habitat

The predominant benthic feeding guilds in both the location and upstream reference transects in the Emory, Clinch, and Tennessee Rivers were gatherers (e.g., burrowing mayflies, oligochaetes, and chironomids) and

predators (e.g., biting and phantom midges), with increasing amounts of filterers (e.g., bivalves) in the Clinch River, and an even greater increase of filterers (e.g., bivalves) in the Tennessee River (Table 4-6; Figure 4-24). The predominant habits displayed by organisms were burrowing and sprawling (organisms that reside on top of sediment matrix) (Table 4-6; Figure 4-24). No statistical tests were conducted on these metrics as the expected response of these to perturbation is often variable (USEPA 1999c).

Similarity Analysis

Similarity analysis on co-located data indicates that differences in the BIC data (taxa and counts; Table 4-4) are correlated more to substrate type and water depth (Table 4-3) than percent ash composition. Similarity analysis is performed on the raw taxa and count data to create resemblance matrices, and is separate from performing multi-metric analyses as presented above. That said, results tend to correspond, and as indicated, both similarity analysis and multi-metric evaluation results showed little relationship between the BIC data and percent ash composition. Comparison between physical (substrate/water depth/ash) and BIC resemblance matrices using Spearman rank correlation showed agreement in multivariate pattern for each river. Results are presented below.

- **Emory River** – The highest correlation for the variables was a Spearman correlation coefficient (r) = 0.562 for water depth, gravel, sand, hard pack clay, fines, detritus, and coal-gravel. The best fit for a single variable was for hard pack clay at r = 0.395. Ash was not highly correlated to the benthic community at r = –0.090.
- **Clinch River** – The highest correlation for the variables was r = 0.568 for bedrock, sand, hard pack clay, and mollusk shells. The best fit for a single variable was for bedrock at r = 0.417. Ash was not highly correlated to the benthic community at r = –0.139.
- **Tennessee River** – The highest correlation for the variables was r = 0.700 for sand, hard pack clay, fines, mollusk shells, and ash. The best fit for a single variable was for fines at r = 0.587. Ash was not highly correlated to the benthic community at r = 0.124.

Similarity analysis on non-co-located data indicated mixed results, with differences in the BIC data (taxa and counts; Table 4-4) being weakly correlated to non-co-located physical variables (including ash), chemical variables, or both. Geospatial analyses (nearest neighbor [NN] and inverse distance weighted [IDW]) were used to assign non-co-located physical and chemistry data to individual BIC sampling locations. Note that the lack of co-located data confounds the similarity analysis, and results should be interpreted cautiously. This is true particularly in the instances where the same estimated value per variable/COPEC would apply to many benthic locations or over the entire transect.

Nearest Neighbor

NN analysis (Thiessen polygons) was conducted to select the closest sediment sample location to each BIC sampling location for comparison (Figures 4-25 through 4-30). The analyses presented here focus on barium, mercury, and zinc because no relationships or only weak relationships were found for other constituents when compared to the benthic invertebrate community data.

- **Emory River** – The highest correlation for the variables was r = 0.335 for gravel, which is also the best fit for a single variable. Ash was not highly correlated to the BIC at r = 0.064.
- **Clinch River** – The highest correlation for the variables was r = 0.441 for mercury, zinc, gravel, and silt. The best fit for a single variable was for gravel at r = 0.336. Ash was correlated to the BIC at r = 0.208.
- **Tennessee River** – The highest correlation for the variables was r = 0.310 for barium, zinc, and ash. The best fit for a single variable was for zinc at r = 0.286. Ash was the next highest correlated single variable to the BIC at r = 0.270.

Inverse Distance Weighted

IDW was conducted to estimate the chemical and physical spatial gradients along the entire river for comparison to the benthic community data (Figures 4-31 through 4-37).

- **Emory River** – The highest correlation for the variables was r = 0.408 for gravel, which is also the best fit for a single variable. Ash was correlated to the BIC at r = 0.181.
- **Clinch River** – The highest correlation for the variables was r = 0.410 for copper, strontium, and gravel. The best fit for a single variable was for strontium at r = 0.358 (silt was the next best fit at r = 0.349 followed by gravel and sand). Ash was correlated to the BIC at r = 0.196.
- **Tennessee River** – The highest correlation for the variables was r = 0.466 for barium, strontium, and molybdenum. The best fit for a single variable was for barium at r = 0.316 (arsenic was the next best fit at r = 0.312 followed by silt and ash). Ash was correlated to the BIC at r = 0.307.

4.2.1.2 Risk Estimation

Collectively, the results of the 2010 BIC sampling in the Emory, Clinch, and Tennessee Rivers indicates that these organisms are present in areas that were affected by the Kingston ash release, and that their community structure and function in these areas is generally similar to that found at the upstream reference transects which were un-impacted by the release of ash. These results are promising and suggest that the BIC is in the process of recovering. It is also evident that habitat plays an important role in organizing the BIC at this complex site, as river conditions transition from channelized upstream areas to impounded lake-like areas downstream. Although the potential for some site-related impairment may be present based on the results of the multi-metric, linear regression, and similarity analysis, by and large, the BIC data in the vicinity

of the Site are favorable and are indicative of a community that might be expected to inhabit finer-grained sediments of a large river system.

A bullet list summarizing the findings is presented below.

- **Habitat** – The Emory, Clinch, and Tennessee Rivers are relatively large rivers with deep, slow-moving water. The predominant substrate type and benthic habitat was fines. The upstream reference transects were located in more channelized portions of the rivers, with smaller cross-sectional areas, while the downstream site transects were often located in more impounded lake-like areas, with larger cross-sectional areas. These differences could not be controlled, given the physical nature of the Site.
- **Benthic Community** – BIC composition between location and reference transects was similar. Four major BIC groups represent the majority of the overall taxa in the Emory, Clinch, and Tennessee Rivers. These are the dipterans (fly larva, including chironomids), oligochaetes (aquatic worms), ephemeropterans (burrowing mayflies), and bivalves (mussels/clams). These organisms are adapted to live in finer-grained sediments.
- **Benthic Metrics** – Most BIC metric results for the Site were within the range of upstream reference conditions, and results between rivers were generally comparable. Almost no statistical differences in metric results were noted for the Emory River between site and reference transects. However, some statistical differences were noted for the Clinch and Tennessee Rivers. Habitat differences appear to be the primary reason for these results as indicated in the similarity analysis.
- **Linear Regression** – Two of ten metrics appeared to decrease with increasing ash concentrations for the Emory River, but the correlations were weak. Furthermore, the results of these two metrics were still within the range of the upstream reference transect results and were not statistically different. No trends were apparent between the metrics and ash data for the Clinch and Tennessee Rivers.
- **Similarity Analysis** – Analysis of co-located data indicated that differences in the BIC data are correlated more to substrate type and water depth than percent ash composition. Analysis of non-co-located data indicated mixed results, with differences in the BIC data being weakly correlated to non-co-located physical variables (including ash), chemical variables, or both. The lack of co-located data confounds the analysis; results of this analysis should be interpreted cautiously.

4.2.2 Sediment Toxicity Tests

Laboratory sediment toxicity testing provides an important LOE for evaluating potential risks to benthic invertebrates. Evaluation of bulk sediment concentrations alone does not take into account potential site-specific physical and chemical conditions that may affect the bioavailability and ultimate toxicity of sediment-associated COPECs. Whole sediment toxicity testing attempts to integrate the effects of all sediment conditions, including the interactions (e.g., synergism, additivity, and antagonism) of various chemicals under controlled laboratory conditions. To the extent that laboratory tests with standard test

organisms simulate *in-situ* conditions and responses of indigenous benthic species, in conjunction with chemical analyses of the sediments they can provide insights into potential or observed real-world effects.

The objective of these tests, as stated in the SAP (Jacobs 2010a) was "to estimate the bioavailability and risk relative to the presence of ash-related constituents." Toxicity tests were conducted for the Emory and Clinch Rivers to address the following questions pertaining to risk:

- Are COPECs bioavailable?
- Do COPEC concentrations related to fly ash cause any adverse effects?
- Do either physical or chemical properties of ash show relationships to observed effects in test organisms?

The results of the toxicity tests were analyzed along with sediment chemistry data to evaluate any potential relationships that might exist between ash-related COPECs and toxicity.

4.2.2.1 Analysis of Exposure and Effects

Toxicity Test Methods

Sediment toxicity tests were conducted with three species, *Ceriodaphnia dubia, Hyalella azteca*, *and Chironomus dilutus.* Each of these species evaluates a different benthic exposure. For example, *H. azteca*, is an amphipod that lives primarily at the sediment/water interface and is often found grazing on vegetative matter; it sometimes burrows into the upper few millimeters of sediment. It is considered a sensitive species that has been shown to be a good indicator of effects from chemical and physical properties of sediment. *C. dilutus* is a species of midge in which the larval stage organisms burrow in sediments and represent potential dietary and respiratory exposures to sediments and interstitial waters. In contrast, *Ceriodahnia dubia* is a species of water flea frequently used in water and effluent toxicity testing. Inclusion of this species allowed for evaluation of potential toxicity associated with the sediment-water interface. Toxicity tests were conducted in accordance with USEPA and ASTM International (formerly American Society for Testing and Materials) toxicity test guidelines. The toxicity testing approach relied on a combination of short-term chronic and longer duration tests as summarized in Figure 4-38. The results of these tests, combined with the surface sediment chemistry results, provide an indication of the potential toxicity resulting from ash-associated COPECs.

C. dubia were evaluated by Environmental Enterprises USA, Inc., in a 3-brood static renewal test (ASTM E 1706-05 Annex A2) measuring effects on survival and reproduction. Organisms were exposed to sediments from 18 locations, including eight site locations in both the Emory and Clinch Rivers, as well as one reference location associated with each river. For each location, a series of dilutions were evaluated, with organisms exposed to 100, 80, 60, 40, 20, and 10 percent dilutions of site sediment with the appropriate river reference sediment. Water from the respective river reference was used as overlying water for each

test. Details regarding the testing methodology and results are summarized in (Appendix N). Results from the *C. dubia* toxicity tests found no significant differences on survival or reproduction in samples from either river site sediment samples relative to the river reference treatment (Appendix N).

H. azteca and *C. dilutus* were evaluated using a tiered approach. Initially, each species was evaluated in 10-day whole sediments tests (USEPA Methods 100.1 and 100.2; USEPA 2009) evaluating survival, growth, and biomass. For each species, eight site sediments and one reference sediment sample per river were evaluated (Figure 2-17). Based on the results of these screening tests (Appendix O [Emory River] and Appendix P [Clinch River]), a subset of locations was selected for longer term tests (USEPA Method 100.1 modified for duration and *H. azteca* age at test start, USEPA Method 100.5 modified for test duration and endpoints for *C. dilutus*; USEPA 2009). Of the four samples per species selected from each river for longer term toxicity testing, at least one was selected from among the locations where effects were not observed in the 10-day screening tests, and three were selected from among locations for which effects were observed. As with *C. dubia*, these additional tests evaluated exposures to 100, 80, 60, 40, 20, and 10 percent dilutions of site sediment with the appropriate river reference sediment. For the longer term tests, both *H. azteca and C. dilutus* tests were evaluated statistically to yield NOEC, LOEC, and IC_{25} (25 percent inhibition concentration) estimates based on percent site sediment.

For *C. dilutus*, endpoints of the long-term test were growth (ash-free dry weight), biomass, survival, and emergence. Survival was evaluated at the end of the test and in four random replicates per treatment on day 20. For the *H. azteca* test, the endpoints were 28-day growth, biomass, and survival.

Results

The endpoints of growth, biomass, survival, and emergence were all included in statistical evaluations; however, for the purpose of this risk assessment, growth is excluded from the following discussion. While useful as a measurement endpoint in toxicity tests, growth only provides evidence of individual organism effects. Biomass offers the best estimate of effects on the tested population because this measurement is based on the number of original organisms exposed to the sediment and combines the survival and growth results. Biomass, survival, and emergence all lend themselves well to estimating risks based on population effects.

Toxicity test results were evaluated statistically using ToxCalc (Appendix O and P). For the endpoints, a treatment was considered statistically significant when significantly lower ($p \leq 0.05$) than what was observed in the reference sample from the river being tested. Some treatments resulted in greater survival, biomass, or emergence relative to the reference (show as a negative percent reduction value in the exhibits below). A summary of the results for each river is provided below.

Toxicity test results for the Emory River are presented in Tables 4-11 through 4-14, the Clinch River toxicity test results are presented in Tables 4-11 and 4-15 through 4-17, and definitive test endpoints are summarized for both rivers in Tables 4-18 and 4-19.

Emory River

Results from the *C. dubia* toxicity tests found no significant differences on survival or reproduction in samples from the Emory River relative to the reference. Effects were observed in both *C. dilutus* and *H. azteca* at the Emory River locations evaluated. Effects on *C. dilutus* were observed in the 10-day screening tests for all ERM locations except ERM 1.0. Where effects were observed, biomass was always reduced; however, survival was only reduced in sediments from ERM 2.5 and 3.5. For *H. azteca,* biomass was significantly reduced in sediments from ERM 0.5, 2.5, 3.0, 3.5, 5.0, and 5.5. Survival was also reduced in sediments from ERM 2.5, 3.0, 3.5, and 5.5.

ERM 0.5 sediment was only tested with *C. dilutus* for long-term testing. This treatment resulted in a significant reduction in biomass at 20 days, and significant reduction for both survival and emergence at the termination of the test. The IC_{25} for *C. dilutus* survival was 71 percent ERM 0.5 sediment; for emergence it was 64 percent ERM 0.5 sediment.

ERM 1.0 sediment caused no significant effects in the 10-day tests and was included as the one sample not causing toxicity for longer term testing. However, in the long-term testing effects were observed for both *H. azteca* and *C. dilutus*. The observed effects on *H. azteca* was a reduction in biomass. *C. dilutus* had an observed effect on biomass at 20 days, and a reduction in survival and emergence at the termination of the test relative to the reference. The IC_{25} for *H. azteca* biomass was 77 percent site sediment, the IC_{25} for *C. dilutus* survival and emergence were 67 percent and 65 percent, respectively.

Of all Emory and Clinch River samples tested, ERM 2.5 sediment had the greatest effect on *H. azteca* survival and biomass in the 10-day tests (49 percent and 71 percent reductions, respectively). ERM 2.5 sediment had the greatest effect on *C. dilutus* survival and biomass in 10-day tests (29 percent and 58 percent reductions, respectively) relative to the reference sample. ERM 2.5 sediment also had the greatest effect on *H. azteca* survival and biomass in the longer term tests (55 percent and 78 percent reductions, respectively). *C. dilutus* experienced the greatest reduction inbiomass at 20 days, as well as a 72 percent reduction in emergence and survival. The IC_{25} values for *H. azteca* survival and biomass were 85 percent and 43 percent ERM 2.5 sediment, respectively. The IC_{25} for *C. dilutus* emergence and survival were 45 percent and 46 percent, respectively.

In screening tests, ERM 3.5 sediment had significant reductions to *H. azteca* survival (73 percent) and biomass (81 percent). This treatment also had significant reductions to *C. dilutus* survival (11 percent), and biomass (40 percent). Long-term definitive tests resulted in significant reductions of *H. azteca* biomass

(60 percent) and survival (38 percent). *C. dilutus* biomass was reduced after 20 days, and there was a 66 percent reduction in survival and emergence. The IC_{25} for *H. azteca* survival and biomass at ERM 3.5 was 92 percent and 38 percent site sediment, respectively. The IC_{25} for *C. dilutus* survival and emergence were 49 percent and 47 percent, respectively.

Exhibit 4-1. Emory River 10-Day Toxicity Test % Reductions Relative to the Reference

Endpoints for *Chironomus dilutus*	**ERM 0.5**	**ERM 0.8**	**ERM 1.0**	**ERM 2.5**	**ERM 3.0**	**ERM 3.5**	**ERM 4.0**	**ERM 5.5**
% Reduction – Survival	-1	4	1	29	-1	11	-1	-1
% Reduction Biomass	25	17	5	58	19	40	23	14
Endpoints for *Hyalella azteca*	**ERM 0.5**	**ERM 0.8**	**ERM 1.0**	**ERM 2.5**	**ERM 3.0**	**ERM 3.5**	**ERM 4.0**	**ERM 5.5**
% Reduction – Survival	1	-1	5	49	48	73	7	21
% Reduction Biomass	13	0	1	71	56	81	22	44

Note:
Shading denotes statistically significantly lower survival and/or biomass relative to the reference.
Negative values indicate greater survival and/or biomass relative to the reference.

Acronyms and Abbreviations:
ERM – Emory River mile

Exhibit 4-2. Longer Term Test % Reductions Relative to the Reference for 100% Sediment

Endpoints for *Chironomus dilutus*	**ERM 3.5**	**ERM 2.5**	**ERM 1.0**	**ERM 0.5**
% Reduction Biomass (20-day)	53	68	32	43
% Reduction Survival (20-day)	6	49	6	4
% Reduction PLC survival	66	72	50	38
% Reduction PLC emergence	66	72	50	38
Endpoints for *Hyalella azteca*	**ERM 5.5**	**ERM 3.5**	**ERM 2.5**	**ERM 1.0**
% Reduction Biomass	29	60	78	31
% Reduction Survival	0	38	55	5

Note:
Shading denotes statistically significantly lower survival, emergence, and/or biomass relative to the reference.

Acronyms and Abbreviations:
ERM – Emory River mile
PCL – partial life cycle

Sediments in the Emory River were more variable than those in the Clinch River. Varying amounts of sand, silt, and in some samples clay and gravel were noted among the samples. The reference was comprised of

sand and silt, whereas the site-related sediments had varying proportions of mostly silt and sand with some clay and gravel. The ash content of these samples ranged from 1 to 88 percent in site sediments (Figure 4-39).

Clinch River

No significant effects were associated with *C. dubia* at the Clinch River locations evaluated. With respect to the *H. azteca* and *C. dilutus* tests effects were observed in the sediments from CRM 1.5, CRM 2.0, CRM 2.5, and CRM 4.0 during the 10-day tests; however, no effects were observed at CRM 2.0, CRM 2.5, and CRM 4.0 in the longer term tests for either species. Effects were observed in the long-term tests in the sediments from CRM 1.5 and CRM 4.5. Specifically, *C. dilutus* exposed to CRM 1.5 sediment experienced a 32 percent reduction in survival and emergence; whereas, *H. azteca* experienced a 31 percent reduction of biomass and no effect on survival. The IC_{25} for *C. dilutus* emergence and survival were 87 percent and 90 percent site sediment, respectively, and the IC_{25} for *H. azteca* biomass was 62 percent. In sediments from CRM 4.5, a 15 percent reduction in *H. azteca* biomass was observed during 10-day exposures, and a 32 percent reduction in biomass during longer term test exposures. The IC_{25} for biomass was estimated at 66 percent site sediment.

Exhibit 4-3. Clinch River 10-Day Toxicity Test % Reductions Relative to the Reference

Endpoints for *Chironomus dilutus*	**CRM 0.0**	**CRM 1.5**	**CRM 2.0**	**CRM 2.5**	**CRM 3.0**	**CRM 3.5**	**CRM 4.0**	**CRM 4.5**
% Reduction - Survival	0	10	3	1	1	0	8	3
% Reduction Biomass	-5	1	15	9	-15	-5	34	4
Endpoints for *Hyalella azteca*	**CRM 0.0**	**CRM 1.5**	**CRM 2.0**	**CRM 2.5**	**CRM 3.0**	**CRM 3.5**	**CRM 4.0**	**CRM 4.5**
% Reduction- Survival	-5	-4	-5	-5	-1	-4	8	-5
% Reduction Biomass	-3	20	11	14	13	7	14	15

Note:
Shading denotes statistically significantly lower survival and/or biomass relative to the reference.
Negative values indicate greater survival and/or biomass relative to the reference.

Acronyms and Abbreviations:
CRM – Clinch River mile

Exhibit 4-4. Longer Term Test % Reductions Relative to the Reference for 100% Sediment

Chironomus dilutus	**CRM 4.0**	**CRM 3.0**	**CRM 2.0**	**CRM 1.5**
% reduction Biomass	-8	-44	-18	-22
% reduction Survival	-2	-7	2	-2
% reduction PLC survival	23	3	5	32
% reduction PLC emergence	23	7	5	32
Hyalella azteca	**CRM 4.5**	**CRM 3.0**	**CRM 2.5**	**CRM 1.5**
% reduction Biomass	32	12	14	31
% reduction Survival	-7	-7	-4	1

Note:
Shading denotes statistically significantly lower survival, emergence, and/or biomass relative to the reference.
Negative values indicate greater survival, emergence, and/or biomass relative to the reference.

Acronyms and Abbreviations:
CRM – Clinch River mile
PCL – partial life cycle

With respect to sediment conditions at the locations tested, there was little variation in sediment type, with the greatest difference occurring at the reference location. The reference was comprised of mostly sand, whereas the site-related sediments had similar proportions of mostly silt with some sand and clay. The ash content of these samples also did not vary greatly, with a range of 20 to 41 percent in site sediments (Figure 4-39).

Overall the effects observed in the Clinch River were relatively low. Statistically significant reductions in biomass and growth were noted during the 10-day test, but not always replicated in the longer term evaluations. In general very limited effects on survival were observed. Greatest effects appear to be associated with sediments from CRM 1.5 and CRM 4.5. In summary, potential risks to benthic invertebrates are higher in the Emory River than in the Clinch River, particularly at ERM 3.5 and ERM 2.5. As noted with *C. dilutus*, effects were most evident on sub-lethal endpoints such as biomass and emergence.

4.2.2.2 Risk Estimation

The objective of the sediment toxicity tests was to determine whether ash-related COPECs in the sediments may be causing measurable and significant effects on benthic invertebrates. The primary questions being addressed were:

- What concentration ranges of ash and ash-related COPECs cause adverse effects in laboratory organisms?

- In relation to the Emory and Clinch Rivers, what is the geographical location (i.e., reaches and river miles) and extent of the impacted sediments causing unacceptable risk to laboratory organisms?

The purpose of the following analysis methods were two-fold: to identify parameters that are most related to effects observed during the toxicity tests and to identify outliers not necessarily related to ash, but potentially contributing to the overall observed effects.

As a first step in evaluating the potential link between observed toxicity and the ash release, principal component analyses (PCAs) were used to evaluate the relationships between percent ash and metals concentrations as well as looking at percent ash, metals, and toxicity test results (Figures 4-40 through 4-44). To address potential correlations between sediment chemistry and observed toxicity, PCA was performed on the bulk sediment data, including but not limited to metal and metalloid COPECs, legacy constituents (i.e., PCBs and PAHs), and physical parameters (e.g., percent ash, TOC, and grain size). PCA is a multivariate technique in which a set of correlated variables are transformed, by matrix algebra, into a set of uncorrelated variables or principal components, providing a mechanism for evaluating the potential relationships between multiple variables in a dataset. It is often used to reduce the dimensionality of the dataset (i.e., to produce a smaller set of principal components that adequately represent the total variance of the dataset). The correlation of the original variables to the principal components (component loadings) can be examined to determine which variables drive the variance within each component. The values of each sample in the new coordinate space defined by the principal components (often referred to as factor scores) can be plotted and examined to determine what variables and covariance structures explain them. PCA can be used to identify outliers, samples that do not fit well within the multivariate structure of the dataset.

In addition to the PCA, those COPECs that were either related to ash or to toxicity test endpoints were evaluated using Pearson correlations (Tables 4-20 through 4-22). These correlations are a tool for investigating the relationships between two quantitative variables, whereas PCA looks at many variables at once. Pearson correlation coefficients (r) are measures of the strength of the association between the two variables, and r^2 is the proportion of explained variance. For the purpose of this analysis, emphasis was placed on Pearson correlation coefficients greater than 0.6 as an indication that the identified variables were likely related. The statistical significance of a correlation is based on the sample size, and shading in the correlation tables (Tables 4-20 through 4-22) indicates p-value < 0.05. Finally, scatter and probability plots (Figures 4-45 through 4-54) were used to evaluate the sediment sample results for outliers when comparing the COPECs to percent ash or to toxicity testing endpoints.

The PCA plots for the 10-day tests compared sediment physical and chemistry results to toxicity results and highlight the following:

- The 10-day Emory and Clinch River PCA plots combine mortality, chemistry, and ash results in order to observe any trends (Figure 4-40).
  - Aluminum, cobalt, iron, lead, magnesium, manganese, mercury, PAHs, PCBs, pesticides, potassium, TOC, and zinc are not correlated with percent ash, other metals, or toxicity endpoints. These COPECs are found at greater concentrations in samples with low concentrations of other metals and causing the least effects in toxicity tests.
  - Percent ash is most closely associated with arsenic and strontium, both of which are associated with mortality.

- The 10-day Emory River PCA plots combine toxicity test endpoints for percent reduction of biomass relative to the Emory River reference and chemistry and ash results (Figure 4-41).
  - Percent ash is mostly clustered and correlated with arsenic, boron, strontium, and toxicity test endpoints.
  - ERM 2.5 is found along the same position in the graph as the metals and has the highest metals concentrations and greatest effects on *H. azteca* and *C. dilutus*.

- The 10-day Clinch River PCA plots combine toxicity test endpoints for percent reduction of biomass relative to Clinch River reference and chemistry and ash results (Figure 4-42).
  - CRM 1.5 and the reference appear as outliers with other site sediments grouping together.
  - CRM 1.5 is positioned at the same location on the axis and correlates most with manganese, mercury, PAHs, and zinc. It does not appear to have a relationship with percent ash.
  - *H. azteca* biomass is clustered with and related to many metals and pesticides and less with percent ash.

The PCA plots for definitive tests compared variables for the Emory and Clinch together due to the low sample size of four out of eight original bulk sediment samples for each river. Each organism was examined separately as the subsets of samples selected for longer term testing were not the same for both organisms. COPECs such as PAHs, pesticides, PCBs, lead, mercury, manganese, and zinc were not closely associated with either ash or toxicity test endpoints and were not included in the longer term test plots. The longer term plots (Figures 4-43 and 4-44) demonstrated that:

- Longer term *H. azteca* and *C. dilutus* endpoints were correlated with metals in that the greater concentrations of metals translate to a lower percent site sediment required to elicit an effect.
  - Ash is most closely associated with arsenic and strontium.
  - CRM 1.5 appears again as an outlier, while ERM 2.5 is mostly associated with ash, arsenic, and strontium.

The Pearson correlations (Tables 4-20 through 4-22) compared those variables that appeared correlated to ash or toxicity test endpoints in the PCA plots. The strongest relationships from this evaluation were:

- Percent ash was found to correlate strongest with arsenic, barium, beryllium, boron, chromium, copper, selenium, strontium, and vanadium.
- Ash content also explained 25 percent to 90 percent of the variability (r^2) in toxicity test endpoints depending on the test and test endpoint.
- Arsenic was found to explain 19 percent to 92 percent of the variability in toxicity test endpoints. Of the arsenic species evaluated, arsenate appeared most closely correlated to toxicity test endpoints.
- Barium, beryllium, boron, chromium, selenium, strontium, and vanadium explain less of the variability in the toxicity testing.

A more detailed evaluation of the results specific to each river is provided below with particular focus on COPECs identified to correlate either with ash or toxicity test endpoints.

Emory River

In contrast to the Clinch River, the Emory River site sediments tested for longer-term testing had a measurable level of effect in either sub-lethal, lethal, or both endpoints. In addition, the Emory River toxicity results had strong relationships with percent ash and ash-related COPECs as shown in the Pearson correlations (Tables 4-20 through 4-22). The strongest relationships to toxicity were between percent ash and arsenic (Figure 4-45).

Specifically, the most pronounced effects on both organisms occurred in sediment samples from ERM 2.5 and ERM 3.5, which had approximately 88 and 64 percent ash, respectively. In addition, sediments from ERM 2.5, ERM 3.0, and ERM 3.5 were the only samples to exceed the consensus-based probable EC of 33 milligrams per kilogram (mg/kg) for arsenic (MacDonald et al. 2000); concentrations in sediment samples from ERM 2.5 and ERM 3.5 were actually twice as high (Figure 4-45). Throughout every analysis, these locations show the greatest relationships between toxicity and percent ash or ash-related COPECs, particularly percent ash and arsenic (Figures 4-40 through 4-44). An evaluation of AVS/SEM data found no correlations to toxicity test endpoints (Tables 4-20 through 4-22); however, these samples had relatively low TOC when compared to other Emory River samples, potentially indicating increased bioavailability (Figure 4-46).

Sequentially extracted metals and TOC data were evaluated for all bulk sediment samples collected. TOC may play a role of decreasing the overall bioavailability of metals (Horowitz 1985; Jackson et al. 1978). TOC concentrations provided further resolution to differences in toxicity among sites with the relationship of increasing toxicity and metal concentrations with decreasing TOC concentrations. Sequentially extracted metals provided resolution to determining the main COPECs related to ash and toxicity test results; the methods for analysis of the data and detailed information are presented in Attachment 2 of Appendix W.

In short, sediments from ERM 2.5 and ERM 3.5 had the highest total metals concentrations and also had the most metals available post-water extraction and in the pH 1.5 extractions (Table 4-23). The pH 1.5 extractions are most related to those metals that would be available in the gut of an organism for potential uptake. These results suggest that selenium, vanadium, and arsenic appear to desorb quite readily from ash. In contrast, strontium and calcium were less bioavailable when extracted from sediments containing higher ash content (Tables 4-23 and 4-24 and Figure 4-55). Copper exhibited a very slight increase in extracted metals concentrations with higher ash content. Barium, although related to ash and toxicity endpoints, showed no clear trends of bioavailability based on the sequentially extracted metals data.

The TOC and sequentially extracted metals data supported the conclusions drawn from the total metals, ash, and toxicity test results. Where sites had the highest total metals concentrations, they also had the greatest toxicities; lowest TOCs; and greatest bioavailability for selenium, vanadium, and arsenic. These results suggested arsenic and percent ash may explain toxicity results more than strontium, which was not as bioavailable. The TOC and sequentially extracted metals data, however, offered no clear resolution to the differences in toxicity test results observed for sites with low-to-mid-range percent ash or total metals concentrations.

The results of this analysis indicate the potential for effects, especially sub-lethal impacts (i.e., reduction in biomass and emergence) on benthic invertebrates in the Emory River as a result of the ash release. Effects significantly different from reference were observed at every location sampled and correlated to some extent with percent ash and ash-related COPECs.

Clinch River

As discussed above, effects observed in Clinch River samples were primarily associated with sediments from CRM 1.5 and CRM 4.5. Specifically, *C. dilutus* survival and emergence at CRM 1.5 as well as *H. azteca* biomass were reduced relative to reference. *H. azteca* biomass was reduced at CRM 4.5 (*C. dilutus* were not evaluated at CRM 4.5).

Evaluating sediment chemistry, both of these locations have elevated concentrations of numerous COPECs, especially metals (Figures 4-45 through 4-54). For example, CRM 1.5 sediments had anomalously high concentrations of copper (60.8 mg/kg) when compared with other locations within the Clinch River. Copper appears as an outlier in bivariate plots between ash and copper for CRM 1.5 (Figure 4-45). This value is within the range of concentrations that have been associated with adverse effects (MacDonald et al. 2000). Similarly, CRM 4.5 sediments had the highest concentrations of arsenic, boron, and vanadium reported in the Clinch River, with arsenic measured at concentrations well above those expected to cause effects (MacDonald et al. 2000) (Figure 4-45). Although these locations had elevated concentrations of specific metals and observed effects, ash concentrations were among the lowest

measured in the Clinch River. Both of these locations are identified as outliers on the bivariate plots evaluating the relationships between these chemicals and ash (Figures 4-45 through 4-49).

In addition to having the highest concentrations for most COPECs, sediments from both CRM 1.5 and CRM 4.5 had the lowest TOC when compared to other Clinch River sediments (Figure 4-46 and Tables 2-33 through 2-35). While TOC may help explain differences in laboratory responses due to differences in metal bioavailability, ash and sediment composition results did not vary greatly among sites (Figures 4-39 and 4-50). The results of this analysis suggest that site-related risks to benthic invertebrates in the Clinch River are low; toxicity was only observed at two locations with maximum concentrations of arsenic and copper and very low TOC. Furthermore, there was no clear evidence to suggest that the elevated concentrations may be associated solely with ash as opposed to possible inputs of copper and arsenic from other sources, as the ash content in sediments from CRM 1.5 and CRM 4.5 was very low.

Conclusions

The results of the toxicity testing analysis indicate the potential for impacts related to the ash release within the Emory River. Toxicity was observed at the locations evaluated and appears to be correlated to ash and ash-related COPECs. In contrast, limited toxicity was observed in the Clinch River, and there is evidence to suggest that the chemicals driving risk may be associated with other sources.

Based on estimates of ash content at the IC_{25}, sub-lethal and lethal level effects can potentially occur when the ash content ranges from 22 percent to 78 percent. The majority of effects to *C. dilutus* emergence were centralized around approximately 40 percent ash content (Figures 4-56 through 4-60) and approximately 30 mg/kg arsenic. Similarly, the *H. azteca* biomass IC_{25} corresponded to an average of 40 percent ash and 34 mg/kg arsenic (Figures 4-56 through 4-58, 4-61, and 4-62; Exhibit 4-5). In addition, a greater than 25 percent reduction in biomass was observed at Emory River locations ERM 0.5 (*C. dilutus* only), ERM 2.5, ERM 3.5, and ERM 5.5 (*H. azteca* only) in the screening tests and at the four locations in the longer term test using 100 percent site sediment (Figures 4-56 and 4-57). Samples containing 40 percent or less site sediment had less than 25 percent reduction in biomass.

Exhibit 4-5. *C. dilutus* and *H. azteca* IC_{25} of Percent Ash[1] and Arsenic (mg/kg)[2] for the Emory River

Organism	***C. dilutus***									
Sample Location	**ERM 0.5**		**ERM 1.0**		**ERM 2.5**		**ERM 3.5**		**Average**	
Endpoint	IC_{25}		IC_{25}		IC_{25}		IC_{25}		IC_{25}	
Constituent	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic
PLC Survival	29.89	15.72	27.74	19.98	47.68	42.77	37.83	37.97	35.79	29.11
PLC Emergence	27.13	14.44	27.23	19.65	47.16	42.32	36.84	37.01	34.59	28.35

Organism	***H. azteca***									
Sample Location	**ERM 1.0**		**ERM 2.5**		**ERM 3.5**		**ERM 5.5**		**Average**	
Endpoint	IC_{25}		IC_{25}		IC_{25}		IC_{25}		IC_{25}	
Constituent	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic	Ash	Arsenic
28-Day Survival	>100%	>100%	77.93	68.98	>100%	>100%	>100%	>100%	77.93	68.98
28-Day Biomass	32.16	22.90	44.78	40.25	57.81	57.31	22.48	16.65	39.31	34.28

Notes:
Table uses only IC25 values and those samples with no measurable effect have an IC25 > 100% site sediment.
[1] Percent ash was calculated by using the IC_{25} provided by the toxicity testing lab (% site sediment) and the 100% site sediment percent ash results determined by PLM analysis.
[2] Arsenic (mg/kg) was calculated by using the IC_{25} provided by the toxicity testing lab (% site sediment) and the 100% site sediment arsenic concentration determined by the analytical laboratory.

Acronyms and Abbreviations:
ERM – Emory River mile
IC_{25} – inhibition concentration, 25 percent
PLM – polarized light microscopy

The observed effects from the tested Emory River sediments containing > 40 percent ash are statistically and biologically significant for *H. azteca* and *C. dilutus*. A 25 percent reduction on *H. azteca* growth has been noted to cause reductions in fecundity, as well as biomass in the environment, thus decreasing the food supply that higher trophic levels may depend on (Kubitz et al. 1996). Effects in the laboratory are also indicative of effects in the environment for chironomids as well. At a 30 percent reduction in growth, chironomid colonization and community structure were significantly impacted (Giesy et al. 1988). This information suggests that reductions in growth above 25 or 30 percent associated with samples containing greater than 40 percent ash may have impacts on the overall population health of benthic invertebrates.

Based on the Emory and Clinch Rivers toxicity studies, there appears to be a potential relationship between ash content, arsenic concentrations, and evidence of toxicity. However, while correlations may suggest

evidence of toxicity, they are not definitive proof of cause and effect. Therefore, it is not possible to determine whether the elevated arsenic is causing the response observed in the tests, or whether the organisms may be responding to stress associated with the presence of the ash or to some other variable. Arsenic is known to have adverse effects on aquatic species. In general, inorganic arsenic is more toxic to aquatic biota than its organic forms. Arsenic toxicity varies between species, and there is little evidence of biomagnification along the aquatic food web (Eisler 1988). The concentrations in sediment samples from ERM 3.5 and ERM 2.5 exceeded the probable EC of 33 mg/kg arsenic (MacDonald et al. 2000) where effects are thought to be likely. IC_{25} concentrations averaged close to this consensus-based criteria, providing support for the conclusion that arsenic may be causing an effect.

In contrast, there is very little research on the effects of ash on aquatic invertebrates. A previous investigative study included a toxicity identification evaluation (TIE) procedure (anionic resin treatment) to test differences between physical effects from ash compared with metals from the ash (Sherrard et al. 2010). It is worth noting that ash presents organisms with a substrate unlike those typically used for toxicity testing. *H. azteca* and *C. dilutus* can tolerate a wide range of substrates, from fines to coarser sediments, providing the organisms are fed during the exposures (ASTM International 2005). Ash is dissimilar to typical substrates as it may resemble silt or clay in appearance, is made of mostly silica, and when it comes into contact with water it may assume a cement-like form. When ash becomes hard-packed, the ability for an organism to burrow or seek refuge is limited. TVA research conducted with *H. azteca* 10-day exposures using whole ash samples from the embayment near the failed Dredge Cell indicated that metal/metalloid COPECs caused effects in *H. azteca* growth, and the sediment physical properties caused effects on survival (Sherrard et al. 2010). Clinch and Emory toxicity tests resulted in effects on growth that were often greater than the effects on survival. Based on the Sherrard et al. study, this would translate to both percent ash and metals concentrations contributing to the overall effects in test organisms.

In summary, there were minimal effects in the Clinch River tests and low confidence in those effects being attributed to ash. In the Emory River, effects were moderate with stronger relationships spatially to the Site and to ash content. Previous TIE studies and effects values for metals provide strong evidence to suggest percent ash and arsenic contributed to the overall effects seen in these tests. Overall confidence in these effects translating to the benthic community in the rivers is moderate based on the many confounding factors controlling bioavailability and species differences.

4.2.3 Benthic Invertebrate Tissue Concentrations of COPECs

Evaluation of tissue concentrations in representative invertebrate species provides a measure of the potential bioavailability of ash-related metals and metalloids. In addition, comparing COPEC concentrations to CBRs helps determine whether uptake is significant enough to adversely affect the health of the organisms. As noted for the screening benchmarks derived for the abiotic media, there is substantial uncertainty associated with the development of CBRs, so comparisons should be cautiously interpreted. As

with the abiotic media, concentrations below these values can be interpreted as not causing an effect, but occurrence of concentrations above the CBRs indicates only that the potential for impact exists and does not necessarily indicate a direct cause and effect.

This assessment focused on benthic invertebrate tissue data collected in 2009 and 2010 by ORNL (Appendix Q). Specifically, six primary sampling locations were selected in 2009 to collect snails (*Pleurocera canaliculatum*) and mayflies (adults and nymphs of *Hexagenia bilineata*) from the Emory River (ERM 6.0, ERM 2.5, and ERM 1.0), Clinch River (CRM 6.0 and CRM 1.5), and Little Emory River (LERM 1.0). Sites at ERM 6.0, CRM 6.0, and LERM 1.0 served as upstream reference locations for both species. In 2010, five more sites were added: ERM 4.0; CRM 3.5; TRM 560.8, TRM 566.3, and TRM 572.5. The TRM 572.5 site served as a reference for the other Tennessee River sites (Figure 4-1). Tissues were analyzed for 26 metals. Figure 4-63 presents photographs of the three species of benthic invertebrates collected for this evaluation.

Snails were collected by hand from submerged gravels, rocks, and various other solid, stable structures along the shoreline at each sampling location during the summer months. At each location, snails were located along transects separated by a distance of approximately 164 to 820 ft. Approximately 35 snails were collected from each transect for depuration prior to analysis, and in 2010, approximately 25 additional snails were collected from transects from a subset of sites for analysis without depuration.

Mayfly nymphs were collected from each location with a Peterson dredge across three (2010) or four (2009) transects perpendicular to the river channel. Each transect was sampled until about 80 to 120 nymphs were collected. Depurated and non-depurated samples of nymphs were analyzed.

Adult mayflies were collected opportunistically with sweep nets from vegetation along the shoreline. Efforts were made in both years to collect adult mayflies from the primary sampling locations, but because of the uncertainty and unpredictability in the timing and location of their emergence, samples were collected wherever available. As a result, samples of adult mayflies were not collected from the primary sampling locations in either year, nor were samples collected from the same locations in both years.

Adult mayflies were sorted by gender and adult stage (i.e., subimago and imago) for analysis, which provided up to four possible subgroups of adult mayflies at a given site: male subimagos, female subimagos, male imagos, and female imagos.

ORNL conducted preliminary analysis of the spatial distribution of the tissue concentrations prior to the availability of the other (i.e., abiotic) analytical chemistry data. Their evaluation found that the highest concentrations of most metals/metalloids for these invertebrates occurred downstream of the ash release site (Appendix Q). In addition, aluminum, iron, boron, and vanadium appeared to increase in concentrations in various benthic invertebrate tissues from 2009 to 2010.

Comparing results for depurated and non-depurated organisms, concentrations of most metals/metalloids were lower in depurated than non-depurated snails. The highest concentrations of several analytes in depurated snails, including arsenic, mercury, molybdenum, and strontium, were observed in samples from CRM 6.0. Except for copper and selenium, concentrations of the other analytes were lower in depurated mayfly nymphs than non-depurated ones. The highest concentrations of most analytes in both depurated and non-depurated mayfly nymphs were found in samples from ERM 1.0 (Appendix Q).

Concentrations of many analytes were statistically different between adult male and female mayflies. In addition to gender differences in concentrations of most analytes, differences were also found between adult life stages (within gender) for several metals.

The higher concentrations of metals/metalloids in benthic invertebrates in the vicinity of the ash release suggest the following analytes are associated with ash: antimony, selenium, arsenic, thallium, barium, vanadium, molybdenum, strontium, chromium, cobalt, and nickel. Aluminum, iron, manganese, and zinc in the invertebrates appear to primarily reflect natural background conditions (Appendix Q).

This BERA analysis builds upon these preliminary results by incorporating the results from the sediment and porewater sampling. Potential adverse effects to benthic invertebrates from bioaccumulation of COPECs associated with ash were evaluated for this BERA using the following methods:

- To evaluate the extent of bioaccumulation above background levels, tissue concentrations of analytes in snails, adult mayflies, and mayfly nymphs near the Site (near-field) were compared to tissue concentrations from reference locations.
- To evaluate the potential for adverse effects, maximum concentrations of COPECs determined to be statistically greater in near-field samples than in reference samples were compared to literature-derived CBR data.
- To evaluate potential relationships among tissue and site-media, correlation coefficients were calculated using tissue/sediment, tissue/porewater, and tissue/ash regressions.

Each step of the evaluation is described in detail in the following sections.

4.2.3.1 Analysis of Exposure and Effects

Since the focus of this BERA is on bioaccumulation of inorganic COPECs (i.e., metals/metalloids) in the tissues of benthic invertebrates, only depurated benthic invertebrate tissue data were evaluated. Depurated data exclude the gut contents of the invertebrates, which contains recently-ingested food and sediments, and wastes about to be excreted by the organism. Therefore, depurated data provide a more accurate measure of the actual body burdens of constituents incorporated into tissues accumulated as a result of exposure.

Furthermore, because this BERA focuses on the potential for adverse ecological effects following remedial activities that occurred at the Site in 2009-2010, only data from 2010 were included in the evaluation. Statistical summaries of depurated 2010 data for snails, mayfly adults, and mayfly nymphs are provided in Tables 4-25 through 4-32, respectively. Box plots of the data are available in Appendix R.

The bioaccumulation endpoint focused on 21 of the 26 inorganic COPECs analyzed. Calcium, iron, magnesium, potassium, and sodium are considered essential nutrients and were eliminated from evaluation.

Comparison of Tissue Concentrations: Near-field Locations versus Reference Locations

For each river system and reach (where possible), near-field invertebrate tissue data were statistically compared to data from that river's upstream reference locations. One-way ANOVA methods were performed using SAS software (SAS Institute, Inc. [SAS] 2010) to compare concentrations among multiple locations; t-tests were run in cases where there were only two locations. Data were log-transformed to meet the assumptions of parametric methods, and Levine's test was used to check the homogeneity of variance among the groups. The residuals of each ANOVA model were checked for normality. In the case of ANOVA, post-hoc testing was conducted using Dunnett's t-test to compare potentially impacted locations to reference locations. When log-transformed data still failed to meet the assumptions of parametric comparison methods, non-parametric tests (e.g., one-sided Wilcoxon Rank Sum Test) were applied to the original data to compare potentially impacted locations to reference locations. In particular, non-parametric tests were used to evaluate the adult mayfly dataset (Table 4-28), as log transformation was not adequate to meet the ANOVA assumptions, and many COPECs displayed bi-modal distributions (Appendix R – mayfly boxplots).

Statistical results for each tissue dataset are presented in Tables 4-25 through 4-32. Results considered significant (i.e., $p \leq 0.05$) are shaded. Results that may be significant ($0.05 < p < 0.1$) are presented in bold font. Figures 4-64 through 4-72 present a graphical depiction of each site's relative difference from reference, using average concentrations. Dark shading on the graphs indicates that mean near-field tissue concentrations are higher than the reference location; no shading indicates that near-field tissue concentrations are lower than the reference location. Significant differences ($p \leq 0.05$) are marked with an asterisk. Additional discussion is provided by tissue type below.

Snails

Tables 4-25 through 4-27 and Figures 4-64 through 4-66 present the statistical results for depurated snail tissue data collected in 2010.

Emory River

When evaluated as an entire site/river system (right-hand portion of Figure 4-64), 2010 depurated snail tissue concentrations of aluminum, arsenic, cadmium, thallium, and vanadium were significantly higher at near-field Emory River locations compared to the reference location. Cobalt and zinc had p-values less than 0.1 but greater than 0.05, so these also may be significant (Table 4-25; Figure 4-64). Concentrations of mercury in snail tissues are significantly higher at the reference location than near the release.

When evaluated by reach (left-hand portion of Figure 4-64), aluminum and thallium are significantly higher in snail tissues from the near-field reaches of the Emory River as compared to the reference location. ER_A also shows statistically higher concentrations of arsenic, vanadium, and zinc than the reference location. Strontium and zinc are statistically higher in ER_C than in the reference location. Antimony in ER_A and vanadium in ER_C both have p-values that are less than 0.1, but greater than 0.05, so these may be significantly enriched (Table 4-25 and Figure 4-64). Statistical results indicate unequal variances with the results for aluminum.

Clinch River

Snail tissue concentrations of manganese and thallium are significantly higher in the Clinch River as a whole when compared to the reference location. Aluminum and vanadium both have p-values that are less than 0.1, but greater than 0.05, so these metals also may be significantly enriched (Table 4-26 and Figure 4-65). Beryllium had low detections in snail tissue (14 percent), which caused the statistics for beryllium to be skewed because they are based on RLs.

When evaluated by reach, only snail tissues from CR_A have significantly higher concentrations of manganese and thallium than tissues from the reference location. These same COPECs border on significance in CR_B with p-values less than 0.1, but greater than 0.05.

Tennessee River

Thallium and beryllium were not detected in any snail tissues in the Tennessee River. The following COPECs are significant at the $p \leq 0.05$ level: arsenic, barium, cadmium, chromium, mercury, molybdenum, nickel, selenium, vanadium, and zinc (Table 4-27).

Adult Mayflies

Tables 4-28 and 4-29 and Figures 4-67 through 4-69 present the statistical results for adult mayfly tissue data collected in 2010. One-sided Wilcoxon Rank Sum Tests were performed on the datasets for the Emory, Clinch, and Tennessee Rivers. No adult mayflies were collected at the Emory River reference

location, so the Clinch River reference location is used to compare data collected from the Emory River. Major conclusions are discussed below, by river system.

Emory River

Adult mayfly tissue concentrations are significantly higher at the Emory River location compared to the Clinch River reference location for the following COPECs: arsenic, barium, boron, molybdenum, selenium, strontium, vanadium, and zinc. Manganese has a p-value of 0.1, which may or may not be significant (Table 4-28 and Figure 4-67). When compared to the reference location on a reach basis, the same COPECs show significance for both reaches of the Emory River, with the exception of manganese (significant only in ER_B) and zinc (significant only in ER_A) (Table 4-28 and Figure 4-67).

Clinch River

Adult mayfly tissue concentrations are significantly higher at the Clinch River location compared to the reference location for arsenic, molybdenum, selenium, and vanadium. Boron, manganese, and zinc have p-values less than 0.1, but greater than 0.05 (Table 4-28 and Figure 4-68).

When evaluated by reach, arsenic, selenium, and vanadium are significantly higher in adult mayfly tissues in the Clinch River when compared the reference location. Cadmium and cobalt show statistical significance in CR_A. Molybdenum may be significant with a p-value of 0.054. Boron, manganese, molybdenum, selenium, and zinc are statistically significantly greater in CR_B than in reference (Table 4-28 and Figure 4-68).

Tennessee River

Separating the evaluation by reach shows the same COPECs being significant; with vanadium only potentially significant in TR_B (p = 0.074) (Table 4-29 and Figure 4-69).

Mayfly Nymphs

Tables 4-30 through 4-32 and Figures 4-70 through 4-72 present the statistical results for depurated mayfly nymph tissue data collected in 2010. Major conclusions are discussed below, by river system.

Emory River

The following COPECs are significantly higher in mayfly nymph tissue in the Emory River area compared to the reference location: antimony, arsenic, barium, beryllium, mercury, molybdenum, selenium, and strontium. The p-value for vanadium is less than 0.1, but greater than 0.05, so this may be significant (Table 4-30 and Figure 4-70).

When evaluated by reach, antimony, arsenic, barium, molybdenum, selenium, and strontium are significantly higher in mayfly nymph tissues from the Emory River than the reference location. Aluminum, chromium, lead, nickel, and vanadium are statistically significantly greater in ER_A. Beryllium and mercury concentrations exceed reference in ER_C. The p-values for beryllium in ER_A, mercury in ER_B, and vanadium in ER_C are less than 0.1, but greater than 0.05 (Table 4-30 and Figure 4-70). Throughout the Emory River, beryllium has a low detection frequency in mayfly nymph tissue (44 percent); as such confidence in the statistical results for beryllium is low.

Clinch River

The following COPECs are significantly higher in mayfly nymph tissue in the Clinch River site compared to the reference location: aluminum, antimony, arsenic, barium, chromium, manganese, molybdenum, nickel, selenium, strontium, and vanadium (Table 4-31 and Figure 4-71).

When evaluated by reach, the same COPECs are significantly higher in mayfly nymph tissues in one or more of the Clinch River reaches than the reference location. Aluminum, antimony, barium, chromium, manganese, molybdenum, selenium, strontium, and vanadium are significant in the reaches. Arsenic and nickel are significant in CR_B. In addition, p-values for arsenic in CR_A and cobalt in CR_B are greater than 0.05, but less than 0.1, indicating potential significance (Table 4-31 and Figure 4-71).

Tennessee River

In the Tennessee River, only boron, molybdenum, and selenium have significantly higher concentrations in mayfly nymph tissue. Strontium and copper have p-values that are less than 0.1, but greater than 0.05 (Table 4-32).

A summary of benthic invertebrate tissue concentrations, by river system, that are statistically greater than their respective reference locations (at $p < 0.05$) is provided in Table 4-33 (Emory River), Table 4-34 (Clinch River), and Table 4-35 (Tennessee River). The COPECs that were greater in at least one reach of one river system were carried forward through the next phase of the analysis, which included a screen against literature-derived CBR values.

Tissue Concentrations from the Site Compared to Literature-Based Critical Body Residue Data

For the COPECs that have higher benthic invertebrate tissue concentrations near the release than in the corresponding upstream reference locations, tissue concentrations were compared to literature-derived CBR data to evaluate whether the concentrations of COPECs that have accumulated in benthic invertebrates are at levels shown to cause adverse effects to growth, survival, or reproductive potential.

Although the bioaccumulation of a constituent is an important consideration, it is important to remember that bioaccumulation in and of itself is not a hazard. It is necessary to determine the body burdens that result in adverse effects on the organism, or the CBR. This approach assumes that a whole body residue is a useful surrogate measurement of the amount of chemical at the site of toxic action within the organism, and that toxic responses can be predicted from whole body concentrations (USEPA 2000b). CBRs may be estimated by establishing the relationships between tissue concentrations and effects.

CBR data were selected from literature-derived values from the ERED (Appendix D). The selection process included only whole body data for the closest relevant species (i.e., mayfly or snail/mollusk/crustacean) and life stages (e.g., adult selected over egg) for growth, mortality, or reproductive endpoints. Combined or absorbed doses were preferred over water only exposures. If the data were unpaired (i.e., only a NOAEL or LOAEL was available), either the highest NOAEL or the lowest LOAEL was selected. The corresponding value was extrapolated from the available value by a factor of 10. If only effects concentrations were available (e.g., LC_{50}, ED_{25}, etc.), the lowest effects concentration was selected as the LOAEL, and the estimated NOAEL was set at 1/10th the LOAEL value. CBR data are unavailable for the following COPECs: antimony, barium, beryllium, boron, molybdenum, and strontium; therefore, these COPECs cannot be assessed using CBR data. Table 4-36 presents the selected CBR data for both mayflies and snails. The full ERED database outputs are provided in Appendix D.

Because CBR data are provided in wet weight, site-specific snail and mayfly tissue concentrations, which were freeze-dried and, therefore, presented as dry weights, were converted to wet weight using the following equation:

$$\text{Wet Weight} = \text{Dry Weight} / (100 / [100 - \text{percent moisture}])$$

A percent moisture content of 80 percent was assumed for the tissue types.

As a conservative measure of risk, the maximum detected COPEC concentrations in each river reach or location were compared to the selected CBRs to derive tissue-based HQs. Both NOAEL- and LOAEL-based CBRs were evaluated to provide a range of HQs. Results are presented below by tissue type.

Snails

Figure 4-73 graphically presents the maximum snail tissue concentrations and their respective CBRs, where available. HQs for snails are presented in Table 4-37 and discussed in detail by river system below.

Emory River

In the Emory River, HQs in the reaches are below 1 for the following metals: aluminum, arsenic, thallium, and vanadium. Only zinc exceeds the NOAEL, with HQ values ranging from 5.6 in ER_B to 12 in ER_A. The LOAEL HQ for zinc only slightly exceeds 1 in ER- A (1.2) (Table 4-37 and Figure 4-73).

Clinch River

In the Clinch River, manganese is the only metal that exceeds an HQ of 1 in both reaches with a NOAEL HQ of 13 and 9 for CR_A and CR_B, respectively. However, the LOAEL HQ for CR_A and CR_B were both less than or equal to 1 (Table 4-37 and Figure 4-73).

Tennessee River

In the Tennessee River, only chromium and zinc slightly exceed the NOAEL HQ of 1. All other metals, with available CBRs, have both NOAEL and LOAEL HQs less than 1.

Adult Mayflies

Figure 4-74 presents the maximum concentrations of metals in adult mayflies and the corresponding CBRs for each river system. HQs are presented in Table 4-38. No CBRs are available for barium, boron, molybdenum, and strontium so potential effects from these metals could not be evaluated.

Emory River

Manganese, vanadium, and zinc have HQs less than 1. The selenium HQ is greater than 1 for the NOAEL HQ and LOAEL HQ, but HQs were similar between the reaches. Arsenic slightly exceeds 1 for the NOAEL HQ in both reaches (Table 4-38 and Figure 4-74).

Clinch River

Results in the Clinch River follow a similar pattern to those in the Emory River. HQs for manganese, vanadium, and zinc are below 1. Selenium exceeds 1 for both the NOAEL HQ and LOAEL HQ in both reaches, but HQs are similar between the reaches. Arsenic slightly exceeds 1 for the NOAEL HQ in both reaches. Cobalt slightly exceeds 1 for the NOAEL HQ in CR_A (Table 4-38 and Figure 4-74).

Tennessee River

In the Tennessee River, mercury, selenium, and vanadium were detected and have CBRs. The HQs for mercury and vanadium are below 1. HQs for selenium exceed 1 in both reaches, but values are similar (Table 4-38 and Figure 4-74).

Mayfly Nymphs

Figure 4-75 presents the maximum concentrations of mayfly nymphs and the corresponding CBRs for each river system. HQs are presented in Table 4-39 and discussed in detail by river system below. No CBRs are available for antimony, barium, beryllium, molybdenum, and strontium so potential effects from these metals could not be evaluated.

Emory River

HQs exceed 1 in the Emory River for arsenic and selenium. The NOAEL HQ for arsenic ranges from 27 in ER_B to 334 in ER_A; the LOAEL HQ ranges from 3 in ER_B to 33 in ER_A. The NOAEL HQ for selenium ranges from 16 in ER_C to 34 in ER_A; the LOAEL HQ ranges from 2 in ER_C to 3 in ER_A. HQs for mercury and lead are below 1. The NOAEL HQ in ER_A for aluminum, chromium, nickel, and vanadium exceed 1. The LOAEL HQ in ER_A slightly exceeds 1 for aluminum (1.4) (Table 4-39 and Figure 4-75).

Clinch River

Aluminum, chromium, and manganese exceed the NOAELHQ, but not the LOAEL HQ in both reaches of the Clinch River; the NOAEL HQs are below 10. Nickel and vanadium both slightly exceed the NOAEL HQ in CR_B, while the LOAEL HQ is less than 1 for both metals. HQs are exceeded in both reaches for selenium, although the NOAEL and LOAEL HQ values are similar in both reaches. Arsenic exceeds 1 for both the NOAEL HQ (43) and LOAEL HQ (4) in CR_B (Table 4-39 and Figure 4-75).

Tennessee River

Both the NOAEL and LOAEL HQ exceed 1 for selenium in the Tennessee River.

Risk Estimation – Based on the evaluation of maximum tissue concentrations against literature-derived CBR data, selenium and arsenic appear to have the highest HQs and, therefore, present the greatest potential for adverse effects to benthic organisms (specifically mayflies). Although concentrations of molybdenum and strontium are greater than reference, no CBRs are available to evaluate potential adverse effects. Additional information regarding potential relationships among tissue and environmental media are provided below.

Tissue/Sediment, Tissue/Ash, and Tissue/Porewater Relationships

To evaluate potential relationships between tissue and abiotic media concentrations, Spearman Rank correlation coefficients were calculated in SAS (2010) using average concentrations across the entire site. Because snails are epifaunal and may be intermittently exposed to sediments along the shoreline, snail tissue concentrations (27 samples) were only evaluated against the seasonally-exposed sediments data (70 samples).

Separate evaluations were run using mayfly adults (93 samples) and nymphs (27 samples) against the surface (i.e., 0 to 0.5 ft) sediments data, which included bulk, submerged, and seasonally-exposed datasets (total of 170 sediment samples). Sediment data were used for the evaluation because of the infaunal nature of the mayfly nymphs, which have the potential to be exposed to all types of sediments, regardless of collection method or spatial location throughout the site. Because the nymph stage of mayflies is assumed to be exposed to all types of sediments, the subsequent adult mayfly is also assumed to be exposed to all sediment types.

In addition, to focus the evaluation and exclude differences in spatial location of the samples, the nymph data were grouped with bulk and submerged sediment samples collected within 500 ft. Correlations were run using only the grouped samples, consisted collection of 22 sediment samples and 24 nymph samples.

Mayfly nymphs were also evaluated against the porewater results. Finally, both adult mayflies and mayfly nymphs were evaluated against percent ash. Figures 4-76 and 4-77 present the sediment and benthic tissue sampling locations for snails/seasonally-exposed sediments and nymphs/submerged sediments, respectively. The call-out boxes on Figure 4-77 indicate which samples were included in the grouped analysis for the nymph samples. Figure 4-78 presents the locations of the percent ash samples and results in relation to the tissue samples. Results of the correlation analysis are presented in Tables 4-40 through 4-47, and Figures 4-79 through 4-85 show statistically significant bivariate plots; significant results are discussed in detail below.

Snails

For analytes whose near-field tissue concentrations exceeded reference location concentrations, average concentrations in snail tissue data were run against seasonally-exposed sediments results. To be conservative, non-detects were set at the RL for that analyte. Correlation results are presented in Table 4-40. Arsenic shows significant correlation at the $p \leq 0.05$, with a correlation coefficient of 0.89 (Figure 4-79). Vanadium is slightly correlated with a p-value of 0.0724 and a correlation coefficient of 0.77.

Adult Mayflies

For analytes where the near-field tissue concentrations exceeded those of the reference location, average concentrations in adult mayflies were run against the sediment data and percent ash data. Non-detects again were set at the RL. Results are presented in Table 4-41. In sediments, only arsenic is significantly correlated at $p \leq 0.05$, with a correlation coefficient of 0.93 (Figure 4-79).

In terms of percent ash, arsenic, molybdenum, selenium, and zinc concentrations in adult mayflies are significantly correlated at $p \leq 0.05$ (Table 4-42 and Figures 4-80 and 4-81). Selenium shows the best relationship, with a correlation coefficient of 0.96. Boron and vanadium have a potentially significant correlation with percent ash at $0.05 < p < 0.1$ (Table 4-42).

Mayfly Nymphs

Average concentrations of metals in mayfly nymphs were run against sediment concentrations and percent ash, with non-detects conservatively set at the RL. Results are presented in Tables 4-43 through 4-45. In sediments, arsenic and strontium are the only COPECs considered significantly correlated with their concentrations in mayfly nymphs at $p \leq 0.05$, with correlation coefficients of 0.73 and 0.96, respectively (Figure 4-82). Mercury is slightly correlated, with a p-value of 0.0671 and a correlation coefficient of 0.63. However, mercury was only detected in 64 percent of the sediment samples (Table 4-43).

More COPECs in mayfly nymphs correlate with percent ash data than with sediment COPEC concentrations. Concentrations of antimony, arsenic, barium, molybdenum, selenium, and strontium in mayfly nymphs correlate with percent ash at the $p \leq 0.05$ (Table 4-44 and Figures 4-83 through 4-85). Strontium shows the strongest relationship with a correlation coefficient of 0.92.

Of the three tissue types analyzed, mayfly nymphs have the greatest potential to bioaccumulate site-related COPECs due to their close association with river channel sediments and their limited mobility. However, because nymph tissue samples were not co-located with sediment samples, site-specific biota sediment accumulation factors could not be developed. Attempts to better correlate the nymph tissue and sediment data by grouping sediment samples located within 500 ft of a nymph sample location for correlation analysis yielded the same correlations as observed without the groupings. Arsenic and strontium were the only two analytes to show significant correlations, with correlation coefficients of 0.81 and 0.89, respectively. Mercury in sediments was not significantly correlated with nymph tissue mercury concentrations (Table 4-45).

No significant correlations between nymph tissue and porewater concentrations were observed at $p \leq 0.05$. No porewater samples were collected from the Tennessee River, so the evaluation only included the Emory and Clinch Rivers. Antimony showed a slight correlation with a p-value of 0.077 and a correlation coefficient of 0.70. However, detection of antimony was low in both nymph tissue and porewater (71 percent and 59 percent, respectively). Results of the porewater/nymph tissue correlations are presented in Table 4-46.

A summary of the relationships among site-related, abiotic media and benthic invertebrate tissue concentrations is provided in Table 4-47.

4.2.3.2 Risk Estimation

The benthic invertebrate bioaccumulation LOE consisted of three evaluations:

- Near-field tissue data (from various study areas) were compared to reference location tissue data.
- For the analytes with near-field concentrations significantly greater than reference, maximum tissue concentrations were compared to literature-derived CBR data.
- For the analytes with near-field concentrations significantly greater than reference, tissue/sediment, tissue/porewater, and tissue/percent ash data correlations were examined to identify statistically significant correlations of COPEC concentrations in tissues with COPEC concentrations in sediments or porewaters, or with percent ash in sediments.

Mayfly nymphs show the strongest associations with COPECs in river sediments, likely because they live in the sediments and have limited mobility. Although both arsenic and strontium in nymph tissues showed significant correlations with sediments, potential risk can only be evaluated for arsenic. The comparisons with CBR values suggest this is possibly a high risk. Additionally, although correlations for selenium between sediment and mayfly nymph tissue concentrations are weak, comparison with CBR values indicates some potential risk from selenium. Correlations between mayfly nymph tissue and ash data also indicate significant relationships for arsenic, selenium, and strontium.

Adult mayfly concentrations also correlate with arsenic in sediments and have HQs > 1. HQs are even higher for selenium, but there is no significant correlation between tissue and sediment concentrations. Correlations between adult mayfly tissue concentrations and percent ash indicate relationships for arsenic, molybdenum, selenium, and zinc.

Compared to infauna mayfly larvae, snails are relatively mobile and are less closely associated with river sediments. Arsenic concentrations in snail tissues show a significant correlation with seasonally-exposed sediments, but corresponding HQs are below 1.

Risk Estimation – The three evaluations presented for the bioaccumulation endpoint indicate arsenic has the greatest potential to cause adverse effects on benthic invertebrates. Arsenic is the only COPEC with near-field tissue concentrations greater than reference locations, HQs > 1, and statistically significant correlations with arsenic in sediments and percent ash in sediments.

Potential adverse effects from selenium are also possible, based on significant correlations between tissue selenium concentrations and percent ash, as well as HQs > 1. Potential adverse effects from strontium may

pose risks to benthic invertebrates based on significant correlations with percent ash and sediment concentrations, but any risks cannot be quantified due to the lack of reliable dose-response information (i.e., CBR values). A significant correlation was observed between percent ash and adult mayfly tissue for molybdenum; however molybdenum was not detected in sediments and there are no available CBRs against which to evaluate potential adverse effects.

4.2.4 Abiotic Media

Another LOE for evaluating potential risks to benthic invertebrates from ash-related COPECs is comparison of COPEC concentrations in near-field abiotic media (i.e., benthic waters, porewaters, groundwaters, and sediments) with reference locations and with screening benchmarks. Comparison to reference concentrations may suggest upstream sources of COPECs; near-field concentrations that are comparable to reference concentrations are unlikely to be release-related.

This LOE focuses primarily on documenting the constituents that are present at levels of potential concern; it provides no real measure of cause and effect. Screening benchmarks are intentionally conservative estimates of risk, designed to overestimate toxicity to ensure that no potential risk drivers are overlooked. They typically do not take into account site-specific characteristics that may reduce bioavailability or organism-specific parameters that might limit uptake. As a result, when constituent concentrations are below the benchmarks there is high confidence that risks are low. Exceedance of benchmarks indicates that further evaluation is required, rather than a measure or prediction of adverse effects.

4.2.4.1 Epi-Benthic Water Chemistry

Epi-enthic water data from near-field and from reference reaches were compared to acute and chronic criteria (i.e., Ambient Water Quality Criteria (AWQC)) developed by USEPA (2009). These benchmarks were derived based on data for many species of aquatic organisms and are considered protective of benthic invertebrates. Because the criteria must be protective of a wide range of species exposed under a variety of conditions, they are conservative. As a result, their application to a BERA likely result in risk estimates that are generally higher than would occur for individual species in the field. This assessment focused on data collected in 2010, consisting of inorganic COPEC concentrations in epi-benthic water samples collected 1.5 ft above the water-sediment interface. Samples were analyzed for metals/metalloids as both filtered (i.e., dissolved) and unfiltered (i.e., total) concentrations. In general, toxicity testing has shown that dissolved measurements are better predictors of toxicity than total recoverable measurements because only the dissolved fraction is normally considered to be bioavailable (USEPA 1992b). Use of the unfiltered data in this analysis is conservative because the total concentration includes some portion of the analyte that is bound to suspended sediment particles, thus is not readily bioavailable.

Several AWQC (cadmium, chromium, copper, lead, nickel, and zinc) are hardness-dependent. The USEPA-recommended hardness adjustments were made to these criteria. Both filtered (i.e., dissolved) and unfiltered (i.e., total) data were compared to criteria; when only criteria based on total concentrations were available, they were adjusted (based on USEPA guidance; USEPA 2009) by a conversion factor to obtain dissolved criteria.

Epi-benthic water sampling locations are shown on Figure 4-1. A summary of the ambient water criteria and comparison to site-specific benthic water concentrations is presented in Table 3-32. The results are summarized below:

- For total metals, aluminum exceeds the chronic criterion in each of the nine reaches (including the reference locations). Arsenic exceeds the chronic criterion at one Clinch River reach (CR_A). Mercury exceeds the chronic criterion in two reaches along the Emory River (ER_A and ER_Ref).
- For dissolved metals, the concentrations are below both the acute and chronic criteria.

Collectively, the epi-benthic water data indicate that total (unfiltered) concentrations for aluminum, and to a lesser extent arsenic and mercury, exceed the chronic values. However, exceedances for aluminum and mercury also occur for upstream (reference) epi-benthic waters.

These data indicate that the ash release has had no significant impact on epi-benthic water quality in the Clinch, Emory, or Tennessee Rivers. Because dissolved concentrations provide a more reliable predictor of bioavailability (and risk) to invertebrates, the LOE represented by the epi-benthic water data indicate no significant risks to benthic invertebrates.

4.2.4.2 Porewater and Groundwater Chemistry

Porewater is the water occupying the interstitial spaces of sediments and is considered to contain the most bioavailable fractions of metals and other COPECs. This is due to the fact that there is greater opportunity for chemical equilibrium to be established between the solid phase (incorporated into or adsorbed on sediments) and dissolved phase (porewaters) of constituents. This results in higher concentrations of constituents in pore waters, and a greater fraction present as dissolved forms.

Porewater was collected from the centrifugation of bulk sediment collected from the Clinch and Emory Rivers. Porewater was only evaluated in a subset of samples, as described in Section 2.5.4.4. Maximum dissolved and total porewater concentrations are presented in Tables 2-36 through 2-42. Total metals/metalloids in porewater were only analyzed from CR_A (Table 2-41); the remaining data are in the dissolved form. Results focus on the dissolved metals/metalloids and indicate the following exceedances:

- ER_B: arsenic and boron
- ER_C: aluminum and lead
- CR_A: lead
- CR_B: copper and lead.

In general, porewater concentrations were higher in the Emory River than in the Clinch River, which is consistent with the origin of the ash release in the Emory River. The ash release occurred in ER_B and the results of the porewater analysis indicate that most AWQC exceedances occur in ER_B and ER_C.

COPECs in groundwaters also have the potential to cause adverse effects to benthic invertebrates because groundwater seeps can occur at the sediment-surface water interface. Therefore, groundwater data was conservatively screened against AWQC protective of aquatic life. This is a very conservative assessment because concentrations in groundwater do not reflect the concentrations to which the aquatic organisms for which the AWQC were developed would be exposed; there is a process of natural attenuation and dilution that occurs as groundwater COPECs move into sediments and surface water.

The most recent round of groundwater sampling results available at the time of this analysis from upgradient, nearby permanent wells (6AR and 22) occurred during October, November, and December 2010 (Figure 2-20). The maximum detected concentrations of metals from these sampling events was compared with chronic surface water quality criteria for protection of fish and aquatic life (Tables 2-70 and 2-71). This assessment indicates that aluminum and cadmium in well 6AR are the only compounds for which the dissolved concentrations exceed AWQC criteria.

In addition to data from the groundwater wells, groundwater flux summaries were developed by Geosyntec et al. (2011) (Tables 2-72 through 2-74). These models estimate the contributions of groundwater to concentrations of selenium and arsenic in the Swan Pond Embayment, the Emory River, and the intake channel in a 30-year and 100-year time frame. Estimates were calculated using different diffusion coefficients for both arsenic and selenium. Estimated concentrations are below both the acute and chronic.

Risk Estimation – Concentrations of metals/metalloids in groundwaters at wells 6AR and 22 are generally below AWQC. In addition, modeled potential fluxes of groundwater for 30 years or 100 years indicate no potential exceedances of AWQC. Consequently, there is very little potential for adverse effects to benthic invertebrates from concentrations of constituents in groundwater.

4.2.4.3 Sediment Chemistry

Sediment samples were collected via two sampling methods. Specifically, discrete samples were collected using VibeCore™ techniques for evaluating the nature and extent of ash and other COPECs. These samples were further classified as consistently submerged or seasonally-exposed sediments. In addition,

multiple surface grab samples were collected in the vicinity of the locations targeted for toxicity testing and were composited to form bulk samples representing a larger spatial area. Sample locations can be found on Figures 2-9 through 2-17.

To evaluate potential adverse effects to benthic invertebrates from exposure to ash-related COPECs in sediments, the bulk and submerged sediment datasets representative of exposure for infaunal organisms were screened against sediment benchmarks (Tables 2-19 through 2-35). The seasonally-exposed sediment data, representative of exposure media for epi-benthic organisms, was screened separately against the same sediment benchmarks (Tables 2-13 through 2-18). The screening involved comparing the maximum detected concentrations from each reach to the sediment benchmarks. In addition to comparing individual chemical concentrations to screening benchmarks, PAHs were also evaluated using the equilibrium partitioning sediment benchmark toxic unit (EPSBTU). This approach assumes that the toxicity of PAHs can be additive and, therefore, evaluates them as a mixture. An EPSBTU greater than 1 is indicative of potential effects.

To visualize the distribution of COPECs in sediments between the individual river reaches and reference locations as well as in relation to conservative screening benchmarks, cumulative frequency distribution plots were generated for both bulk and VibeCore™ sediment samples. Due to low sample numbers for many COPECs, these were generated for metals/metalloids only. Results from this screening can be found in Appendix S.

Lastly, ash content was plotted against metal concentrations for submerged sediment samples collected via VibeCore™. These plots (Appendix S) were generated to help visualize potential relationships between ash content and metal/metalloid concentrations. Correlation coefficients were also generated and can be found in Appendix S.

The overall results of the sediment analysis are summarized below.

- None of the detected organic constituents in seasonally-exposed and submerged sediments were identified as COPECs. All were either not statistically different from reference, below benchmarks, or both. PAH EPSBTUs were below 1.
- Seasonally-exposed sediments: arsenic, chromium, copper, lead, manganese, mercury, nickel, and zinc were detected in at least one reach at concentrations above conservative screening benchmarks and were also determined to be detected at statistically significantly greater concentrations versus reference.
- Submerged sediments: arsenic, copper, mercury, nickel, and zinc were detected in at least one reach at concentrations above conservative screening benchmarks and were also determined to be detected at statistically significantly greater concentrations versus reference. These COPECs were further analyzed in comparison to their respective ash content and three show a positive relationship with ash content (correlation coefficients in parentheses): arsenic (0.82), copper (0.61), and nickel (0.51).

This analysis assumes that only those COPECs in site-related seasonally-exposed and submerged sediments that are statistically greater than reference and higher than conservative sediment screening benchmarks pose a potential risk to benthic invertebrates. Based on this analysis, no potential impacts to sediment due to ash were identified for any sampled areas of the Tennessee River with the exception of mercury. Sampled areas of the ER_A and ER_B were identified as presenting potential impacts to sediment-dwelling receptors due to the presence of arsenic, chromium, copper, lead, mercury, nickel, and zinc. Sampled areas of the CR_A and CR_B were identified as presenting potential impacts to sediment-dwelling receptors due to the presence of arsenic, chromium, copper, lead, manganese, mercury, nickel, and zinc.

4.2.4.4 Risk Estimation

Based on this assessment, groundwater and benthic water were eliminated from further evaluation; however, a number of metals were found in sediments and porewaters at concentrations of potential concern for benthic invertebrates:

- Sediments – arsenic, chromium, copper, lead, manganese, mercury, nickel, and zinc
- Porewaters – aluminum, arsenic, boron, copper, lead.

All of these compounds have been associated with ash to some degree, and correlations with ash content have been noted at the Site for several (e.g., arsenic, chromium, copper, lead, and nickel). As previously noted, the presence of these COPECs at concentrations above screening benchmarks does not indicate that adverse effects have or will occur, but rather that there is the potential for impacts and that further assessment is warranted.

4.3 Risk Characterization

Risks to benthic invertebrates were evaluated using multiple LOEs each aimed at providing a unique measure of exposure and potential effect. To integrate these discrete assessments, a qualitative WOE approach was applied. Greater emphasis was placed on those LOEs that provide direct measures of actual effect.

4.3.1 Weight of Evidence

A summary of the conclusions associated with each LOE is presented below and summarized in Exhibits 4-6 and 4-7.

Exhibit 4-6. Benthic Invertebrate Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Community Surveys	Negligible	High	Low	High
Sediment Toxicity Tests	Moderate (ER) Low (CR)	High	High	Moderate
Tissue Concentrations	Moderate	Low	High	Moderate
Porewater Chemistry	Low	Moderate	Low	Moderate
Sediment Chemistry	Low	High	High	Low
Weight of Evidence	**Moderate (ER) Low (CR)**	**High**	**High**	**--**

Acronyms and Abbreviations:
CR – Clinch River
ER – Emory River

Exhibit 4-7. Benthic Invertebrate Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Community Surveys	High	Negligible	High	--
Sediment Toxicity Tests	Moderate	Moderate (ER) Low (CR)	High	Arsenic, ash
Tissue Concentrations	Low	Moderate	Low	Arsenic, selenium
Porewater Chemistry	Moderate	Low	Moderate	--
Sediment Chemistry	Low	Low	High	Arsenic (ER, CR); Copper (CR); Nickel (CR); Mercury (ER)
Weight of Evidence	**--**	**Moderate (ER) Low (CR)**	**High**	**Arsenic, selenium, ash**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern
CR – Clinch River
ER – Emory River

4.3.1.1 Benthic Invertebrate Community Surveys

BIC assessments provide a direct measure of the quality of the biological community. Unlike laboratory toxicity tests, they evaluate conditions in situ and, therefore, integrate all potential exposure pathways. Interpretation of the results, however, can be complicated because variation in other physical conditions must be considered.

Evaluation of standard BIC metrics calculated for the Site provides no evidence that the BIC composition has been negatively impacted. Rather, the BIC composition appears to be strongly correlated with substrate type rather than ash or ash-related chemicals. Based on this assessment, variability in the BIC is likely explained by the heterogeneity inherent in the river system. The confidence in these conclusions is high, and the relative weight given to this assessment endpoint is high.

4.3.1.2 Sediment Toxicity

There are uncertainties introduced by sediment handling methods and by laboratory conditions; however, in general toxicity tests provides a good measure of the potential bioavailability and toxicity of sediment associated chemicals, and their ability to predict adverse effects is relatively good. The results of the toxicity testing analysis indicate that sediments in the Emory River have a moderate potential for impacts, largely associated with percent ash and arsenic. This is based on the fact that toxicity was observed at the locations evaluated and appears correlated to ash and ash-related COPECs, particularly arsenic. In contrast, toxicity observed in the Clinch River was lower, with fewer locations exhibiting adverse effects and correlations to ash and ash-related COPECs not as strong. In general, the majority of statistically significant effects observed were sub-lethal impacts (i.e., reductions in growth, biomass, or emergence) indicating that effects are not likely to be immediate and severe, but could result in long-term impacts to the population over time. The confidence in these conclusions is high, and the relative weight given to this assessment endpoint is moderate.

4.3.1.3 Benthic Invertebrate Tissue

The accumulation of chemical COPECs in the tissues of benthic invertebrates provides direct evidence of bioavailability. However, bioaccumulation alone is not an indication that an adverse effect has occurred or is occurring to the organism. CBRs provide a way to evaluate the possibility of an adverse effect; however, as with the discussion for abiotic media, the confidence in the ability of these screening values to predict toxicity is relatively low. In general, CBRs are conservative and are designed to overestimate effects. Both snail and mayfly tissue data indicate potential relationships with ash and sediment concentrations of arsenic and selenium; however, the tissue concentrations are low compared to the CBRs. In addition, arsenic and selenium are the only constituents with tissue concentrations greater than the CBRs that are also statistically different from reference. Of these two metalloids, arsenic also demonstrates a relationship between tissue

and sediment concentrations. The lack of strong relationships between tissue and sediment concentrations is consistent with the bioaccumulative nature of selenium. Both arsenic and selenium concentrations in mayfly tissues were found to covary with percent ash and the spatial distribution of arsenic and selenium in tissues is consistent with the distribution of ash from the release. Based on this assessment, it is concluded that arsenic and selenium associated with the ash release may be accumulating in the tissues of benthic invertebrates at concentrations that could result in an adverse effect. Confidence in this conclusion is relatively low because of the uncertainty associated with the CBRs; the relative weight given to this LOE is low.

4.3.1.4 *Abiotic* Media

Evaluation of abiotic media relative to reference and conservative screening benchmarks provides a conservative estimate of potential exposures to benthic invertebrates. These benchmarks, for sediments in particular, can vary widely in their ability to predict toxicity, but in general provide a very conservative estimate of potential risk because they do not take into account site-specific factors that may affect the bioavailability and ultimate toxicity of the measured COPECs. As a result, the relative weight given to these LOE is low because they are not considered accurate predictors of actual effects but rather tend to overestimate potential impacts. However, the confidence associated with findings of low risk is high given the conservative nature of the benchmarks. A summary of the conclusions associated with the abiotic media is provided below:

Porewater

Porewater concentrations represent one of the likely routes of exposure for benthic invertebrates as the portion of the constituent that partitions into porewater is believed to represent the bioavailable fraction. However, there are a limited number of porewater samples; therefore, it is difficult to draw conclusions regarding correlations with concentrations of constituents in sediments or tissues. Results of the porewater data review indicate that the Emory River has higher concentrations of COPECs in porewater that exceed AWQC than does the Clinch River. Maximum concentrations of aluminum, arsenic, boron, and lead are higher than AWQC in porewater samples from the Emory River, which is consistent with the origin of the ash release in the Emory River. The ash release occurred in ER_B and the results of the porewater analysis indicate that most AWQC exceedances occur in ER_B and ER_C, which may be indicative of a spatial gradient near-field to far-field reaches. Based on this assessment, there is a low potential for risk associated with porewater.

Sediment Chemistry

Based on this analysis, no potential impacts to sediment-dwelling receptors (i.e., benthic invertebrates) due to ash were identified for any sampled areas of the Tennessee River. Sampled areas of the ER_A and ER_B were identified as containing potential impacts to sediment-dwelling receptors due to the presence of arsenic, chromium, copper, and lead. Sampled areas of the CR_A and CR_B were identified as containing

potential impacts to sediment-dwelling receptors due to the presence of arsenic, chromium, copper, lead, and nickel. Many of these chemicals are known to be associated with ash and showed a positive correlation in site sediments with ash content. The HQs associated with these comparisons were low, often just above 1, which when combined with the conservative nature of the benchmark values gives high confidence in the determination that risk is low. In addition, because of the uncertainty associated with the benchmark values, the relative weight of this LOE is low when compared to more direct measures such as the toxicity test results of the community data.

4.3.2 Uncertainties

Uncertainties that may lead to either overestimates or underestimates of risk are associated with each stage of risk assessment. It is important to understand the major uncertainties and how they might affect the outcome of the risk assessment. The uncertainties associated with the three LOEs for benthic invertebrates are summarized below.

4.3.2.1 Benthic Community Analyses

The benthic community analysis relies on the premise that the presence of COPECs can alter the structure of the community by affecting the diversity and abundance of species. However, physical content/conditions of the sediment (e.g., grain size, organic content, and pH) may also cause differences in the BIC structure that are unrelated to chemical contamination. For this evaluation, physical parameters of the sediment were considered to reduce these potential uncertainties.

The BIC evaluation also makes assumptions about the interpretation of the individual metrics that may not reflect actual conditions for the organisms. For example, a decrease in abundance or diversity may be seen as a negative impact when in reality it may actually be beneficial for the long-term health of the population. To reduce this uncertainty, the evaluation focuses on comparison to reference, rather than relying on absolute measures presumed to be indicative of a healthy community.

In addition to these general uncertainties, there are several site-specific issues or methods that contribute to the uncertainties associated with the result. These are summarized below.

- Sediment chemistry, toxicity testing, physico-chemical (including quantitative grain size, TOC, and ash), and water quality measurements were collected at different locations than the BIC samples. Nearby samples were grouped together for the purpose of evaluating potential correlations between these parameters and the observed community metrics; however, direct effects of these measures on the BIC could not be assessed and, therefore, were inferred.

- As noted by TVA (Appendix M), the cross-sectional area increases in the downstream direction in each river and conditions change from riverine to lacustrine. Upstream to downstream changes in river morphology, including the presence of river confluences, can influence BIC composition.
- Although 10 upstream background samples each were collected in the Emory and Tennessee Rivers, and 30 upstream background samples were collected in the Clinch River, these samples came from only 1 or 2 transects each per river. As a result, these data are limited in terms of providing representative spatial coverage to approximate the background benthic community. However, it is also recognized that the availability of comparable upstream background locations from which to select is limited. This is due to the increasing riverine conditions that exist upstream of the Site, as the river channels narrow and becomes less influenced by the impounding effects of the Watts Bar Dam.
- Transect sampling provides good cross-sectional data, but is limited in terms of providing directly comparable benthic datasets by habitat type. That said, qualitative substrate data collected by TVA (Appendix M) suggests that most habitats were dominated by finer-grained substrates and that gross habitat differences were not that large.
- A portion of the benthic invertebrate data was collected during or shortly after dredging was completed. Dredging occurred only in Emory reaches B and C, and engineering controls were used to minimize migration of disturbed sediment and ash. Therefore, the potential influence of dredging is likely to be limited.

Emory River BIC data collected in January 2012 was evaluated and compared with 2010 results. Emory River BIC habitat, composition, diversity, abundance, and metric results were generally consistent between the 2 years. Differences that were noted in 2012 data were also noted in the reference transect. Similarity analysis on co-located data indicates that differences in the 2012 Emory River BIC data (taxa and counts) are correlated more to substrate type and water depth, similar to 2010, than percent ash composition or sediment chemistry (26 metals) (Appendix C). Some negative relationships with metal concentrations may exist, although these were few and appeared to be negligible. Therefore, this preliminary evaluation does not change the overall interpretation of risk.

4.3.2.2 Toxicity Endpoint

As with all toxicological studies, issues that create uncertainty with the toxicity data include the individual species tested and the fact that toxicity studies are conducted in a controlled laboratory setting. The toxicity evaluation for benthic invertebrates used toxicological data that were collected on individual species, not at the community level (such as in a mesocosm). Therefore, the potential risks identified in these tests are to individual species of organisms (i.e., *Ceriodaphnia dubia, Hyalella azteca*, and *Chironomus dilutus*) and not to a population or community comprised of populations of multiple species. It is assumed that these species are suitable and representative sentinels for the benthic community at the Site, and potential risks could be extrapolated to include the entire invertebrate population as a whole. In general, species selected for study

are those with a high degree of sensitivity to the COPECs being evaluated, resulting in endpoints that are protective of certain sensitive species.

In addition, the toxicity data were derived from laboratory studies conducted in settings that do not truly mimic field conditions. Laboratory studies typically control various factors in order to isolate one particular parameter. Although such controlled experiments make it easier to interpret isolated parameters or relationships, uncertainty is associated with assuming that laboratory exposure conditions are equivalent to in situ exposure conditions. For example, Sample et al. (1998) reports that laboratory studies are designed to maximize exposure of the test organism and result in scenarios unlikely to be found in the natural environment (e.g., 100 percent bioavailability, chemical present in only its most toxic form). In an attempt to combat some uncertainty and have the toxicological exposures more closely resemble site-specific conditions, site water was used for the toxicity studies. However, other variables (e.g., aeration, pH, diet, light, and temperature) remained controlled.

4.3.2.3 Bioaccumulation Endpoint

Uncertainty with the bioaccumulation endpoint is mainly attributed to the literature-derived CBR data. CBRs were generally based on either the lowest bounded or unbounded tissue concentration associated with significant effects on growth, survival, or mortality. In the case of an unbounded NOAEL, the highest concentration was used. In the case of an unbounded LOAEL, a 10-fold extrapolation factor was applied to estimate a NOAEL. The approach used here is a conservative one because the CBRs were generally based on the lowest reported tissue EC for the species most relevant to the study. Even greater uncertainty regarding the degree of conservatism exists for some metals for which few or no CBR effect data exists. Additionally, sample specific percent moisture determinations were unavailable at the time of data evaluations for the BERA. The default of 80 percent moisture agrees with the actual sample specific percent moisture determined for snails and mayfly nymphs. However, the mayfly adult percent moisture average was approximately 67 percent and as a result, the HQs for mayfly adults may have been under-estimated. This difference is considered to have a negligible impact on the overall conclusions (Smith pers. comm. 2012). Specifically, antimony, barium, beryllium, boron, molybdenum, and strontium could not be evaluated for potential adverse effects due to lack of CBRs and, therefore, potential risks are unknown.

Adams et al. (2010) cautions against the use of CBRs because there is no precise CBR for a constituent at which an individual in a population is significantly affected. The uptake rate of a constituent into an organism depends on the bioavailable concentration as well as site-specific biotic and abiotic conditions. Furthermore, it is unlikely that a laboratory-derived whole-body residue-effect relationship can be applied to evaluate the potential for effects in situ, or that a relationship defined in one situation can be applied to another without extensive validation (Adams et al. 2010). As a result of the various uncertainties associated with the CBR evaluation, the bioaccumulation endpoint received a low score in terms of confidence in the WOE evaluation.

Snail and mayfly bioaccumulation data collected in 2011 from the Emory, Clinch, and Tennessee Rivers was evaluated and compared with 2010 results. While fluctuations in arsenic and selenium concentrations occurred at several reaches in the Emory, Clinch and Tennessee Rivers, this preliminary evaluation does not change the overall interpretation of risk (Appendix AC).

4.3.3 Summary and Conclusions

Based on the overall WOE, the ash release resulted in impacts to the BIC, particularly in lower reaches of the Emory River. These risks appear to be primarily associated with ash and ash-related COPECs, especially arsenic. This conclusion is based on the finding of concentrations of ash-related COPECs in sediments and benthic tissues at concentrations potentially associated with adverse effects; combined with the observation at many locations of statistically significant reductions in biomass of test organisms in laboratory toxicity tests. However, the community structure and function in areas affected by the release is generally similar to that found at the upstream reference locations, suggesting that the BIC is recovering. The observed differences between sampling locations also appear to be related to variation in available substrate and habitat rather than the presence or absence of ash. Therefore, despite evidence that ash and ash-related chemicals are associated with potentially significant effects, the overall current condition of the BIC is compelling evidence that risks to the BIC are moderate.

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5. Aquatic Vegetation

The aquatic plant community (vegetation) is evaluated in this BERA because it provides habitat to resident biota and represents an important food resource to herbivorous and omnivorous organisms. Aquatic vegetation also has the potential to be affected from direct exposure to fly ash-related constituents. Plant communities are useful indicators of localized exposure because they are immobile and can bioaccumulate COPECs from localized areas in their tissues (i.e., stems, roots, leaves).

5.1 Problem Formulation

Aquatic vegetation refers to plants that live in direct contact with sediments and/or surface waters at least part of the year. These aquatic plants include macrophytes, algae, and periphyton. Macrophytes are vascular aquatic plants that may be emergent (e.g., cattails, rushes, and sedges), submergent (e.g., *Elodea* and milfoil), or floating (e.g., duckweed). Emergent and floating macrophytes can comprise a large portion of the diets of waterfowl and herbivorous mammals that feed in aquatic systems. Because emergent and submerged macrophytes are rooted in sediment, they are in direct contact with sediment, sediment porewater, and surface water; whereas, floating macrophytes are not rooted in sediment and, thus, are only in direct contact with surface water.

Algae are non-vascular plants that may be single celled, multi-cellular, or filamentous. Periphyton refers to the complex of algae, bacteria, and detritus attached to submerged surfaces. Periphyton and suspended algae account for a significant portion of primary productivity in many freshwater systems and are an important food source for various food web organisms including benthic invertebrates, amphibians, fish, birds, and mammals. Epi-benthic algae and periphyton may also be in contact with sediments, porewaters, and surface waters. As a result, aquatic plants and algae may be an important pathway for the trophic transfer of COPECs in water and sediment to aquatic biota and wildlife feeding in aquatic habitats.

Within large river systems, periphyton and macrophytes are generally restricted to shoreline habitats. Periphyton assemblages occur within these shallow shoreline habitats where the photic zone reaches available substrates. Similarly, emergent and shoreline vegetation are mostly restricted to shallow shoreline habitats. As described previously (Jacobs 2010a), wetland areas along the Emory and Clinch Rivers are generally limited to narrow fringe wetlands. Riparian habitats along the Clinch and Tennessee Rivers are varied in nature and include mature deciduous (or mixed) forests, scrub/shrub, mixed herbaceous vegetation, rock or concrete retaining walls, and manicured lawns.

Pool elevation in the Watts Bar Reservoir varies annually (e.g., 740 to 741 ft amsl in summer and 735 to 737 ft amsl in winter) resulting in seasonally-exposed sediments during the winter months due to the higher pool elevation. The relatively steep topography along much of the shoreline limits the areas where soils remain saturated, limiting suitable habitat for aquatic plants to overbank areas; shallow inlets fed by springs

or small tributaries; and point bars, shallow coves, and islands. As described in the SAP (Jacobs 2010a), aquatic vegetation described herein refers to those plants growing along the shoreline and emerging from the summer pools (primarily rushes and cattails).

Aquatic vegetation and periphyton in the Emory, Clinch, and Tennessee Rivers may be directly exposed to inorganic and other chemical COPECs in sediments, surface waters, and/or sediment porewaters. Concentrations of ash-related COPECs may bioaccumulate in aquatic plants and periphyton over time. Toxic effects from these COPECs can result in necrosis, decreased plant productivity, and death. On a community level, adverse effects include reduced structure and diversity of the aquatic plant community. In addition, plants can bioaccumulate these COPECs, resulting in subsequent exposures to fish and wildlife species via trophic transfer.

5.1.1 Assessment Endpoints

As discussed in Section 2.2, the assessment endpoint for aquatic vegetation used to develop this BERA is the survival and production of the aquatic plant communities in the Emory, Clinch, and Tennessee Rivers. Based on this assessment endpoint, the ecological risk question established for the evaluation of aquatic vegetation at the Site is as follows:

- Do concentrations of ash-related COPECs in surface water, submerged sediment, shallow sediment (seasonally-exposed), or sediment porewater pose unacceptable risks to aquatic plant communities?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective must be evaluated in the WOE assessment. These risk questions help focus subsequent steps of the aquatic vegetation assessment, including the selection of measurement endpoints.

5.1.2 Measurement Endpoints

The measurement endpoints for aquatic vegetation selected for the BERA are as follows.

Measures of exposure:

- COPEC concentrations in surface waters (total and dissolved) and speciation of selenium and arsenic
- COPEC concentrations in sediments, speciation of selenium and arsenic, and TOC
- COPEC concentrations in sediment porewaters
- Tissue concentrations of COPECs.

Measures of effects:

- Literature-derived toxicity data (e.g., NOAELs, LOAELs, lowest chronic values, and screening values) based on sediment, aqueous, and residue-effect levels.

5.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to aquatic plants. The LOEs used to evaluate aquatic plants include:

- Periphyton tissue data
- Shoreline and emergent aquatic vegetation tissue data
- Surface water and sediment chemistry data evaluation.

Each LOE is summarized below.

5.2.1 Periphyton Tissue Concentrations of COPECs

As summarized in the SAP (Jacobs 2010a), periphyton tissue samples were collected from periphytometers deployed in the Clinch and Emory Rivers between May 31 and June 24, 2011 (post-dredging), in two reaches per river downstream of the ash release, and in one upstream, un-impacted reach (reference reach) from the Clinch and Emory Rivers (Figure 2-23). Each periphytometer consisted of four glazed ceramic tiles attached to a foam float by a nylon rope. Ceramic tiles and rope were decontaminated prior to deployment. At each river reach, three periphytometers were attached to a single anchor and deployed to provide a single composite sample per each river reach. Periphyton samples were not collected in the Tennessee River. The sampling program yielded a sample size of n ≤ 2 per reach.

A total of 26 metal and metalloid COPECs were measured in periphyton tissue samples collected in the Clinch and Emory Rivers. Out of the 26 metals, 18 were detected at 100 percent frequency in samples collected from reference and impacted reaches. Spatial trends among reaches were largely absent for the COPECs, which is either a reflection of natural system variability, the small sample size, and/or no detectable impact of the ash on uptake of metal and metalloid COPECs.

5.2.1.1 Analysis of Exposure and Effects

Trace element concentrations in periphyton were not statistically compared among locations due to small samples sizes (n ≤ 2 per reach) in the *Trace Element Concentrations in Aquatic Vegetation and Periphyton: 2011* report (Appendix T). Analysis of risks involved comparison of tissue concentrations in reference reaches to downstream reaches. Due to the small sample size and no apparent spatial trends among downstream reaches, periphyton tissue COPEC concentrations measured in each reference reach were grouped into one “reference population” and compared to tissue concentrations measured in the downstream reaches of the two rivers combined (“downstream population”). A one-sided t-test was conducted to determine if tissue COPEC concentrations in the downstream population were greater than tissue COPEC concentrations in the reference population. Although no comparisons yielded a significant

relationship (α = 0.05), arsenic (p = 0.085), copper (p = 0.086), and strontium (p = 0.086) approached the specified level of significance with mean concentrations slightly higher in grouped impacted reaches relative to the grouped reference reaches (Table 5-1). Consequently, the significance of these tissue concentrations was determined from a community perspective.

Arsenic was detected in the periphyton samples collected from downstream and reference reaches. Mean arsenic concentration in periphyton tissue collected from downstream reaches (57.5 mg/kg dry weight) was about 1.8 times greater than concentrations measured in the reference population (31.9 mg/kg dry weight) (Figure 5-1). The higher concentration in the downstream population is primarily attributed to a single Emory River sample (ERM 2.5) that contained 116.3 mg/kg dry weight (Table 2-183). These arsenic levels are within ranges previously reported for aquatic plant and algae species. In a North Dakota stream, a periphyton community that contained arsenic concentrations between 1,465 and 1,931 mg/kg dry weight had an autotrophic index typical of an unstressed community (Kuwabara and Fuller 2004). This suggests that arsenic tissue concentrations in the downstream reaches of the Emory and Clinch Rivers (19.43 to 116.3 mg/kg dry weight) are not within the range associated with adverse effects to the periphyton community.

Phyto-accumulation of inorganic arsenic forms (arsenite and arsenate) is known to vary depending on a variety of environmental variables including pH, humic acid, and phosphate concentrations. Arsenic is accumulated from the water by a variety of organisms, including periphyton and algae, but there is no evidence of magnification in the aquatic food web (Eisler 1988). Tissue concentrations measured in the downstream reaches were below dietary levels in rainbow trout associated with adverse effects (120 mg/kg dry weight) (Hood 1985 as reported in Eisler 1988). Aqueous concentrations of arsenic were not detected above water quality criteria in any samples of surface water collected from impacted or reference reaches on the Emory, Clinch, or Tennessee Rivers.

Copper was detected in the periphyton samples collected from downstream and reference reaches (Figure 5-2). Mean copper in periphyton tissue collected from downstream reaches was about 2.2 times greater than the reference population (Figure 5-2). The downstream Clinch River periphyton samples had higher copper concentrations than reference, but the downstream periphyton in the Emory River was comparable to the reference population (Tables 2-182 and 2-183). Background and impacted tissue concentrations in periphyton vary widely (e.g., Behra et al. 2002 and Besser et al. 2001). A critical concentration range for copper in terrestrial vascular plant tissue is between 25 and 90 mg/kg dry weight (Kumar 2008). In a study on microalgae, De Schamphelaere et al. (2005) reported median effect levels for microalgae based on tissue-bound copper that were more than an order of magnitude greater than the reference and Emory River reaches and more than seven times greater than Clinch River impacted reaches. Effect levels reported by De Schamphelaere et al. (2005) were reported as a mass per cell basis for internal and external copper (external copper was operationally defined as the fraction removable from the algal cell by short-term contact with ethylenediaminetetraacetic acid). To compare these values with

tissue levels reported herein, external and internal copper was combined and concentrations were converted to a mass per mass basis using the average algal cell weights reported by Kolts et al. (2009). Mean concentrations for the Clinch River impacted reaches were less than 20 percent of the minimum reported tissue EC_{50}s reported by De Schamphelaere et al. (2005), which suggests that periphyton exposures in the Clinch River are below levels associated with adverse effects.

Strontium was detected in the periphyton samples collected from downstream and reference reaches. Mean strontium concentrations were about 1.7 times greater in periphyton collected from downstream reaches (158.2 mg/kg dry weight) compared to the reference population (94.4 mg/kg dry weight) (Figure 5-3). Strontium concentrations were greatest in the Clinch River downstream reaches (Table 2-185). No strontium tissue concentrations associated with adverse effects in freshwater periphyton were found in the literature. Accumulation of strontium varies considerably across species and can be dependent on the co-occurrence of other metals including calcium, magnesium, and barium (Haghiri 1964). The EC_{25} for algae exposed to aqueous concentrations of strontium is 53 mg/L (Pacholski 2009). Maximum detected concentrations of strontium in surface waters were similar between impacted and reference reaches and below 0.130 mg/L. This suggests aquatic plants and algae are not adversely affected by site-specific strontium exposure in the Emory, Clinch, and Tennessee Rivers.

5.2.1.2 Risk Estimation

Tissue concentrations in periphyton in downstream reaches were significantly different at $p < 0.1$ for arsenic, copper, and strontium compared to the reference population. Analysis of the significance of tissue concentrations to the community suggests that these COPECs are not expected to pose unacceptable risk to the periphyton community. Uncertainties associated with this characterization are described in Section 5.3.2.

5.2.2 Shoreline and Emergent Vegetation Tissue Concentrations of COPECs

As summarized in the SAP (Jacobs 2010a), aquatic vegetation was collected from June 21 through June 24, 2011. Aquatic vegetation samples were collected in three reaches per river downstream of the impacts, and in one upstream, un-impacted reach (“reference reach”) per river (Figure 2-23). In each reach, a sample of emergent vegetation was collected from a summer pool and a second sample was collected from vegetation growing along the shoreline. Both samples were sampled using Kyocera ceramic scissors that were decontaminated between each use. The part of the plant collected was limited to the portion of the plant that was emerging from the water; roots and other submerged parts of the plant were not included in the sample. Appendix T details this vegetation study including the study objectives, location of samples, methods, results (including tables and figures), and interpretation. Below is a summary and interpretation of the results.

5.2.2.1 Analysis of Exposure and Effects

COPEC concentrations measured in aquatic vegetation tissues were not statistically higher in downstream reaches compared to the reference population (Appendix T - Tables 3 and 4). The only significant differences between study reaches were attributed to greater concentrations in reference reaches. As a result, ash-related impacts to aquatic plants are not indicated based on the available data and absence of elevated tissue concentrations in downstream reaches.

5.2.2.2 Risk Estimation

Available data do not indicate COPEC concentrations are higher in aquatic vegetation growing within downstream reaches compared to the reference population (Tables 2-186 through 2-203). Therefore, ash-related risks to shoreline and emergent vegetation are unlikely. Uncertainties associated with this characterization are discussed in Section 5.3.2.

5.2.3 Surface Water Chemistry

Surface water was evaluated as an LOE to support assessment of potential risks to aquatic vegetation and periphyton. Surface water samples were collected over eight weeks in 2010 from 11 fixed-station monitoring locations in the Emory, Clinch, and Tennessee Rivers (Figure 2-19). At each location, samples were collected from mid-depth and approximately 1.5 ft from the bottom.

5.2.3.1 Analysis of Exposure and Effects

Surface water concentrations of ash-related COPECs were compared to USEPA Region 4 Chronic Freshwater Screening Values (USEPA 2002b) to estimate potential exposure to aquatic plants. Only maximum detected concentrations of aluminum, mercury, and lead exceeded these screening values (only epi-benthic water exceeded the lead screening value). Surface water screening values were not available for barium, cobalt, manganese, molybdenum, strontium, or specific arsenic or selenium species.

Aluminum concentrations (total) in the reaches except ER_Ref exceeded the screening value of 87 micrograms per liter (µg/L). However, this criterion is only applicable for acidic water (pH < 5) and was developed for fish in low water hardness environments. Thus, adverse effects are not expected for aquatic plants based on the higher pH and higher water hardness in the reaches assessed. In a study on a diatom, growth was inhibited at 0.81 mg/L, which is above concentrations reported for the downstream reaches (USEPA 1988).

Maximum concentrations of total recoverable lead collected from epi-benthic water exceeded the chronic screening values for ER_B and CR_A. However, lead concentrations in these reaches were less than

2 µg/L, which is substantially below the 800 µg/L maximum permissible concentration (MPC) reported for an aquatic macrophyte (Wang 1986 – reported MPC calculated as 10 percent of the median EC [EC_{50}]).

Barium, cobalt, manganese, and molybdenum screening values for surface water were not available from USEPA Region 4 (USEPA 2002b). For these COPECs, alternative values were developed based on a literature-review. For barium, Wang (1986) derived an MPC of 2.6 mg/L, which is more than an order of magnitude greater than the maximum detected barium concentration of any study reach (e.g., 0.0862 mg/L at ER_B).

Naumann et al. (2007) reported cobalt EC_{20} values for growth and production of *Lemna minor* between 0.069 and 0.117 mg/L. These values were more than 100 times greater than maximum detected concentrations of cobalt in downstream reaches (0.00033 to 0.00046 mg/L). Maximum concentrations of manganese were detected in ER_C (0.225 mg/L total recoverable) and ER_Ref (0.196 mg/L total recoverable). Wang (1986) reported a MPC of 3.1 mg/L for manganese, which is more than an order of magnitude greater than maximum concentrations measured across the Site.

Finally, Swain (1986) reported deleterious effects to algae at molybdenum concentrations in surface water > 0.02 mg/L, which is greater than surface water concentrations in downstream reaches. No adverse effects associated with strontium exposure were found in the literature. However, maximum concentrations of strontium were comparable across downstream and reference reaches, ranging from 0.107 to 0.126 mg/L, which likely indicates natural system variability instead of contamination from ash.

5.2.3.2 Risk Estimation

Based on the screening analysis provided in Section 2.5.4.5, in conjunction with the above analysis of exposure and effects, risks associated with exposure of aquatic plants to COPECs in surface water are unlikely. Surface water COPECs that did not exceed USEPA Region 4 screening values (USEPA 2002b) were not carried forward into further evaluation of exposure and effects. These screening values are based on aquatic animal toxicity values and are generally consistent with USEPA's nationally recommended AWQC (USEPA 2009). These benchmarks were derived based on data for many species of aquatic organisms and will likely protect aquatic plants (USEPA 1985). Plant studies are considered in USEPA's derivation of water quality criteria, although plant toxicity data suitable for criteria derivation are typically limited. Because the criteria must be protective of a wide range of species exposed under a variety of conditions, they are conservative. As a result, their application to a BERA likely result in risk estimates that are generally higher than would occur for individual species in the field.

5.2.4 Sediment and Porewater Chemistry

Seasonally-exposed sediments, submerged sediments, and sediment porewaters were evaluated as LOEs to support assessment of potential risks to aquatic plants. Seasonally-exposed sediment refers to sediment that is exposed in shoreline habitats during low reservoir levels in the winter months (i.e., December to April). These sediments are submerged under shallow waters during the remainder of the year and can provide habitat to rooted aquatic plants and periphyton. Submerged sediment refers to the sediments that are below water year round. Aquatic vegetation and periphyton can be expected to occur in submerged sediments that are relatively shallow water (e.g., less than 3 ft deep). Sediment porewater refers to the interstitial water that occupies the space between sediment grains and is a primary exposure source to rooted aquatic macrophytes.

5.2.4.1 Analysis of Exposure and Effects – Sediment

Sediment COPEC concentrations were compared to USEPA Region 4 sediment screening values (USEPA 1994b) to estimate potential exposure to aquatic plants. As described in Sections 2.5.4.2 and 2.5.4.3, several metal and metalloid COPECs exceeded available screening values including: arsenic, chromium, copper, lead, mercury, nickel, and zinc. As a result, these metals were carried forward for further evaluation of exposure and effects to aquatic plants.

Potential risk associated with aquatic plant exposure to sediment concentrations of COPECs was evaluated using USEPA's Ecological Soil Screening Levels (Eco-SSLs) (USEPA 2005) developed for terrestrial plants. The Eco-SSLs for plants have limited applicability to aquatic systems because preference was given to studies with high bioavailability of the COPECs in upland soils when Eco-SSL values were derived (USEPA 2005). As a result, application of these values to support exposure and effects assessment of aquatic plants may be conservative because river sediment contains higher OC content and decreased bioavailability relative to soils used in studies to derive Eco-SSL values. Aquatic plants may, however, be more sensitive to certain metals than upland plants, which may not necessarily be conservative. Eco-SSL values used to assess aquatic plant exposure, and corresponding HQs, are provided in Tables 5-2 through 5-5 for maximum and mean detected COPECs in seasonally-exposed sediment. HQs based on maximum and mean COPEC concentrations in submerged sediment are provided in Tables 5-6 through 5-11.

Based on maximum detected concentrations, concentrations of arsenic in seasonally-exposed sediment and submerged sediment in downstream reaches exceeded USEPA Region 4 sediment criteria (USEPA 1994b) and were significantly higher than reference reach sediment concentrations at ER_A, ER_B, CR_A, and CR_B. At each of these locations, maximum detected arsenic concentration also exceeded Eco-SSL values (HQs ranged from 2 to 5). For submerged sediment, maximum detected arsenic concentrations were also above Eco-SSL values at ER_C (HQ = 4). When average concentrations of arsenic were compared to the Eco-SSL plant value, HQs were less than 1.4 for these locations (Tables 5-2 through 5-11).

Mean cobalt concentrations were statistically greater than reference locations at CR_A and CR_B (both sediment types) and at ER_A and ER_B in seasonally-exposed sediment (Tables 5-4 and 5-5 and 5-9 through 5-11). However, these concentrations did not exceed USEPA Region 4's sediment screening value (USEPA 1994b) of 50 mg/kg dry weight cobalt. The Eco-SSL plant value for cobalt is 13 mg/kg dry weight, as a result, potential cobalt exposure from sediment was evaluated further for plants. Based on maximum submerged sediment concentrations and the plant Eco-SSL value, HQs were less than or equal to 2 at these locations except the ER_Ref (HQ = 3). When submerged sediment concentrations were evaluated based on mean concentrations, HQs were less than 2. Maximum seasonally-exposed sediment concentrations exceeded the plant Eco-SSL value (HQs = 2 to 7) in these locations except the ER_Ref (Tables 5-2 through 5-11). However, when evaluated on an average basis, HQs were equal to or less than 1.1.

Copper concentrations in both sediment types exceeded USEPA Region 4 screening values (USEPA 1994b) and were significantly greater at ER_A, ER_B, CR_A, and CR_B relative to reference reaches. However, HQs for copper in seasonally-exposed sediment were less than or equal to 1 (Tables 5-2 and 5-3 and 5-6 through 5-8).

Manganese concentrations in seasonally-exposed sediment in downstream reaches in the Clinch River (CR_A and CR_B) were statistically higher relative to the ER_Ref. (Note: seasonally-exposed sediment was not collected at a CR_Ref.) Based on maximum concentrations at these locations and the Eco-SSL plant value, HQs were 8 (CR_A) and 16 (CR_B). However, maximum concentrations of manganese in submerged sediments at the ER_Ref, CR_Ref, and TR_Ref exceeded the Eco-SSL value by a similar magnitude (HQs = 2 to 17). Manganese concentrations in the sediment types and at each location (including reference reaches) exceeded the Eco-SSL plant values when mean and maximum concentrations were evaluated. This suggests that the Eco-SSL plant value is probably not relevant of potential exposure and effects to aquatic plants growing in river sediment. Further, because studies used to develop the manganese plant-based Eco-SSL value used soils with less organic content compared to river sediment, the bioavailability of manganese is expected to be greater than in river sediment. No manganese sediment concentrations associated with adverse effects in aquatic plants were found in the literature. Manganese deficiency and toxicity has been described for terrestrial plants, and based on this information, toxicity is generally observed in acidic soils below pH 5.5 (Schulte and Kelling 1999), which is not typical in these river sediments.

Selenium was detected in seasonally-exposed and submerged sediment collected from several downstream reaches and was not detected in reference reaches. The available Eco-SSL for selenium is highly conservative (0.52 mg/kg dry weight) and based on terrestrial crop species of vegetation. However, adverse effects associated with sediment exposure of selenium to aquatic plants are not available in the literature. Based on maximum detected concentrations of selenium in seasonally-exposed sediments, HQs ranged from 3 to 7 in the downstream reaches. For maximum selenium concentration in submerged sediments, HQs

ranged from 4 to 10 in the downstream reaches. However, application of these effect levels to sediment and aquatic plants is highly uncertain because they were derived in upland soils using terrestrial crop species. Accumulation and toxicity of selenium to plants is widely variable across species. For example, a 10 percent reduction in growth corresponded to a tissue concentration of 2 mg/kg dry weight in rice and 330 mg/kg dry weight in white clover (Terry et al. 2000). Mean tissue concentrations of selenium in emergent vegetation was less than 0.8 mg/kg dry weight in the vegetation samples (selenium was not detected in shoreline vegetation). This suggests that aquatic plants are not adversely affected by selenium exposure.

5.2.4.2 Analysis of Exposure and Effects – Porewater

Sediment porewater concentrations were also compared to USEPA Region 4 freshwater screening values (USEPA 2002b) to estimate potential exposure to aquatic life. Only aluminum, arsenic, lead, and mercury in sediment porewater exceeded these screening values. These COPECs were carried forward to evaluate potential exposure and effects to aquatic plants. This comparison showed that porewater concentrations were below levels associated with adverse effects to aquatic plants, as described below.

Porewater concentrations of aluminum exceeded the USEPA Region 4 chronic criteria (USEPA 2002b) in the downstream and reference reaches. Acute criteria were also exceeded at ER_B (1.2 m/L) and ER_C (0.83 mg/L). Based on the available literature, no adverse effects to aquatic plants associated with this range of aluminum exposure are expected. For example, the growth of the floating aquatic macrophyte *Lemna* was stimulated at aqueous concentrations of 4 mg/L aluminum over a 15-day exposure period (Radić et al. 2009).

Porewater concentrations of lead only marginally exceeded the chronic screening level at CR_A and ER_B. Maximum detected concentrations of total lead at these locations were less than 2 µg/L, which are substantially below levels associated with adverse effects in aquatic plants as described above in Section 5.2.3.1.

Porewater concentrations of arsenic only marginally exceeded the chronic screening level at CR_A. The maximum detected concentration of total arsenic at this location was 198 µg/L. This value is within the range associated with adverse effects to some sensitive green microalgae species (CCME 2001). A 14-day EC_{50} for the green alga, *Scenedesmus obliquus*, demonstrated growth inhibition at a concentration of 50 µg/L (arsenic). Other acute (96-hour) EC_{50}s have been reported as 79 µg/L and 159 µg/L (as Arsenic) for arsenite and arsenate, respectively. However, these values are derived from laboratory toxicity tests using synthetic laboratory water that is typically low in organic content and suspended particulate matter. Dissolved arsenic concentrations at CR_A were below 1 µg/L, which indicates the concentration of the bioavailable form was considerably less than the total recoverable arsenic concentration of 198 µg/L. Available toxicity data indicate that macrophytes are less sensitive to arsenic: 14-day EC_{50}s, based on

growth inhibition, of 0.63 mg/L (arsenite) and 22.2 mg/L (arsenate) have been reported for the floating macrophyte *Lemna* (Jenner and Janssen-Mommen 1993).

5.2.4.3 Risk Estimation

Exposure to submerged sediment, seasonally-exposed sediment, and sediment porewater was evaluated as separate LOEs to support risk evaluations for aquatic plants and periphyton. Eco-SSL values developed for soil and terrestrial plants were used to evaluate potential adverse effects to aquatic plants associated with sediment exposures. Application of Eco-SSL values to aquatic sediment may be a conservative approach because these screening values are based on studies that used soil consisting of little organic content and, thus, increased bioavailability compared to sediment-bound COPECs. As a result, sediment exceedance of a soil-based Eco-SSL value does not necessarily indicate risks to aquatic plants. Arsenic, cobalt, copper, and manganese were statistically greater in select downstream reaches when compared to reference reaches and were, thus, compared to Eco-SSL values to evaluate potential risks. Maximum concentrations of these COPECs in reaches that were identified as being statistically higher compared to reference reaches were generally only marginally above the Eco-SSL values (i.e., HQs < 5). However, the average sediment COPEC concentration was generally comparable with the Eco-SSL values (i.e., HQs < 2). Manganese concentrations in seasonally-exposed and submerged sediments exceeded the Eco-SSL value in the locations (including reference), and HQs based on maximum concentrations of up to 18.7 were determined. However, submerged sediment collected from the TR_Ref location contained manganese concentrations of 3,660 mg/kg dry weight, which corresponds to an Eco-SSL-based HQ of 17. This suggests that manganese measured in sediment collected from downstream reaches might be at least partly related to legacy metals contamination.

Porewater concentrations that exceeded the screening values were also evaluated to support estimates of exposure and effects to aquatic plants. Based on this analysis, the only potential risk identified to be associated with porewater exposure was based on total recoverable arsenic concentrations at CR_A. Species of microalgae have been reported to be adversely effected when exposed to similar arsenic concentrations; however, those exposures occurred in synthetic laboratory water that probably contained low concentrations of DOC and suspended particles, leading to a relatively high percentage of the total recoverable arsenic being bioavailable in the laboratory waters. Therefore, it is not valid to compare effects concentrations based on high concentrations of mostly bioavailable total recoverable arsenic in the laboratory exposure waters to the most likely lower bioavailability arsenic concentrations measured at CR_A. Other species of aquatic macrophytes are not expected to be adversely affected based on this exposure range.

Potential risk to the aquatic plant community is unlikely, based on the above analysis. Marginal exceedances of conservative soil-based Eco-SSL values are not interpreted as being indicative of unacceptable risk to the aquatic plant communities. In addition, comparison of sediment porewater

concentrations of COPECs to effect levels reported for aquatic plants and algae does not indicate unacceptable levels of risk. Because sediment porewater represents a primary exposure pathway to rooted aquatic plants, it can be concluded that these species of aquatic plants are not likely to be adversely exposed to sediment COPECs.

5.3 Risk Characterization

Risk estimates to aquatic plants are based on comparisons of trace element concentrations measured in biotic and abiotic media to available effect data. Spatial correlation of biotic and abiotic media concentrations was also used to distinguish ash-related COPECs from potential legacy contamination. LOEs are interpreted below in a WOE context to characterize potential risk to aquatic plants.

5.3.1 Weight of Evidence

Trace element concentrations in surface water, sediment, and sediment porewater were considered as exposure pathways to evaluate potential risks to aquatic plants. Exposure to aqueous concentrations of trace elements in the surface water column, epi-benthic water, and sediment porewater did not indicate moderate or unacceptable risks to aquatic plants. Because aquatic plants absorb elements primarily through surface water and porewater, results from this assessment do not indicate adverse levels of exposure through the uptake of surface water COPECs. However, sediment can also be an exposure source to aquatic plants because COPECs can desorb from sediment and become available for biotic uptake over time. Exposure to sediment concentrations of trace elements in seasonally-exposed and submerged sediment did not indicate moderate or unacceptable risks to aquatic plants.

Tissue concentrations of trace elements in periphyton and aquatic vegetation were also evaluated to estimate potential bioaccumulation of COPECs from the exposure pathways described above. Available data do not indicate COPEC concentrations are higher in emergent or shoreline aquatic vegetation growing within downstream reaches compared to the reference population. Therefore, ash-related risks to shoreline and emergent vegetation are unlikely. Only arsenic, copper, and strontium were higher in periphyton tissue collected from downstream reaches. However, tissue concentrations of these COPECs are not expected to pose unacceptable risk to the periphyton communities based on a comparison of tissue concentrations to available literature effects values.

For each LOE, the evidence of causality, estimate of magnitude of adverse effects, confidence in magnitude of adverse effects, and ecological relevance is integrated and presented in Exhibits 5-1 and 5-2. Overall, evidence of causality is moderate with a low estimate of magnitude of adverse effects and low confidence in the magnitude of adverse effects. Uncertainties associated with these descriptions are described below.

Exhibit 5-1. Aquatic Vegetation Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Periphyton Tissue Concentrations	Low	Low	Moderate	Moderate
Aquatic Vegetation Tissue Concentrations	Negligible	Low	Moderate	Moderate
Surface Water Chemistry	Negligible	Moderate	Moderate	Low
Sediment Chemistry	Low	Low	High	Low
Weight of Evidence	**Low**	**Low**	**Moderate**	**--**

Exhibit 5-2. Aquatic Vegetation Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Periphyton Tissue Concentrations	Moderate	Low	Moderate	Arsenic
Aquatic Vegetation Tissue Concentrations	Moderate	Low	Moderate	--
Surface Water Chemistry	Low	Low	Moderate	--
Sediment Chemistry	Low	Low	Moderate	Arsenic, Manganese, Nickel, Zinc
Weight of Evidence	**--**	**Low**	**Moderate**	**--**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

5.3.2 Uncertainties

Uncertainty is inherent in many aspects of field surveys and BERA, particularly in systems with complex COPEC mixtures. Additionally, the Site consists of confluences of three river systems, potential legacy metals contamination in upstream reaches, and potential upstream migration and deposition of COPECs. Each of these factors can confound interpretation of spatial trends, which subsequently introduces uncertainty into estimates of risks. Other sources of uncertainty related to the characterization of potential risks to aquatic plants include: inter-species differences in bioaccumulation and sensitivities to constituents, lack of available toxicological benchmarks for several COPECs, bioavailability of COPECs, and study design limitations. Focused discussions of the key uncertainties for evaluating aquatic plant risks are discussed below.

5.3.2.1 Inter-Species Differences in Bioaccumulation Potential and Sensitivity to Trace Elements

Aquatic vegetation sampling locations were selected based on the availability of emergent vegetation, resulting in the collection of a variety of emergent and shoreline species. Periphyton communities may have also varied across sampling locations due to physical and chemical habitat differences. Thus, inter-species differences in bioaccumulation of metals typically observed for aquatic plant species may bias the spatial trends observed in these results. As discussed above, aquatic plant species also exhibit wide ranges in their sensitivity to trace elements. This introduces uncertainty in the characterization of risks to aquatic plants because effect levels reported in the literature may not adequately represent the range of aquatic plant sensitivities.

5.3.2.2 Lack of Toxicological Benchmarks

Most of the available effect data relevant to aquatic plants are based on aqueous exposures. Fewer effect values are available that report adverse effects associated with tissue or sediment concentrations of trace elements. This introduces uncertainty in characterizing potential risk associated with tissue and sediment exposures. Eco-SSL values, which are based on contaminant exposures in upland soil, were used as conservative effects levels to characterize potential risk associated with sediment exposures, which likely results in an overestimation of risks.

5.3.2.3 Bioavailability of COPECs

Bioavailability of trace elements can be influenced by several environmental conditions including pH, TOC content, suspended solids, and presence of other co-occurring elements. Because the bioavailability of COPECs in environmental media is expected to be less than in exposure media used in toxicity tests, risks estimates based on comparing media concentrations to laboratory toxicity tests likely overestimates risks.

5.3.2.4 Study Design Limitations

Aspects of the present study design also limit interpretation of spatial exposure trends, including small sample sizes ($n = 3$ for emergent and shoreline vegetation per location; $n \leq 2$ for periphyton per location) that decrease statistical power and, thus, decreases the ability to detect significant differences when differences may truly exist. Another factor contributing to uncertainty is the collection of different species of vegetation across sampling locations, which may bias spatial bioaccumulation trends. In addition, only plant biomass emerging from the ground or water surfaces were analyzed for metals. Because plant roots can be a significant source of accumulated metals relative to aboveground biomass (Outridge and Noller 1991), it is possible that these tissue results underestimate concentrations in aquatic vegetation.

5.3.2.5 COPEC Selection

Some trace elements are essential for normal plant growth (e.g., potassium, calcium, magnesium, iron, boron, manganese, zinc, copper, molybdenum, nickel, and selenium) (Epstein 1965), although both essential and non-essential metals (e.g., mercury, cadmium, and lead) can result in growth inhibition and toxicity (Vardanyan et al. 2008).

5.3.2.6 Analytical Uncertainties

A rigorous QA/QC program was implemented to minimize the inherent uncertainties associated with chemical and physical analyses. The majority of chemical analyses were performed using the best commercially available methods and the results were reported to the lowest technically defensible level. Even so, the detection limits for some constituents, notably selenium in water and sediment, were higher than potentially ecologically important concentrations. This is a well recognized and largely unavoidable issue for some bioaccumulative constituents like selenium. This may lead to underestimation of exposures. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty.

5.3.3 Summary and Conclusions

In summary, the evaluation of risks to aquatic plants uses a combination of conservative and central tendency estimates for the exposure assessment and conservative estimates of effects. As a result, risk estimates to aquatic plants are likely overestimated. Based on evaluation of multiple exposure pathways including surface waters, sediment porewaters, seasonally-exposed sediments, and submerged sediments, this BERA demonstrates that risks to aquatic plants are low.

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6. Aquatic- and Riparian-Feeding Birds

Aquatic- and riparian-feeding birds are an important component of a functioning ecosystem at the Site. These birds represent a group of species dependent on aquatic organisms potentially affected by the release. These species are exposed to COPECs from the ash in the sediments at the Site through the aquatic and riparian food web and through direct ingestion of sediments and water. Birds have been shown to be sensitive to toxic effects of many of the COPECs in the ash. Therefore, aquatic- and riparian-feeding birds were a focus in this BERA.

6.1 Problem Formulation

Aquatic- and riparian-feeding birds in the impacted reaches of the Emory, Clinch, and Tennessee Rivers may be directly exposed to both inorganic and organic COPECs through drinking the water in the rivers and incidentally ingesting sediments while feeding. They may be indirectly exposed through feeding on organisms that have bioaccumulated COPECs. Such exposure, whether direct or indirect, may cause toxicity in the form of reduced reproduction or survival, leading to a decline in the bird populations and bird species diversity. Foraging habitat must be sampled to evaluate the potential for such declines. The types of habitat sampled that support these species, the sampling media, assessment and measurement endpoints, and risk questions to be answered by this assessment are discussed below.

The riparian habitat where these birds forage is classified as Appalachian oak forest. Pine beetle infestations and land development create a mosaic of forest types, shrub/brush, and early successional habitats. Land uses such as recreational and industrial parks, barge terminals, and residential shoreline developments and marinas occur along the shoreline. Warm-season grasses have been planted in some shoreline areas to support grassland bird species (TVA 2009a). Tree cover is approximately 59 to 64 percent within 100 ft of the shoreline.

Wetlands are another habitat type where these birds forage. Wetlands are productive habitats for birds and are found in embayments and along the aquatic to terrestrial shoreline gradient (TVA 2009a). Forested wetlands are the most common wetland type followed by scrub-shrub and emergent wetlands. When the water is drawn down with release of water from the dam, mudflats form exposing sediment.

The aquatic habitat for feeding is largely affected by surrounding land use and bathymetry. In residential areas, seawalls and riprap dominate the shoreline. In undeveloped areas, fallen trees, bedrock outcrops, shallow shoreline gravel beds, and rubble and cobble beds on steeper shorelines are present. Aquatic plant beds are rare (TVA 2009a).

The exposure media sampled in these habitats to evaluate risk to aquatic- and riparian-feeding birds are surface waters, shallow sediments (seasonally-exposed), tissue samples of diet items, and tissue

concentrations of piscivorous bird eggs. Organisms sampled that could be part of the diet of herbivores, omnivores, and carnivores include emergent and near-shore vegetation, benthic and emergent invertebrates, amphibians, and fish.

6.1.1 Assessment Endpoints

As described in Section 2.2 of this BERA, assessment endpoints are used in the BERA process to identify focus areas for the risk assessment and develop a method for evaluating ecological risks. The initial selection of the assessment endpoints was presented in BERA Methodology of the SAP (Jacobs 2010a). As discussed in that methodology, the assessment endpoints for aquatic- or riparian-feeding birds used to develop this BERA are key trophic levels of herbivores and carnivores and those in-between that are omnivores. The assessment endpoints selected are:

- Aquatic- or Riparian-Feeding Herbivorous Bird Populations
- Aquatic- or Riparian-Feeding Insectivorous (Omnivorous) Bird Populations
- Aquatic- or Riparian-Feeding Carnivorous (Piscivorous) Bird Populations.

Each of these groups has important ecological functions that have the potential to be impaired by the ash. Aquatic- and riparian-feeding herbivorous bird populations, such as wood duck (*Aix sponsa*), found on the Site rely primarily on aquatic vegetation as forage. They inhabit the shallow or seasonally-exposed sediments of riverbanks and other water bodies, feeding primarily along the shore and on mudflats. These herbivores play an essential role in the aquatic ecosystem by transferring energy available in plant tissue (primary producers) to animal tissue, in turn making it available to upper trophic-level organisms. These birds potentially regulate vegetation density, species abundance, and diversity and cycle nutrients through feeding, in addition to serving as prey items for upper trophic-level predators. Therefore, aquatic- and riparian-feeding herbivorous birds are vital to energy flows and nutrient cycling in a balanced ecosystem and may influence the abundance or composition of the vegetation or carnivorous communities.

Aquatic- and riparian-feeding herbivorous birds not only have the potential for direct and indirect exposure producing adverse effects, but are a potential source for bioaccumulation and transfer of COPECs to higher trophic-level consumers. Because they make up a large component of the aquatic food web, providing an important food resource for carnivorous organisms such as larger birds and mammals, they could put their predators at risk.

Aquatic- and riparian-feeding omnivorous bird populations consume mostly plants and invertebrates, but in the definition of this BERA, they are referred to as insectivorous (omnivorous) bird populations because they include not only omnivorous ducks, but many birds that are almost exclusively insectivorous such as small shorebirds. This group, however, does not include aerial insectivores, which are addressed in a separate section of the BERA. In addition to linking energy pathways in an aquatic ecosystem, these omnivores

forage on shallow or seasonally-exposed sediments of riverbanks and other water bodies, feeding in the water, along the shore, and on mudflats. Common species at the Site include the mallard (*Anas platyrhynchos*) and killdeer. Omnivorous birds may serve as the prey items for upper trophic-level predators. As a result, aquatic- and riparian-feeding omnivorous birds are vital to energy flows and nutrient cycling in a balanced ecosystem, and possibly influence the abundance or composition of aquatic vegetation and invertebrate communities. Similar to the herbivores, aquatic- and riparian-feeding omnivorous birds have the potential for direct and indirect exposure causing adverse effects, as well as the potential for bioaccumulation and transfer of COPECs to higher trophic levels.

Aquatic- and riparian-feeding carnivorous bird populations that are not insectivorous rely primarily on fish as forage at the Site but also consume small mammals, amphibians, and reptiles. For this BERA, they are referred to as aquatic- and riparian-feeding carnivorous (piscivorous) bird populations. Carnivorous species that are piscivorous, such as the osprey (*Pandion haliaetus*), feed by hovering over the water, often shallow water, and diving feet first to catch their prey with their talons. Other species, such as the great blue heron (*Ardea herodias*), feed on fish by wading in the shallow or seasonally-exposed sediments of riverbanks and other bodies of water. Piscivores play an essential role to an ecosystem by transferring the energy available in animal tissue to upper trophic levels. These populations of birds potentially regulate fish and other aquatic organisms' density, species abundance, and diversity, but may also serve as prey items for the next upper trophic-level predators and carrion-feeders and may influence the structure of those communities. Similar to the other trophic levels, aquatic- and riparian-feeding piscivorous birds are vital to a balanced ecosystem.

Piscivorous birds that are upper trophic-level predators are especially susceptible to exposure to certain COPECs that have accumulated in the tissues of their prey organisms. Increase of concentrations within an individual organism is bioaccumulation. Concentrations of some organic COPECs in tissue, such as PCBs, further increase with each trophic level due to a process known as biomagnification. Fish in particular have been shown to biomagnify some COPECs that are present in aquatic ecosystems such as organic COPECs and organic forms of selenium, mercury, cadmium, and arsenic. The inorganic forms of some of these constituents also can bioaccumulate. Birds that consume fish and invertebrates have the potential to biomagnify concentrations of COPECs in their tissues.

The species selected to represent each assessment endpoint (described in Appendix U), referred to as the receptor species, are as follows:

- Aquatic- or Riparian-Feeding Herbivorous Bird Populations:
  - Wood duck
- Aquatic- or Riparian-Feeding Insectivorous/Omnivorous Bird Populations:
  - Mallard – omnivore
  - Killdeer – insectivore

- Aquatic- or Riparian-Feeding Carnivorous (Piscivorous) Bird Populations:
  - Great blue heron
  - Osprey.

The ecological risk questions established for the evaluation of aquatic- or riparian-feeding birds at the Site are as follows:

- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or foods pose unacceptable risks to aquatic- or riparian-feeding herbivorous bird populations?
- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or foods pose unacceptable risks to aquatic- or riparian-feeding omnivorous bird populations?
- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or foods pose unacceptable risks to aquatic- or riparian-feeding carnivorous (piscivorous) bird populations?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective needs to be evaluated in the WOE assessment. These risk questions are used to help focus subsequent steps of the aquatic- or riparian-feeding bird assessment, including the selection of measurement endpoints.

6.1.2 Measurement Endpoints

The measurement endpoints for each assessment endpoint of aquatic- or riparian-feeding birds selected for this BERA are as follows:

Measures of exposure:

- COPEC concentrations (total) in surface waters.
- COPEC concentrations in sediments, with speciation of concentrations for arsenic and selenium to account for relative bioavailability of the different forms.
- COPEC concentrations in tissues of dietary items.
- Modeled COPEC dietary doses in mg/kg-day from diet, accounting for bioaccessible fractions in sediments (sum of concentration of four steps of sequential metals extraction ending at the manganese-iron oxide-bound fraction extracted at pH of 1.5) of vanadium, aluminum, manganese, copper, and lead (Appendix W).
- COPEC concentrations in eggs of piscivorous birds (osprey and great blue heron).

Measures of effects:

- Screening-level literature-derived toxicity dietary doses representing NOAELs and LOAELs for bird growth or reproduction for each assessment endpoint
- For COPECs that did not pass the screen, refined literature-derived, dietary doses tailored to be more site-specific representing NOAELs and LOAELs for bird growth or reproduction
- TRVs for egg concentrations representing benchmarks for toxic effects to the embryo (EC_{10} or LOAEL)
- Egg health metrics (weight, volume, length, width) of great blue heron and osprey eggs.

6.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to aquatic- or riparian-feeding birds. The LOEs used to evaluate aquatic- or riparian-feeding birds include:

- Heron and osprey egg tissue data and associated clutch size and egg health metrics
- Dietary exposure models for wood duck, mallard, killdeer, great blue heron, and osprey.

The heron and osprey egg results mainly apply to the carnivorous (piscivorous) bird assessment endpoint, whereas the dietary exposure results apply to the three assessment endpoints. Each LOE is summarized below.

6.2.1 Piscivorous Egg Tissue Concentrations of COPECs and Egg Health Metrics

Egg concentrations of metals can help identify whether adverse effects of the ash to the bird population are likely. If metals/metalloids exceed concentrations known to be associated with adverse effects to embryos or chicks and the exceedances can be linked to the ash, this would provide one LOE for estimating the level of risk to aquatic- and riparian-feeding bird populations at the Site. Eggs were collected from heron and osprey nests in impacted and reference locations in 2009 and 2010 to evaluate spatial and temporal variations in egg content concentrations bioaccumulated through the diet (discussed in Section 6.2.2) and subsequently maternally transferred to the egg. Egg tissue was analyzed to obtain concentrations of 26 metal and metalloid COPECs expected in the ash. Eggs were measured for weight, volume, length, width, and clutch size to evaluate whether constituents were impacting egg development. Appendix V details this egg study including the study objectives, location of nests, methods, results (including tables and figures), and interpretation. Below is a summary and interpretation of the results.

6.2.1.1 Analysis of Exposure and Effects

Eggs from heron and osprey nests may be representative of carnivorous (piscivorous) birds' exposures to ash COPECs because these birds are common at the Site and reference locations. Locations of heron and osprey eggs collected (one per nest) are shown in Figure 2-24. For each species, egg concentrations were

statistically compared between impacted and reference locations during spring 2009 and 2010, a period before dredging was completed. The analysis of heron egg concentrations collected post-dredging in spring 2011 (osprey eggs were not collected) were not available for statistical evaluations at the time of this assessment. Therefore, the 2011 egg data are discussed after a quick review of maximum egg concentrations in the uncertainty section. Overall, the evaluation of this LOE for residual risk after the dredging operation is conservative because the data analyses do not yet incorporate the positive effects of dredging. Also, exposures during dredging activities could be higher because of resuspension and enhancement of bioaccessibility and bioavailability of ash-related COPECs.

A two-way ANOVA (or non-parametric equivalent) was used to assess the effects of location (reference, impacted) and year (2009 and 2010) on egg concentration. If the results showed a significant location effect (where $p \leq 0.05$) or almost significant effect (where $0.05 \leq p \leq 0.1$) with higher concentrations in the ash-impacted locations, ash may be responsible for the higher concentrations. If, additionally, the interaction term between year and location is significant, the increase above reference location values was greater in one year than the other. A conservative one-sided post-hoc Dunnett's test comparing impacted locations to reference locations was employed to identify if any impacted site had greater COPEC concentrations than the reference site. Results that were close to showing a significant difference were evaluated for risk to be conservative; however, power to detect biologically meaningful differences from the reference sites using two-way ANOVA was generally 80 percent with 95 percent confidence, even with small sample sizes (great blue heron egg n = 5 in each location in 2009 and n = 9 and 4 in 2010; osprey egg n = 2 to 3 in 2009 and 1 to 5 in 2010, Appendix V). Power was less for detecting clutch size differences of one, reducing confidence in those results. Meaningful differences in concentrations (delta to be detected) were set at the difference between the reference value and benchmark for adverse effects or lower, if possible, and was between 2 to 5 mg/kg (with the exception of zinc, which was 30 mg/kg).

Of the 26 analytes, only a few were higher than the reference location for the heron eggs and none were higher for the osprey. Selenium and copper in heron eggs had significantly higher concentrations in ash-impacted locations (Appendix V, Figures 16 and 13, respectively). Mercury was almost significantly higher in the ash-impacted locations (Appendix V, Figure 15). In no case was the interaction term significant for either the osprey or heron egg concentrations, although the interaction term for copper was almost significant ($p = 0.10$), showing a much smaller increase above the reference location in 2010 compared to 2009 (Appendix V, Figure 13).

In all of these cases, the magnitudes of differences were small and not likely biologically meaningful to population viability. Mean concentrations of impacted heron eggs had less than 1 mg/kg dry weight more selenium than reference eggs in 2009 and in 2010. For copper, one outlier in 2009 with very high copper concentrations (18.5 mg/kg dry weight) caused the significantly higher average copper in the impacted location (Appendix V, Figure 6). The other eggs had copper concentrations < 9.6 mg/kg, including ten eggs collected in 2011 in the impacted reach with a maximum of 5.2 mg/kg dry weight. Reference heron eggs

were less than 7.1 mg/kg copper dry weight, which is about the upper end of background concentrations in dry weight for piscivorous birds reported in the literature (2.1 mg/kg wet weight equals approximately 7 mg/kg dry weight, King et al. 1983). For mercury, mean concentrations of impacted eggs were less than 0.1 mg/kg dry weight higher than reference eggs in 2009, while 2010 impacted eggs were approximately 0.4 mg/kg dry weight higher than reference eggs.

The egg health metrics and clutch size did not significantly differ between impacted and reference locations for the heron. However, for osprey, egg length and volume were close to being significantly lower in the impacted locations (0.3 and 0.1 centimeter shorter in 2009 and 2010, respectively), but no analytes were higher relative to the reference locations to implicate ash as a potential cause. Moreover, no analytes were negatively correlated with egg health metrics or clutch size for either the heron or osprey eggs. Osprey clutch size was significantly different among locations, but mean clutch size was higher at the most impacted Emory and Clinch Rivers locations than at the reference or more downstream Tennessee River locations, a finding that does not implicate ash as decreasing egg production.

6.2.1.2 Risk Estimation

Selenium, copper, and mercury concentrations in 2009 and 2010 appeared to be slightly higher in heron eggs at the ash-impacted reaches relative to the reference reaches, but not in osprey eggs. The increase is from maternal transfer to eggs, which has been documented in various species of wild aquatic birds for selenium, copper, and mercury (Agusa et al. 2005; Heinz et al. 1989; Heinz et al. 1987; King 1988; King et al. 1994; Ohlendorf et al. 1986). Selenium concentrations above toxic thresholds can reduce hatchability, increase teratogenic deformities of embryos, and increase offspring mortality. Similarly, mercury concentrations above toxic thresholds have been documented to reduce hatchability, cause eggshell loss, and egg infertility (Shore et al. 2011). Excessive copper fed to the laying bird can result in maternal transfer and reduced hatchability, egg fertility, and egg weight (Elsayed and Wakwak 2010).

None of the egg concentrations for these three metals for osprey or heron eggs at the Site exceeded known toxic thresholds. Mean background concentrations of selenium in eggs are generally thought to be below 3 mg/kg dry weight, with individual egg concentrations generally less than 5 mg/kg dry weight (Ohlendorf and Heinz 2011). Adverse effects to hatchability begin at concentrations > 7.7 mg/kg dry weight for the mallard, the most sensitive species (Janz et al. 2010). A black-necked stilt (*Himantopus mexicanus*) field study that suggested lower levels than this threshold was found to need re-analysis. The re-analysis indicated a higher threshold for stilts than for mallards (Adams et al. 2003). Sensitivity varies greatly among bird species and may be much lower for piscivores (Attachment 3 of Appendix W) than omnivorous ducks. Effects on reproduction did not occur until > 16.5 mg/kg dry weight of selenium in eggs for the piscivore, black-crowned night heron (*Nycticorax nycticorax*) (Janz et al. 2010), which might have similar sensitivity to the great blue heron. Mean concentrations in heron eggs of selenium (2.9 mg/kg dry weight in 2009 and

3.64 mg/kg dry weight in 2010) fall into the background concentration range, and no heron or osprey egg exceeded any adverse threshold (maximum was 5 mg/kg dry weight).

Thompson (1996) reviewed ranges of mercury concentrations in eggs causing adverse effects and concluded egg concentrations up to 2.5 mg/kg dry weight appear to have little detrimental effect on reproduction. The maximum concentration of mercury in heron or osprey eggs was 1.99 mg/kg dry weight, below this threshold of 2.5 mg/kg dry weight.

Similarly, most piscivorous egg copper concentrations at the Site fell in the range of background concentrations (< 7 mg/kg dry weight, King et al. 1983), except three eggs in the impacted reach that had 7.5, 9.6, and 18.3 mg/kg dry weight, respectively. Unlike selenium and mercury, threshold egg concentrations for adverse effects from copper are not well documented. One study injected eggs with copper and identified a median lethal dose (LD_{50}) for the embryos at about 20 mg/kg dry weight (converted to dry weight with 80 percent moisture; Ridgway and Karnofsky 1952) (Appendix V). Such a median threshold is well above all but one egg concentration at the Site, but the lowest adverse effect level is unknown. Egg injection studies often underestimate the median lethal or lowest adverse threshold compared to studies with natural maternal transfer (Heinz et al. 2009), and it is quite possible eggs with < 10 mg/kg dry weight of copper, true of the piscivorous eggs collected in 2010 and 2011, are not adversely affected. The egg with 18.5 mg/kg dry weight copper seems high, but was found in 2009 and may not represent conditions in 2010 or in 2011 post-dredging. Thus, risk from copper to piscivorous bird populations appears to be low.

Table 6-1 summarizes analytes exceeding reference areas in piscivorous bird eggs and the assessment of the level of risk of the residual ash to the piscivorous receptor group. In summary, risk from the 26 metal and metalloid COPECs evaluated is too low to be of concern, although some uncertainty exists for copper. Unlike mercury and selenium, adverse effects levels are not established for copper in eggs, probably because, once nutritional needs for copper are met, increasing concentrations in the diet increases adverse effects on reproduction, yet maternal transfer of copper to eggs is not proportionally increased. For example, a diet with 450 mg/kg of copper reduced egg production of chickens, but egg concentrations of those birds were similar to birds fed a control diet (Jackson and Stevenson 1981). Thus, copper effects are best evaluated using dietary exposure models.

6.2.2 Dietary Exposure Models

Aquatic- and riparian-feeding birds are exposed to ash COPECs through dietary uptake from the food web and abiotic media. A dietary exposure model calculates the dose predicted for each of the receptor bird species representative of the three assessment endpoints, herbivores, omnivores, and carnivores (piscivores). The five representative species for which a dose was modeled are the herbivorous wood duck, omnivorous mallard, omnivorous (insectivorous) killdeer, and piscivorous great-blue heron and osprey. The calculated dose is compared to a TRV, which is a threshold above which adverse effects are likely. The

exact TRV is uncertain and quantified from the literature as the NOAEL and LOAEL. It is assumed the actual threshold is between the two. The quotient of calculated dose divided by TRV is the HQ, which helps quantify risk. An HQ > 1 has potential for adverse effects on the individual and possibly the population. Of the 26 COPECs, 23 were inorganic and three were organic (organic only sampled in fish) and were assessed for risk. Additionally, radionuclide exposure and risk was evaluated. Details on the dietary exposure modeling, including the objectives, methods, results (including tables and figures), interpretation, and uncertainty of the results and interpretation are provided in Appendix W. The radionuclide evaluation using the RESRAD-Biota model is detailed in Appendix X. The dietary exposure modeling and results in those appendices are summarized below to assess risk from this LOE.

First, a screening model with highly conservative parameters was run to screen out COPECs that have a NOAEL HQ < 1, indicating negligible risk even under highly conservative assumptions and TRVs. Second, a refined model was run that had more site-specific exposure parameters and NOAEL and LOAEL TRVs. Third, for COPECs remaining with HQ > 1, a probabilistic model was run that incorporated the variability and uncertainty of the parameters and TRVs in the model (last step not performed for radionuclides) to estimate the probability of an HQ > 1.

The deterministic (not probabilistic) dietary exposure model equation is described in USEPA (1999b). Appendix U describes the source of the species-specific exposure parameters in the model, specifically body weight, ingestion rate, and diet. The model was run for each reach of the Emory, Clinch, and Tennessee Rivers. Impacted reaches were paired with reference reaches in the same river to evaluate if doses and, thus, HQs, are higher in the impacted reaches relative to the reference reaches. Input parameters into the dietary exposure model for each COPEC, receptor species, and reach include concentration of COPEC in food items, drinking water, and incidentally ingested sediment; proportion of diet in each food item; food, water, and sediment ingestion rates; bioavailability of the COPEC in food and sediment; and site foraging frequency. The modeling equation and selection of input parameters are provided in Appendix W.

Exposure parameters that changed between the screening and refined analyses are the bioavailability factor and, for the radio-nuclides, site foraging frequency. In the screening model, the bioavailability of COPECs, from both tissue and incidentally ingested soil, was set at 100 percent. In the refined models, the bioavailability of COPECs from tissue was retained at 100 percent for most COPECs. The exceptions are the fish and invertebrate gut for iron and aluminum (most likely bound with incidentally-ingested sediment, rather than food in gut) and the invertebrate body for iron (most iron encrusted on body is not bioavailable), which were treated similar to sediment. Additionally, iron in plants was assigned a bioavailability of 20 percent (high end of range of 2 to 20 percent, USNIH 2012). The bioavailablity of COPECs from incidentally ingested sediment was estimated using bioaccessibility fractions. Bioaccessibility fractions were estimated from sequential extraction procedures, using the summed concentrations of the fractions of metals released that were water soluble, exchangeable, carbonate-bound, and Mn-Fe-bound. The last step was at pH 1.5 to prevent metals from re-adsorbing and to model the function of a bird stomach. A bird stomach has a pH of

2 to 3 which makes metals less bioavailable than at 1.5, but the conservative pH might offset the SEP method that underestimates metals bound to the organic-fraction. The accuracy of this method is uncertain due to reagent non-selectivity and re-adsorption (Filgueiras et al. 2002), but Karadas and Kara (2012) found these fractions produce results approximately similar to in vitro bioavailability tests simulating animal stomachs for many metals with some exceptions (chromium and to some extent copper). Sequential extraction procedures were performed on all the bulk sediment (18 samples) that were collected for toxicity testing from the Emory and Clinch Rivers. More detail about the bioaccessibility estimates is provided in Attachment 2 of Appendix W.

Site foraging frequency was set to 1 (always forages within the Site) in the screening and refined analysis except for the radionuclides, where it was set to 0.5 in the refined analysis because riparian- and aquatic-feeding animals are assumed not to be physically exposed to the radionuclides in sediment/water about half the time (Appendix X). Some input parameters to the model are variable and, thus, were allowed to randomly vary in the probabilistic version of the model, according to the mean and standard deviation and distribution of that parameter. The variability in the calculated dose (and resultant probability of exceeding HQ of 1) in the probabilistic model mostly captures measured variability in the parameters used to calculate dose; whereas, the variability in the TRVs captures uncertainty in the TRVs.

The Monte Carlo probabilistic analysis was conducted following USEPA (1997b) guidance, but only for COPECs exceeding HQs > 1 in the refined analysis with higher dose in the impacted reaches than the reference reaches. Input parameters that were varied included COPEC concentrations in food, water, and sediment; body weight (which will vary ingestion rate in a Nagy [2001] allometric equation); and bioavailability of the COPEC. The probabilistic model was run for each reach and then for each river (reaches combined to increase sample size). The locations of sampled food items and abiotic media that could be ingested (exposed sediment and mid-column river water) are shown in Appendix W (Figures 1 and 2, respectively) for each reach in the Emory River (ER_Ref, ER_A, ER_B, and ER_C), Clinch River (CR_Ref, CR_A, and CR_B), and Tennessee River (TR_Ref, TR_A, and TR_B). The timing of sample collections used in the model is shown in Table 2-12, which indicates many tissue samples used were collected prior or during dredging except plant tissue. Water and sediment samples were collected after dredging. Only 2010 samples for abiotic and biotic media were used in the dietary exposure models. The output of the deterministic modeling was an HQ, and the outcome of the probabilistic modeling was the probability of HQ exceeding 1. The probabilistic model also produced tornado diagrams that indicate which parameters in the model most influence the results, which helps evaluate the relative uncertainty of parameters in the dietary exposure model.

In Appendix W, Tables 11-1 to 11-12 provide the HQ results of the screening analysis, Tables 12-1 through 12-10 provide the HQ results of the refined analysis, and Tables 13-1 through 13-6 provide the results of the probabilistic analysis. Radionuclide results for riparian wildlife are in Appendix X. The results are summarized by receptor below.

6.2.2.1 Herbivores (Wood Duck)

The screening-level HQs of COPECs for the wood duck, a receptor representative of herbivores, were below 1 except for aluminum, iron, radionuclides (radium-228, radium-226, potassium-40), and vanadium. In the refined analysis, no HQ exceeded 1 (Table 6-2). The results suggest ash is not likely causing adverse effects to herbivorous aquatic- and riparian-feeding birds, and risk was negligible.

6.2.2.2 Omnivores (Mallard)

The screening-level HQs of COPECs for the mallard, a receptor representative of omnivores feeding on benthic invertebrates and vegetation, were below 1 except for aluminum, arsenic, iron, radionuclides (radium-228, radium-226, potassium-40), selenium, and vanadium. In the refined analysis, selenium HQ still exceeded 1 in ER_A, ranging from 0.8 to 2 (Table 6-2).

The deterministic NOAEL HQ for a mallard in the refined analysis is based on conservative laboratory studies and conservative exposure point concentrations (EPCs), resulting in this HQ only slightly above 1. Using less conservative assumptions, the probability of the selenium HQ exceeding 1 for mallards in the probabilistic model is essentially 0 percent (Table 6-3). It is near zero (0 to 2 percent) because the vegetation in its diet dilutes the dose of selenium from the invertebrates. The results from the model support the conclusion that risk of the residual ash to mallards appears to be low.

6.2.2.3 Insectivores (Killdeer)

The screening-level HQs of COPECs for the killdeer, a receptor representative of insectivores feeding on benthic invertebrates (conservatively assumed to feed 100 percent on such invertebrates), were below 1 except for aluminum, arsenic, lead, iron, radionuclides (radium-228, radium-226, potassium-40), selenium, and vanadium. In the refined analysis, arsenic HQ still exceeded 1, but only in ER_A (Table 6-2). In contrast, high aluminum and selenium doses were more ubiquitous, with HQs exceeding 1 and HQs of the reference reaches in most reaches of the Emory and Clinch Rivers (A and B, Table 6-2). The HQs of NOAELs and LOAELs ranged from 0.5 to 3 for aluminum, 1 to 5 for selenium, and 1 to 3 for arsenic.

The probability analysis indicates killdeer HQs for aluminum have high uncertainty. The probability of aluminum HQ exceeding 1 ranges from 3 percent (LOAEL) to 100 percent (NOAEL) in the four impacted reaches with exceedances > 1 (ER_A, ER_B, CR_A, and CR_B; Table 6-3). The probability analyses are more informative estimates of risk than the refined, deterministic models, because they evaluate the variability in EPCs of abiotic and biotic media, rather than the conservative upper confidence limit (UCL) or maximum concentration. They also consider the variability of TRVs across many species, not just the most sensitive species tested, and incorporate the variability in TRVs reported in the literature. Aluminum TRVs

are not well-defined and lack strong, sound studies, resulting in a large range and standard deviation, which in turn increases uncertainty.

Notable, although aluminum is a COPEC, comparisons of sediment concentrations of aluminum and ash for the assessment of the killdeer's prey, the benthic invertebrates found no evidence that aluminum concentrations in the prey are due to the ash release because aluminum in sediments is poorly correlated to ash in sediments (Section 4.2.2.2 and Appendix S, Exhibit S-1). Furthermore, even though aluminum is elevated in sediments in impacted reaches, aluminum in mayfly nymphs was statistically significantly higher than reference ($p<0.05$) in the Clinch River (CR_A and CR_B) and lower Emory River (ER_A), but not in samples collected closer to the release area (ER_B or ER_C) (Section 4.2.3.1). This suggests that bioaccumulation of aluminum is higher in benthic invertebrates living in habitats with a higher proportion of Clinch River sediments, and the aluminum in ash might have less bioaccumulative potential.

The probability of exceeding an HQ of 1 for selenium and arsenic is not as variable as aluminum. In ER_A, the probability for selenium ranges from 10 percent to 24 percent, and for the other three reaches ranges from 5 percent to 19 percent (Table 6-3). The probability of exceeding an HQ of 1 for arsenic in ER_A is 5 percent to 10 percent.

The tornado diagrams show that the parameter that most affects these probabilities in the most impacted reach for aluminum and selenium (ER_A) by far is the TRV uncertainty (Appendix W, Figures 11 through 17). The variability in insect concentrations is next most important and has about 20 to 40 percent of the influence as the TRV (Appendix W, Figures 11 through 17). In contrast, insect concentration variability is the most influential for arsenic. The uncertain TRV for arsenic only has about 80 percent as much influence as the insect concentrations. For these three COPECs, variability in sediment concentrations is of least importance.

The invertebrates in the killdeer diet were collected in spring 2010 when dredging was not yet complete, thus the estimated risk might be lower if 2011 data had been available to use in the model. Overall, based on the HQs, risks to birds that feed on benthic invertebrates is estimated to be low from the three COPECs of aluminum, arsenic, and selenium. Although aluminum is retained as a COPEC for this LOE, data for sediments and mayfly nymphs indicate that observed aluminum bioaccumulation in the vicinity of the Site does not appear to be due to the ash release.

6.2.2.4 Piscivores (Heron and Osprey)

The screening-level HQs of the 26 COPECs and radionuclides for the piscivores (great blue heron and osprey) were below 1 except for total PCBs, aluminum, radionuclides (radium-228, radium-226, potassium-400), and iron. In the refined analysis, no COPEC HQs were > 1. The results suggest ash is not likely causing adverse effects to piscivorous birds and that risk is low.

6.3 Risk Characterization

A WOE approach is used to evaluate overall risk to aquatic- and riparian-feeding birds. The risk for each LOE is evaluated, and both the confidence in the risk estimate and the ecological relevance of each LOE is considered in assessing overall risk.

6.3.1 Weight of Evidence

Exhibit 6-1 evaluates each LOE for each receptor, dividing the two major LOEs into subcategories that were evaluated independently. For the egg tissue LOE, the subcategories were exposure (tissue concentrations) and effects of exposure (clutch size and egg health metrics). For the dietary exposure model, the subcategories were the four wildlife receptor species of the three assessment endpoints evaluated, which are herbivores, omnivores (one omnivore and one insectivore), and piscivores (carnivores).

Exhibit 6-1. Aquatic- or Riparian-Feeding Bird Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Piscivore – Heron and Osprey				
Egg health metrics	Low	Low	Low	Moderate
Egg concentrations	Low	High	Low	Moderate
Dietary exposure models	Low	High	Low	Low
Insectivore – Killdeer				
Dietary exposure models	Low	Moderate	Moderate	Low
Omnivore – Mallard				
Dietary exposure models	Low	Moderate	Moderate	Low
Herbivore – Wood duck				
Dietary exposure models	Low	High	Low	Low
Weight of Evidence	**Low**	**Moderate**	**Low**	**--**

Exhibit 6-1 summarizes the magnitude of adverse effects, confidence in the magnitude, evidence of causality, and ecological relevance of each LOE. Magnitude of adverse effects, indicated by amount of exceedance of benchmarks, is low for all LOE because HQs were no higher than 5 in the refined analysis, mean egg concentrations were below known benchmarks, and only one great blue heron egg might have had high concentrations (18 mg/kg in copper). Confidence in the magnitude for dietary exposure is moderate to high because the magnitude is based on the screening or refined model analysis. Confidence

in being protective is high for screening dietary exposure models, moderate for refined models, and low for probabilistic models, based on the level of conservatism incorporated into the assumptions for each. Magnitude of adverse effects is not negligible for piscivorous or herbivorous birds because screening levels of dietary exposure were exceeded for a number of COPECs (aluminum, arsenic, lead, iron radionuclides, selenium, PCBs, and vanadium), but confidence that the exceedance magnitude was no more than low is high, especially since most (except selenium, arsenic, and aluminum) were screened out in the refined analysis. Omnivorous and insectivorous birds, on the other hand, were not screened out in the refined analysis, and only have moderate confidence. For egg concentrations, confidence was high because comparison of tissue concentrations had high power (80 percent) to detect differences with high confidence (95 percent). Clutch size, egg weight and egg volume had low power, and clutch size had measurement error (sometimes estimated after hatch), resulting in low confidence in the magnitude estimate of clutch size and egg health metrics.

Because they are site-specific, measured concentrations in eggs and clutch size provide moderate ecological relevance. Clutch size and egg concentrations would have higher relevance if combined with estimates of hatching and fledgling success, but such data could not easily be collected and are unavailable. The dietary exposure models are based on site-specific biotic and abiotic concentrations and estimates of bioavailable fractions, but rely on the literature to parameterize most of the rest of the model and to evaluate effects. Thus, the results of such models are of low ecological relevance. Causality is moderate for the insectivore and omnivore that had elevated selenium or arsenic exposure above benchmarks in the dietary models but is weaker for the egg study or herbivore model in which COPECs were not clearly above benchmarks of concern.

Exhibit 6-2 summarizes the weight of each LOE, level of risk, confidence in the risk level assigned, and COPECs driving the risk. Weight is based on ecological relevance and quality of the data and analysis. Risk level is based on the refined and probabilistic analysis of the dietary models. If a HQ for the refined NOAEL is below 1, risk is negligible. If above 1 and magnitude is low, then risk is low. If sample sizes are inadequate, risk cannot be negligible but can be low with low confidence if magnitude is low. Confidence that the risk is no higher than the level indicated in Exhibit 6-2 is moderate based on refined and probabilistic results.

Exhibit 6-2. Aquatic- or Riparian-Feeding Bird Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Piscivore – Heron and Osprey				
Egg health metrics	Low	Negligible	Low	--
Egg concentrations	Moderate	Negligible	Moderate	--
Dietary exposure models	Low	Negligible	Moderate	--
Insectivore – Killdeer				
Dietary exposure models	Low	Low	Moderate	Aluminum, arsenic, selenium
Omnivore – Mallard				
Dietary exposure models	Low	Low	Moderate	Selenium
Herbivore – Wood duck				
Dietary exposure models	Low	Negligible	Moderate	--
Weight of Evidence	**--**	**Low**	**Moderate**	**Arsenic, Selenium**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

Risk is estimated to be negligible and acceptable for all but two of the subcategories of LOEs (Table 6-2). The killdeer dietary exposure model indicates that risk may exist but is low for insectivorous birds that feed in the riparian and aquatic zone in the Emory and Clinch Rivers. Confidence in this assessment of negligible (for birds not feeding on invertebrates) or low risk is moderate for all but clutch size and egg weight because the former are based on refined dietary model results and adequate sample sizes for egg concentrations. In contrast, clutch size does not have sufficient sample sizes and is assigned low confidence, and how much egg volume or weight change is required to show detrimental effects is difficult to assess effects confidently. Also, the egg results should have low weight except when applied to piscivores because they do not necessarily apply to omnivores or insectivores. Piscivores appear to be less exposed than birds feeding on invertebrates.

Aluminum concentrations in the prey items that are closely associated with sediments (i.e., mayfly nymphs) are poorly correlated with the ash release and ash in sediments. Therefore, aluminum is not identified as a final COEC (bottom row of Exhibit 6-2) based on the WOE for insectivorous birds.

Overall, the WOE supports with moderate confidence that risk is negligible for piscivores and herbivores. Risk is estimated with moderate confidence to be present but low for omnivores as represented by the mallard. Risk is estimated with low confidence to be present but low for insectivores. Risk overall, if defined

by the endpoint with the highest risk level (insectivores), is estimated with low confidence to be low in the impacted reaches of the Emory and Clinch Rivers and to be negligible in the impacted reaches of the Tennessee River. Arsenic and selenium are identified as COECs based on dietary models using site-specific invertebrate tissue concentrations to estimate exposures for insectivores.

6.3.2 Uncertainties

Uncertainties that may lead to either overestimates or underestimates of risk are associated with each stage of risk assessment. It is important to understand the major uncertainties and how they might affect the risk assessment. The uncertainties for the two LOEs for the aquatic- and riparian-feeding birds are reviewed below.

6.3.2.1 Heron and Osprey Egg Tissue Concentrations of COPECs

Issues creating uncertainty associated with evaluation of the egg tissue concentrations from heron and osprey eggs are applicability of literature-derived adverse effects levels for eggs, effects of complex COPEC mixtures, limited egg data, unknown concentrations below detection limits, seasonality of receptors, timing of egg collections, and representativeness of the reference locations. Unfortunately, reproductive measures such as hatching and fledgling success could not be documented to interpret potential effects of observed egg concentrations on productivity of piscivorous populations. Thus, interpretation relied on literature-derived values of adverse effects levels typically determined for individuals; these may not apply to this area or be applicable to populations that have adaptive and compensatory responses to stressors. Information on toxicity of copper concentrations in eggs is limited, creating uncertainty in the choice of a low adverse effect level.

Complex COPEC mixtures can have antagonistic or synergistic effects not elucidated in single-analyte toxicity studies. For example, selenium and arsenic have antagonistic modes of action whereby doses of both in combination have lower toxicity than doses of each individually (Hoffman et al. 1992). Thus, the risk estimate may be biased high for COPECs such as selenium and arsenic, both elements are predicted to bioaccumulate in these birds in some reaches.

Chemical analyses were performed using the best commercially available methods and results were reported to the lowest technically defensible level. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty. This approach increased the likelihood of exceeding a benchmark for adverse effects but could have decreased the likelihood of detecting differences between impacted and reference sites, of concern only if detection limits are near the benchmark. Detection limits for eggs were below the adverse benchmark of COPECs with readily available benchmarks (arsenic, boron, copper, lead, mercury, molybdenum, selenium, zinc, cadmium, chromium, strontium, and manganese), which provides some confidence in the results.

Benchmarks for aluminum are unknown, but aluminum detection limits were as high as 265 mg/kg in some eggs, which may or may not be well above elevated or adverse levels, given aluminum considered elevated in domestic quail eggs is 67 mg/kg (Abduljaleel and Shuhaimi-Othman 2011) and is often lower in wild birds at < 5 mg/kg (Nelson and Esmoil 1999).

Percent moisture was analyzed in biological samples so that concentrations reported in wet weight could be converted to dry weight if necessary. Insufficient sample volume was available for some biological media. Some rejected percent moisture results could be reanalyzed, but the results were qualified due to holding time exceedances. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

Safely collecting eggs from heron colonies and osprey nests high in the air over deep water is challenging without disturbing the birds. Thus, sample sizes are low, likely reducing the ability to detect differences between control and reference locations, which could bias the risk low for egg health metrics. Additionally, nests were visited only once each year, and not always during the optimum window of time to verify clutch size (sometimes just-hatched nestlings were found), thus clutch size data is compromised to some extent. A portion of the heron and osprey samples were collected during or shortly after dredging was completed. Dredging occurred only in Emory reaches B and C, and engineering controls were used to minimize migration of disturbed sediment and ash that might affect the birds.

Seasonality of receptors can change results since some piscivorous species are not year-round residents, thus are less exposed than year-round residents. Results based on year-round residents such as the heron and osprey bias the results high for migratory piscivorous birds. The reference reaches are not ideal controls for the impacted locations, especially on rivers, which by their nature change with distance downstream. Differences in egg tissue concentrations attributed to the ash could also be from natural variations in sediment conditions and food items, which are assessed in the second LOE, the dietary exposure models. A risk assessment of the residual ash post-dredging is very uncertain, given that eggs and nestlings were collected in 2009 and 2010 from birds that may have not yet benefitted from ash removal by the dredging.

These uncertainties reduced confidence in the assessment of risk from heron and osprey egg tissue to moderate or low confidence. Confidence might be increased if 2011 results were included in the risk assessment. For the 2011 heron eggs, of the 13 COPECs evaluated with sufficient detections (barium, calcium, cobalt, copper, iron, magnesium, manganese, mercury, potassium, sodium, selenium, strontium, and zinc) in the impacted heron colony of the Emory River, all but manganese had lower maximum concentrations than in 2010, and selenium, mercury, and copper maximum concentrations did not exceed potential adverse effect thresholds. In contrast, maximum concentrations of iron, manganese, and selenium increased in the reference colony on the upper Tennessee River.

6.3.2.2 Dietary Exposure Model

Many uncertainties exist in the dietary exposure model that are detailed in Appendix W. Of variables tested in a sensitivity analysis, the results generally are most sensitive to the literature-derived TRVs and secondly to the measured concentrations (food EPCs) in the food, particularly the insects. In contrast, variability in body weight, bioavailable estimates of COPECs, and sediment concentrations (sediment EPCs) have smaller effects on the results. Sediments and water in the diet provide small contributions to risk relative to biotic media (for example, Table 6-4 shows the proportions of selenium and aluminum contributions; 98 percent of the arsenic contribution for killdeer is from invertebrates).

Although variability is not high for bioavailable estimates, the accuracy of the estimates is uncertain due to reagent non-selectivity and re-adsorption of the sequential extraction procedure used to develop the estimates (Filgueiras et al. 2002). However, Karadas and Kara (2012) found the fractions used to produce bioavailable estimates produce results approximately similar to in vitro bioavailability tests simulating animal stomachs for many metals, with some exceptions (chromium and to some extent copper), although there is high individual sample variability. Nevertheless, they did not test many of the metals used in this model and it is possible the bioavailable estimates underestimate risk by some unknown amount.

Assumptions for other parameters that are fixed in the model (i.e., not subject to sensitivity analysis) also may affect the results. For example, except for EPCs and bioavailable fraction of COPECs, exposure parameters were obtained from the literature and were assumed to be applicable to the Site, which may not be accurate.

The literature-based TRVs have a significant degree of uncertainty (Dale et al. 2008). This uncertainty was incorporated into the probabilistic analysis for COPECs, but only for COPECs identified as possible risk drivers during the refined analysis. Toxicity data are only available for a limited number of species, mostly laboratory test species under a strictly defined set of test conditions that do not necessarily match natural conditions (Sample et al. 1996; Suter 1996). The ranges of tolerance had to be inferred from the data for the range of species tested. Tolerance and adaptation of the receptors are not considered for laboratory studies (Grant 2002; Millward and Klerks 2002), but do play a role in field TRV studies included in some of the Monte Carlo probabilistic models (arsenic and selenium).

Relatively few studies evaluate the effects of toxicity at the population scale that integrate reproduction and survival or incorporate concepts of carrying capacity, intraspecific competition, density-dependence, or dispersal and, thus, the TRV studies used for this receptor group rarely represent population-level effects. The large uncertainty in TRVs for aluminum and arsenic (Attachment 3 of Appendix W) makes assessment of risk from these COPECs very uncertain. For these reasons, it was critical to evaluate uncertainty of such TRVs in a probabilistic assessment. That assessment indicates with moderate confidence very little risk from arsenic, but unknown risk from aluminum. TRVs for selenium are best established for the mallard and,

hence, the confidence in the accuracy of the risk assessment for omnivorous birds such as the mallard is highest and is lower for the other assessment endpoints. However, confidence in being protective is often desired over confidence in accuracy of the assessment. There is low uncertainty in being protective with the screening level TRVs, moderate uncertainty with the refined TRVs, and high uncertainty with the TRV for the probabilistic evaluation. Such uncertainty was used to determine confidence in the risk assessment for endpoints. High uncertainty is also associated with the probabilistic model because the average and standard deviation were not always based on a robust data set. In several cases, the standard deviation and type of distribution of TRVs was assumed (e.g., aluminum). Consequently, the probability analysis is not always more accurate.

For the deterministic analysis, the EPCs used in the dietary exposure model were conservative, biasing the dose high, because the EPCs were the 95 UCL or maximum concentration for a reach (Appendix W), rather than the average. The probability analysis used the average and standard deviation of these concentrations, which is designed to be less biased.

The food item EPCs were obtained from tissues of one or a few species that may not represent the full set of food items of a receptor, particularly because groups that included terrestrial, less impacted food items were replaced with aquatic food items, biasing risk estimates high. Specifically, the diet component identified as “terrestrial invertebrates” was assessed using tissue data for aquatic invertebrates. The diet component referred to as “small mammals and birds” was represented solely by fish tissue data. Incidental sediment ingestion substituted for the incidental ingestion of soil and sediment. Additionally, body weight and the percentages of diet from the literature have some inherent uncertainty because data are neither available for an entire population nor available for all ages and all seasons.

Whole body tissue concentrations for fish and mayfly larvae were used to derive EPCs used in the exposure calculation. Using whole body prey tissue concentrations in addition to incorporating incidental sediment ingestion rates (calculated based on gut or scat contents by Beyer et al. 1994) is a conservative approach and may overestimate the risk to wildlife.

The detection limits for some constituents, notably selenium in water and sediment, were higher than potentially ecologically important concentrations. This is a well recognized and largely unavoidable issue for some bioaccumulative constituents like selenium. This may lead to underestimation of exposures and obscure the identification of exposure pathways. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty.

Percent moisture was analyzed in biological samples so that concentrations reported in wet weight could be converted to dry weight if necessary. Insufficient sample volume was available for some biological media (e.g., eggs). Although some rejected percent moisture results could be reanalyzed, the results were

qualified due to holding time exceedances. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

Mercury analyses of fish muscle tissue were found to be biased low based on QA/QC evaluations using standard reference materials. A detailed study by the project QA/QC team determined that the use of a CF of 1.3 was appropriate for fish fillet samples, but that sample type was not used in the dietary exposure models. It is yet unknown whether other tissues were similarly biased and, if so, what CF should be used. However, it is unlikely that a 30 percent increase in the reported concentrations would alter the characterization of risks in this BERA, given the low concentrations and associated low risk of mercury.

Only the leaves of emergent and shoreline vegetation were collected even though many receptors (ducks), can also feed on the roots of emergent vegetation. Roots tend to concentrate more selenium and arsenic than other plant parts (Afton et al. 2009). Risk to herbivorous birds could be biased low but not enough that if corrected for roots, it would increase HQs to above 1. Aluminum and iron do not appear to be ash-related (not correlated to ash). Vanadium HQs were all at least an order of magnitude below 1. This suggests it is unlikely that if corrected for consumption of roots it would increase HQs to above 1.

Tissue sample collection was often clustered and limited in each reach (Appendix W, Figure 2). Good representation of reach exposure concentrations and their distribution is uncertain and may under or over represent exposure (depending on if hotspots were sampled). The timing of sample collections varied from pre-dredging to post-dredging, making representation of post-dredging conditions difficult. Analysis of 2011 results would improve the evaluation but is not yet fully available. A quick review of the 2011 results recently released indicates arsenic decreased in both depurated and undepurated mayflies from 2010 to 2011. Selenium is decreasing in 2011 in undepurated larval mayflies, but increased in depurated mayflies. The selenium results suggest bioavailable selenium in tissues is still elevated in 2011.

In addition, species differ with respect to water, sediment, and food ingestion rates and absorption, metabolism, distribution, and excretion of chemicals. In particular, sediment ingestion studies have not been completed on the receptor species. Such data from other species had to be used. Because of these various uncertainties for the differing LOEs, the WOE approach was used to evaluate if the sum of the information can lead to valid conclusions, which are summarized below.

6.3.3 Summary and Conclusions

Based on the overall WOE for aquatic- and riparian-feeding birds, a low risk to bird species that feed on benthic invertebrates exists at the ash-impacted locations of the Emory and Clinch Rivers (Reaches A and B). Insectivorous species are most at risk, but the risk is low given low estimates of magnitude of adverse effects and that the invertebrate tissue concentration results which contribute the most to this conclusion were collected prior to completion of dredging. COPECs with potential adverse effects that exceeded

screening levels from the dietary exposure model or potential egg benchmarks are aluminum, arsenic, lead, iron, radionuclides (radium-228, radium-226, potassium-40), selenium, PCBs, copper, and vanadium. PCBs are a legacy COPEC not associated with the ash. COPECs driving risk based on a refined analysis were reduced to selenium, arsenic, and aluminum. Aluminum does not appear to be correlated to the ash and is not considered a COEC. The final COECs for aquatic- and riparian-feeding birds are selenium and arsenic in Reaches A and B of the Emory and Clinch Rivers.

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7. Aquatic- and Riparian-Feeding Mammals

Similar to aquatic- and riparian-feeding birds, aquatic- and riparian-feeding mammals are an important component of a functioning ecosystem at the Site, with their diets consisting partly of aquatic organisms that may have bioaccumulated ash-related constituents. These mammalian species are potentially exposed to COPECs from the ash in the sediments at the Site through the aquatic and riparian food web and through ingestion of sediments and water. Mammals have been shown to be sensitive to the toxicity of many of the COPECs in ash. Therefore, aquatic and riparian-feeding mammals were a focus of this BERA.

7.1 Problem Formulation

Aquatic- and riparian-feeding mammals in the Emory, Clinch, and Tennessee Rivers may be directly exposed to both inorganic and organic COPECs through drinking the water in the rivers and incidentally ingesting sediment while feeding. They may be indirectly exposed through feeding on organisms that bioaccumulate these COPECs. Bioaccumulation of the COPECs in the mammals from these direct and indirect sources may cause toxic effects in the forms of reduced reproduction or survival, leading to a decline in the mammalian populations and species diversity. Foraging habitat must be sampled to evaluate the potential for such declines. The types of habitat sampled that support these species, the sampling media, assessment and measurement endpoints, and risk questions to be answered by this assessment are discussed below.

Riparian, wetland, and aquatic habitat where these mammals occur and feed is described in Section 6.1 under Problem Formulation for Aquatic- and Riparian-Feeding Birds. The exposure media sampled in these habitats to evaluate risk to aquatic- and riparian-feeding mammals are surface waters, shallow sediments (seasonally-exposed), and tissue samples of diet items and adult mammals. This group includes herbivores, omnivores, and carnivores. Organisms sampled that could represent the diets of these mammals include emergent and near-shore vegetation, benthic invertebrates, amphibians, and fish. The raccoon, a mammalian species common throughout the Site, was selected to represent adult mammal tissue for this receptor group. This group, however, does not include aerial-feeding insectivores (i.e., bats) that are addressed in Section 8.

7.1.1 Assessment Endpoints

As described in Section 2.2 of the BERA, assessment endpoints are used in the BERA process to identify focus areas for the risk assessment and develop a method for evaluating ecological risks. The initial assessment approach was presented in BERA Methodology of the SAP (Jacobs 2010a). As discussed in that methodology, the assessment endpoints for aquatic- or riparian-feeding mammals used to develop this BERA are:

- Aquatic- or Riparian-Feeding Herbivorous Mammal Populations
- Aquatic- or Riparian-Feeding Insectivorous (Omnivorous) Mammal Populations
- Aquatic- or Riparian-Feeding Carnivorous (Piscivorous) Mammal Populations.

Each of these endpoints has important ecological functions that have the potential to be impaired by the ash. Aquatic- and riparian-feeding herbivorous mammal populations, such as muskrats (*Ondatra zibethicus*), rely primarily on aquatic and riparian vegetation as forage. They feed primarily along shorelines and within emergent vegetation, and make their burrows in river banks, tree root cavities, or rock or brush piles found along aquatic systems. Herbivores play an essential role in an ecosystem by transferring the energy available in plant tissue (primary producers) to animal tissue, in turn making it available to upper trophic-level organisms. These organisms may affect density, species abundance, and diversity of vegetation through feeding, in addition to serving as prey items for upper trophic-level predators and possibly influencing those communities. Therefore, aquatic- and riparian-feeding herbivorous mammals are vital to energy flows and nutrient cycling in a balanced ecosystem. Aquatic- and riparian-feeding herbivorous mammals have the potential for direct exposure resulting in adverse effects, as well as the potential for bioaccumulation and transfer of COPECs to higher trophic-level consumers. They make up a large component of the aquatic and riparian food web base, providing an important food resource for carnivorous organisms such as larger birds and mammals.

Aquatic- and riparian-feeding omnivorous mammal populations (i.e., raccoons), consume mostly plants and animals such as mollusks, crustaceans, and other invertebrates. They feed primarily along the shoreline, within the shallow or seasonally-exposed sediments of riverbanks and other water bodies, and make their dens in tree cavities or rock or brush piles found adjacent to aquatic systems. In addition to linking energy pathways in aquatic and riparian ecosystems, omnivores foraging on insects, invertebrates, and vegetation may affect the population density and composition of these food items, or may affect higher trophic-level communities because omnivorous mammals also serve as prey for higher trophic-level predators. As a result, aquatic- and riparian-feeding omnivorous mammals are vital to energy flows and nutrient cycling in a balanced ecosystem. Aquatic- and riparian-feeding omnivorous mammals have a potential for direct exposure resulting in adverse effects, as well as a potential for bioaccumulation and transfer of COPECs to higher trophic levels.

Aquatic- and riparian-feeding carnivorous mammal populations (excluding aerial-feeding insectivores), such as mink (*Neovison vison*), rely primarily on fish as forage at the Site, but may also consume small mammals, amphibians, and reptiles. For this BERA, they are referred to as aquatic- and riparian-feeding carnivorous (piscivorous) mammal populations, and the focus is on piscivores. They feed primarily along shorelines and within emergent vegetation and make their burrows in river banks, tree root cavities, or rock or brush piles found along aquatic systems. Mammalian piscivores play an essential role in an ecosystem by transferring the energy available in animal tissue to upper trophic levels. These populations of mammals may influence fish and other aquatic organisms' density, species abundance, and diversity, but may also

influence upper trophic-level predator and carrion-feeder abundance and composition by serving as their prey. As a result, aquatic- and riparian-feeding piscivorous mammals are vital to a balanced ecosystem.

Like piscivorous birds, piscivorous mammals are upper trophic-level predators, and as a result, they are especially susceptible to exposure to COPECs that may accumulate in their prey. Increase of concentrations within an individual organism is bioaccumulation. Mammals that consume fish have the potential to further biomagnify concentrations of COPECs in their tissues, particularly if a COPEC biomagnifies up the food web. Fish, in particular, have been shown to biomagnify some COPECs present in aquatic ecosystems, such as organic COPECs and organic forms of selenium, mercury, cadmium, and arsenic. The inorganic forms of these constituents also can bioaccumulate with trophic levels.

The species selected to represent each assessment endpoint of this group (Appendix U), referred to as the receptor species, are as follows:

- Aquatic- or Riparian-Feeding Herbivorous Mammal Populations:
  - Muskrat
- Aquatic- or Riparian-Feeding Omnivorous Mammal Populations:
  - Raccoon
- Aquatic- or Riparian-Feeding Carnivorous (Piscivorous) Mammal Populations:
  - Mink.

The ecological risk questions established for the evaluation of aquatic- or riparian-feeding mammals at the Site are as follows:

- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or food pose unacceptable risk to aquatic- or riparian-feeding herbivorous mammal populations?
- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or food pose unacceptable risk to aquatic- or riparian-feeding omnivorous mammal populations?
- Do concentrations of ash-related COPECs in surface waters, shallow sediments (seasonally-exposed), or food pose unacceptable risk to aquatic- or riparian-feeding carnivorous (piscivorous) mammal populations?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective needs to be evaluated in the WOE assessment. These risk questions are used to help focus subsequent steps of the aquatic- or riparian-feeding mammal assessment, including the selection of measurement endpoints.

7.1.2 Measurement Endpoints

The measurement endpoints for aquatic- or riparian-feeding mammals selected for the BERA are as follows.

Measures of exposure:

- COPEC concentrations (total) in surface waters
- COPEC concentrations in sediments, with speciation of concentrations for arsenic and selenium to account for relative toxicity of the different forms
- COPEC concentrations in tissue of dietary items
- Modeled COPEC dietary doses in mg/kg-day from diet, accounting for bioaccessible fraction in sediments (sum of concentrations of four steps of sequential metals extraction ending at the manganese-iron oxide-bound fraction extracted at pH of 1.5 [Appendix W]) of vanadium, aluminum, manganese, copper, and lead
- COPEC concentrations in tissues of raccoons
- Health metrics of exposure in raccoons, specifically gross lesions observed during necropsy, histopathology, blood count, and plasma biochemistry (considered a secondary LOE).

Measures of effects:

- Screening-level literature-derived toxicity dietary doses representing NOAEL and LOAEL for mammal growth or reproduction for each assessment endpoint
- For COPECs that did not pass the screen, refined literature-derived, dietary doses tailored to be more site-specific representing NOAELs and LOAELs for mammal growth or reproduction
- TRVs for raccoon concentrations representing benchmarks for toxic effects to mammals (EC_{10} or LOAEL).

7.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to aquatic- or riparian-feeding mammals. The LOEs used to evaluate aquatic- or riparian-feeding mammals include:

- Raccoon tissue data
- Raccoon health metrics
- Dietary exposure models for muskrat, raccoons, and mink.

Each LOE is summarized below.

7.2.1 Raccoon Tissue Concentrations of COPECs

Tissue concentrations of COPECs in the various organs and parts of an animal present evidence of exposure and can be compared to tissue benchmarks for known adverse effects to the individual animal or population. The omnivorous raccoon was selected for tissue analysis because it is common on the Site and its riparian- and aquatic-feeding habits potentially expose it to the COPECs in the sediment and bioaccumulated through the food web. Ten animals were captured from the impacted locations and five from the reference locations each year, 2009 and 2010, for tissue analysis. The objective was to compare tissue concentrations of exposed animals to reference animals over the 2 years and compare observed concentrations to thresholds of concern. Details of this study including the objectives, methods, results, and interpretation are in Appendix Y. Table 2 in Appendix Y provides the blood and tissue concentrations of the metal and metalloid COPECs analyzed that had sufficient detection frequency.

7.2.1.1 Analysis of Exposure and Effects

Raccoons were trapped and euthanized on locations impacted by the ash along the plant intake channel along the Emory River, East and North Embayments of the Emory River, and Clinch River (CRM 2.5). In 2009, the reference location was a positive control (Melton Hill Dam) that had historical ash contamination, and can serve as a reference location for the more historically contaminated Clinch River location. In 2010, the reference locations were in other locations in Knox and Grainger Counties believed to be uncontaminated by ash that may best serve as reference locations for the Emory River (Appendix Y, Figure Y in Attachment 1). The number of animals caught was 12, 7, 1, 5, and 5 in the Emory River embayments, Emory River plant intake, Clinch River, positive control, and reference location, respectively. Raccoons in 2009 were collected during dredging and in 2010 after dredging (Table 2-12). The focus of this assessment is on risk of residual ash after dredging in 2010. As a result, emphasis is placed on the 2010 data when evaluating risk. The 2010 data include an ash-unimpacted reference location helpful for evaluating potential ash effects.

Blood was collected for metal (and metalloid) concentrations of 26 COPECs. Tissues collected for analysis included hair, brain, gonad, kidney, liver, muscle, and blood. An ANOVA (or non-parametric equivalent) with post-hoc testing was performed to compare tissue and blood concentrations of four groups of raccoons: the 2009 impacted animals, 2009 reference animals, 2010 impacted animals, and 2010 reference animals (Appendix Y). The sample sizes reported for statistical analyses in Table 2 of Appendix Y were small (n = 1 to 5 for reference, n = 3 to 10 for impacted) and varied because samples with concentrations below the detection limit were omitted from the analysis. Power to detect differences may have been low, but data were not statistically analyzed unless there were at least 15 animals total in the four groups combined. Eliminating samples below the detection limit inflates the averages for groups, which is conservative when evaluating whether means exceed threshold means. However, power of detecting differences between reference and impacted locations reduces if the reference locations have more non-detects (typical of uncontaminated locations). Many tissues did not have enough detections in samples to statistically

compare, lowering the confidence in conclusions of no difference from the reference. Significant differences at the 0.05 alpha level in Appendix Y were discussed, but due to low samples sizes and to be conservative, almost all significant differences at the 0.1 level were also evaluated for potential toxicity in this BERA if the metal or metalloid was higher in the 2010 impacted location sample than one or more reference location samples.

Concentrations are reported in wet weight rather than dry weight due to percent moisture criteria not being validated (Attachment 1 of Appendix Y). COPECs near the release area that were significantly higher in concentration in at least one tissue relative to the uncontaminated reference locations were arsenic, chromium, and selenium in hair; mercury and iron in muscle; and zinc and barium in the brain.

7.2.1.2 Risk Estimation

Risk can be evaluated by comparing metal or metalloid COPEC concentrations to thresholds of concern beyond which adverse effects occur. Of the 26 COPECs measured, no tissue or blood concentrations of any COPEC measured in the raccoons exceeded any known thresholds of concern.

However, some COPECs do not have information on toxic thresholds. For example, barium and zinc concentrations were higher in the brains of exposed raccoons compared to reference raccoons. No known toxicity thresholds for the brain for barium are known, even in the well-studies omnivore, humans. No clinical signs associated with toxicity (e.g., tremors or paralysis) were observed, which suggests barium toxicity is unlikely. Zinc threshold concentrations in brain are unknown, but blood in humans has zinc concentrations typically averaging about 10 mg/kg wet weight (Friberg et al. 1979). The mean concentration of zinc in the blood of the raccoons examined in this study was less than this threshold at 8.01 mg/kg wet weight. No clinical signs or pathologic lesions associated with zinc exposure were observed in any of the animals.

Other COPECs have benchmarks for background levels or adverse effect levels for omnivorous mammals, particularly for omnivorous humans. Reported arsenic concentrations in hair of unexposed humans generally do not exceed 1 mg/kg wet weight (ATSDR 2007). The maximum in raccoon hair in impacted location samples was lower than this threshold (0.8 mg/kg). Friberg et al. (1979) reported median concentrations in hair of populations of humans ranging from 0.174 to 0.460 mg/kg wet weight. The highest mean for exposed raccoons (0.455 mg/kg) fell within this range. Additionally, the necropsy and histopathological results did not reveal any signs of arsenic toxicity.

The maximum chromium hair concentration in impacted location samples was 5.14 mg/kg wet weight in 2009, but was much lower (0.73 mg/kg wet weight) in 2010 samples collected after the dredging was completed. The median concentrations for normal human hair range from 0.2 to 2 mg/kg wet weight (Friberg et al. 1979), and the means (0.40 and 0.91 mg/kg wet weight) of the impacted location samples fell within this range. No signs of chromium toxicity were observed in the raccoons. Humans with 17 mg/kg of

chromium in their hair with long-term exposure exhibited no clinical signs of chromium toxicity (Saner et al. 1984). It is unknown how long-term exposure of chromium at the Site may affect reproduction or survival in wild raccoon populations subject to various stressors that differ from human population stresses, but individuals examined were not suffering from chromium toxicity.

The maximum selenium hair concentration in impacted location samples was 8 mg/kg wet weight in 2010, averaging 4.7 mg/kg wet weight. Concentrations in raccoon hair at a selenium-contaminated location, Kesterson National Wildlife Refuge, were much higher, averaging 21 mg/kg wet weight (maximum of 46 mg/kg wet weight, converted from dry weight assuming 10 percent moisture in hair), and yet no adverse effects in the raccoon population were observed (Clark et al. 1989). Burger et al. (2002) reported raccoons in South Carolina had mean selenium background concentrations of 2.38 mg/kg wet weight and 2.01 mg/kg wet weight for the kidney and liver, respectively. At selenium-contaminated Kesterson National Wildlife Refuge, liver selenium concentrations averaged about 3 mg/kg wet weight (converting from dry weight, assuming 85 percent moisture in liver). The highest mean selenium concentrations of raccoons in this study of 1.9 mg/kg wet weight in the kidney and 1.3 mg/kg wet weight in the liver were below the background levels and well below the Kesterson National Wildlife Refuge levels.

Iron was higher in muscle tissues relative to reference location samples, but excess iron typically concentrates in blood, liver, and spleen. Those tissues did not significantly differ from reference locations. Blood and liver concentrations of iron range from 500 to 800 mg/kg in humans (Friberg et al. 1979). Iron concentrations for raccoons were generally below this range in the study locations (means of 306 to 482 mg/kg wet weight). It is unknown what the typical iron concentrations are in raccoon blood and tissues.

Blood mercury levels of 0.2 mg/kg wet weight in humans are not likely to lead to neurologic clinical signs (Friberg et al. 1979). The mean blood mercury concentration of the nine raccoons with tissue above the detection limit was lower than this threshold at 0.01 mg/kg wet weight. The measured concentrations of mercury are unlikely to lead to toxic effects.

Aluminum was not statistically significantly ($P<0.05$) higher in impacted than reference location samples, although power to detect differences may be low. In 2010, the mean was 3.5 times higher in hair from impacted location samples than in the reference location samples. “Normal” humans have aluminum concentrations in hair up to 67 mg/kg wet weight (Yokel 1982), and the maximum hair concentration in raccoons in reference location samples in 2010 was 76 mg/kg wet weight. Exposed raccoons in 2010 had much higher mean aluminum levels in hair (126 mg/kg wet weight) than these background levels with a maximum of 474 mg/kg. Over 70 mg/kg in hair is associated with hair falling out in humans and depression (Rees 1979). A month of injecting adult rabbits subcutaneously with aluminum at sublethal doses of 25 µmol/kg body weight was required to increase new hair concentrations by 10 times to levels dose to or higher than seen in the site samples (up to a mean of 224 mg/kg wet weight and maximum of 716 mg/kg). Such hair concentrations were not associated with adverse effects in the rabbits or their offspring, even when

dosing continued for one month (Yokel 1987), although longer-term effects are unknown. Hair aluminum concentrations associated with adverse effects in these studies were over ten times higher (4,432 mg/kg) than the maximum raccoon hair in the impacted sites.

Although hair is bioaccumulating aluminum, other tissues of most raccoons in impacted site samples have aluminum concentrations lower than the detection limit. Some aluminum detection limits for blood, liver, and brain were higher than concentrations considered normal or no-effect levels. For example, 0.25 mg/kg wet weight is normal concentration in the human brain, and 5 mg/kg wet weight can cause neurological disease (McLaughlin et al. 1962). Detection limits for raccoon brains ranged higher than normal for humans from 0.82 to 4.2 mg/kg wet weight. Detection limits for raccoon livers were 0.84 to 4.1 mg/kg wet weight, too high to be certain of no effects, because background levels are typically no more than 2.6 mg/kg wet weight in humans (Hamilton et al. 1972). The lack of aluminum detections in most of the tissue data could support that toxicity of aluminum in raccoons may not have an adverse effect, but it also may be due to inadequate detection limits.

Observed elevated aluminum in raccoon tissues is likely not from the ash. Comparisons of sediment concentrations of aluminum and ash for the assessment for benthic invertebrates found no evidence that aluminum concentrations are due to the ash release. For example, in Section 4.2.2.2, it is shown that aluminum and ash percentage were not statistically significantly ($p < 0.10$) correlated in sediment samples used for toxicity testing (Tables 4-20 and 4-21). The associated correlation coefficients were low for the Emory River dataset (0.47) and the combined Emory and Clinch River dataset (0.09). Considering the VibeCore™ sample data for submerged sediments from all three rivers as a single dataset yields a statistically non-significant ($p < 0.05$) correlation coefficient of 0.072 for aluminum and ash (Appendix S, Exhibit S-1).

Aluminum is elevated in the impacted sediments compared to reference reaches. Interestingly, the mayfly nymph tissue data evaluated in Section 4.2.3.1 suggest that bioaccumulation of aluminum in this sediment is higher in animals living in Clinch River sediments. Aluminum was statistically significantly higher than reference ($p < 0.05$) in the Clinch River (ER_A and ER_B) and lower Emory River (ER_A), but not in samples collected closer to the release area (ER-B or ER_C) or in the Tennessee River (Tables 4-30 through 4-32). The similarity of ER_A results with CR_A and CR_B results is consistent with the deposition of Clinch River sediments in the lower Emory River, which is the accepted source of DOE-related legacy constituents (e.g., celsium-137) in ER_A. Thus, the ash aluminum may not be bioaccumulating very much and the raccoon hair bioaccumulation may reflect bioaccumulation of aluminum from legacy sources.

In summary, except for aluminum, the measured concentrations of metal and metalloid COPECs from remaining ash at the Site are unlikely to lead to toxic effects in the raccoon population at the Site. Although uncertainty exists for the potential effects of aluminum, the data for sediments and for mayfly nymphs

suggest that aluminum bioaccumulation in the vicinity of the Site is not due to the ash release. Risk to omnivorous mammals from the ash is likely negligible.

7.2.2 Raccoon Health Metrics

The raccoons captured for tissue analysis in impacted and reference areas in 2009 and 2010 were also evaluated for indicators of health. Blood was collected for a complete blood count and plasma biochemistry panel. Raccoons were euthanized, necropsied, and gross lesions were noted. Liver, kidney, lungs, brain, gonads, and adrenal glands were collected for histopathology and microlesions were noted. Details of the health metrics study for raccoons are in Appendix Y. Similar to the raccoon tissue concentrations, emphasis is placed on the 2010 health and metric data when evaluating risk because the 2010 data were collected post-dredging and, unlike the 2009 reference location, the reference location in 2010 was considered uncontaminated by ash historically and appropriate for evaluating potential effects of the ash for the Emory River, where most raccoons were trapped.

7.2.2.1 Analysis of Exposure and Effects

Because of the small number of animals with pathologic lesions, control animals were compared to exposed animals regardless of year. Prevalence of pathologic lesions was analyzed with a chi square test or Fischer's Exact test where appropriate with significance set at $p > 0.05$. Significant differences are reported in Appendix Y. Summaries of the concentrations in each tissue type for reference and impacted locations in 2009 and 2010 are presented in Tables 2-214 through 2-244. A comparison of reference and impacted locations for 2010 are presented side-by-side in Tables 7-1 through 7-7 for each tissue type.

The health metrics showed few significant differences between impacted and reference raccoon samples, and none could be attributed to the ash. Microscopic lesions observed in the animals included: pulmonary anthracosis (n = 19), eosinophilic pneumonia (n = 11), portal fibrosis (n = 16), gonadal interstitial hypertrophy (n = 11), and pulmonary fibrosis/granuloma (n = 7), but there was no significant difference in the occurrence of lesions between exposed animals and control animals when compared by year or when years were combined. The lesions can be attributed to other mechanisms than ash. Although reproductive fitness was not specifically examined in this study, there was no pathology of the gonads.

The only significant difference observed in blood counts was the hematocrit was higher in the 2010 exposed animal samples when compared to the 2009 exposed animal samples, but the mean values for the raccoon groups fell within the normal range. Significant differences in hematocrit were not reported between exposed and reference groups. Significantly lower cholesterol was observed in exposed animal samples caught in 2010 compared to reference animal samples caught the year before, but the means of both groups fell within the normal range of levels for raccoons, and such levels would not likely lead to disease (Fowler and Miller 2003). Creatinine, an indicator of renal disease if it and blood urea nitrogen are elevated, was

significantly lower in exposed animal samples in 2009 than exposed animal samples in 2010, and both years had lower levels than the reference range for captive raccoons in another study (Fowler and Miller 2003). Kidney disease and muscle mass loss were not apparent; thus, the creatinine difference could not be explained. Creatinine was not reported as significantly differing between 2010 reference and exposed animals and, thus, ash is not implicated as causing the difference.

Control animal samples from 2009 in the positive control had lower plasma calcium concentrations than the other study locations, below normal ranges for captive raccoons (Fowler and Miller 2003). However, post-mortem examination indicated no signs of diseases that might cause the hypocalcemia. No raccoons from the ash-impacted locations showed signs of hypocalcemia.

Although no significant differences were identified in raccoon health metrics between impacted and reference areas, sample sizes were small, particularly for reference animals, reducing power to detect differences. Additionally, correlations of health metrics with tissue concentrations are not available from the raccoon study to evaluate if observed lesions are related to COPECs. Thus, conclusions based on these data are uncertain.

7.2.2.2 Risk Estimation

No clinically significant differences were observed in prevalence of lesions, blood counts, or plasma biochemistry that could be attributed to the ash. In particular, exposed raccoons collected after the dredging in 2010 had most health metrics within the normal range. Therefore, risk to omnivorous mammals, as represented by the raccoon is negligible. This result can not necessarily be extrapolated to other mammalian assessment endpoints, such as piscivorous and herbivorous mammals, but is suggestive that mammals feeding on a variety of foods in the impacted area are as healthy as mammals elsewhere. Nonetheless, uncertainty is high due to low sample sizes, and there is low confidence in this conclusion. Thus, risk is estimated to be at most low from this LOE.

7.2.3 Dietary Exposure Models

Aquatic- and riparian-feeding mammals are exposed to ash COPECs through dietary uptake from the food web and abiotic media. A dietary exposure model calculates the dose predicted for each of the receptor mammal species representative of the three assessment endpoints, herbivores, omnivores, and carnivores (piscivores). The three representative species for which a dose was modeled are the herbivorous muskrat, omnivorous raccoon, and piscivorous mink. The calculated dose is compared to a TRV, which is a threshold above which adverse effects are likely. The modeling methods used for this receptor group are the same as those described for aquatic- and riparian-feeding birds and are detailed in Appendices W and X. Only 2010 EPCs for abiotic and biotic media were used in the dietary exposure models.

In Appendix W, Tables 11-1 through 11-11 provide the HQ results of the screening analysis, Tables 12-1 through 12-10 provide the HQ results of the refined analysis, and Tables 13-1 through 13-6 provide the results of the probabilistic analysis. Radionuclide results for riparian wildlife are in Appendix X. The results are summarized by receptor below.

7.2.3.1 Piscivores (Mink)

The screening-level HQs of COPECs for the piscivorous mink were below 1 except for aluminum, radionuclides (radium-228, radium-226, potassium-40), iron, and PCBs. In the refined analysis, aluminum HQ still exceeded 1 (0.1 to 3) as well as reference reach HQs in ER_B and Reaches A and B of the Clinch and Tennessee Rivers. However, no fish or sediment were sampled in the Tennessee River (Clinch River fish and sediment were used in model), making results in that river uncertain. The Iron HQ also exceeded 1 (HQ = 1 to 2) in the refined analysis, but was not higher than the reference reach (HQ = 2, Table 6-2).

The mink HQs for aluminum are quite uncertain. The probability of aluminum HQ exceeding 1 ranges from 5 percent (LOAEL) to 60 percent (NOAEL) in ER_B, CR_A, and CR_B (Table 6-3). Approximately half of the fish representing the mink diet were collected in spring 2010. Aluminum concentrations in these gizzard shad samples were generally higher than in the fall 2010 samples. This was notable for all locations and both sample types (i.e., whole body and whole body without guts or gut contents) in the Emory River, where aluminum concentrations were typically an order of magnitude lower in the fall than in the spring (Appendix L); the results for the Clinch River were less consistent. Dredging was not yet complete when the spring 2010 fish were collected and may have contributed to the observed temporal trend. However, this trend was also evident for whole body gizzard shad in the Emory River reference reach (ER_R) and in two of three Clinch River reaches (CR_R and CR_B).

The factor most affecting the probabilities for aluminum by far is the TRV (NOAEL) uncertainty. The model output is about two times more sensitive to the TRV parameter than the next most important parameter of fish tissue concentrations in the most impacted reach (ER_A; Appendix W, Figures 11 through 17). Variability in sediment concentration is of least importance. Despite the uncertainty, the low magnitude of the HQs relative to the reference reaches with HQs > 1 (Table 6-2) suggests a low level of risk from aluminum in the ash.

7.2.3.2 Omnivores (Raccoon)

The screening-level HQs of COPECs for the omnivorous raccoon were below 1 except for aluminum, radionuclides (radium-228, radium-226, potassium-40), and iron. In the refined analysis, only the aluminum HQ exceeded 1 (range of 0.2 to 3) and was larger in impacted than reference reaches in Reach A of the Clinch and Tennessee Rivers. The probability of aluminum HQ exceeding 1 is uncertain, ranging from 6 percent to 70 percent in the Clinch and Tennessee Rivers (though sediment ingestion, substantial in

raccoons, in the Clinch River was used in the model for the Tennessee River). The most sensitive factor in the probability model was the uncertainty of the TRVs, followed by insect concentration variability, which was about one-third as important at influencing the probabilities than the TRV. Given the low magnitude of HQs relative to the reference reaches with HQs > 1 (Table 6-2), risk of aluminum to omnivorous mammals such as the raccoon is estimated to be low.

Also, as noted above in Section 7.2.1.2, aluminum concentrations in sediment are poorly correlated with ash in sediment, and aluminum in mayfly nymphs is not concordant with proximity to the release area. That is, the data for sediment and mayfly nymphs suggest that the observed aluminum bioaccumulation in the vicinity of the Site is not due to the ash release. Uncertainties associated with aluminum are discussed in Section 7.3.2.

7.2.3.3 Herbivores (Muskrat)

The screening-level HQs of COPECs for the herbivorous muskrat were below 1 except for aluminum, radionuclides (radium-228, radium-226, potassium-40), iron, and manganese. In the refined analysis, the aluminum HQ still exceeded 1 (0.1 to 2) in a few reaches, but was not greater for impacted than reference reaches (Table 6-2). Thus, a low risk from ash was suggested for herbivorous mammals.

7.3 Risk Characterization

A WOE approach is used to evaluate overall risk to aquatic- and riparian-feeding mammals. The risk for each LOE is evaluated, the confidence in the risk estimate and the ecological relevance of each LOE is considered when assessing overall risk.

7.3.1 Weight of Evidence

Exhibit 7-1 evaluates each LOE for each receptor, dividing the dietary exposure model into subcategories of LOEs. The subcategories were the three wildlife receptor species of the three assessment endpoints evaluated, which are herbivores, omnivores, and piscivores (carnivores).

Exhibit 7-1. Aquatic- or Riparian-Feeding Mammal Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Ecological Relevance
Omnivore – Raccoon				
Tissue Concentrations	Low	Low	Low	Moderate
Dietary Exposure Models	Low	Moderate	Low	Low
Health Metrics	Low	Low	Low	Low
Piscivore – Mink				
Dietary Exposure Models	Low	Moderate	Low	Low
Herbivore – Muskrat				
Dietary Exposure Models	Low	Moderate	Low	Low
Weight of Evidence	**Low**	**Moderate**	**Low**	**--**

Exhibit 7-1 summarizes the magnitude of adverse effects, confidence in the magnitude, evidence of causality, and ecological relevance of each LOE. Uncertainty in being protective is low for screening dietary exposure models, moderate for refined models, and high for probabilistic models. Magnitude of adverse effects, indicated by amount of exceedance of benchmarks, is low for all LOE because HQs were no higher than 3 in the refined analysis, and confidence in the magnitude is moderate. Also, screening levels were exceeded for a number of COPECs—aluminum, radionuclides (radium-228, radium-226, potassium-40), iron, and manganese, and not all were screened out in the refined analysis, making confidence in the dietary exposure models moderate. Causality is low because elevated aluminum in tissue and diet is potentially not from the ash.

Because they are site-specific, measured concentrations in raccoon tissue provide moderate ecological relevance. Metrics of health measure the effect, rather than the exposure and might be of higher ecological relevance. However, health metrics that are not related to reproductive fitness do not always extrapolate well to population-level effects and, thus, are assigned low ecological relevance. The dietary exposure models are based on site-specific biotic and abiotic concentrations and estimates of bioavailable fractions, but rely on the literature to parameterize most of the rest of the model and to evaluate effects. Thus, the results of such models are of low ecological relevance.

Exhibit 7-2 summarizes the weight of each LOE, level of risk, confidence in the risk level assigned, and COPECs driving the risk. Weight is based on ecological relevance and quality of the data and analysis. Risk level is based on the refined and probabilistic analysis of the dietary models. If a HQ for the refined NOAEL is below 1, risk is negligible. If above 1 and magnitude is low, then risk is low. If sample sizes are

inadequate, risk cannot be negligible but can be low with low confidence if magnitude is low. Confidence that the risk is no higher than the level indicated is moderate to high based on refined and probabilistic results.

Exhibit 7-2. Aquatic- or Riparian-Feeding Mammal Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Omnivore – Raccoon				
Tissue Concentrations	Moderate	Low	Low	Aluminum
Dietary Exposure Models	Low	Low	Moderate	Aluminum
Health Metrics	Low	Low	Low	--
Piscivore – Mink				
Dietary Exposure Models	Low	Low	Moderate	Aluminum, iron
Herbivore – Muskrat				
Dietary Exposure Models	Low	Low	High	Aluminum
Weight of Evidence	**--**	**Low**	**Moderate**	

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

Potential risk is estimated to be at most low for all the subcategories of LOEs. The tissue concentration data for raccoons and dietary exposure models indicate that a low risk from aluminum may exist, for mammals that are predators and incidentally ingest sediment in the ER_B, CR_A, CR_B, and TR_A. Additionally, a low risk from iron from the dietary exposure model is suggested for piscivores, but with high confidence it is no more than low for iron because HQs in the impacted reaches are similar to the reference reaches for iron. Confidence is moderate for aluminum being low risk for piscivores because impacted reaches had elevated HQs relative to reference reaches. A low risk may also exist for herbivorous mammals because of HQ > 1 for aluminum, but is less likely because reference reach exposures for these COPECs were the same as Site reach exposures, resulting in high confidence the risk is no higher than low. Although probabilities of risk to aluminum range from 5 percent to 70 percent and TRV studies were of low quality, confidence in the risk being no higher than low is moderate because other TRV studies support higher effect levels, and the TRVs used for the refined analysis were conservative.

The raccoon, representing omnivorous mammals, was evaluated for risk two ways:

- With a dietary exposure model
- Tissue concentrations and health metrics in trapped raccoons.

As discussed above, the dietary model suggests, at most, low risk to this group from the ash. Both the tissue and health metrics results from the raccoon study similarly suggest low risk, but with low confidence. The low confidence is because aluminum detection limits in some tissues were too high to be certain levels were consistent with background or no adverse effect levels. For one tissue that had sufficient detection limits, hair, aluminum was over three times higher in hair of study location raccoon samples than reference animal samples, but such levels were not significantly higher and may not be associated with adverse effects of aluminum bioaccumulation on reproduction. In combination, the raccoon health and tissue data suggest low risk from aluminum to omnivorous mammals.

Risk levels from the dietary exposure model are similar for the raccoon and mink, which suggests the raccoon tissue data may apply to piscivores (such as mink), and may further support that risk is low for mink. However, much uncertainty exists around aluminum risk with few toxicity studies on growth effects upon which the TRV was based, and the effects of COPECs on reproduction were not measured in raccoons. Overall the WOE supports that risk is, at most, low for aquatic- and riparian-feeding mammals with moderate confidence in the impacted reaches of the Emory, Clinch, and Tennessee Rivers.

Aluminum concentrations in the prey items that are closely associated with sediments (i.e., mayfly nymphs and gizzard shad) are poorly correlated with the ash release and ash in sediments. Therefore, aluminum is not identified as a COEC based on the WOE for mammals. Similarly, HQs for iron were the same for the impacted and reference reaches, and therefore iron was not identified as a COEC.

7.3.2 Uncertainties

Uncertainties that may lead to either an overestimate or underestimate of risk are associated with each stage of risk assessment. It is important to understand the major uncertainties and how they might affect the risk assessment. The uncertainties associated with the two LOEs for the aquatic-and riparian-feeding mammals are reviewed below.

7.3.2.1 Raccoon Tissue Concentrations of COPECs

Issues that create uncertainty associated with the evaluation of the raccoon tissue concentrations are applicability of literature-derived background levels to interpret tissue concentrations, limited datasets for reference areas, unknown concentrations below detection limits, effects of complex COPEC mixtures, timing of tissue collection, and representativeness of the reference locations. Reproductive measures and survival rates were not documented to interpret the effect of the tissue concentrations observed on productivity of omnivorous mammal populations. As a consequence, except for selenium, interpretation depended primarily on literature-derived values of background levels for human populations. Such background concentrations for humans may not represent raccoon populations, nor is it certain they represent concentrations with no adverse effects to raccoon populations. However, raccoon populations have been studied for adverse effects

of selenium at a selenium-contaminated site (Kesterson National Wildlife Refuge; Clark et al. 1989) and none were found, supporting the results showing low risk from selenium in the ash.

Aluminum risk, identified as of potential concern by the dietary exposure models for raccoons, is more uncertain because the reference location in 2009 for the raccoon study is a positive control that may have higher aluminum risk from previous ash contamination. The reference location in 2010 has no such history and should have negligible aluminum risk, but sample size in that location was low (n = 5), making power to detect differences at alpha levels of 0.05 or 0.1 likely low (p-value = 0.19, n = 5 and 10, Appendix Y), particularly given non-detects were eliminated from the dataset. Mean aluminum concentrations show aluminum in raccoon hair is 3.5 times higher in the impacted location samples than the reference location samples in 2010 (Appendix Y), at levels above background levels for humans and at levels that may be correlated with human neurological disorders (Rees 1979). This result is not driven by outliers. If more raccoons had been trapped, this large difference may be significant.

Results of studies also depend on how non-detects are treated. Unlike the piscivorous and aerial-feeding bird egg tissue and dietary studies, non-detects were not set to the RL, but instead were eliminated from the samples, which inflates means and likely reduces power to detect differences from reference locations. Another uncertainty is that complex COPEC mixtures that include aluminum often have antagonistic or synergistic effects (Carson 2002). Finally, differences in tissue concentrations attributed or not attributed to the ash may be from natural variation in sediment conditions and food items between reference and impacted reaches. Such variations are better captured in the second LOE, the dietary exposure models.

A risk assessment of the residual ash post-dredging is uncertain given post-dredging raccoon data were collected in 2010 not long after dredging ended. The 2011 data may show different results and will be evaluated when they become available. These various uncertainties reduced confidence in the assessment of risk being negligible to omnivorous mammals based on raccoon tissue (Exhibit 7-2).

7.3.2.2 Raccoon Health Metrics

Confidence in the health metrics assessment showing negligible risk to omnivorous mammals is higher than the tissue assessment, because it is less dependent on detection levels. Confidence is rated moderate. Issues that create uncertainty associated with the raccoon health metrics that prevent high confidence in the conclusion are low power to detect differences between reference and impacted locations using limited datasets, representativeness of reference locations, and extrapolation of observations to reproduction and population viability.

Reference locations were relied upon rather than health benchmarks from the literature in this health metric assessment. Problems with this approach are that the 2009 reference location was a positive control subject to historical ash contamination, making comparisons difficult to interpret that year. The 2010 reference

locations may be more representative of background conditions, but a description of the habitat in these areas and similarity to the Site is missing from Appendix Y and its attachment.

Lesions in individual animals and blood characteristics indicating health were documented, but reproductive measures and survival rates were not documented to interpret the potential effect of ash on productivity of omnivorous mammal populations. Nevertheless, the finding of no abnormalities indicative of disease as well as tissue concentrations not likely to be of concern supports with low confidence that risk is low.

7.3.2.3 Dietary Exposure Model

Many uncertainties exist in the dietary exposure model and are detailed in Appendix W. Of variables tested in a sensitivity analysis, the results are generally most sensitive to the literature-derived TRVs selected and secondly to the measured tissue concentrations (EPCs) of the food items. In contrast, variability in body weight, bioavailable estimates of COPECs, and sediment concentrations have a smaller effect and contribution to the results. Sediment and water provide small contributions to risk for mink and raccoon relative to biotic media in the diet (Table 6-4).

Although variability is not high for bioavailable estimates, the accuracy of the estimates is uncertain due to reagent non-selectivity and re-adsorption of the sequential extraction procedure used to develop the estimates (Filgueiras et al. 2002). However, Karadas and Kara (2012) found the fractions used to produce bioavailable estimates produce results approximately similar to in vitro bioavailability tests simulating animal stomachs for many metals, with some exceptions (chromium and to some extent copper), although there is high individual sample variability. Nevertheless, they did not test many of the metals used in this model and it is possible the bioavailable estimates underestimate risk by some unknown amount.

Assumptions for other parameters not subject to a sensitivity analysis (fixed in the model) may also affect the results. For example, exposure model parameters (except EPCs, bioavailable fraction of COPECs, and raccoon weights) were obtained from the literature and assumed to be applicable to the Site, which may not be true.

The TRVs are from the literature and are associated with significant uncertainty (Dale et al. 2008), which was incorporated into the probabilistic analysis for COPECs, but only for COPECs identified as possible risk drivers during the refined analysis. Nevertheless, toxicity data are only available for a limited number of species, mostly laboratory test species, under a strictly defined set of test conditions that deviate from natural conditions (Sample et al. 1996; Suter 1996). The aluminum studies relied upon and discussed by Sample et al. (1996) and used to refine TRVs were poorly conducted or documented. Carson (2002) documents other chronic aluminum toxicity tests that focus more on reproduction rather than growth. Reproductive effects occur at higher thresholds than growth effects, but may be more closely related to population-level effects. If reproductive endpoints were used for TRVs (e.g., NOAEL of 97 mg/kg-day,

Bernuzzi et al. 1989), aluminum HQs would be less than 1 for the mammals. Additionally, tolerance and adaptation of the receptors are not considered for laboratory studies (Grant 2002; Millward and Klerks 2002), but do play a role in field studies included in Monte Carlo probabilistic models for selenium.

Relatively few studies evaluate the effects of toxicity at the population scale that integrate reproduction and survival endpoints or incorporate concepts of carrying capacity, intraspecific competition, density-dependence, or dispersal. None of the TRV studies used for this receptor group modeled population-level effects. Thus, the large uncertainty in TRVs for COPEC drivers such as aluminum and selenium (Attachment 3 of Appendix W) makes assessment of their risk very uncertain. For these reasons, it was critical to evaluate uncertainty of TRVs in a probabilistic assessment.

For the deterministic analysis, the EPCs used in the dietary exposure model were conservative, biasing the dose high, because the EPCs were the 95 UCL or maximum concentration for a reach (Appendix W), rather than the average. The probability analysis used the average and standard deviation of these concentrations, which is more accurate if based on a robust dataset.

However, confidence in being protective is often desired over confidence in accuracy of the assessment. There is low uncertainty in being protective with the screening level TRVs, moderate uncertainty with the refined TRVs, and high uncertainty with the TRV for the probabilistic evaluation. Such uncertainty was used to determine confidence in the risk assessment for endpoints. High uncertainty is also associated with the probabilistic model because the average and standard deviation were not always based on a robust data set. In several cases, the standard deviation and type of distribution was assumed (e.g., aluminum). Consequently, the probability analysis is not always more accurate.

The food item EPCs were obtained from tissues of one or a few species that may not represent the full set of food items of a receptor, particularly because food groups that included terrestrial, less impacted items were replaced with aquatic food items, biasing risk estimates high. Specifically, the diet component identified as “terrestrial invertebrates” was assessed using tissue data for aquatic invertebrates. The diet component referred to as “small mammals and birds” was represented solely by fish tissue data. Incidental sediment ingestion substituted for the incidental ingestion of soil and sediment. Additionally, the percentages of diet and body weights from the literature have some inherent uncertainty because data are neither available for an entire population nor available for all ages and all seasons.

Whole body tissue concentrations for fish and mayfly larvae were used to derive EPCs used in the exposure calculation. Using whole body prey tissue concentrations in addition to incorporating incidental sediment ingestion rates calculated based on gut contents (Beyer et al. 1997; Beyer et al. 1994) is a conservative approach and may overestimate the risk to wildlife.

The detection limits for some constituents, notably selenium in water and sediment, were higher than potentially ecologically important concentrations. This is a well recognized and largely unavoidable issue for some bioaccumulative constituents like selenium. This may lead to underestimation of exposures and obscure the identification of exposure pathways. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty.

Percent moisture was analyzed in biological samples, but insufficient sample volume was available for some biological media. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

Mercury analyses of fish muscle tissue were found to be biased low based on QA/QC evaluations using standard reference materials. A detailed study by the project QA/QC team determined that the use of a CF of 1.3 was appropriate for fish fillet samples, but that sample type was not used in the dietary exposure models. It is yet unknown whether other tissues were similarly biased and, if so, what CF should be used. However, it is unlikely that a 30 percent increase in the reported concentrations would alter the characterization of risks in this BERA, given the low concentrations and associated low risk of mercury.

Only the leaves of emergent and shoreline vegetation were collected even though many receptors (muskrats and raccoons) can also feed on the roots of emergent vegetation, and roots tend to concentrate more selenium and arsenic than other plant parts (Afton et al. 2009). Risk to herbivorous mammals could be biased low, but it is unlikely that removal of the bias would increase HQs to greater than 1 because the HQs are quite low.

Tissue sample collection was often clustered and limited in each reach (Appendix W, Figure 2). Good representation of reach exposure concentrations and their distribution is uncertain. The timing of sample collections varied from pre-dredging to post-dredging, making representation of post-dredging conditions difficult.

In addition, species differ with respect to water, sediment, and food ingestion rates and absorption, metabolism, distribution, and excretion of chemicals. In particular, sediment ingestion studies have not been completed on the receptor species, and data from other species had to be used. Because of these various uncertainties for the differing LOEs, the WOE approach was used to evaluate if the sum of the information can lead to valid conclusions, which are summarized below.

7.3.3 Summary and Conclusions

Based on the overall WOE for aquatic- and riparian-feeding mammals, a low risk to mammalian species that feed on benthic invertebrates exists at the ash-impacted locations of the Emory, Clinch, and possibly Tennessee Rivers (Reaches A and B, Table 6-2). Omnivorous and piscivorous species are most at risk, but

the risk is low, particularly since not all fish and invertebrate data upon which this conclusion was based were collected after dredging. COPECs with potential adverse effects that exceeded screening levels from the dietary exposure model were aluminum, iron, manganese, PCBs and radionuclides (radium-228, radium-226, potassium-40). PCBs are a legacy COPEC not related to the ash. COPECs driving risk based on a refined analysis were aluminum and iron. Neither of these are considered COECs because they are not correlated with ash (aluminum), or because risk (HQ) from the refined exposure models is the same for reference and impacted reaches (iron). No COECs are identified for the aquatic- and riparian-feeding mammals.

8. Aerial-Feeding Birds and Mammals

Aerial-feeding birds and mammals are evaluated separately in this BERA because they represent a group that may be at risk to COPECs such as selenium, which can become concentrated in aerial insects. This group is also an important component of a functioning ecosystem at the Site, and includes a federally-listed endangered species, the gray bat.

8.1 Problem Formulation

Aerial-feeding birds and mammals in the impacted reaches of the Emory, Clinch, and Tennessee Rivers may be directly exposed to both inorganic and organic COPECs through drinking the water in the rivers. They may be indirectly exposed through feeding on flying insects that bioaccumulated these COPECs. Bioaccumulation of the COPECs in the aerial-feeding birds and mammals from these direct and indirect sources may cause toxicity in the form of reduced reproduction or survival, leading to a decline in these populations and species diversity. Foraging habitat must be sampled to evaluate the potential for such declines. The types of habitat sampled that support these species, the sampling media, assessment and measurement endpoints, and risk questions to be answered by this risk assessment are discussed below.

Riparian, wetland, and aquatic habitat where these mammals feed is described in Section 6.1 under Problem Formulation for Aquatic- and Riparian-Feeding Birds. Additionally, bats, such as the gray bat, use caves for roosting and hibernation. Six caves are found along the reservoir; 24 total caves are found within the four-county area (TVA 2009a). The gray bat is the only species of this group that is federally-listed. It is listed as endangered because this species has been reduced to a small population at risk of extinction.

8.1.1 Assessment Endpoints

The assessment endpoints for aerial-feeding insectivores used to develop this BERA are:

- Aerial-Feeding Insectivorous (Omnivorous) Bird Populations
- Aerial-Feeding Insectivorous (Omnivorous) Mammal Populations.

Both of these endpoints have important ecological functions that have the potential to be impaired by the ash. Aerial-feeding insectivorous birds, such as tree swallows, feed primarily on emergent aquatic insects such as mayflies, true flies, dragonflies, and caddisflies. They are referred to as aerial-feeding insectivorous (omnivorous) bird populations because this group represents aerial feeders that also feed on some seeds and berries. However, to be conservative, this assessment assumes they feed solely on aquatic insects.

In addition to linking energy pathways in an aquatic ecosystem, aerial-feeding birds foraging on insects may regulate the population density of these insects. Conversely, birds may also serve as the prey items for upper trophic-level predators, potentially influencing the density and composition of those populations. As a result, aerial-feeding insectivorous birds are vital to a balanced ecosystem and may affect the vigor of vegetation communities indirectly by reducing insect population outbreaks. Aerial-feeding insectivorous birds have the potential for exposure resulting in adverse effects, mostly through the food web, as well as the potential for bioaccumulation and transfer of COPECs to higher trophic levels, potentially harming their predators.

Aerial-feeding insectivorous mammals, such as gray bats, also feed primarily on emergent aquatic insects and provide the same ecosystem functions identified for aerial-feeding birds. Similarly, aerial-feeding insectivorous mammals have the potential for exposure resulting in adverse effects, mostly through the food web, as well as the potential for bioaccumulation and transfer of COPECs to higher trophic levels.

The species selected to represent each assessment endpoint of this group (Appendix U), referred to as the receptor species, are as follows:

- Aerial-Feeding Insectivorous (Omnivorous) Bird Populations: Tree Swallow
- Aerial-Feeding Insectivorous (Omnivorous) Mammal Populations: Gray Bat.

Based on these two assessment endpoints, the ecological risk questions established for the evaluation of aerial-feeding insectivores at the Site are as follows:

- Do concentrations of ash-related COPECs in surface water or food items pose unacceptable risk to aerial-feeding insectivorous bird populations?
- Do concentrations of ash-related COPECs in surface water or food items pose unacceptable risk to aerial-feeding insectivorous mammal populations?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective needs to be evaluated in the WOE assessment. These risk questions are used to help focus subsequent steps of the aerial-feeding bird and mammal assessment, including the selection of measurement endpoints.

8.1.2 Measurement Endpoints

The measurement endpoints for aerial-feeding birds and mammals are as follows:

Measures of exposure:

- COPEC concentrations (total) in surface water
- COPEC concentrations in tissues of dietary items
- Modeled COPEC dietary doses in mg/kg-day from diet
- COPEC concentrations in tree swallow eggs, eggshells, and nestlings.

Measures of effects:

- Screening-level literature-derived toxicity dietary doses representing NOAELs and LOAELs for individual growth or reproduction for each assessment endpoint
- For COPECs that did not pass the screen, refined literature-derived, dietary doses tailored to be more site-specific representing NOAELs and LOAELs for individual growth or reproduction
- TRVs for egg, eggshell, and nestling concentrations representing benchmarks for toxic effects to the embryo or nestling (EC_{10} or LOAEL)
- Tree swallow egg health metrics (weight, length, width, volume)
- Tree swallow nestling health metrics (nestling weight, tarsus length, feather length)
- Productivity metrics including clutch size, hatching success, nestling survival, and fledglings produced per nesting female.

8.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to aerial-feeding insectivores. The LOEs used to evaluate aerial-feeding insectivores include:

- Reproductive data for tree swallow populations
- Tree swallow egg, eggshell, and whole body nestling tissue data
- Dietary exposure models for tree swallows and gray bats.

Each LOE is summarized below.

8.2.1 Tree Swallow Tissue Concentrations of COPECs and Reproduction

Because tree swallows represent the aerial-feeding insectivorous birds, the reproductive success of this bird species in relation to potential metal and metalloid COPECs effects was assessed at the Site. Appendix Z details the study objectives, methods, results (including tables and figures), and interpretation of the tree swallow reproductive and egg tissue study. The study is summarized below to assess risk to the aerial-feeding bird insectivores. The results do not necessarily apply to aerial-feeding mammal insectivores for COPECs that have different mechanisms of toxic action for birds compared to mammals (e.g., selenium).

Tree swallow eggs were collected from nest boxes constructed at the impacted location and reference areas to evaluate the spatial and temporal variation in egg concentrations of inorganic ash COPECs bioaccumulated into tree swallows through the diet and maternally transferred to eggs. The tree swallow study additionally evaluated metal and metalloid COPECs in nestlings obtained from both the diet and maternal transfer. In addition to analyzing egg content, eggshell concentrations were also evaluated because some elements such as strontium tend to bioaccumulate in the eggshell, weakening its integrity and hatching success.

If metal and metalloid COPECs exceed concentrations causing adverse effects to the embryo or nestling, and the exceedance can be linked to the ash, the concentrations can then be evaluated for their effect on measures of tree swallow reproductive success, specifically clutch size, egg health metrics, hatching success, and nestling survival. A key reproductive measure that integrates these measures is number of fledglings produced per nesting female (expressed as female fledglings produced per nesting female to evaluate replacement potential). The correlation between egg concentrations and reproductive measures and differences in these measures between impacted and reference reaches were quantified. Egg, eggshell, and nestling tissues were analyzed to obtain concentrations of 26 metal and metalloid COPECs.

8.2.1.1 Analysis of Exposure and Effects

Over 500 nest boxes were constructed for the tree swallow study. Tree swallow egg, eggshell, and nestling concentrations, egg and nestling health metrics, and reproductive measures were statistically compared between impacted and reference locations during spring 2009 and 2010, a period before dredging was completed. Analysis of 2011 results is pending completion of chemical analyses by a contract laboratory. The evaluation of this LOE for residual risk after the dredging operation is conservative because the data analysis does not yet incorporate the potentially positive effects of dredging. Locations of tree swallow eggs collected (one per nest) in impacted and reference locations are shown on Figure 8-1. One reference location, Melton Hill Dam, is a positive control because it has known historical contamination from ash. Reference location locations varied by year with additional locations added in 2010 and are discussed in Appendix Z.

A two-way ANOVA (or non-parametric equivalent) was performed to evaluate the effect of location (reference and impacted) and year (2009 and 2010) on egg and nestling concentration, egg and nestling health metrics, and clutch size. A one-way ANOVA assessed measures that were collected in 2010 only, which were eggshell concentrations, reproductive measures, and the parameters measured at new reference locations added in 2010. If the results showed a significant location effect (where $p \leq 0.05$) or almost significant effect (where $0.05 \leq p \leq 0.1$) with higher concentrations in the ash-impacted locations, ash could be responsible for the higher concentrations. Additionally, if the interaction term between year and location is significant, the increase was greater in one year than the other. Results that were close to showing a significant difference were also evaluated for risk to be conservative, and for eggshells because

power to detect differences was not always high depending on the sample size (n = 2 to 7 in 2010) (Appendix Z). Power was high (80 percent) with high confidence (95 percent) for other comparisons of eggs, nestlings, and productivity measures.

Ash flowed upstream and downstream from the failed dredge cell, impacting the embayments and Emory River most, followed by the downstream Clinch River, and then the most downstream Tennessee River (Jacobs 2010a). If comparisons of the locations within the impacted area (using post-hoc pairwise comparisons of the ANOVA) indicated higher egg concentrations or lower reproduction in the least impacted Tennessee River than the most impacted Emory River or embayments, the difference was not attributed to ash. Similarly, if parameters measured were worse in the impacted location than the positive control but not worse than at any of the other reference areas that do not have historical contamination, the difference was not attributed to the ash. Due to low sample size at the Little Emory River reference location (LERM 1.0), it was not statistically compared to the impacted locations.

Of the 26 metal and metalloid COPECs, only a few concentrations were significantly or almost significantly higher than the reference locations for the tree swallow eggs, eggshells, and nestlings. Selenium and mercury concentrations were higher in eggs and nestlings in impacted location samples than reference location samples, but neither was higher in eggshells (Appendix Z; Figures 36, 37, and 45). Copper, strontium, and manganese were higher in eggs from impacted location samples than in samples from one reference location, but not all reference location samples (Appendix Z; Figures 31, 33, 34, and 38). Strontium was also higher in eggshells and nestlings in impacted location samples than in reference location samples (Appendix Z; Figures 41 and 47). Nestlings had higher cadmium concentrations in impacted location samples relative to reference location samples (Appendix Z, Figure 42), but that relationship was not observed for eggs or eggshells. Egg weight, hatching success, and nestling survival were significantly lower in some impacted location samples than the reference location samples (Appendix Z; Figures 51 and 52). Fledglings produced per nesting female were not significantly lower, however.

None of the metals or metalloids COPECs that were higher in the impacted reaches in egg, eggshell, or nestling samples were significantly and negatively correlated to the reproductive measures of hatching success, nestling survival, or fledglings produced per nesting female, except strontium, barium, and manganese. The negative correlations with productivity measures were weak (Appendix Z), and only significant using the Spearman rank correlations that do not assume a linear relationship (Figure 8-2). Specifically, strontium and barium in eggs were negatively correlated to nestling survival. Barium also was negatively correlated to clutch size. Manganese was negatively correlated to fledglings per female. Only copper and strontium in eggs were associated with decreases in egg weight. Copper in nestlings was also associated with decreases in nestling weight (Appendix Z).

The calculated selenium dose for a tree swallow was significantly and negatively correlated to fledglings produced per nesting female within the impacted location samples (Pearson correlation $r = -0.34$, $p = 0.03$,

Figure 8-3), but not significantly correlated to hatching success or nestling survival. Reproductive measures were not monitored in any Tennessee River locations and, thus, this correlation only applies to the Clinch and Emory Rivers. Manganese, mercury, strontium, and copper were also evaluated for significant correlations of calculated doses with reproductive measures, and only strontium had a significant and negative relationship with fledglings produced pre nesting female (Pearson correlation $r = -0.34$, $p = 0.03$) and not with the other measures.

8.2.1.2 Risk Estimation

The metal and metalloid COPECs with concentrations in egg, eggshell, or nestling that were higher than reference location samples were compared to the reported LOAEL to assess the risk of the observed concentrations. If some concentrations are above the benchmark suggesting potential adverse effects, evidence of those concentrations coming from the ash and adversely impacting the population was evaluated. In particular, the correlation of tissue concentrations or calculated doses of the metal or metalloid COPEC with the reproductive measures were examined.

None of the elevated metals or metalloids COPECs exceeded known benchmarks for adverse effects, except for a few eggs that exceeded a selenium benchmark. The adverse effect levels for egg concentrations for mercury, selenium, and copper are estimated to be 2.5 mg/kg, 7.7 mg/kg, and 10 mg/kg dry weight, as discussed in Section 6.2.1.2 (but the 10 mg/kg for copper is an “educated guess,” not a known benchmark). The mercury benchmark was not exceeded by any eggs from the Site (maximum was 0.37 mg/kg dry weight), and the selenium benchmark was exceeded by only 1.7 percent of the eggs from the impacted reaches (all in the Emory River). The maximum copper concentration in the impacted location samples was a single outlier at 11.96 mg/kg dry weight and exceeded the copper “benchmark,” but mean concentrations were within background levels (2.43 to 3.61 mg/kg dry weight for 2009 and 2010, King et al. 1983). Notably, one reference location (Tennessee River) and positive control also had one high copper outlier each (16.3 mg/kg and 19 mg/kg dry weight, respectively) (Appendix Z, Figure 6). This suggests ash is not necessarily responsible for the copper outlier in the impacted location sample since similar outliers were observed in the reference location samples. Copper is negatively correlated with egg and nestling weights, but such a correlation is not important because copper cannot be linked to changes in egg hatchability or nestling survival, which affect population viability. Nor was copper concentration in eggs or calculated dose negatively correlated to fledglings produced per nesting female, further suggesting little risk to aerial insectivorous bird populations from copper.

The adverse effect levels for nestling concentrations of selenium and mercury are not well-defined, but can be estimated from a couple of studies. Whole body tissue concentrations of selenium that likely adversely affect nestlings range from 8 to 12 mg/kg (range of common grackle (*Quiscalus quiscula*) nestling carcass, liver, and feather concentrations, Bryan et al. 2012). No nestlings from the impacted location samples exceeded this threshold, although the positive control had one nestling sample with 9 mg/kg dry weight.

Whole body nestling concentrations of mercury of about 5 mg/kg dry weight (8 mg/kg in feathers) were not associated with a decrease in productivity (nestlings per occupied nest) in an osprey population in a mercury-polluted area (Hakkinen and Hasanen 1980). The maximum whole body mercury concentration in nestlings of 0.15 mg/kg dry weight in the impacted location samples was well below this mercury benchmark.

While no toxicity reference value has been established for barium in avian eggs, field studies published on barium in egg tissue have indicated barium concentrations are associated with embryo deformities at 2.2 mg/kg dry weight in clapper rail eggs (Schwarzbach et al. 2006). This effect level of 2.2 mg/kg dry weight was from a field study with a mixture of metals. As a result, the deformities associated with this concentration of barium potentially could be attributed to the effects from other metals found in the embryos. Another field study reported concentrations of barium in passerine birds up to 14.3 mg/kg dry weight without reporting that concentration as a level of concern (Mora 2003), although the threshold that conclusion was based upon is unclear. Reference locations had up to 40 mg/kg barium in eggs. If 40 mg/kg barium is the reference level, then only one outlier egg, located at an impacted location (ERM 3.0) exceeds that threshold and may have been affected by the ash. Barium in eggs is not significantly and negatively correlated to fledglings produced per female, which suggests it is not an important factor affecting tree swallows. Barium concentrations in nestlings were negatively correlated to nestling survival, but barium in nestlings was no higher than reference sites. Barium in eggs was also negatively correlated to clutch size, as well as barium in nestlings to feather length, but clutch size and feather length were not significantly lower than the reference locations. Overall, the results for barium do not strongly implicate ash as affecting tree swallow productivity.

Cadmium concentrations in nestlings were higher in impacted location samples. In high concentrations, cadmium can reduce growth of chicks. Cadmium concentrations in whole body nestling samples of < 2 mg/kg dry weight are well below levels of concern (Bryan et al. 2012). The maximum observed concentration of cadmium in a nestling sample from an impacted location was 0.09 mg/kg, which is much lower than the NOAEL concentrations for nestlings.

Strontium was elevated relative to reference locations in eggs, eggshells, and nestlings. High egg content of strontium may increase embryo deformities (Mora et al. 2007) or embryotoxicity (Ridgway and Karnofsky 1952). Adverse effect thresholds are not well-studied for strontium. Schwarzbach et al. (2006) reported mean concentration of strontium in normal clapper rail (*Rallus longirostris*) embryos was 66 mg/kg dry weight (range 30 to 95 mg/kg dry weight) compared to 121 mg/kg dry weight (range 82.2 to 176 mg/kg dry weight) in deformed embryos. However, many other contaminants co-occurred in these eggs, and strontium may not be strongly teratogenic, but rather mainly embryotoxic (Ridgway and Karnofsky 1952). Ridgway and Karnofsky (1952) injected strontium into chicken (*Galliformes*) eggs to calculate an LD_{50} for strontium. The median LD for the embryo was 73 mg/kg dry weight (assuming 55 gram egg at 80 percent moisture). The threshold effect level for that study was not determined, however. Mean strontium concentrations in egg samples from

locations at the Site impacted by the ash release ranged from 4.78 to 17.85 mg/kg dry weight, which are lower than the LD_{50} threshold mean of normal clapper rail eggs. They are also lower than means for yellow-breasted chat (*Icteria virens*) and willow flycatcher (*Empidonax traillii)* eggs that might be elevated in strontium in Arizona (24 and 35 mg/kg, Mora 2003), although no adverse effects were clearly documented. The maximum strontium concentration in eggs of the ash-impacted locations at the TVA site was 51.45 mg/kg dry weight, which is below Ridgway and Karnofsky's (1952) LD_{50} and the mean for normal clapper rail eggs in Schwarzbach et al. (2006).

Strontium in eggs was weakly but negatively correlated to nestling survival (r=0.27, Figure 8-2), mostly at low strontium concentrations (Figure 8-2). Two nests monitored with eggs containing the highest concentrations (>27 mg/kg) still had 100 percent nestling survival and the plot of nestling survival against strontium egg concentrations shows no clear trend of adverse effects as strontium concentrations increase.

Similar to strontium in eggs, thresholds for adverse effects from strontium in eggshells are not defined. Elevated levels of strontium in eggshells have been found to reduce eggshell strength and increase the number of cracked eggs, in turn reducing hatching success and increasing embryonic mortality (Mraz et al. 1967, Mora et al. 2011). Reduced egg production in Peking ducks (*Anas peking*) was reported when eggshell strontium concentrations were 120,000 mg/kg dry weight (Wheeler 1919), but it is unknown at what threshold adverse effects begin. Mean strontium in eggshells of non-raptor birds sampled in five states (California, Idaho, Arizona, Texas, and Kansas) ranged from 197 to 2,666 mg/kg dry weight (Mora et al. 2011). The mean concentration of strontium in eggshells from impacted reaches at the TVA site were below these levels (102 mg/kg in the Emory River, 123 mg/kg in the embayments, and 125 mg/kg in Tennessee River), and even the maximum (176 mg/kg) was lower than these mean. While Mora et al. (2011) found that means were lower for raptors in Texas and Michigan (39 to 98 mg/kg), in general raptors have the lowest and insectivores the highest strontium concentrations (Mora et al. 2011). Most importantly, strontium in eggshells (and eggs) of the insectivore, the tree swallow, at the Site was not negatively correlated to hatching success, and thus likely was not causing excessive eggshell cracking. Concentrations of strontium in eggshells were not negatively and significantly correlated to any productivity measures, which suggest eggshell concentrations likely are not of great concern.

Mean concentrations of strontium in nestling samples (9.4 to 14 mg/kgdry weight) from the impacted locations were lower than the mean of common grackle carcasses (39 mg/kg dry weight) and estimates of whole body concentration (approximately 32 mg/kg) of nestlings that had feather strontium concentrations (4 mg/kg) below that of healthy, normal heron populations. The healthy heron populations had geometric mean feather concentrations of 5.1 and range of 2.8 to 9.3 mg/kg (Golden et al. 2003), higher than the 4 mg/kg in Bryan et al. (2012). Pipping chicks with mean strontium concentrations of 11.3 mg/kg dry weight had hepatic oxidative stress in black-crowned night herons (Rattner et al. 2000). Unlike egg content, nestling strontium concentrations at the Site were not negatively and significantly correlated to any productivity measure.

Similar to the tree swallow nestlings at the Site, Bryan et al. (2012) found elevated selenium, strontium, and cadmium in nestling grackles nesting near an ash basin in South Carolina and indicated levels accumulated were below thresholds of concern for all but selenium. Unlike the Site, arsenic was also elevated in the South Carolina birds, but not above levels of concern.

Means of manganese concentrations in tree swallow egg samples from impacted locations ranged from 5 to 14 mg/kg dry weight with a maximum of 44 mg/kg dry weight and were almost always significantly higher than samples from one of the reference locations (Fort Loudon Dam). Excessive manganese uptake can cause nervous system dysfunction. Unfortunately, effects levels for manganese in eggs have not yet been determined (Burger and Gochfeld 2003), but mean concentrations in bird eggs measured at a variety of geographical locations ranged between 1 and 5 mg/kg dry weight (Howe and Malcolm 2004).

Manganese concentration in eggs was negatively and weakly correlated to fledglings produced per female (Figure 8-2) and has the potential to have a negative effect if too high in eggs. Mean concentrations of manganese in impacted eggs were 6.61 and 7.98 mg/kg dry weight in 2009 and 2010, respectively. These concentrations were only slightly higher than mean concentrations in reference eggs from Fort Loudoun Dam of 4.94 mg/kg dry weight. Thus, manganese concentrations are probably not high enough in eggs to be of great concern, particularly given the fledglings produced per female were similar between impacted and reference sites.

Dose, as modeled by dietary exposure models, may be a better indicator of effects than egg, eggshell, or nestling concentrations because it averages food items across the impacted locations; whereas, eggs or young in the same nest may have too high variability in concentration for one egg or nestling to accurately represent the entire nest and the mother's reproductive success. Calculated selenium dose is negatively correlated with tree swallow reproductive success (female fledglings per nesting female), which supports that high selenium doses may have adverse effects on reproduction in the tree swallow population. (Note: Swallow nest reference locations were not included in the correlation, because dietary data were unavailable for those locations.) Strontium dose shows the same negative relationship. However, number of fledglings per nesting female was not significantly lower in the impacted locations than the reference locations. Rather, depending on the analysis, hatching or nestling survival were both significantly lower; but oddly they were not significantly correlated to selenium dose. Nestling survival was, however, weakly correlated to strontium dose.

The data suggest there may be some small risk in the Emory River from selenium, strontium, or manganese, but such a conclusion is very uncertain. For selenium, only three of 64 eggs sampled in the Emory River area have selenium concentrations exceeding the LOAEL for the mallard. Sensitivity varies greatly among bird species and adverse effect levels may be higher for aerial insectivores than the mallard (Attachment 3 of Appendix Z). Effects on reproduction do not occur until 22 mg/kg selenium dry weight in eggs for the red-winged blackbird (*Agelaius phoeniceus*; Janz et al. 2010), a bird that, when near water,

mostly feeds on emergent aquatic insects in the air and insects on vegetation during the breeding season (Bird and Smith 1964); thus, it may be similar to the tree swallow in sensitivity. Tree swallow eggs had selenium concentrations below 22 mg/kg, which supports low risk of selenium to the aerial-feeding bird receptor group. Additionally, the small difference in fledglings produced per female between reference and impacted reaches was not statistically significant.

Strontium does not have well-established thresholds, and thus conclusions of risk are much more uncertain with strontium than selenium. The finding that strontium dose is correlated to fledglings produced per female suggests dietary dose has elevated concentrations in eggs, eggshells, and nestlings, and the literature supports that high elevations of strontium may lower reproduction. Although elevated in eggshells as well as eggs and nestlings, the weak correlations of egg concentrations with dietary dose and nestling survival, no correlations of eggshells and nestlings to productivity measures, low strontium concentrations compared to other studies of impacted or reference areas, and lack of significant difference in fledglings produced per female between reference and impacted areas suggest no more than low magnitude of effects and risk from strontium.

Manganese concentrations in eggs are negatively correlated to fledglings produced per female, but, unlike selenium and strontium, manganese is not correlated to dietary dose, nor elevated in nestlings or eggshells. Thresholds for adverse effects from manganese are not established, creating high uncertainty in conclusions in regard to manganese. Effects are no more than low because manganese in eggs in impacted reaches is only slightly elevated over reference concentrations, and fledglings produced per female did not significantly differ between reference and impacted areas.

Table 6-4 summarizes metal and metalloid COPECs concentrations exceeding those in reference areas in tree swallow eggs and the assessment of the level of risk of the residual ash to the aerial-feeding insectivorous bird receptor group. In summary, risk from the 26 COPECs evaluated based on tissue alone is too low to be of concern, although some uncertainty exists for selenium, strontium, and manganese. This risk assessment does not include any tree swallow post-dredging egg data collected in 2011, which may change the results if included. Like the two previous years, the 2011 productivity data do not support significant differences between impacted and reference sites in tree swallow reproduction.

Some COPECs (e.g., copper) may not be adequately evaluated for adverse effects using tissue benchmarks because the COPEC may not bioaccumulate in eggs. Thus, dietary exposure models were employed to further assess risk to aerial-feeding insectivorous birds and mammals.

8.2.2 Dietary Exposure Models

Aerial-feeding insectivorous birds and mammals are exposed to ash COPECs through dietary uptake from the food web, specifically flying insects, most of which are emergent aquatic insects at the Site. A dietary

exposure model calculates the dose predicted for each of the receptor species representative of the two assessment endpoints for this receptor group, mammals and birds. The two representative species for which a dose was modeled are the tree swallow and gray bat. The calculated dose is compared to a TRV, which is a threshold above which adverse effects are likely. The modeling methods used for this receptor group are the same as those described for aquatic and riparian-feeding birds and are detailed in Appendix W. The models assume tree swallows and gray bats feed exclusively on aerial insects, which are represented by concentrations in adult mayflies collected in 2010. This is a relatively conservative assumption, since some portion of both species diet is comprised of insects with less exposure to ash. Data collected in 2011 on mayfly concentrations were not available for incorporation into the model but recently became available for a cursory assessment of potential trends in exposure to aerial-feeding insectivores over time.

In Appendix W, Tables 11-1 through 11-11 provide the HQ results of the screening analysis, Tables 12-1 through 12-10 provide the HQ results of the refined analysis, and Tables 13-1 through 13-6 provide the results of the probabilistic analysis. The results are summarized by receptor below.

8.2.2.1 Birds (Tree Swallow)

The screening-level HQs of COPECs for the tree swallow were below 1 except for copper and selenium. In the refined analysis, copper and selenium HQs still exceeded 1, but copper did not exceed the reference reach HQs (Table 6-2). Selenium HQs ranged from 2 (LOAEL) to 5 (NOAEL) in impacted reaches in the Emory and Clinch Rivers and exceeded reference reach HQs. The Tennessee River impacted reaches had a slightly lower HQ range of 1 to 3 for selenium. For comparison, of the three reference reaches, CR_Ref exceeded HQ of 1 for selenium, but the downstream Tennessee River reaches did not. It is unknown if ER_Ref would exceed 1 because no insect data were collected in the ER_Ref in 2010 (or in ER_C, see Table 1 in Appendix W for substitutions used to calculate HQs in Table 6-3). Data on mayfly concentrations collected in 2011 show that the ER_Ref tends to have lower selenium concentrations and, thus, likely a lower dose and HQ than ER_A and ER_B (Figure 8-4).

The tree swallow HQs for selenium have a relatively low probability of exceeding 1. The probability of selenium HQ exceeding 1 ranges from 10 percent (LOAEL) to 30 percent (NOAEL) in the Emory and Clinch Rivers and 4 percent to 10 percent in the Tennessee River (Table 6-3). These probabilities were determined with the probabilistic analysis that has higher uncertainty of being protective than the screening and refined deterministic models. The probabilities are most influenced by the TRV uncertainty, and secondly by the variability in insect concentrations (Appendix W, Figure 16). The results suggest low risk to aerial insectivorous birds from selenium in the residual ash.

8.2.2.2 Mammals (Gray Bat)

The screening-level HQs of COPECs for the gray bat were below 1 except for aluminum and selenium. In the refined analysis, in the impacted reaches selenium HQs still exceeded 1 as well as the reference reach HQs. The HQs ranged from 3 to 10 (Table 6-2).

The gray bat HQs for selenium from the refined analysis are very conservative, based on reduced growth of laboratory animals. As discussed in detail in Appendix W (Attachment 3), field studies do not support adverse effects to small mammal or raccoon populations from selenium at places known to have elevated selenium such as Kesterson National Wildlife Refuge (Clark et al. 1989). Mink populations might possibly suffer adverse effects but likely at much higher doses than observed in laboratory toxicity studies of domestic mammals (Attachment 3 of Appendix W). Incorporating population-level TRVs into the range of uncertainty for TRVs, the probability of selenium HQs exceeding 1 for gray bats is near 0 percent (Table 6-3). This probability is most influenced by the TRV uncertainty, followed by the variability in insect concentrations. The low probability of high HQs suggests gray bats and other aerial-feeding mammalian populations are not likely at any risk of decline from selenium at the Site, although the probabilistic model has the highest uncertainty in being protective. Thus, risk is estimated as, at most, low and likely negligible.

8.3 Risk Characterization

A WOE approach is used to evaluate overall risk to aerial-feeding insectivores. The risk for each LOE is evaluated, and the confidence in the risk estimate and the ecological relevance of each LOE is considered when assessing overall risk.

8.3.1 Weight of Evidence

Exhibit 8-1 summarizes the magnitude of adverse effects, confidence in the magnitude, evidence of causality, and ecological relevance of each LOE of the aerial feeding insectivorous receptor group. Uncertainty in being protective is low for screening dietary exposure models, moderate for refined models, and high for probabilistic models. Magnitude of adverse effects, indicated by amount of exceedance of benchmarks, is low for all LOE. This result is supported: 1) because few eggs had concentrations above benchmarks for adverse effects for COPECs and nestling concentrations of COPECs were low; 2) concentrations of strontium in eggs, eggshells, and nestlings were not very high, and egg correlations with productivity measures were weak; 3) manganese concentrations were not much above background levels; 4) HQs were no higher than 5 in the refined analysis for the tree swallow; and 5) probability of risk for gray bats was near zero in the probability analysis. Confidence in the magnitude is moderate to high for the tree swallow egg, productivity, and dietary exposure results and low for the gray bat dietary model (because the latter is based on the probabilistic model). Causality and confidence is high for the egg study because

the study evaluated correlations of productivity/egg measures with tissue concentrations or doses. Confidence was moderate for the other tree swallow LOEs, which had less evidence for causality.

Reproductive measures are closest to evaluating population-level effects and have high ecological relevance. Because concentrations are site-specific but do not measure effects, measured concentrations in tree swallow egg, eggshell, and nestling tissues provide moderate ecological relevance. The dietary exposure models are based on site-specific biotic and abiotic concentrations and estimates of bioavailable fractions but rely on the literature to parameterize most of the rest of the model and to evaluate effects. Thus, the results of such models are of low ecological relevance.

Exhibit 7-2 summarizes the weight of each LOE, level of risk, confidence in the risk level assigned, and COPECs driving the risk. Weight is based on ecological relevance and quality of the data and analysis. Risk level is based on the egg studies and refined and probabilistic analysis of the dietary models. Potential risk was low for all LOE for one or more of the COPECs, including selenium, strontium, manganese, and copper. Sample sizes of the egg studies (except eggshells) were adequate to detect differences with 80 percent power and 95 percent confidence, and the COPECs identified in eggs had well-studied LOAEL TRVs for selenium but no available TRVs for strontium in eggs or eggshells or manganese in eggs. Selenium increased in mayflies in impacted reaches in 2011 and thus selenium results have moderate rather than high confidence that risk is no more than low. Strontium and manganese are also assigned moderate confidence of no higher than low risk.

Risk to strontium in eggshells is assigned low risk because dose was negatively correlated to reduced fledgling production. Eggshell concentrations may have increased from this dose. However, it is more likely risk is negligible based on eggshell results because of the low concentrations relative to background eggshells in the United States, and lack of correlation of eggshell concentrations to clutch size, hatching success, nestling survival or fledglings produced. This suggests eggshell cracking from strontium is not occurring.

On the other hand, strontium in the egg, not eggshell, might be creating risk, because egg content of strontium in eggs was negatively correlated to nestling survival, although the relationship is weak. While the dietary exposure model did not support the conclusion of risk from strontium dose, the TRV for the dietary study is not based on accepted Eco-SSL studies (Appendix W – Attachment 3). Additionally, the selenium dose could be causing the correlation with fledglings produced instead of strontium. Note that nestling concentrations of strontium were assigned low risk but were not correlated with any productivity measures and concentrations observed in nestlings were not considered high in the literature. The same is true of selenium in nestlings. Furthermore, changes in fledglings produced between the reference and impacted sites were not statistically significant. Thus, evidence supporting strontium in eggshells or nestlings and selenium in nestlings as decreasing productivity of aerial-feeding birds is minimal.

The evidence is also weak for manganese, which was elevated only in eggs, showed no negative correlation with dose, but had a negative correlation with fledglings produced per female. While a threshold for adverse impacts is unknown, the eggs in the impacted sites were not much higher than background levels reported around the United States, which suggests little to no impacts from manganese.

For the dietary exposure models, if a HQ for the refined NOAEL of the dietary model is below 1, risk is negligible. If higher, risk is low, even if not greater than reference HQs, although similarity to the reference will cause the COPEC to not be considered a COEC. For example, copper had a HQ > 1 for tree swallows in the refined dietary exposure analysis and is identified as a COPEC, but is not a COEC because HQs in impacted reaches were the same as reference reaches. Confidence that the risk is no higher than the level indicated for dietary exposure models is moderate for the tree swallow (based on refined analysis) and low for gray bat (based on almost no risk predicted with probabilistic analysis).

Exhibit 8-1. Aerial-Feeding Bird and Mammal Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Bird – Tree Swallow				
Reproductive Data	Low	High	High	High
Egg Tissue Concentrations	Low	High	High	Moderate
Eggshell Concentrations	Low	Moderate	Moderate	Moderate
Nestling Tissue Concentrations	Low	Moderate	Moderate	Moderate
Dietary Exposure Models	Low	Moderate	Moderate	Low
Mammal – Gray Bat				
Dietary Exposure Models	Low	Low	Moderate	Low
Weight of Evidence	**Low**	Moderate	**High**	**--**

Exhibit 8-2. Aerial-Feeding Bird and Mammal Risk Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Bird – Tree Swallow				
Reproductive Data	High	Low	Moderate	--
Egg Tissue Concentrations	Moderate	Low	Moderate	Selenium, strontium, manganese
Eggshell Concentrations	Moderate	Low	Moderate	Strontium
Nestling Tissue Concentrations	Moderate	Low	Moderate	Selenium, strontium
Dietary Exposure Models	Low	Low	Moderate	Selenium, copper
Mammal – Gray Bat				
Dietary Exposure Models	Low	Low	Low	Selenium
Weight of Evidence	**--**	**Low**	**Moderate**	**Selenium**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

As discussed in previous sections, risk is estimated to be present but, at most, low for tree swallow and gray bat. Such risk for tree swallow is assessed with moderate confidence based on reproductive measures that show a small, if any, reduction in female fledglings produced per nesting female and that probability of HQs exceeding 1 are low (10 to 30 percent) in the dietary exposure model and only implicates selenium. No evidence exists for adverse effects of selenium on small mammal populations in the field at the concentrations occurring in aerial insects at the Site using the dietary model (probability of exceeding 1 is near 0), although laboratory studies show reduced growth in young at such concentrations. Risk to gray bat is considered, with low confidence, to be present from selenium but low. Unlike birds, the field studies of effects of selenium on mammal populations show no clear adverse effects but are limited and uncertain.

Overall, the WOE supports with low to moderate levels of confidence that risk is low for aerial feeding birds and mammals on the Emory, Clinch, and Tennessee Rivers. Support for strontium and manganese causing adverse impacts is weak, given threshold benchmarks for concentrations of eggs and eggshells are unavailable. Selenium adverse effects, on the other hand, have been extensively studied, and have well-defined benchmarks that are exceeded by a very small percentage of eggs in the Site, and have minimal, if any effect on reproduction. Dietary doses are exceeded, supporting low risk. Thus, selenium is identified as a COEC.

Strontium is not listed as a COEC because the dietary model does not support impacts, the decrease in fledglings produced with increasing dose could be due to other metals or metalloids in the eggs (like

selenium, which shares the identical correlation), no evidence supports that hatching success was decreased by eggshell cracking from strontium replacing calcium in the eggshell, and strontium in eggs plotted against all productivity measures shows no clear pattern that increasing strontium is reducing survival or fledglings produced. Thresholds in eggs or eggshells for impacts to reproduction are not available in the literature, which creates too much uncertainty to warrant including strontium as a COEC.

8.3.2 Uncertainties

Uncertainties that may lead to overestimates or underestimates of risk are associated with each stage of risk assessment. It is important to understand the major uncertainties and how they might affect the risk assessment. The uncertainties associated with the two LOEs for the aerial-feeding birds and mammals are reviewed below.

8.3.2.1 Tree Swallow Tissue Concentrations of COPECs and Reproduction

Issues associated with uncertainties of the evaluation of the egg tissue concentrations from tree swallow eggs are applicability of literature-derived adverse effects levels for tissue concentrations, effects of complex COPEC mixtures, unknown concentrations below detection limits, seasonality of receptors, timing of sample collections, and representativeness of the reference locations. Reproductive measures such as hatching and fledgling success help interpret the effects of egg concentrations on productivity of aerial-feeding insectivorous bird populations but are difficult to confidently link directly to a COPEC stressor. Negative correlations found between egg, nestling, or eggshell COPEC concentrations with any of the three reproductive measures (hatching success, nestling survival, or fledglings produced per female) were weak. But the finding of a negative correlation between dose and fledglings produced per female, even though the measure of fledglings produced per female was not significantly lower than the reference locations, provides more support for some COPECs, but still demonstrates much uncertainty because the results are inconsistent (e.g., selenium is correlated to dose but not any other reproductive measures).

Chemical and physical analyses were performed using the best commercially available methods and results were reported to the lowest technically defensible level. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty. For percent moisture, insufficient sample volume was available of some biological media. Although some rejected percent moisture results could be reanalyzed, the results were qualified due to holding time exceedances. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

Interpretation also depended on literature-derived values of adverse effect levels for tissue concentrations typically determined for individuals. Such TRVs may not apply to the Site or be applicable to populations that often have adaptive and compensatory responses to stressors. Moreover, complex COPEC mixtures

can have antagonistic or synergistic effects not elucidated in single-analyte toxicity studies. For example, selenium and arsenic, which both are elevated in invertebrates at the Site, have antagonistic modes of action, whereby doses of both in combination have lower toxicity than doses of each individually (Hoffman et al. 1992). Thus, these risk estimates may be biased high for COPECs. The mixture of metals in tree swallow eggs makes it difficult to be certain that strontium or selenium are the cause of adverse effects.

Compared to heron and osprey nests, tree swallow boxes were easy to visit and monitor, producing more reliable estimates of clutch size and estimation of reproductive measures. The simpler logistics allowed sample sizes, particularly in 2010 for eggs (n = 10 to 15), to be higher for the tree swallow egg tissue study than for the heron and osprey egg tissue study, which increases power to detect differences. Moreover, egg concentrations below the analytical detection limit were assumed to be at the detection limit, which biases the risk conservatively high because this approach increased the likelihood of exceeding a benchmark for adverse effects. However, it could have decreased the likelihood of detecting differences between impacted and reference sites, of concern only if detection limits are near the benchmark. All detection limits for eggs were below the adverse benchmark of COPECs with readily available benchmarks (arsenic, lead, mercury, molybdenum, selenium, zinc, cadmium, chromium, and strontium) for all but copper, boron, and manganese. Manganese in eggs (a few, mostly in positive control, had detection limit over 5 mg/kg), nevertheless was identified as significantly different from impacted sites, and conclusions were not affected. Copper was also identified as significantly different, and only 1 non-detect in the Tennessee reference site had a high detection limit at 10 mg/kg, not enough to change conclusions. The comparison to reference for boron was influenced by the detection limit (over 20 mg/kg a level of concern for eggs, U.S. Bureau of Reclamation et al. 1998) in 2009 but detection limits were lowered in 2010 to $\leq$ 7 mg/kg, making the finding of little boron in the impacted sites more certain. Benchmarks for aluminum are unknown, but aluminum detection limits were as high as 265 mg/kg in some eggs, which may or may not be well above elevated or adverse levels, given aluminum considered elevated in domestic quail eggs is 67 mg/kg (Abduljaleel and Shuhaimi-Othman 2011) and is often lower for wild birds at < 5 mg/kg (Nelson and Esmoil 1999).

Power to detect nestling or hatching success absolute differences of 15 to 20 percent between impacted and reference reaches (or a difference of 1 fledgling produced per female), respectively was high at 80 percent at 95 percent confidence with the sample sizes used in the tree swallow productivity study. Power was similarly high for detecting differences in egg and nestling concentrations, but required slightly higher sample sizes for eggshells to achieve that level of power and confidence.

Seasonality of receptors can change results because some aerial-feeding insectivorous bird species are not year-round residents and are less exposed than year-round residents at the Site. Results based on year-round residents bias the results high for migratory aerial insectivorous birds such as the tree swallow. The reference reaches are not true controls for the impacted locations, especially on rivers, which by their nature change with distance downstream. Differences in egg tissue concentrations attributed or not

attributed to the ash may be from natural variation in sediment conditions and food items, which are incorporated into the second LOE, the dietary exposure models.

The risk assessment of the residual ash post-dredging is uncertain, given that eggs and nestlings were collected from birds that may not have had opportunity to benefit from ash removal by dredging because they were collected in 2009 and 2010. However, the 2011 mayfly data do not show a reduction in selenium concentrations (Figure 8-4), and the benefit may not yet be observable until selenium begins to be depurated from the food web. These uncertainties reduce confidence in the assessment of risk from the tree swallow results from high to moderate confidence (Exhibit 8-2).

The study of tree swallow reproductive success and trace elements in tissue was continued in 2011 by Virginia Polytechnic Institute and State University. Preliminary results are available on reproductive success (Hopkins et al. 2012). Analysis of trace element concentrations, monitored in eggs and nestling blood, has not been completed. The 2011 study evaluated egg, nestling, and survival parameters as in the previous year but also included endocrine responses to stress of fledglings and composition of the nestling diet. The study of boluses fed to the young indicated 67 percent of the invertebrates in the diet had an aquatic life stage, supporting the assumption that tree swallows depend on food from the river, although the dietary exposure model conservatively assumes 100 percent of the food is from the river. Impacted and reference locations did not differ in the proportion of young that survived to fledging (day 17), and nestlings in the spill area were generally in good health. One of the two nesting sites in the impacted Emory River, as well as a historically contaminated site (Melton Hill Dam) and a reference site (Fort Loudon Dam) experienced twice the reproductive impacts of other monitored locations during an unusual cold-weather event in May. It is premature to draw firm conclusions in the absence of trace element data, but this preliminary evaluation does not change the overall interpretation of risk to aerial-feeding insectivorous birds.

8.3.2.2 Dietary Exposure Model

Many uncertainties exist in the dietary exposure model and are detailed in Appendix W. Of variables tested in a sensitivity analysis, the results are generally most sensitive to the literature-derived TRVs selected and secondly to the measured tissue concentrations (EPCs) of the food item, adult mayflies. In contrast, variability in body weight, the only other parameter varied in the tree swallow and gray bat probabilistic models, has a smaller effect on the results.

Assumptions for other parameters not subject to a sensitivity analysis (fixed in the model) may also affect the results. For example, exposure parameters except EPCs were obtained from the literature and assumed to be applicable to the Site, which may not be true.

The TRVs are from the literature and are associated with significant uncertainty (Dale et al. 2008), which was incorporated into the probabilistic analysis for COPECs, but only for COPECs identified as possible risk

drivers during the refined analysis. Nevertheless, toxicity data are only available for a limited number of species, mostly laboratory test species, under a strictly defined set of test conditions that deviate from natural conditions (Sample et al. 1996; Suter 1996). Tolerance and adaptation of the receptors are not considered for laboratory studies (Grant 2002; Millward and Klerks 2002) but do play a role in field studies included in some of the Monte Carlo probabilistic models for selenium. Relatively few studies evaluate the effects of toxicity at the population scale that integrate reproduction and survival endpoints or incorporate concepts of carrying capacity, intraspecific competition, density-dependence, or dispersal, and none were available to evaluate selenium effects on population size and growth. However, the reproductive study discussed in this BERA is closer to evaluating population-level effects and shows a small, if any (because statistically insignificant), reduction in reproduction (fledglings produced per female). Because of the TRV uncertainty in the dietary exposure model, it was critical to evaluate uncertainty of TRVs that have a deterministic HQ exceeding 1 (in this case, selenium) in a probabilistic assessment.

For the deterministic analysis, the EPCs used in the dietary exposure model were conservative, biasing the dose high, because the EPCs were the 95 UCL or maximum concentration for a reach (Appendix W), rather than the average. The probability analysis used the average and standard deviation of these concentrations, which is more accurate.

The detection limits for some constituents, notably selenium in water and sediment, were higher than potentially ecologically important concentrations. This is a well recognized and largely unavoidable issue for some bioaccumulative constituents like selenium. This may lead to underestimation of exposures and obscure the identification of exposure pathways. Non-detected concentrations of all constituents were conservatively assumed to be present at the RL to compensate for this uncertainty.

Percent moisture was analyzed in biological samples, but insufficient sample volume was available for some biological media. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

The use of the mayfly concentration data to represent the insects is uncertain, but data for other insects were unavailable. Likely, aerial-feeding insectivores feed on some terrestrial insects that provide a lower dose of selenium. Additionally, the percentages of diet and body weights from the literature have some inherent uncertainty because data are neither available for an entire population nor available for all ages and all seasons.

Tissue sample collection was often clustered and limited in each reach (Appendix W, Figure 2). Good representation of reach exposure concentrations and their distribution is uncertain. The timing of sample collections was during or just after dredging completion, and the adult mayflies captured may have been exposed to some pre-dredging conditions. Dredging occurred only in Emory River reaches B and C, and engineering controls were used to minimize migration of disturbed sediment and ash. The 2011 selenium

concentrations in mayflies were similar or higher than 2010 (Figure 8-4), which would not likely significantly change the conclusions if used in the dietary exposure models.

In addition, species represented by this assessment endpoint differ with respect to water and food ingestion rates, absorption, metabolism, distribution, and excretion of chemicals. With these uncertainties in the dietary exposure model, confidence in the risk conclusions was low. Because of these various uncertainties for the differing LOEs, the WOE approach was used to evaluate if the sum of the information can lead to valid conclusions, which are summarized below.

8.3.3 Summary and Conclusions

Based on the overall WOE for aerial-feeding insectivorous birds and mammals, a low risk to species that feed on aerial insects exists at the ash-impacted locations of the Emory and Clinch Rivers (reaches A and B, Table 8-2), based on data collected mostly before completion of dredging.

COPECs with potential adverse effects that exceeded screening levels from the dietary exposure model or potential egg benchmarks of adverse effects are copper, selenium, barium, manganese, and strontium. The latter four had weak correlations to tree swallow productivity measures. In the refined dietary analysis, only copper and selenium exceeded benchmarks, but copper was eliminated from concern because HQs were not different than the reference reach. COPECs remaining that might drive risk based on a refined dietary analysis and some evidence that ash is causing adverse effects based on the egg study were selenium and strontium. The evidence for strontium driving risk in eggs is highly uncertain and not well-supported (no available benchmarks for adverse effects), and therefore only selenium is the final COEC for aerial feeding insectivorous birds and mammals.

9. Amphibians

Amphibians are a receptor of particular concern for the BERA because 2.51 acres of wetland habitats in the embayments were covered with ash (Jacobs 2009). Amphibians have been considered a "sensitive sentinel to environmental change" due to their unique life cycle, which exposes them to aquatic dissolved metals, as well as metals sequestered to sediments (Hopkins and Roe 2009 as cited in Sparling et al. 2010). Not only are dermal and ingestion exposure pathways to constituents significant in wetlands, but amphibians are also exposed to constituents in a variety of life stages. As a result, higher concentrations of metal and metalloid COPECs have the potential to elicit adverse effects in amphibians.

Accumulation of these ash-related metal and metalloid COPECs may start early on for anurans (frogs and toads), as literature studies have suggested maternal transfer of some constituents into the egg. Shortly after an egg is deposited into the aquatic environment, aqueous uptake of constituents into the egg can also occur. Both larval and adult anurans experience dermal and respiratory uptake pathways (Birge et al. 2000; Kadokami et al. 2002, 2004; and Hopkins et al. 2006 as cited in: Sparling et al. 2010). In addition, for metals such as mercury and selenium, exposure from food sources and sediment is more substantial than concentrations found in the aqueous phase. Other metals such as aluminum and copper are typically taken up via exposures to the water (Pickhardt et al. 2006 as cited in Sparling et al. 2010).

Some species (e.g., toads and tree frogs) may have high exposures during specific early life stages, but may migrate to less-contaminated habitats as adults. Other species, such as ranid frogs, may be more continuously exposed to impacted surface water and sediment.

9.1 Problem Formulation

The environmental setting for amphibians consists of both riverine (e.g., Emory, Clinch, and Tennessee Rivers) and ephemeral aquatic habitats. However, with the exception of ranid frogs, most species of amphibians at the Site breed in ephemeral aquatic habitats or small streams.

Three amphibian target species (anurans) were collected from five locations (Table 9-1; Figure 2-27). Anurans were sampled from impacted locations (West and North Embayments), locations just north of the ash release identified as un-impacted (Dawson Farm and Rocky Top Farm), and a reference location pond in Knox County, Tennessee (Timberlake Subdivision).

Impacted locations include the West and North Embayments. Both embayments were wetlands on TVA land that were previously monitored as part of the Reservoir Operation Study Design (TVA 2004). Both embayments have hydric soils with vernal pools and are considered important to various amphibian species. The embayments were surrounded by farm areas, residential areas, and roadways (Jacobs 2009). The West Embayment included 1.55 acres identified as palustrine emergent wetland and 0.65 acres identified as

palustrine forested wetland using the Cowardin classification system (1979) (Jacobs 2010b). The North Embayment included 0.30 acres of scrub/shrub wetland. Both embayments were covered by ash during the release and had ash in the collection areas during the 2009 sampling event. However, ash in the West Embayment was removed in the summer of 2009, and the area was remediated prior to the 2010 sampling event. The North Embayment was covered with ash and inundated with water during the 2009 and 2010 sampling events (Jacobs 2011b).

Both Dawson Farm and Rocky Top Farm are located approximately 0.5 to 1 mile from the North Embayment (Figure 2-27) and were not considered to be impacted by the ash release. The Dawson Farm is classified as a palustrine, scrub-shrub, broad-leaved deciduous, temporarily flooded wetland (PSS1A; U.S. Fish and Wildlife Service [USFWS] 2004). Rocky Top Farm is a wetland in a floodplain, adjacent to the northern portion of the North Embayment. Sampling in both farms was conducted in scrub-shrub wetland areas with open water and adjacent hayfields.

The reference location pond was a man-made subdivision pond located in Knox County, Tennessee (Timberlake Subdivision). Amphibians were sampled from the pond in 2009 and 2010 in the wetland areas, as well as from small drainage areas nearby. The pond is in a depression area surrounded by suburban development; however, it has no deliberate anthropogenic point sources of contaminants. Thirteen species of amphibians commonly occur in the vicinity of Watts Bar Reservoir (Table 2-6).

There are four amphibian species listed for Roane County as state-endangered species by the Tennessee Natural Heritage Program (http://tn.gov/environment/na/pdf/county.pdf): Eastern hellbender, green salamander (*Aneides aeneus*), Berry cave salamander, and four-toed salamander. The four-toed salamander and eastern hellbender are listed as in need of management (Table 2-10).

There are approximately 22 species of salamanders and 14 species of frogs and toads that can be found in Roane County, Tennessee (Klein 1989; Redmond and Scott 1996; Scott and Redmond 2008). A complete list of the representative amphibian species of potential occurrence observed in this area is provided in Table 9-2 and generally confirms that the potential receptors selected for the BERA are relevant.

Amphibians can inhabit both aquatic and riparian habitats, depending on their life stage. Within their early life stages, amphibians are generally aquatic, consuming mainly plants and insects. However, as they mature, amphibians develop lungs and their diet changes to a variety of aquatic and terrestrial organisms. Amphibians, such as frogs and salamanders, can include herbivores, omnivores, invertivores, and carnivores. Throughout their lifespan, amphibians can act as both predators to smaller species, as well as being prey items for higher trophic-level predators. As a result, amphibians are vital to a balanced ecosystem by helping to maintain aquatic and riparian vegetation, invertebrate and fish communities, as well as regulating other mid to upper trophic-level organisms. Amphibians have the potential for direct exposure and adverse effects, as well as the potential for accumulation and transfer of constituents to higher trophic-level

consumers. They can make up a large component of the aquatic and riparian food web, providing an important food resource for organisms such as larger fish, birds, and mammals.

The primary exposure media considered for amphibians include surface water, shallow sediment (seasonally-exposed), and tissue. Based on the shoreline and embayment wetland habitat preference of most amphibians that occur within the Site, submerged sediment was not considered to be a significant exposure medium for amphibians. Although the dietary exposure pathway for amphibians is complete, dietary-based TRVs for quantitative risk assessment are not available, and the dietary pathway was not evaluated.

The COPECs for amphibians were identified in Section 2.2 and were based on literature suggesting evidence for adverse effects or the potential for such effects (USEPA 1997a). COPECs include a suite of 26 metal and metalloid COPECs (Table 9-3).

COPEC fate and transport considerations for amphibians were focused on the general behavior of the metals COPECs in surface water. Sources of metals contamination to surface waters include both natural processes and industrial discharges. Metals are transported in surface water either as dissolved species or as an integral part of suspended sediments (Meade 1995). Volatilization and sorption to riverbed sediments are dominant fate processes.

A literature review of the potential toxicity of metal and metalloid COPECs in amphibians revealed significant effects on survival, development, and reproduction of amphibians. Direct contact of surface water with the gills and dermis is considered a primary route of exposure. Therefore, exposures to aquatic eggs, larval stages, and breeding adults of amphibians are considered. In general, toxicity data for adult amphibian exposures are lacking. Although ingestion of food and sediments are likely exposure routes, these routes were not considered due to the lack of available toxicity benchmarks.

9.1.1 Assessment Endpoints

The assessment endpoint for amphibians in this BERA is the viability of the amphibian community.

Based on this assessment endpoint, the ecological risk question established for the evaluation of amphibians at the Site is as follows:

- Do concentrations of ash-related COPECs in surface water, submerged sediment, shallow sediment (seasonally-exposed), or sediment porewater pose unacceptable risks to amphibian populations?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective needs to be evaluated in the WOE assessment. This risk question is used to help focus subsequent steps of the amphibian assessment, including the selection of measurement endpoints.

9.1.2 Measurement Endpoints

The measurement endpoints for amphibians selected for the BERA are as follows:

Measures of exposure:

- COPEC concentrations in surface water (total and dissolved)
- COPEC concentrations in sediment
- Health metrics of exposure – measured whole body tissue COPEC concentrations of selenium (and other bioaccumulators) in amphibians.

Measures of effects:

- Literature-derived toxicity data.

9.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to amphibians. The LOEs used to evaluate amphibians include:

- Whole body frog and toad tissue data
- Surface water and sediment chemistry data evaluation.

Each LOE is summarized below.

9.2.1 Amphibian Tissue Concentrations of COPECs

Tissue data provide a direct measure of exposure of amphibians to COPECs from surface water, sediment, and food items at the Site. Amphibians were sampled in order to determine their exposure to metals and to establish a comparative baseline for any future studies. Target species included the American toad, spring peeper, and the upland chorus frog that were found in wetland areas near the ash release area. Study objectives were to:

- Quantify tissue concentrations of COPECs in three different species for among location and between-year comparisons
- Assess risk to amphibians by relating concentrations measured at the impacted locations to reference location concentrations and literature derived effects values, when available.

Appendix AA details the study objectives, methods, results (including tables and figures), and interpretation of the amphibian tissue study. The study is summarized below to assess risk to amphibians.

9.2.1.1 Analysis of Exposure and Effects

Over 180 amphibians representing three species were sampled for the tissue concentrations study. COPECs concentrations in tissues were statistically compared among impacted and reference locations during 2009 and 2010, a period before dredging activities were completed. Data collected post-dredging during 2011 were not statistically analyzed, but are presented and discussed qualitatively in the uncertainty section. The evaluation of the tissue concentration LOE for residual risk is conservative because it does not take into account the lower concentrations post-dredging.

COPECs with relatively high frequency of detection and present in statistically higher concentrations at the ash-impacted locations (ANOVA; Appendix AA) than at the reference locations were barium, copper, selenium, and strontium. Of the COPECs with high detection frequency, only barium, selenium, and strontium have been reported at higher concentrations in other ash-impacted areas (Rowe et al. 1996).

Barium was only significantly higher in the spring peeper, and this was only for pairwise comparisons between West Embayment samples and reference location samples; however, there are no clear discernible differences when evaluating differences between both years and locations (Appendix AA, Figure 25). Based on the limited literature background concentrations, this may be due to natural variation in the river system from year to year. Although previous studies have not reported copper statistically higher in anuran tissues at ash-impacted areas, the present study revealed that differences between ash-impacted and reference locations existed (Appendix AA, Figure 26). American toad and spring peeper samples from the 2009 West Embayment had the highest mean concentrations for copper. Copper was statistically higher than the reference locations at the West Embayment compared with average concentrations in American toad and spring peeper samples of 8.73 mg/kg and 27.18 mg/kg, respectively.

Few effects levels have been established for copper in anurans, partly because of the protective mechanism by which excess copper is sequestered in lysosomes in the liver, which has been noted to protect the cell from toxic effects from copper (Goldfischer et al. 1970 as cited in Eisler 1998). NOEC CBRs for copper have been reported as 16 to 79 mg/kg wet weight in *Rana* tadpoles and 93 mg/kg in *Bufo* tadpoles for both survival and sub-lethal effects on length and weight (NAVFAC 2004). No effects have been reported in frogs with liver concentrations of > 2,000 mg/kg dry weight of copper, and background concentrations have been noted for whole cricket frog (*Acris crepitans*) tadpoles (9.8 to 15.7 mg/kg dry weight), whole body gray treefrog (*Hyla versicolor*) (7.4 to 12.6 mg/kg), and whole body adult southern toad (*Anaxyrus terrestris*) (20.80 mg/kg dry weight) (Eisler 1998; Hopkins et al. 1998). Three species of anurans had higher levels of selenium, and two had higher levels of strontium compared with the reference locations (Appendix AA, Figures 29 and 30).

American toad, spring peeper, and upland chorus frog samples had selenium concentrations within previously reported reference/control site ranges (Hopkins et al. 1998; Hopkins et al. 2006; Snodgrass et al. 2004). This suggests that significant bioaccumulation of selenium has not occurred and that the statistical differences observed may be a reflection of natural variability. Concentrations at the locations were lower than those concentrations of selenium known to be embryotoxic to some fish and birds at 4 to 16 mg/kg dry weight (Lemly 1996; Ohlendorf 2003). While upland chorus frog samples had no differences in strontium, concentrations in spring peeper samples from the 2009 and 2010 West Embayment area and American toad samples from the 2009 West Embayment and 2010 North Embayment areas were closer to the contaminated site concentrations from previous studies. Reduced survival to metamorphosis in anurans (Snodgrass et al. 2004) and salamanders (Roe et al. 2006) has been linked to exposure to coal combusted waste. However, literature effects values are lacking for selenium and strontium. In the mole salamander (*Ambystoma talpoideum*), whole body sample strontium concentrations of 250 mg/kg dry weight were associated with reduced survival to metamorphosis by 57 to 77 percent (Roe et al. 2006). Larval mole salamanders are more sensitive than adult anurans and, therefore, provide a protective comparison level. Concentrations of strontium from amphibians at the locations were below this conservative level, as the highest concentrations were from the 2009 and 2010 West Embayment for American toad samples of 206 mg/kg dry weight and 241 mg/kg dry weight, respectively (Table 3 in Appendix AA).

9.2.1.2 Risk Estimation

The COPECs with concentrations in amphibian whole body tissue that were higher than reference locations were compared to adverse effect levels reported to assess the risk of such higher concentrations. If some concentrations are above the benchmark, suggesting potential adverse effects, evidence of those concentrations coming from the ash and adversely impacting the population was evaluated.

HQs were used to compare amphibian tissue COPEC concentrations to literature-derived thresholds deemed protective of assessment endpoints. Adverse ecological effects are possible if any HQ exceeds 1 (i.e., exposure exceeds threshold level). Separate HQs were calculated for each species at each location (Tables 9-4 through 9-7). With the exception of copper in spring peeper and chorus frog samples in 2009 at the West Embayment (Table 9-5), the mean HQ was < 1 for the species from the other locations. The lower end of the NOEC range (16 to 79) for copper was conservatively used. When considering the higher end of the NOEC, the mean HQs would be less than 1. Although zinc in amphibian tissue was not statistically higher than reference locations, the maximum HQ was less than or equal to 1 in chorus frog tissues from the north embayment in 2009 (Table 9-4) and in chorus frog tissues from the West Embayment in 2010 (Table 9-7).

Tissue benchmarks were not available for selenium and strontium. However, the mean concentrations of selenium and strontium in amphibian whole body tissue are presented in Table 9-8 and Figures 9-1 and 9-2, respectively, to illustrate the variability of concentrations and the lack of a clear trend of bioaccumulation

over time. The evaluation of COPEC concentrations in amphibian whole body tissue supports the conclusion that negligible risks exist for amphibians at the Site.

9.2.2 Surface Water Chemistry

One LOE to evaluate potential risks to amphibians is the comparison of surface water data to acute and chronic water quality criteria. This method utilizes the existing surface water data from the near-location and reference reaches and compares the data to regulatory benchmarks that were developed to protect aquatic resources and to amphibian-specific screening values. The ratios of the measured concentration to the regulatory benchmarks are used to calculate HQs for individual chemicals and individual reaches.

9.2.2.1 Analysis of Exposure and Effects

Although surface water samples were collected from river locations, it was conservatively assumed that theses sample locations are representative of amphibian exposure areas. However, primary amphibian habitats for most species at the Site include ephemeral wetlands and embayments. Ranid frogs are the only amphibian species that primarily use riverine habitat.

For an initial evaluation, available USEPA Region 4 water quality screening levels (USEPA 2001b) were used as generic effects levels. Amphibian-specific chronic water TRVs were developed for surface water COPECs based on the approach presented in NAVFAC (2004) and on the LC_{10} values developed by Westerman et al. (2003).

Amphibian screening values were developed for cadmium, copper, mercury, and zinc by the U.S. Navy using both laboratory testing and a literature review of the amphibian ecotoxicological literature (NAVFAC 2004). Preliminary effects concentrations were based on the 10th percentile and 50th percentile of the toxicity distribution calculated using methods described by Solomon et al. (2001). With the exception of the chronic/10th percentile values for zinc, the threshold values calculated using available amphibian mortality data were higher than their respective acute and chronic AWQC (NAVFAC 2004). It should be noted that this approach only considered lethal effects data, and that the resulting values are not directly comparable to AWQC. However, the U.S. Navy further evaluated toxicity of the four metals to larval amphibians (*Bufo* and *Rana*) exposed to sediment/hydric soil in the laboratory. NOECs and LOECs were determined for both lethal and sub-lethal endpoints in amphibian tissue, sediment, and overlying water. Sediment, water, or tissue concentrations above the laboratory-derived NOECs require additional evaluation. Concentrations below the NOECs are unlikely to cause adverse impacts to the local amphibian population (NAVFAC 2004).

9.2.2.2 Risk Estimation

Surface water COPEC concentrations were below USEPA Region 4 water quality screening levels (USEPA 2001b) (Table 9-9) and amphibian-specific literature based Naval Facilities Engineering Command NOECs and LOECs (NAVFAC 2004) (Table 9-10). The AWQC for aluminum was developed using toxicity data from acidic waters and, therefore, is not considered applicable to the habitats available within the Site. Similarly, the LC_{10} value for iron likely overestimates toxicity of iron to amphibian species (Westerman et al. 2003) and an embryo-larval LC_{50} value of 0.470 mg/L was reported for *Bufo fowleri* (Birge et al. 2000). Additionally, aluminum and iron concentrations were higher in background samples. The evaluation of COPEC concentrations in surface water supports the conclusion that negligible risks exist for amphibians at the Site.

9.2.3 Sediment Chemistry

Another LOE to evaluate potential risks to amphibians is the comparison of seasonally-exposed sediment data to sediment quality screening levels and amphibian-specific sediment NOECs and LOECs. The ratios of the measured concentration to the regulatory benchmarks are used to calculate HQs for individual chemicals and individual reaches.

9.2.3.1 Analysis of Exposure and Effects

Only shallow sediment (seasonally-exposed) data were considered to be representative of amphibian exposure areas. Additionally, primary amphibian habitats for most species at the Site include ephemeral wetlands and embayments. Ranid frogs are the only amphibian species that primarily use riverine habitat, but these species typically use shoreline areas.

For an initial evaluation, available USEPA Region 4 sediment quality screening levels (USEPA 1994b) were utilized as generic effects levels. Chronic sediment TRVs were developed for sediment COPECs based on the approach presented in NAVFAC (2004).

Amphibian-specific sediment NOECs and LOECs were determined for both lethal and sub-lethal endpoints (NAVFAC 2004). Sediment concentrations above the laboratory-derived NOECs require additional evaluation. Concentrations below the NOECs are unlikely to cause adverse impacts to the local amphibian population (NAVFAC 2004).

9.2.3.2 Risk Estimation

Sediment HQs were calculated by comparing COPEC concentrations in seasonally-exposed sediment samples to chronic sediment based TRVs. These TRVs were based on sediment quality screening levels (USEPA 1994b) and amphibian-specific literature-derived TRVs.

Mean sediment HQs exceeded USEPA Sediment Quality Screening Levels (USEPA 1994b) (Tables 9-11 and 9-12) for the following COPECs:

- Arsenic (CR_A and CR_B; ER_A and ER_B)
- Copper (CR_A)
- Lead (CR_A)
- Mercury (CR_A and CR_B)
- Nickel (CR_A; ER_A and ER_B).

Mean sediment HQs were less than or equal to 3. Sediment HQs were below amphibian-specific literature-derived TRVs (Tables 9-13 and 9-14). As a result, the sediment chemistry evaluation indicates no risks at a spatial scale that would be relevant to mobile amphibians. For example, estimates of the home range of amphibians have been reported to be 0.065 hectare (ha) for the American toad (Fitch 1958), and 0.0641 to 0.6024 ha for the upland chorus frog (Kramer 1974).

9.3 Risk Characterization

Risks to amphibian populations were evaluated based on the toxicological assessment of surface water, sediment and tissue chemistry data (Exhibit 9-1). The ecological relevance, potential risk, and confidence in the risk determination are summarized in Exhibit 9-1. Based on concordance of the LOEs, potential risks to amphibians are considered negligible with moderate confidence.

Exhibit 9-1. Amphibian Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Tissue Concentrations	Low	Moderate	High	Moderate
Surface Water Chemistry	Low	Moderate	Moderate	Low
Sediment Chemistry	Low	Moderate	High	Low
Weight of Evidence	**Low**	**Moderate**	**Moderate**	**--**

9.3.1 Weight of Evidence

An integration of multiple LOEs is shown in Exhibit 9-2. For each LOE, the evidence of causality, estimate of magnitude of adverse effects, confidence in magnitude of adverse effects, and ecological relevance is integrated and presented in Exhibit 9-2. Overall, evidence of causality is moderate with a low estimate of magnitude of adverse effects and moderate confidence in the magnitude of adverse effects.

Exhibit 9-2. Amphibian Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Tissue Concentrations	Moderate	Low	Moderate	--
Surface Water Chemistry	Low	Negligible	Moderate	--
Sediment Chemistry	Low	Low	Moderate	--
Weight of Evidence	**--**	**Low**	**Moderate**	**--**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

In summary, there are low risks to the amphibian populations at the Site based on the following LOEs:

- Chemical concentrations in amphibian whole body tissue are substantially below levels associated with adverse effects
- Surface water chemistry data indicate no toxicity to aquatic life, including an evaluation of amphibian-specific surface water benchmarks
- Sediment chemistry data indicate low toxicity to amphibians including an evaluation of amphibian-specific sediment toxicity benchmarks.

9.3.2 Uncertainties

This BERA for amphibians was conducted using conservative exposure parameters to ensure, to the extent possible, that potential risks were not underestimated. General sources of uncertainty in any BERA, including this BERA, include the selection of indicator species, the presence of these species at the Site and the level of activity at the Site, exposure estimation, and the lack of available toxicological benchmarks for several COPECs. Focused discussions of the key uncertainties for evaluating amphibian risks are provided below.

9.3.2.1 Receptor Selection

Differences in feeding habits, habitat, behavior, and activity patterns of amphibians can result in varying exposure to COPECs. The American toad, spring peeper, and upland chorus frog were assumed to be

appropriate indicator species based on knowledge of the potential species that may use the Site and their exposure potential. There is a chance that these species may not represent the most sensitive species that occur at the Site. However, given the fact that the HQ results for these species, under very conservative exposure and effects assumptions, were below 1, it is unlikely that risks to any amphibian species exist.

Direct measures of effects to endangered or threatened species are not usually possible. Therefore, surrogate indicator species are selected to evaluate potential measures of effect to special status species. Of the threatened and/or endangered species that may occur in the vicinity of the Site, the amphibian indicator species selected would be relevant and protective in terms of potential exposure. Based on the LOEs evaluated for amphibian indicator species in this BERA, low adverse effects would be expected. It is unlikely that threatened and/or endangered species would have significantly higher exposures than the surrogate amphibian species examined.

9.3.2.2 COPEC Exposure Estimation

Estimation of the COPEC exposure involves several uncertainties including the measurement of COPEC concentrations and the concentrations estimated to be taken up from media. Based on the conservative assumptions used throughout this BERA, risk results were likely overestimated; not underestimated.

9.3.2.3 Percent Moisture

Percent moisture was analyzed in biological samples so that concentrations reported in wet weight could be converted to dry weight if necessary. Insufficient sample volume was available for some biological media. Some rejected percent moisture results could be reanalyzed, but the results were qualified due to holding time exceedances. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low

9.3.2.4 Bioavailability of COPECs

The assessment of the bioavailability of COPECs to amphibians in this BERA had low to moderate confidence in the risk determination. For most species of wildlife and amphibians, the bioavailability of COPECs in environmental media is generally lower than in the exposure media employed in toxicity tests.

9.3.2.5 Effects on Individual Organisms versus Populations

This evaluation is intended to assess risks to amphibian populations. However, the toxicological evaluation of chemical concentrations generally relies on information that is most applicable to individual organisms. Site-specific population-level data are not available for amphibians. The relationship between individual and population-level effects is, thus, a significant source of uncertainty for amphibians.

As with most field surveys, uncertainties exist when evaluating species that have the ability to migrate or reside in systems of complex COPEC mixtures. The evaluations in the tissue study focused on comparisons among locations, between years, and with literature-derived concentrations in order to assess the relationship between the ash release and COPECs in amphibians. Given the lack of anuran-specific literature-derived effects values for many of the COPECs found in ash, it is difficult to discern level of effect that the anuran populations have potentially experienced due to the ash release. Furthermore, the natural variability or historical contamination associated with the sample locations have potentially contributed to the differences found among locations. A review of 2011 data for selenium and strontium (Figures 9-1 and 9-2) indicate lower concentrations for spring peeper and upland chorus frog samples, but increased concentrations in American toad samples from the West Embayment only. Overall, there does not appear to be a clear trend of bioaccumulation over time in amphibians.

9.3.3 Summary and Conclusions

The risk evaluation for amphibians uses a combination of conservative and central tendency estimates for the exposure assessment and conservative estimates for the effects assessment. This approach very likely results in a significant overestimation of potential risks to amphibians. Even so, this BERA demonstrates that COPECs at the Site pose low risks to amphibians.

10. Reptiles

The sensitivity of reptiles to many environmental chemicals and associated ecological hazards are unknown. However, the position of reptiles in the food web enhances the potential for biological concentration of environmental contaminants and, thus, increases usefulness of this class of vertebrates as a sentinel species of exposure and toxicity of contaminants. Aquatic turtles were selected as a representative reptile receptor species for this BERA. Turtles are either tertiary consumers or secondary consumers depending on the species and their respective food habits. Furthermore, the life-history characteristics of these animals increase their suitability for use as sentinel species for environmental contamination. Limited information from field studies indicates that the metal concentrations reported in tissues did not appear to have adverse effects on turtles (Albers et al.1986; Yu et al. 2011). The lack of toxicity values are a limiting uncertainty in any BERA for reptiles (Sparling et al. 2010).

Concentrations of some ash-related COPECs may bioaccumulate in wildlife over time and adversely affect those wildlife populations. Risks to turtles may be assessed through comparisons of measured concentrations in blood and other tissues with effects values from scientific literature. Literature values are of limited availability for many COPECs and species; therefore, body burden concentrations may be only supplemental evidence of exposure. Metal and metalloid COPEC concentrations in turtles were compared between impacted and un-impacted locations, as well as between years. Elevated concentrations at impacted locations and changes in concentrations over time may indicate increased exposure and, potentially, risk to reptiles.

10.1 Problem Formulation

The environmental setting for turtles consists of both riverine (e.g., Emory, Clinch, and Tennessee Rivers) and ephemeral aquatic habitats. Turtles were sampled in order to determine their exposure to metal and metalloid COPECs and to establish a comparative baseline for any future studies. Sampling locations (Table 10-1; Figure 2-28) included locations in the Emory River, Clinch River to its confluence with the Tennessee River, Tennessee River below the confluence with the Clinch River, and Tennessee River above the confluence with the Clinch River (the 2010 Reference Area). Specimens were also collected from a reference location in nearby Knox County (the 2009 Reference Area). Eleven species of reptiles commonly occur in the vicinity of Watts Bar Reservoir (Table 2-6).

The northern pine snake is listed for Roane County as a state-endangered species by the Tennessee Natural Heritage Program (http://tn.gov/environment/na/pdf/county.pdf). The eastern slender glass lizard is listed as in need of management (Table 2-10).

There are approximately 13 species of turtles, 10 species of lizards, and 23 species of snakes that can be found in Roane County, Tennessee (Klein 1989; Redmond and Scott 1996; Scott and Redmond 2008). A

complete list of the representative reptile species of potential occurrence observed in this area is provided in Table 10-2 and generally confirms that the potential receptors selected for the BERA are relevant.

Reptiles can inhabit both aquatic and riparian habitats. They consume a variety of plants, invertebrates, other reptiles, amphibians, fish, small birds, and small mammals in both ecosystems. Reptiles, such as turtles and snakes, can include herbivores, omnivores, invertivores, and carnivores. Throughout their lifespan, reptiles can act as both predators to smaller species, as well as being prey items for higher trophic-level predators. As a result, reptiles are vital to a balanced ecosystem by helping to maintain aquatic and riparian vegetation, invertebrate and fish communities, as well as regulating other mid- to upper trophic-level organisms. Reptiles have the potential for direct exposure and adverse effects, as well as the potential for accumulation and transfer of COPECs to higher trophic-level consumers. They can make up a large component of the aquatic- and riparian-food web base, providing an important food resource for organisms such as larger fish, birds, and mammals. As a result, the viability of the reptile community was selected as a valid BERA endpoint.

The primary exposure media considered for reptiles include surface water, submerged sediment, shallow sediment (seasonally-exposed), and tissue. Although the dietary exposure pathway for reptiles is complete, dietary-based TRVs for quantitative risk assessment are not available, and the dietary pathway was not evaluated.

The COPECs for reptiles were identified in Section 2.2 and were based on literature suggesting evidence for adverse effects or the potential for such effects (USEPA 1997a). COPECs included a suite of 26 metal and metalloid COPECs (Table 10-3).

COPEC fate and transport considerations for reptiles are similar to that of amphibians and include the general behavior of the metal COPECs in surface water. Metals are transported in surface water either as dissolved species or as an integral part of suspended sediments (Meade 1995). Volatilization and sorption to riverbed sediments are dominant fate processes. In general, toxicity data for turtles are lacking. Although ingestion of food and sediments are likely exposure routes, these routes were not considered due to the lack of available toxicity benchmarks.

10.1.1 Assessment Endpoints

The assessment endpoint for reptiles in this BERA is the viability of the reptile community. Based on this assessment endpoint, the ecological risk question established for the evaluation of reptiles at the Site is as follows:

- Do concentrations of ash-related COPECs in surface water, submerged sediment, shallow sediment (seasonally-exposed), or food items pose unacceptable risks to reptile populations?

If unacceptable risks are determined, the spatial extent and magnitude of these risks from a community perspective needs to be evaluated in the WOE assessment. These risk questions are used to help focus subsequent steps of the reptile assessment, including the selection of measurement endpoints.

10.1.2 Measurement Endpoints

The measurement endpoints for reptiles selected for this BERA are as follows:

Measures of exposure:

- COPEC concentrations in surface water (total and dissolved)
- COPEC concentrations in sediment
- Health metrics of exposure – measured blood COPEC concentrations of selenium (and other bioaccumulators) in reptiles.

Measures of effects:

- Turtle community survey data
- Surface water and sediment quality screening.

10.2 Lines of Evidence

The measurement endpoints were evaluated as separate LOEs in the assessment of risks to reptiles. The LOEs used to evaluate reptiles include:

- Turtle community surveys
- Turtle blood and carapace tissue data
- Surface water and sediment chemistry data evaluation.

Each LOE is summarized below.

10.2.1 Reptile Community Surveys

A turtle community survey was conducted in 2009 and 2010 during the field work for the turtle tissue residue study (Appendix AB). Sampling locations are shown on Figure 2-28. Detailed data were collected for the aquatic turtle species captured within the Site. Turtle species diversity, relative abundance, and gross pathologies were assessed for multiple species.

10.2.1.1 Analysis of Exposure and Effects

A total of seven turtle species were observed during 2009 (Table 10-4). Two species were recorded at the Clinch River (upstream), four species at Clinch River (downstream), five species at the ash release area, and six species from the Emory River. During 2009, observations of the turtle species present at the reference location were not recorded.

Based on the number of turtles captured from each location, relative abundance of turtles was similar among the locations (Table 10-5). Note that during 2009, observations of the turtle species present at the reference location were not recorded. A total of seven turtle species were observed during 2010 (Table 10-6). Seven species were recorded at the Clinch River, Emory River, Tennessee River (downstream), and reference location.

Based on the number of turtles captured per trap-day, relative abundance of turtles was higher at the Clinch River and Tennessee River (downstream) than at the reference location in 2010 (Table 10-7). Relative abundance of turtles at the Emory River was similar to the reference location.

In summary, observations indicated that:

- Species richness was similar when comparing reference and ash-impacted locations
- Relative abundance indicated similar or higher numbers of turtles at both reference and ash-impacted locations
- Qualitative observations suggest that differences of anomalies and diseases were not apparent between reference and ash-impacted locations.

These field measures of turtle community health do not indicate a risk to turtles. However, the uncertainty in such qualitative evaluations is high due to a number of confounding natural factors that may have affected the results.

10.2.1.2 Risk Estimation

Observational results from the turtle community surveys suggest a lack of adverse impacts on reptile populations due to COPECs. Species richness and relative abundance of turtles do not appear to be affected based on the similarity of turtle communities from ash-impacted locations and reference locations. Based on a qualitative (visual) examination of more than 4,000 individual turtles, no significant anomalies or diseases (e.g., tumors, mutations, or cancers) were observed. These qualitative observations suggest that turtle health has not been impacted.

10.2.2 Turtle Tissue Concentrations of COPECs

Tissue data provide a direct measure of exposure of reptiles to COPECs from surface water, sediment, and food items at the Site. Turtles were sampled in order to determine their exposure to metals and to establish a comparative baseline for any future studies. Concentrations of metals and metalloids in blood may be used to predict concentrations also occurring in tissues more relevant to ecological assessment endpoints, such as eggs. Sampling locations (Figure 2-28) included locations in the Emory River, Clinch River to its confluence with the Tennessee River, Tennessee River below the confluence with the Clinch River, and Tennessee River above the confluence with the Clinch River (the 2010 Reference Area). Specimens were also collected from a reference location in nearby Knox County (the 2009 Reference Area). Turtles were collected during the summer and fall months of 2009 and 2010. Study objectives were to:

- Quantify tissue concentrations of COPECs in three different species of turtles for among location and between-year comparisons
- Assess risk to turtles by comparing concentrations measured at the study locations with literature-derived effects values, when available.

Appendix AB details the study objectives, methods, results (including tables and figures), and interpretation of the turtle tissue study. The study is summarized below to assess risk to reptiles.

10.2.2.1 Analysis of Exposure and Effects

Over 260 turtles representing three species were sampled for the tissue concentrations study. COPEC concentrations in tissues were statistically compared among impacted and reference locations during 2009 and 2010, a period before dredging activities were completed. Data collected post-dredging during 2011 were not statistically analyzed, but are presented and discussed qualitatively in the uncertainty section. The evaluation of the tissue concentration LOE for residual risk is conservative because it does not take into account the lower concentrations post-dredging.

COPECs including arsenic, barium, cobalt, mercury, selenium, and strontium were found to be statistically higher in blood from at least one of the three turtle species from the ash-impacted locations when compared with those from the reference locations. Three COPECs (i.e., arsenic, barium, and mercury) were statistically higher but detected in less than 50 percent of samples. Tissue concentrations of these metals lack a clearly distinguishable pattern for any of the three turtle species, which suggests that the observed statistical differences in arsenic, barium, and mercury are likely a reflection of inherent variability of these naturally-occurring elements. Limited metal and metalloid reference blood concentrations from turtles exist for comparison, so data from other reptiles were also reviewed.

Arsenic in blood was only detected in 15 percent of samples (Appendix AB, Table 2). Common musk/mud turtle samples (2010) had concentrations that were higher at CRM 2.5 (0.06 mg/kg ± 0.03 SD), and eastern spiny softshell turtle samples had concentrations that were higher in the Emory River (0.06 mg/kg ± 0.06 SD). Arsenic tissue concentrations associated with adverse effects in turtles were not found in the literature. However, the Emory River and Clinch River arsenic concentrations in turtle blood were within the range of blood concentrations reported for aquatic snakes from reference locations in the southeast. Concentrations of arsenic in blood averaged from 0.015 parts per million in water snakes from Tennessee (Burger et al. 2007) to 0.3 mg/kg and 0.4 mg/kg dry weight in aquatic snakes from South Carolina (Burger et al. 2006). In addition to inherent variability of naturally-occurring COPECs, higher arsenic concentrations could be the result of other types of anthropogenic input. For example, arsenic is common on farms due to organoarsenicals used for herbicidal applications or as feed additives for poultry and swine, which provides a pathway for arsenic to the nearby environment (NAS 1977 as cited in Eigenbrod et al. 2008).

Barium was only detected in 38 percent of blood samples (Appendix AB, Table 2), and only common musk/mud turtle samples (2010) had concentrations that were higher at CRM 2.5 (0.25 mg/kg ± 0.29 SD). Barium tissue concentrations associated with adverse effects in turtles were not found in the literature. Concentrations of barium in blood from aquatic snakes averaged from 0.3 mg/kg to 0.6 mg/kg dry weight (Burger et al. 2006). When compared to barium blood levels reported in aquatic snakes (Burger et al. 2006), common musk/mud turtle samples from the Clinch River do not appear to have significantly accumulated barium.

Mercury was only detected in 44 percent of blood samples across the three species of turtles (Appendix AB, Table 2), and only mercury concentrations in common musk/mud turtle samples were higher in the Emory River when compared to the reference location. Concentrations of mercury in blood of musk/mud turtles collected in 2010 from the Emory River had an average concentration of 0.03 mg/kg (± 0.03 SD) wet weight and a maximum concentration of 0.17 mg/kg wet weight. Mercury tissue concentrations associated with adverse effects in turtles were not found in the literature. However, the concentrations in this BERA are lower than reference location concentrations of 0.050 mg/kg reported for other aquatic turtles (Bergeron et al. 2007). Mercury can bioaccumulate in organisms, biomagnify up the food web, and is a ubiquitous metal (Bergeron et al. 2011). Studies have found mercury blood concentrations to be higher in aquatic turtles from impacted locations with concentrations up to 3.6 mg/kg (Bergeron et al. 2007). Concentrations of mercury in blood from the three species of turtles evaluated in this BERA are well below this level. COPECs with the highest frequency of detection that were statistically higher than the reference locations include cobalt, selenium, and strontium.

Cobalt was statistically higher than the reference location only in common musk/mud turtle samples at CRM 2.5 with an average concentration of 0.17 mg/kg. Cobalt tissue concentrations associated with adverse effects in turtles were not found in the literature. Concentrations of cobalt in blood from aquatic snakes averaged from 0.08 mg/kg to 0.1 mg/kg dry weight (Burger et al. 2006). Based on the limited literature

background concentrations, the higher concentrations of cobalt may be due to natural variation in the river system. The lack of significantly higher concentrations of cobalt in common snapping turtle samples and eastern spiny softshell turtle samples also suggest that natural variability may explain the statistical elevation of cobalt in common musk/mud turtle samples. Alternatively, cobalt in musk/mud turtle samples from the Clinch River might be associated with upstream legacy sources at the USDOE's Oak Ridge Reservation or may be due to the ash-impacted sediment.

Only selenium and strontium have been found at higher concentrations associated with ash-impacted areas (Sparling et al. 2010). Selenium was higher in common musk/mud turtle and common snapping turtle samples, while only common musk/mud turtle samples had higher levels of strontium when compared to the reference locations (Appendix AB; Figures 16, 17, and 22). More specifically for selenium, common musk/mud turtle samples from CRM 2.5 and CRM 4.0 and common snapping turtle samples from the 2009 ash-impacted areas (Emory River, Clinch River, and ash release area) were higher compared to the reference areas. Strontium in the 2010 CRM 2.5 common musk/mud turtle samples was higher compared to reference areas. Previous studies have illustrated the ability for both selenium and strontium to bioaccumulate in reptiles (Sparling et al. 2010).

In this BERA, common musk/mud turtles and common snapping turtles had maximum selenium concentrations of 1.6 mg/kg and 4.5 mg/kg, respectively. Selenium tissue concentrations associated with adverse effects in turtles were not found in the literature. Concentrations of selenium in blood from four aquatic turtle species (including common snapping turtle and common musk/mud turtle) in Virginia ranged between 0.260 and 0.339 mg/kg (Bergeron et al. 2007). This suggests that bioaccumulation of selenium may be occurring in these species. Although selenium concentrations were not statistically significant for eastern spiny softshell turtle, maximum concentrations of 1.4 mg/kg were reported at the Emory River and 1.5 mg/kg at the Clinch River.

Common musk/mud turtle was the only turtle species with significantly higher concentrations of strontium with an average concentration of 0.31 mg/kg at CRM 2.5. Strontium tissue concentrations associated with adverse effects in turtles were not found in the literature. Concentrations of strontium in blood from aquatic snakes in South Carolina averaged from 0.14 mg/kg to 0.2 mg/kg dry weight (Burger et al. 2006).

10.2.2.2 Risk Estimation

The COPECs with concentrations in turtle blood that were higher than reference locations were also compared to literature-derived blood reference concentrations to evaluate the potential that such concentrations might contribute to risks. If some concentrations are above the reference concentration, suggesting the potential for increased exposure, evidence of those concentrations coming from the ash and potentially adversely impacting the population was evaluated. In particular the correlation of tissue concentrations of the COPEC with the results of the turtle community survey was examined.

A total of 26 metal and metalloid COPECs were measured in blood samples from common musk/mud turtles, common snapping turtles, and eastern spiny softshell turtles. Out of the 26 metal and metalloid COPECs, ten COPECs were detected infrequently (less than 10 percent), and almost all COPECs had concentration ranges that reflected natural system variability. For those COPECs with high frequency of detections, only selenium and strontium had apparent trends based on comparisons between years and among locations. For strontium, 2010 blood concentrations in common musk/mud turtle samples from CRM 2.5 were higher than the Tennessee River reference. Recognizing that bioaccumulation can potentially take longer than 1 or 2 years, other concentration ranges from literature were used for comparisons as well. When selenium and strontium were compared to literature-derived reptile blood concentrations measured from un-impacted areas, only selenium fell within the "impacted" range and strontium was more comparable to the literature reference areas and natural variability. Therefore, literature-derived concentrations suggest that common musk/mud turtle samples from CRM 2.5 and CRM 4.0 and common snapping turtles from the 2009 ash-impacted areas have accumulated selenium. A consistent pattern of bioaccumulation of strontium is not evident, given that only one species of turtle had higher concentrations. The mean concentrations of selenium and strontium in turtle blood tissue are presented in Table 10-8 and Figures 10-1 and 10-2, respectively, to illustrate the variability of concentrations and the lack of a clear trend of bioaccumulation over time.

Characterization of COPEC exposures in turtles revealed that, in comparison to reference locations, common musk/mud turtles and common snapping turtles may have accumulated levels of ash-related COPECs at locations downstream of the ash release. Although ecological effects of metals on turtles are not well documented, the bioaccumulation of selenium and strontium in the tissues of these animals suggests the potential for increased exposure to this receptor group and higher trophic levels. Continued monitoring may elucidate trends or differences relative to the concentrations in the 2009 and 2010 monitoring.

10.2.3 Surface Water Chemistry

One LOE to evaluate potential risks to reptiles is the comparison of surface water data to acute and chronic water quality criteria. This method utilizes the existing surface water data from the near-location and reference reaches, and compares the data to regulatory benchmarks that were developed to protect aquatic resources. The ratios of the measured concentration to the regulatory benchmarks are used to calculate HQs for individual chemicals and individual reaches.

10.2.3.1 Analysis of Exposure and Effects

Surface water samples were collected from river locations, which represent the primary habitat and exposure areas for turtle indicator species. Available USEPA Region 4 water quality screening levels (USEPA 2001b) were utilized as generic effects levels. Reptile-specific TRVs are not available.

10.2.3.2 Risk Estimation

Surface water COPEC concentrations were below USEPA water quality screening levels (USEPA 2001b) (Table 10-9). The AWQC for aluminum was developed using toxicity data from acidic waters and, therefore, is not considered applicable to the habitats available within the Site. Additionally, aluminum and iron concentrations were higher in background samples. The evaluation of COPEC concentrations in surface water supports the conclusion that negligible risks exist for reptiles at the Site.

10.2.4 Sediment Chemistry

Another LOE to evaluate potential risks to reptiles is the comparison of seasonally-exposed and submerged sediment data to sediment quality screening levels. Reptile-specific screening standards were not available for comparison. The ratios of the measured concentration to the regulatory benchmarks are used to calculate HQs for individual chemicals and individual reaches.

10.2.4.1 Analysis of Exposure and Effects

Shallow sediment (seasonally-exposed) and submerged sediment data were considered to be representative of turtle indicator species exposure areas. Available USEPA Region 4 sediment quality screening levels (USEPA 1994b) were utilized as generic effects levels. Reptile-specific TRVs are not available.

10.2.4.2 Risk Estimation

Sediment HQs were calculated by comparing COPEC concentrations in submerged sediment and seasonally-exposed sediment samples to chronic sediment-based TRVs. These TRVs were based on sediment quality screening levels (USEPA 1994b). Reptile-specific literature-derived TRVs are not available.

For submerged sediment, mean sediment HQs exceeded USEPA sediment quality screening levels (USEPA 1994b) (Tables 10-10 through 10-12) for the following COPECs:

- Arsenic (CR_A and CR_B; ER_A, ER_B, and ER_C; TR_A and TR_B)
- Copper (CR_A and CR_B; ER_A, ER_B; TR_A, and TR_B)
- Lead (TR_A)
- Manganese (CR_A; TR_A, TR_B, and TR_Ref)
- Mercury (CR_A, CR_B, and CR_Ref; TR_A and TR_B)
- Nickel (CR_A and CR_B; ER_A and ER_B; TR_A and TR_B)
- Selenium (CR_A and CR_B; ER_A, ER_B, and ER_C; TR_A and TR_B)
- Zinc (TR_A).

With the exception of mercury and arsenic, mean sediment HQs were typically less than 2. The highest mean HQs for mercury were 4 at CR_A, 3 at CR_B, and 16 at CR_Ref. The highest mean HQs for arsenic were 3 at ER_A and 3 and ER_B.

For seasonally-exposed sediment, mean sediment HQs exceeded USEPA sediment quality screening levels (USEPA 1994b) (Tables 10-13 and 10-14) for the following COPECs:

- Arsenic (CR_A and CR_B; ER_A and ER_B)
- Copper (CR_A)
- Lead (CR_A)
- Mercury (CR_A and CR_B)
- Nickel (CR_A; ER_A and ER_B).

Mean sediment HQs were less than or equal to 3. The generic sediment screening-level values are designed for the protection of benthic invertebrate toxicity and likely overestimate the potential for risks to turtles. More importantly, the results of the amphibian sediment screening that did not indicate risks when using amphibian-specific sediment benchmarks. Direct toxicity to turtles from COPECs with low bioavailability in sediments is not likely based on the limited area of skin contact available for constituent absorption and the lack of significant concentrations of COPECs in surface water. Therefore, low risks from direct contact sediment exposure are likely at a spatial scale that would be relevant to mobile turtles. For example, estimates of the home range of turtles have been reported to be 7.3 to 8.9 hectares for snapping turtles (USEPA 1993), 1.18 hectares for common musk turtles (Ernst 1986), and 0.8 hectares for eastern spiny softshell turtles (Plummer et al. 1997).

10.3 Risk Characterization

Risks to reptile populations were evaluated based on the toxicological assessment of surface water, sediment, tissue chemistry, and biological survey data (Exhibit 10-1). The ecological relevance, potential risk, and confidence in the risk determination are summarized in Exhibit 10-1. Based on concordance of the LOEs, potential risks to reptiles are considered negligible with moderate confidence. Furthermore, when considering amphibians as a surrogate receptor for reptiles, risk conclusions for amphibians support the LOEs considered for reptiles.

Exhibit 10-1. Reptile Risk Characterization Attributes

Line of Evidence	Estimate of Magnitude of Adverse Effects	Confidence in Magnitude of Adverse Effects	Evidence of Causality	Relevance to Assessment Endpoint
Community Survey	Moderate	Moderate	High	Moderate
Tissue Concentrations	Moderate	Low	Moderate	Low
Surface Water Chemistry	Low	Low	Low	Low
Sediment Chemistry	Moderate	Low	Low	Low
Weight of Evidence	**Moderate**	**Low**	**--**	**--**

10.3.1 Weight of Evidence

An integration of multiple LOEs is shown in Exhibit 10-2. For each LOE, the evidence of causality, estimate of magnitude of adverse effects, confidence in magnitude of adverse effects, and ecological relevance is integrated and presented in Exhibit 10-1. Overall, evidence of causality is moderate with a low estimate of magnitude of adverse effects and low confidence in the magnitude of adverse effects.

Exhibit 10-2. Reptile Summary of Risks

Line of Evidence	Relative Weight	Potential Risk	Confidence in Risk Determination	COPECs/COECs
Community Survey	High	Negligible	Moderate	--
Tissue Concentrations	Moderate	Low	Moderate	--
Surface Water Chemistry	Low	Negligible	Low	--
Sediment Chemistry	Low	Low	Low	--
Weight of Evidence	**--**	**Low**	**Moderate**	**--**

Acronyms and Abbreviations:
COEC – constituent of ecological concern
COPEC – constituent of potential ecological concern

In summary, there are negligible risks to reptile populations at the Site based on the following LOEs:

- Turtle community surveys indicate that overall community structure is not altered and did not indicate any obvious anomalies or diseases.
- Surface water chemistry data indicate no toxicity to aquatic life or amphibians and less susceptible turtle indicator species are not likely to be affected.
- Sediment chemistry data suggest the potential for limited low-level toxicity to amphibians and less susceptible turtle indicator species are not likely to be affected.

- Chemical concentrations in turtle blood tissue indicated that concentrations of selenium and strontium were higher documenting increased exposure for this receptor. It is unknown whether the elevated levels of selenium and strontium may cause adverse effects since no adverse TRVs could be found in the literature for these two chemicals.

10.3.2 Uncertainties

This BERA for reptiles was conducted using conservative exposure parameters to ensure, to the extent possible, that potential risks were not underestimated. General sources of uncertainty in any BERA, including this BERA, include the selection of indicator species, the presence of these species at the Site and the level of activity at the Site, exposure estimation, and the lack of available toxicological benchmarks for several COPECs. Focused discussions of the key uncertainties for evaluating reptile risks are provided below.

10.3.2.1 Receptor Selection

Differences in feeding habits, habitat, behavior, and activity patterns of reptiles can result in varying exposure to COPECs. Aquatic turtles were assumed to be appropriate indicator species based on knowledge of the potential species that may use the Site and their exposure potential. There is a chance that these species may not represent the most sensitive species that occur at the Site.

Direct measures of effects to endangered or threatened species are not usually possible. Therefore, surrogate indicator species are selected to evaluate potential measures of effect to special status species. Of the threatened and/or endangered species that may occur in the vicinity of the Site, the turtle indicator species selected would be relevant and protective in terms of potential exposure. Based on the LOEs evaluated for turtle indicator species in this BERA, adverse effects would not be expected. It is unlikely that threatened and/or endangered species would have significantly higher exposures than the surrogate turtle species examined.

10.3.2.2 COPEC Exposure Estimation

Estimation of the COPEC exposure involves several uncertainties including the measurement of COPEC concentration and the concentrations estimated to be taken up from media. Based on the conservative assumptions used throughout the BERA, risk results were likely overestimated; not underestimated.

10.3.2.3 Percent Moisture

Percent moisture was analyzed in biological samples so that concentrations reported in wet weight could be converted to dry weight if necessary. Insufficient sample volume was available for some biological media. Some rejected percent moisture results could be reanalyzed, but the results were qualified due to holding

time exceedances. Default wet weight to dry weight conversion factors were used in the absence of sample-specific results. This may bias the results high or low.

10.3.2.4 Bioavailability of COPECs

Assessment of the bioavailability of COPECs to reptiles in this BERA had low to moderate confidence in the risk determination. For most species of wildlife and turtles, the bioavailability of COPECs in environmental media is generally lower than in the exposure media employed in toxicity tests.

10.3.2.5 Effects on Individual Organisms versus Populations

This evaluation is intended to assess risks to reptile populations. However, the toxicological evaluation of chemical concentrations generally relies on information that is most applicable to individual organisms. Although qualitative, site-specific population-level data for turtles indicates that there are not clear effects associated with COPECs. The relationship between individual and population-level effects is, thus, a significant source of uncertainty for reptiles. As with most field surveys, uncertainties do exist when evaluating species that have the ability to migrate, or in systems of complex COPEC mixtures. The evaluations of this study focused on comparisons among locations, between years, and to literature-derived concentrations in order to assess the relationship between the ash release and COPECs in turtles. Given the lack of literature-derived effects values for many of the COPECs found in ash, it is difficult to discern whether the turtle populations have potentially experienced adverse effects due to the ash release. Furthermore, the natural variability and historical contamination associated with the sample locations have potentially contributed to the differences found among locations.

In 2011, a field study of the effects of the ash spill on turtle populations and reproductive health was conducted by Virginia Polytechnic Institute and State University. A total of 5,077 turtles representing seven different species were captured from the Clinch, Emory, and Tennessee Rivers. No differences in turtle body size, species composition, or relative abundance among river locations were reported that could be attributable to the ash spill. Eggs were collected from gravid females and artificially incubated in the laboratory to assess maternal transfer of trace elements and any resultant effects on early development. Initial findings for turtle reproductive health suggest that reproductive health for stinkpot and common slider turtles were similar at both impacted and non-impacted sites. Although trace element concentration data in tissues were not yet available for a determination of bioaccumulation patterns at the site, this preliminary evaluation of the potential effects on turtles supports the conclusions of the BERA and does not change the overall interpretation of risk to turtles (Hopkins et al. 2012).

10.3.3 Summary and Conclusions

In summary, the risk evaluation for reptiles uses a combination of conservative and central tendency estimates for the exposure assessment and conservative estimates for the effects assessment. This approach had low to moderate confidence in the risk determination for reptiles. Even so, this BERA demonstrates that COPECs at the Site pose low to negligible risks to reptiles.

11. Interpretations and Recommendations

The preceding sections in this Baseline Ecological Risk Assessment (BERA) present in detail the lines of evidence (LOEs) for each receptor, concluding with the characterization of risk based on the weight of evidence (WOE) for each assessment endpoint. The assessment endpoints for the river system BERA include maintenance and reproduction of balanced communities or populations of ecological receptors that may be exposed and sensitive to the ash-related constituents of potential ecological concern (COPECs). Receptor groups evaluated in this BERA include: fish; benthic invertebrates; aquatic plants; and animals that feed in, along, and over the river system (i.e., birds, mammals, amphibians, and reptiles)

Risk characterization consists of risk estimation, risk description, and description of the strengths and limitations (uncertainties) of the available information. Together these provide information to help judge the ecological significance of risk estimates for the baseline condition – in the absence of remedial (activities or, in this case, further remedial activities) (USEPA1997a). To support the decision-making process, the risk assessment identifies the assessment endpoints for which at least some level of risk management may be warranted.

In risk management, the risk assessment results are integrated with other considerations to make and justify risk management decisions. Additional risk management considerations can include the implications of existing background levels of contamination, available technologies, tradeoffs between human and ecological concerns, costs of alternative actions, and remedy selection (USEPA 1997a). Risk management is Step 8 of the U.S. Environmental Protection Agency (USEPA) guidance for designing and conducting ecological risk assessments (USEPA 1997a; Figures 1-1 and 1-2). The Step 8 process for this site is addressed in detail in the Engineering Evaluation and Cost Analysis (EE/CA) for the River System scheduled for completion in June, 2012.

The general response actions evaluated in the EE/CA include monitored natural recovery, sediment capping, and sediment removal. That is, risk management includes a range of actions and alternatives from monitoring to active remediation. A risk management recommendation in this BERA should not be interpreted as an endorsement of a particular response action. Rather, the risk management recommendations in this document are simply suggested ways to further characterize risks, monitor ecological receptors' responses to selected risk management actions, and convey the assessors' best professional judgment as to the likelihood and magnitude of impairment of the assessment endpoint.

The next section provides a high-level summary of the risk determination for each assessment endpoint, identifies the endpoints and geographic areas (i.e., river reaches) for which risk management is recommended, and identifies the constituents of ecological concern (COECs) that drive the need for risk management.

11.1 Risk Summary and Conclusions

Multiple LOEs were available for most receptors. These LOEs were interpreted and integrated using a WOE approach. This was accomplished by first characterizing risk based on each individual LOE, and then characterizing risk based on the entire body of evidence. At both levels of evaluation, ecological risks were characterized by estimating the magnitude and likelihood of potential adverse effects. This included, either implicitly or explicitly, consideration of at least four factors:

- An estimate of severity (magnitude) of potential adverse effects
- Confidence (or uncertainty) in the estimated magnitude of potential effects
- Evidence of causality
- Relevance of the data to the assessment endpoints.

These factors were synthesized together as an estimated potential risk to the assessment endpoint and a level of confidence in that estimate. This integration of information was performed for each individual LOE and then for the entire body of evidence. These risk characterization attributes are summarized in the first exhibit of each receptor-specific section (e.g., Exhibit 3-1 for fish, Exhibit 4-1 for benthic invertebrates, etc.). The result is a holistic and comprehensive characterization of risks for each receptor group that was evaluated.

The WOE approach used in this BERA also entails explicitly assigning a relative weight for each LOE. This was based largely on the degree of association between the measure of potential adverse effects and the actual assessment endpoints to be protected at the Site (i.e., relevance to the assessment endpoint). The uncertainties inherent in the extrapolation from measurement endpoints to assessment endpoints also were considered inassignment of LOE weights. The summary of risks is presented in the second exhibit of each receptor (e.g., Exhibit 3-2 for fish, Exhibit 4-2 for benthic invertebrates, etc.).

The following sub-sections briefly summarize the risks by receptor, identifying those for which risk management appears to be warranted. The WOE risk characterization for each receptor is summarized in Table 11-1, along with the risk management recommendations.

11.1.1 Fish

The assessment endpoint for fish is the survival and maintenance of pelagic and benthic fish populations in the Emory, Clinch, and Tennessee Rivers. Risks to these fish communities were estimated using a variety of complementary types of data. Risks to fish were evaluated for each LOE, as summarized below:

- The fish community survey results suggest that there has been no ecologically significant impairment to the fish communities in these rivers. Similar patterns in population and community metrics were observed before and after the release. The fish assemblage data are representative of the habitat and observed patterns are associated with factors that extend reservoir-wide.
- The fish toxicity tests did not elicit significant effects on survival or growth of tested species. These observations are consistent with the fish community surveys; overt toxic effects were not observed in the population data.

The fish reproduction studies provided no conclusive evidence of ecologically significant adverse impacts on the reproductive condition of female largemouth bass or redear sunfish at the Site in 2009 and 2010 (the latest data set available when this BERA was performed). No adverse effects on reproductive condition of the three tested species were observed in 2009 or 2010, although an apparently localized delay in reproductive development was noted at one location (ERM 3.0) and only in the spring of 2009. This delay was attributed to habitat alterations and food web disruptions, and is not representative of current conditions at the Site. These observations are consistent with the fish community surveys, which found no ecologically significant impacts on these populations and communities.

- The fish health studies provided no conclusive evidence of ecologically significant health effects (i.e., obvious damage or injury) in sentinel fish species. Study results have been reported for fish collected in the spring and fall of both 2009 and 2010. These observations are consistent with the fish community surveys, which found no ecologically significant impacts on the fish community.
- The comparison of fish tissue data to Critical Body Residues (CBRs) indicates several exceedances. However, similar exceedances were frequently observed in fish collected from reference areas. These results indicate that although there has been some uptake of ash-related metal and metalloid COPECs, and levels sometimes exceed values associated in the literature with adverse effects, the highly conservative nature of the screening values is, at best, weak evidence suggesting that effects to the fish population may have occurred. In contrast, the site-specific fish population surveys and toxicity testing results indicate that significant risks to fish are neither likely nor expected.
- The comparison of surface water chemistry data to water quality benchmarks indicates few exceedances of water quality benchmarks. These results indicate limited potential for exposure to ash-related metal and metalloid COPECs in surface water, thus no significant risks to fish are expected.

11.1.1.1 Fish – Weight of Evidence

The WOE suggests that ash and ash-related COPECs pose a negligible risk to the fish communities in the Emory, Clinch, and Tennessee Rivers (Exhibit 3-2 and Exhibit 11-1). The available LOEs provide a moderate level of confidence in this risk determination, primarily due to the overall concordance of the LOEs.

Exhibit 11-1. Weight of Evidence Risk Characterization Summary by Receptor

Receptors	Potential Risk	Confidence in Risk Determination	Risk Management Recommended?	COECs
Fish	⊘	Moderate		
Benthic Invertebrates	◒ (ER) ○ (CR)	High	✓	As, Se, Ash
Aquatic Vegetation	○	Moderate		
Birds				
Piscivore - Heron	⊘	Moderate		
Piscivore - Osprey	⊘	Moderate		
Insectivore - Killdeer	○	Moderate	✓	As, Se
Omnivore - Mallard	○	Moderate		
Herbivore - Wood Duck	⊘	Moderate		
Aerial Insectivore - Tree Swallow	○	Moderate	✓	Se
Mammals				
Carnivore - Mink	○	Moderate		
Omnivore - Raccoon	○	Low		
Herbivore- Muskrat	○	High		
Aerial Insectivore - Gray Bat	○	Low		
Amphibians				
American Toad, Spring Peeper and Chorus Frogs	○	Moderate		
Reptiles				
Musk, Snapping, and Softshell Turtles	○	Moderate		

⊘ = risks are negligible; ○ = risks are low; ◒ = risks are moderate; ● = risks are high; ✓ = risk management is recommended.

COECs = Constituents of ecological concern.

ER = Emory River; CR = Clinch River.

The fish community surveys are given the greatest weight in the estimation of risk, because they are a direct measure of the assessment endpoint to be protected. These surveys provide robust evidence that the ash has not had measurable adverse effects on the fish communities at the Site.

The next most relevant LOEs are the fish toxicity tests and fish reproduction studies. The results of each of these studies are consistent with the fish community survey results. This suggests there are no ecologically important impacts to individuals that might manifest themselves at the population or community level. The fish health studies also suggest that no notable adverse effects are occurring at the organism or sub-organism level. Two types of data were compared with literature-derived effects values:

COPEC concentrations in fish tissues and COPEC concentrations in surface waters. Several COPECs exceeded CBRs, but CBRs are inherently conservative and uncertain effects values that provide only weak evidence of risk to the fish community. There were only a few limited exceedances of water quality effects values. Both of these types of literature-derived effects values are considered to have low relevance to the assessment endpoint. These LOEs are most appropriately used to help explain effects observed in community surveys, toxicity tests, and reproduction studies, rather than to predict effects. Given the lack of effects observed for the community surveys, toxicity tests, and reproductive competence, comparisons with literature-derived effects values contribute little to the overall assessment of risks to fish.

11.1.1.2 Fish – Uncertainties

Uncertainties were conservatively incorporated into the BERA approach, which is generally more likely to overestimate risks to fish than to underestimate risks. Major sources of uncertainty related to the characterization of potential risks to fish are described in Section 3, summarized in Table 11-1, and listed below:

- Risk Assessment Approach
- Site Data
- Analytical Methods
- Fish Community Survey Data
- Fish Health and Reproduction Studies
- Toxicity Test Procedures
- Toxicity Benchmarks

11.1.1.3 Fish – Conclusions

Risks to fish within the Emory, Clinch, and Tennessee Rivers are expected to be negligible, based on the overall WOE and the associated uncertainties. The LOEs that are most supportive of the low estimates of risk are:

- The lack of population-level impairment indicated by the fish community studies
- The lack of effects from fish toxicity testing.

No risk management actions are recommended for the protection of the fish communities at the Site (Exhibit 11-1).

11.1.2 Benthic Invertebrate Community

The assessment endpoint for benthic invertebrates is survival and maintenance of benthic invertebrate communities (BICs) in the Emory, Clinch, and Tennessee Rivers. Risks to benthic invertebrates were evaluated using multiple LOEs, each aimed at providing a unique measure of exposure and potential effect. Risks were evaluated for each LOE, as summarized below:

- BIC survey results provide no substantive evidence that the BIC composition has been negatively impacted. Rather, the BIC composition appears to be strongly correlated with substrate type rather than ash or ash-related COPECs.
- Toxicity test results indicate that sediments in the Emory River have a moderate potential for adverse effects on benthic invertebrates, largely associated with percent ash and arsenic. Toxicity observed in the Clinch River sediments was lower, with fewer locations exhibiting adverse effects and weaker correlations to ash and ash-related COPECs. For both rivers the majority of statistically significant effects were sub-lethal (i.e., reductions in growth, biomass, or emergence), indicating that effects are not likely to be immediate and severe, but may have some potential for population impacts over time.
- Snail and mayfly tissue concentrations suggest that arsenic associated with ash is accumulating in the tissues of benthic invertebrates at concentrations that could result in adverse effects on these organisms. Selenium also is accumulating to potentially toxic levels in these animals. Benthic invertebrate selenium concentrations are greater in the area of the release than in the reference areas, but do not demonstrate a strong relationship with selenium concentrations in sediment.
- Porewater concentrations suggest that ash-related COPECs (e.g., aluminum, arsenic, boron, and lead) pose a low risk to benthic invertebrates, with concentrations that exceed ambient water quality criteria (AWQC) being higher in the Emory River than in the Clinch River. However, the limited number of porewater samples makes it difficult to draw conclusions regarding correlations with sediment chemistry or tissue chemistry.

Sediment concentrations suggest that ash-related COPECs pose a low risk to benthic invertebrates, based on marginal exceedances of conservative benchmarks. This applies to arsenic, chromium, copper, and lead in both the Emory River sediments (reaches ER_A and ER_B) and Clinch River sediments (reaches CR_A and CR_B). This also applies to nickel in the Clinch River sediments (reaches CR_A and CR_B).

11.1.2.1 Benthic Invertebrates – Weight of Evidence

The WOE suggests that ash and ash-related COPECs pose a moderate risk to BICs in the Emory River and a low risk to BICs in the Clinch River (Exhibit 4-7 and Exhibit 11-1). Statistical analysis indicates exposures to ash and arsenic appear to be most closely associated with the observed effects. Selenium also shows

evidence of bioaccumulation in benthic invertebrates. The available LOEs provide a high level of confidence in these risk determinations and high confidence that risks are no greater than estimated.

The BIC surveys are given the greatest weight in the estimation of risk, because they are a direct measure of the assessment endpoint to be protected. Variability in the BIC appears to be explained primarily by the heterogeneity inherent in the river system, rather than the presence of ash and ash-related COPECs. The confidence in these conclusions is high, as is the relative weight given to this LOE.

Three LOEs, sediment toxicity tests, invertebrate tissue concentrations, and sediment porewater concentrations, have moderate relevance to the assessment endpoint. The results for these LOEs are consistent, but contrary to the BIC survey results. All three LOEs indicate the potential for at least low risks to benthic invertebrates with some degree of correlation with arsenic and/or ash. This is particularly true for the toxicity tests for the Emory River, and confidence in these test results is high. Tissue concentrations of arsenic and selenium were higher in the Emory River than at reference locations and were measured at potentially toxic concentrations as compared to the few applicable studies reported in the literature. Although bioaccumulation provides direct evidence of bioavailability, confidence in the ability of this LOE to predict toxicity *in-situ* is relatively low. Porewater concentrations of ash-related COPECs, including arsenic, exceeded literature-reported adverse effects levels, again most notably in the Emory River, but the benthic community results reflect no significant adverse effects. Although the potential for risk appears to be low, these exceedances are consistent with the observed bioaccumulation of arsenic, and with the correlation of arsenic and ash in sediments with adverse effects in the sediment toxicity tests. Confidence in the porewater LOE is moderate.

The least relevant LOE is the comparison of sediment concentrations with literature-derived effects values. These results suggest that ash-related COPECs pose a low risk to benthic invertebrates. There is high confidence that these conservative effects values do not underestimate risks; the observation that arsenic in the Emory River is among the COPECs exceeding benchmarks is consistent with the results of sediment toxicity tests, benthic invertebrate bioaccumulation measurements, and porewater measurements.

11.1.2.2 Benthic Invertebrates – Uncertainties

In general, uncertainties were conservatively incorporated into the assessment for benthic invertebrates. The major sources of uncertainty associated with the three LOEs for benthic invertebrates are described in Section 4, summarized in Table 11-1, and listed below:

- Representativeness of “Co-Located” Samples
- Physical Content/Conditions of the Sediment
- Changes in River Morphology
- Representativeness of Test Species

- Representativeness of Test Conditions
- Effects on Individuals Versus Populations
- Lack of Effects Data
- Critical Body Residues
- Analytical Methods

11.1.2.3 Benthic Invertebrates – Conclusions

BICs appear to be at moderate and low risk in the Emory River and Clinch Rivers, respectively. These risks appear to be primarily associated with ash and ash-related COPECs, especially arsenic, in the lower Emory River. This conclusion is primarily based on the observation of statistically significant reductions in growth and biomass in toxicity tests with Emory River sediments. This is augmented by finding ash-related COPECs concentrations in sediments and benthic tissues at concentrations potentially associated with adverse effects. The fact that the BIC assessment does not show clear and substantial impacts attributable to the ash release is compelling evidence that risks to the BIC are no more than moderate. However, it is possible that over time reductions in growth and biomass could result in measurable impacts on community structure. Risk management actions are recommended for protection of the BICs in the Emory River, but are not recommended for the Clinch or Tennessee River (Exhibits 11-1 and 11-2). The evaluation of alternative actions is addressed in detail in the River System EE/CA, which was submitted in June, 2012.

11.1.3 Aquatic Vegetation

The assessment endpoint for aquatic vegetation is the survival and production of the aquatic plant communities in the Emory, Clinch, and Tennessee Rivers. Risk estimates to aquatic plants are based on comparisons of metal and metalloid COPECs concentrations measured in biotic and abiotic media to available effect data. Risks were evaluated for each LOE, as summarized below:

- Periphyton tissue concentrations in downstream reaches were significantly different at $p < 0.1$ for arsenic, copper, and strontium compared to the reference location. Analysis of the significance of tissue concentrations to the community suggests that these constituents are not expected to pose unacceptable risk to the periphyton community.
- Shoreline and emergent vegetation tissue concentrations were not statistically higher in downstream reaches compared to the reference population. The only significant differences between study reaches were attributed to greater concentrations in reference reaches. Therefore, ash-related risks to shoreline and emergent vegetation are unlikely.
- The comparison of surface water chemistry data to water quality benchmarks indicates few exceedances: only maximum detected concentrations of aluminum, mercury, and lead exceeded conservative screening values. Comparison with additional literature-derived effects values identified no

exceedances for COPECs. These results suggest that risks associated with exposure of aquatic plants to constituents in surface waters are unlikely.

- Porewater concentrations that exceeded the aqueous screening values were also evaluated to support estimates of exposure and effects to aquatic plants. Only arsenic required further evaluation, which determined that risks were unlikely, based on differences in bioavailability *in-situ* relative to in synthetic laboratory water used for the toxicity study reported in the literature.
- Comparison of COPEC concentrations in submerged and seasonally-exposed sediments to soil screening values indicated negligible or low risks to aquatic plants. Arsenic, cobalt, and copper were higher in select downstream reaches compared to reference reaches. Maximum concentrations were only marginally above conservative screening values, and average sediment constituent concentrations were generally comparable with these screening values. Manganese concentrations in seasonally exposed and submerged sediments exceeded the soil screening value at the Site (including reference locations). Submerged sediments collected from the reference location in the Tennessee River had some of the highest manganese concentrations, which suggests that manganese in downstream reaches likely is due to sources unrelated to ash.

11.1.3.1 Aquatic Vegetation – Weight of Evidence

The WOE suggests that ash-related COPECs pose (at most) a low risk to aquatic vegetation in the Emory, Clinch, and Tennessee Rivers (Exhibit 5-2 and Table 11-1). Arsenic appears to be the ash-related COPEC most consistently associated with potential effects. Selenium does not appear to be bioaccumulating in downstream periphyton or other aquatic vegetation. The available LOEs provide a moderate level of confidence in the risk determination, and there is relatively high confidence that risks are not greater than estimated.

Tissue concentrations of COPECs in periphyton and aquatic vegetation were given the most weight in the estimation of risks because they require the least extrapolation from measured concentrations to relevant exposures. However, they are still considered to have only a moderate weight, with moderate relevance to the assessment endpoint. The potential for risks cannot be fully excluded based on these data, but risks are expected to be low, at most. The least weight is given to the LOEs based on ambient media concentrations of COPECs. Of these, concentrations in sediments and porewaters are expected to be the most closely associated with potential ash exposures. The low potential for risks indicated by these data is consistent with the estimated low risks based on tissue concentrations.

11.1.3.2 Aquatic Vegetation – Uncertainties

In general, uncertainties were conservatively incorporated into the assessment, which is more likely to overestimate risks to aquatic vegetation than to underestimate risks. Major sources of uncertainty related to

the characterization of potential risks to periphyton, emergent vegetation, and shoreline vegetation are described in Section 5, summarized in Table 11-1, and listed below:

- River System Complexity
- Inter-Species Extrapolations
- Lack of Toxicological Benchmarks
- Bioavailability of Constituents
- Limited Number of Samples
- Representativeness of Sampled Tissues
- COPEC Selection
- Analytical Methods

11.1.3.3 Aquatic Vegetation – Conclusions

Risks to aquatic vegetation at the Site are expected to be low, based on the combined overall WOE and the associated uncertainties. Concentrations of COPECs were evaluated for surface waters, sediment porewaters, seasonally-exposed sediments, and submerged sediments. These evaluations used a combination of conservative and central tendency estimates of exposure and conservative estimates of effects. As a result, risks to aquatic plants are likely over-estimated. No risk management actions are recommended for the protection of the aquatic plants at the Site (Exhibit 11-1).

11.1.4 Aquatic- and Riparian-Feeding Birds

The assessment endpoints for aquatic- and riparian-feeding birds are survival and maintenance of bird populations foraging at the Site. One or more receptor species were evaluated for each of the following trophic levels:

- Piscivores (heron and osprey)
- Insectivores (killdeer)
- Omnivores (mallard)
- Herbivores (wood duck).

Risks were estimated using dietary exposure models for the receptors. Multiple LOEs were used if available (i.e., for heron and osprey). Risks were evaluated for each LOE and receptor species, as summarized below:

- Egg health metrics (clutch size and egg biometrics) for heron did not significantly differ between impacted and reference locations. These results provide only low evidence of impacts on heron populations.
- Egg health metrics (clutch size and egg biometrics) for osprey were different among locations, but do not provide substantive evidence of impacts on osprey populations. Egg lengths and volumes were close to being significantly lower in the impacted locations. Osprey clutch size was significantly different among locations, but mean clutch size was higher on the Emory and Clinch Rivers than at the reference location or more downstream Tennessee River location.
- Tissue concentrations in heron eggs were higher than the reference location concentrations for three COPECs: selenium and copper were significantly higher at ash-impacted locations; mercury was almost significantly higher at ash-impacted locations. However, the differences were small and likely were not biologically meaningful to population viability. This interpretation is consistent with the observation that clutch sizes and egg health metrics were not significantly reduced at impacted locations.
- Tissue concentrations in osprey eggs were not significantly higher than the reference location concentrations for any COPECs. This is consistent with the apparent lack of impacts on clutch sizes and egg biometrics at impacted locations.
- Dietary exposure estimates for heron suggest that ash-related COPECs are causing at most low adverse effects on piscivorous birds that feed in the river system. Screening-level hazard quotient (HQs) were below 1 for COPECs except for total polychlorinated biphenyls (PCBs) (which is not ash-related), aluminum, iron, and radionuclides (radium-228, radium-226, and potassium-40). Refined-analysis HQs were below 1 for the COPECs. This is consistent with the findings based on heron clutch sizes, egg biometrics, and egg tissue concentrations.
- Dietary exposure estimates for osprey suggest that ash-related COPECs are causing at most low adverse effects on piscivorous birds that feed in the river system. Screening-level HQs were below 1 for COPECs except total PCBs (which is not ash related), aluminum, iron, and radionuclides (radium-228, radium-226, and potassium-40). Refined-analysis HQs were below 1 for the COPECs. This is consistent with the findings based on osprey clutch sizes, egg biometrics, and egg tissue concentrations.
- Dietary exposure estimates for killdeer indicate that risk associated with the ash may exist, but is low for insectivorous birds that feed in the riparian and aquatic zones in the Emory and Clinch Rivers. The refined-analysis HQs of no observed adverse effect level (NOAELs) and lowest observed adverse effect level (LOAELs) ranged from 1 to 5 for selenium and 1 to 3 for arsenic, respectively. Dietary exposure models were the only direct LOE available for estimating risks to killdeer.
- Dietary exposure estimates for mallards indicate that ash-related COPECs pose a low risk to omnivorous birds that feed in the river system. In the refined deterministic analysis, the selenium HQ exceeded 1 in reach ER_A, ranging from 0.8 to 2. However, the estimated probability of the selenium

HQ exceeding 1 is actually near zero (0 to 2 percent) for the mallard. Compared to the killdeer, vegetation in the mallard's diet dilutes the dose of selenium that comes from consumption of benthic invertebrates. Dietary exposure models were the only direct LOE available for estimating risks to mallards.

- Dietary exposure estimates for wood ducks suggest that ash-related COPECs pose a negligible risk to herbivorous birds that feed in the river system. Screening-level HQs were below 1 for COPECs except aluminum, iron, vanadium, and radionuclides (radium-228, radium-226, and potassium-40). Refined-analysis HQs were below 1 for all COPECs. Dietary exposure models were the only direct LOE available for estimating risks to wood ducks.

11.1.4.1 Aquatic- and Riparian-Feeding Birds – Weight of Evidence

The WOE suggests that ash-related COPECs pose a negligible risk to piscivorous and herbivorous birds that forage in and along the river system (Exhibit 6-2 and Exhibit 11-1). Only the insectivorous killdeer appears to be at some risk from exposure to ash-related COPECs. The killdeer dietary exposure model suggests that arsenic and selenium associated with the ash pose a low risk in the impacted reaches of the Emory and Clinch Rivers, and a negligible risk in the impacted reaches of the Tennessee River. Concentrations of these COPECs measured in benthic invertebrates from the Site are the primary drivers for the estimated risk to insectivorous riparian-feeding birds.

There is moderate confidence in the negligible risk determinations for heron, osprey, and wood duck, the three receptor species that do not feed predominantly on invertebrates. Risk to piscivores is based on multiple LOEs, including egg data that has moderate relevance to the assessment endpoints. Dietary exposure models are the only LOE for the other three receptor species. There is moderate confidence in the determination of low risk from selenium (probability $\leq$ 2 percent) for omnivores, as represented by the mallard. There is moderate confidence in the low risk determination for insectivores, though the probability of risk from arsenic and selenium is low (< 25 percent).

11.1.4.2 Aquatic- and Riparian-Feeding Birds – Uncertainties

In general, uncertainties were conservatively incorporated into the screening-level and deterministic dietary assessments for aquatic- and riparian-feeding birds. The probabilistic dietary assessment portrays more realistic potential exposures. The major sources of uncertainty associated with the two LOEs for aquatic- and riparian-feeding birds are described in Section 6, summarized in Table 11-1, and listed below:

Heron and Osprey Egg Tissue Concentrations

- Applicability of Literature-Derived Effects Levels for Eggs
- Sample Number and Timing of Egg Collections

- Representativeness of Exposure Periods
- Recently Collected Egg Data
- Effects of COPEC Mixtures
- Analytical Methods

Dietary Exposure Model

- Applicability of Literature-Derived Dietary TRVs
- Relevance of Dietary TRVs to Populations
- Dietary TRVs for Potential Risk Drivers
- Representativeness of EPCs
- Analytical Methods
- Representativeness of Food-Item EPCs
- Representativeness of Plant Tissues
- Distribution and Timing of Tissue Collections
- Estimation of Incidental Sediment Ingestion

11.1.4.3 Aquatic- and Riparian-Feeding Birds – Conclusions

There is a low risk to bird species that feed on benthic invertebrates in ash-impacted reaches of the Emory and Clinch Rivers (reaches A and B), based on the overall WOE for aquatic- and riparian-feeding birds. The dietary exposure model suggests that arsenic and selenium associated with the ash may pose low risks to insectivorous species (as represented by the killdeer). Actual risk could be negligible (acceptable), given that the invertebrate data upon which this conclusion is based were collected during or immediately following dredging. Risk to omnivores, as represented by the mallard, also are low but confidence is moderate given the low ($\leq$ 2 percent) probability for risk. Risks to the four other aquatic- and riparian-feeding bird species selected as representative receptors appear to be negligible. The WOE suggests that ash-related COPECs pose a low risk to omnivores and negligible risks to piscivorous and herbivorous birds that forage in and along the river system. Risk management actions are recommended only for the insectivorous species (e.g., killdeer) at the Site and only in the lower reaches (A and B) of the Emory and Clinch Rivers (Exhibits 11-1 and 11-2).

11.1.5 Aquatic- and Riparian-Feeding Mammals

The assessment endpoints for aquatic- and riparian-feeding mammals are survival and maintenance of populations that forage at the Site. One representative receptor species was evaluated for each of the following trophic levels: piscivores (mink), omnivores (raccoon), and herbivores (muskrat). Risks were

estimated using dietary exposure models for all these receptors. Multiple LOEs were available and used only for raccoon. Risks were evaluated for each LOE and receptor species, as summarized below:

- Raccoon tissue concentrations that were significantly higher than at the reference locations were arsenic, chromium, and selenium in hair; mercury and iron in muscle; and zinc and barium in the brain. The mean aluminum concentrations in hair were 3.5 times higher in samples from impacted locations than in samples from the reference location. This difference was not statistically significant and data for sediment and mayfly nymphs suggest that aluminum bioaccumulation at the Site is not likely due to exposure to ash. The measured concentrations of metal and metalloid COPECs are unlikely to lead to toxic effects in the raccoon population at the Site.
- Health metrics in exposed raccoons collected after the dredging in 2010 were generally within the normal range. No clinically significant differences were observed in prevalence of lesions, blood counts, or plasma biochemistry that could be attributed to the ash.
- Dietary exposure estimates for mink suggest that aluminum may pose, at most, a low and uncertain risk to piscivorous mammals based on measured concentrations in prey species (including gut contents) collected at the Site. Aluminum concentrations in gizzard shad generally varied seasonally in 2010 by as much as an order of magnitude, including in the reference reaches. Furthermore, aluminum in sediment and benthic organisms at the Site is poorly correlated with ash.
- Dietary exposure estimates for raccoon suggest that aluminum may pose, at most, a low and uncertain risk to omnivorous mammals based on measured concentrations in prey species collected at the Site. Aluminum concentrations in mayfly nymphs appeared to be somewhat associated with Clinch River sediments, but not associated with ash or the ash release. Aluminum is also poorly correlated with ash in sediment.
- Dietary exposure estimates for muskrat suggest that ash-related COPECs are causing at most low adverse effects on herbivorous mammals at the Site. The screening-level HQs of COPECs for the herbivorous muskrat were below 1 except for aluminum, iron, manganese, and radionuclides (radium-228, radium-226, potassium-40). In the refined analysis, the aluminum HQ still exceeded 1 (0.1 to 2) in a few reaches, but was not larger in impacted than reference reaches.

11.1.5.1 Aquatic- and Riparian-Feeding Mammals – Weight of Evidence

Based on the overall WOE for aquatic- and riparian-feeding mammals, a low and moderately confident risk to mammalian species that feed on benthic invertebrates exists from aluminum exposures in some ash-impacted locations of the Emory, Clinch, and possibly Tennessee Rivers (Exhibit 7-2 and Exhibit 11-1). Omnivorous and piscivorous species are most at risk, but the risk is low and could be negligible given that not all fish and invertebrate data, upon which this conclusion was based, were collected after dredging.

Aluminum and the other COPECs appear to pose a low risk to herbivores, with high confidence in the estimation of risk.

The raccoon dietary model suggests, *at most*, low and moderately confident risk from aluminum. The tissue and health metrics results from the raccoon study suggest low risk, with moderate confidence. Aluminum in hair was over three times higher than in reference animals; however, such levels were not statistically significant and may not be associated with adverse effects of aluminum bioaccumulation on reproduction. In combination, these two LOEs suggest low risk from aluminum to omnivorous mammals.

Risk levels from the dietary exposure model are similar for the raccoon and mink, which suggests the raccoon tissue data may apply to piscivores (such as mink) and that risk could be negligible for mink. However, much uncertainty exists around aluminum risk with few toxicity studies available on growth effects upon which the TRV was based. Reproductive studies of aluminum toxicity were not reported in the literature and the effects of COPECs on reproduction were not measured in raccoons at the Site. The WOE supports that risk is, *at most*, low for aquatic- and riparian-feeding mammals, with moderate confidence in the impacted reaches of the Emory, Clinch, and Tennessee Rivers.

11.1.5.2 Aquatic- and Riparian-Feeding Mammals – Uncertainties

In general, uncertainties were conservatively incorporated into the screening-level and deterministic dietary assessments for birds. The probabilistic dietary assessment portrays more realistic potential exposures. Major sources of uncertainty associated with the three LOEs for aquatic- and riparian-feeding mammals are described in Section 7, summarized in Table 11-1, and listed below:

Raccoon Tissue Study

- Relevance of Tissue Concentrations
- Limited Datasets for Reference Areas
- Unknown Concentrations below Detection Limits
- Effects of COPEC Mixtures
- Recently Collected Raccoon Data

Raccoon Health Metrics

- Representativeness of Reference Sites
- Extrapolation to Reproduction and Population Viability

Dietary Exposure Model

- Applicability of Literature-Derived Dietary TRVs
- Relevance of Dietary TRVs to Populations
- Relevance of Dietary TRVs to Populations
- Representativeness of EPCs
- Representativeness of Food-Item EPCs
- Analytical Methods
- Representativeness of Plant Tissues
- Estimation of Incidental Sediment Ingestion

11.1.5.3 Aquatic- and Riparian-Feeding Mammals – Conclusions

Based on the overall WOE for aquatic- and riparian-feeding mammals, there may be a low and moderately certain risk to mammalian species that feed on benthic invertebrates due to aluminum exposures in some reaches of the Emory, Clinch, and possibly Tennessee Rivers. The dietary exposure model suggests that aluminum poses, at most, a low risk to omnivorous and piscivorous species (as represented by mink and raccoon), but the risk could be negligible given that not all fish and invertebrate data upon which this conclusion was based were collected after dredging. Furthermore, aluminum concentrations in the prey items that are closely associated with sediments (i.e., mayfly nymphs and gizzard shad) are poorly correlated with the ash release and ash in sediments. Therefore, aluminum is not identified as a COEC based on the weight of evidence for mammals. The WOE also suggests that ash-related COPECs pose a low risk to herbivorous mammals (muskrat) that forage in and along the river system. No risk management actions are recommended for the protection of mammal populations at the Site, because aluminum risks are low and aluminum exposures are not clearly associated with the released ash (Table 11-1).

11.1.6 Aerial-Feeding Birds and Mammals

The assessment endpoints for aerial-feeding birds and mammals are survival and maintenance of populations or aerial-feeding insectivores that forage at the Site. This feeding guild consumes primarily emergent aquatic insects such as mayflies, true flies, dragonflies, and caddisflies. The representative receptor species are the tree swallow and gray bat. Risks were estimated using dietary exposure models for both receptors. Additional LOEs were available for tree swallows, which were surveyed and sampled at location and reference locations. Risks were evaluated for each LOE and receptor species, as summarized below:

- Reproductive measures for tree swallow show a small, if any, reduction in female fledglings produced per nesting female at impacted locations compared to reference locations. Reduction in female

fledglings integrates other measures of tree swallow reproductive success, such as clutch size, egg health metrics, hatching success, and nestling survival.

- Metal and metalloid COPECs concentrations that were higher in eggs, eggshells, or nestlings at impacted locations did not exceed known benchmarks for adverse effects. The exception is selenium, for which three of 64 eggs sampled in Emory River reaches had selenium concentrations exceeding the selenium TRV, which is the lowest adverse effect level for a wild bird species. However, that benchmark is for an herbivorous bird, the mallard, and aerial insectivores appear to be less sensitive to selenium. During the breeding season the red-winged blackbird has a diet similar to that of the tree swallow and may also be similar in sensitivity. Selenium concentrations in the tree swallow eggs from Emory River locations were well below the lowest adverse effect level for the red-winged blackbird. This suggests a low risk of selenium to the aerial-feeding bird receptor group.
- Dietary exposure estimates for tree swallow suggest that ash-related selenium may pose a low risk to aerial-feeding insectivorous birds that forage over the Emory and Clinch Rivers. Screening-level HQs were below 1 for COPECs except copper and selenium. Refined-analysis HQs for copper and selenium also exceeded 1, but the HQ for copper did not exceed the reference reach HQ. The HQs for selenium ranged from 2 (LOAEL) to 5 (NOAEL) in impacted reaches in the Emory and Clinch Rivers and exceeded reference reaches. The probabilistic dietary model indicates that tree swallow HQs for selenium have a relatively low probability of exceeding 1 (i.e., from 10 percent to 30 percent in the Emory and Clinch Rivers, and from 4 percent to 10 percent in the Tennessee River).
- Dietary exposure estimates for gray bats suggest that ash-related selenium may pose, at most, a low risk to bats that forage over the Emory and Clinch Rivers. Screening-level HQs were below 1 for COPECs except aluminum and selenium. Refined-analysis HQs for selenium also exceeded 1, but the effects value is based on reduced growth of laboratory animals and is very conservative. Incorporating population-level TRVs into the probabilistic dietary model indicates that the probability of selenium HQs exceeding 1 for gray bats is near 0 percent. This suggests that gray bats and other bat populations are probably not at risk of decline from selenium at the Site. Thus, risk is estimated as, at most, low and likely negligible.

11.1.6.1 Aerial-Feeding Birds and Mammals – Weight of Evidence

The WOE suggests that ash-related COPECs pose, at most, low risks to aerial-feeding insectivorous birds and mammals that forage over the river system (Exhibit 8-2 and Exhibit 11-1). Risk for tree swallow is based on site-specific reproductive measures that show a small, if any, reduction in female fledglings produced per nesting female and that the probability of selenium dietary HQs exceeding 1 are low (10 to 30 percent). No evidence exists for adverse effects of selenium on aerial-feeding mammal (bat) populations at the concentrations occurring in emergent insects at the Site using the dietary model (probability of exceeding 1 is near 0), though laboratory studies do show reduced growth in young at such concentrations. Literature-

derived field studies of effects of selenium on mammal populations show no clear adverse effects, but are limited and uncertain.

There is moderate confidence in the low risk determination for tree swallow and low confidence in the low risk determination for gray bats. The moderate level of confidence for risks to tree swallow is based on the results for site-specific reproductive measures in tree swallow and the results of the probabilistic dietary risk model. Unlike for birds, the field studies of effects of selenium on mammal populations are limited and uncertain. The probabilistic dietary risk model based on those field studies indicate the probability of exceeding 1 is near 0, but other laboratory studies show reduced growth in young at such concentrations. When evaluated holistically, the WOE suggests, with moderate confidence, that overall risk to aerial-feeding insectivores (birds and mammals) is, at most, low in the impacted reaches of the Emory, Clinch, and Tennessee Rivers.

11.1.6.2 Aerial-Feeding Birds and Mammals – Uncertainties

In general, uncertainties were conservatively incorporated into the screening-level and deterministic dietary assessments for tree swallows and gray bats. The probabilistic dietary assessment portrays more realistic potential exposures. The major sources of uncertainty associated with the LOEs for aerial-feeding birds and mammals are described in Section 8, summarized in Table 11-1, and listed below:

Tree Swallow Tissue Concentrations and Reproduction

Issues associated with uncertainties of the evaluation reproduction and tissue results reduced confidence in the assessment of risk from high to moderate confidence, as listed below:

- Relationships between Exposure and Reproductive Measures
- Effects of COPEC Mixtures
- Sample Number
- Representativeness of Exposure Periods
- Analytical Methods

Dietary Exposure Model

- Applicability of Literature-Derived Dietary TRVs
- Relevance of Dietary TRVs to Populations
- Representativeness of EPCs
- Representativeness of Food-Item EPCs
- Distribution and Timing of Tissue Collections
- Analytical Methods

11.1.6.3 Aerial-Feeding Birds and Mammals – Conclusions

At most, there is a low risk to species that feed on aerial insects from the ash-impacted locations of the Emory and Clinch Rivers (reaches A and B). This conclusion is based on the overall WOE for aerial-feeding birds and mammals. The dietary exposure model suggests that the probability of selenium dietary HQs exceeding 1 is low (10 to 30 percent) for tree swallow and negligible for gray bat (probability near 0). Literature-derived field studies of effects of selenium on mammal populations are limited and uncertain, but show no clear adverse effects. However, risk management actions are recommended for aerial-feeding insectivorous birds at the ash-impacted locations in the reaches A and B in both the Lower Emory and Clinch Rivers (Exhibits 11-1 and 11-2).

11.1.7 Amphibians

The assessment endpoint for amphibians is survival and maintenance of populations in the Emory, Clinch, and Tennessee Rivers. Risks to amphibians were evaluated using multiple LOEs each aimed at providing a unique measure of exposure and potential effect. Risks were evaluated for each LOE, as summarized below:

- Amphibian tissue concentrations suggest that COPECs pose a low risk for amphibians at the Site. Separate screening-level HQs were calculated for three species (i.e., American toad, spring peeper, and upland chorus frog) at each location. The mean HQ was < 1 for all species and locations, except for copper in spring peeper and chorus frog samples in 2009 at the West Embayment. However, the copper mean HQ in the refined-analysis was less than 1. The maximum HQ for zinc was slightly higher than 1 in chorus frog samples from the North Embayment in 2009 and in chorus frog samples from the West Embayment in 2010. However, zinc concentrations in amphibian tissue at all locations and years were not statistically higher than reference locations. Tissue benchmarks were not available for selenium and strontium. However, mean concentrations of selenium and strontium in amphibian whole body tissue were variable and lacked a clear trend of bioaccumulation over time.
- Surface water concentrations of COPECs support the conclusion that negligible risks exist for amphibians at the Site. COPEC concentrations were below water quality screening levels and amphibian-specific literature-derived TRVs. Aluminum and iron exceeded conservative screening values, but concentrations were also elevated in background samples.
- Sediment concentrations of COPECs indicated low risks at spatial scales that would be relevant to mobile amphibians. COPEC concentrations in seasonally-exposed sediment samples were compared with chronic sediment TRVs. Mean sediment HQs exceeded conservative screening levels for the following COPECs in the Emory and Clinch Rivers: arsenic (ER_B and ER_A; CR_A and CR_B), copper (CR_A), lead (CR_A), mercury (CR_A and CR_B), and nickel (ER_B and ER_A; CR_A). Mean sediment HQs were less than 3. Sediment HQs were below 1 for amphibian-specific literature-derived TRVs.

11.1.7.1 Amphibians – Weight of Evidence

The WOE suggests that ash and ash-related COPECs pose low risks to the amphibian populations at the Site (Exhibit 9-2 and Exhibit 11-1). Chemical concentrations in amphibian whole body tissues are substantially below levels associated with adverse effects. Surface water chemistry data indicate no toxicity to aquatic life including an evaluation of amphibian-specific surface water benchmarks. Sediment chemistry data indicate no toxicity to amphibians including an evaluation of amphibian-specific sediment toxicity benchmarks.

11.1.7.2 Amphibians – Uncertainties

The assessment for amphibians was conducted using conservative exposure parameters to ensure, to the maximum extent possible, that no potential risks were under-estimated. The major sources of uncertainty associated with the assessment for amphibians are described in Section 9, summarized in Table 11-1, and listed below:

- Representativeness of Selected Receptors
- COPEC Exposure Estimation
- Percent moisture
- Bioavailability of COPECs
- Effects on Individual Organisms versus Populations
- Representativeness of Exposures
- Recently Collected Data

11.1.7.3 Amphibians – Conclusions

Amphibian populations at the Site appear to be at low risk from exposure to ash-related COPECs. This conclusion is primarily based on the low concentrations for COPECs as compared to general and amphibian-specific effects values. The risk evaluation for amphibians uses a combination of conservative and central tendency estimates for the exposure assessment, and conservative estimates for the effects assessment. This approach very likely results in a significant over-estimation of potential risks to amphibians. Even so, this evaluation demonstrates that COPECs at the Site do not pose substantial risks to amphibians. No risk management actions are recommended for the protection of the amphibian populations at the Site (Exhibit 11-1).

11.1.8 Reptiles

The assessment endpoint for reptiles is survival and maintenance of populations in the Emory, Clinch, and Tennessee Rivers. Risks to reptiles were evaluated using multiple LOEs each aimed at providing a unique measure of exposure and potential effect. Risks were evaluated for each LOE, as summarized below:

- Turtle community survey results indicate a lack of adverse impacts on reptile populations related to COPEC exposures. Species richness and relative abundance of turtles do not appear to be affected based on the similarity of turtle communities from ash-impacted locations and reference locations. A qualitative (visual) examination of more than 4,000 individual turtles observed no significant anomalies or diseases (e.g., tumors, mutations, or cancers). These qualitative observations indicate that turtle health has not been impacted.
- Turtle tissue concentrations revealed that, in comparison to reference locations, common musk/mud turtles and common snapping turtles may have accumulated levels of ash-related COPECs (selenium and strontium) at locations downstream of the ash release. However, no adverse effects have been correlated with such concentrations in the literature. This is consistent with the turtle community surveys, which found no clear evidence of community impacts associated with increased exposure to ash and ash-related COPECs.
- Surface water concentrations of COPECs support the conclusion that negligible risks exist for reptiles at the Site. COPEC concentrations were below water quality screening levels, and reptile-specific TRVs are not available from the literature. Aluminum and iron exceeded conservative screening values, but concentrations also were elevated in background samples.
- Submerged sediment concentrations of COPECs indicated low risks from direct contact sediment exposure at spatial scales that would be relevant to mobile turtles. Mean sediment HQs exceeded conservative sediment screening levels for the following COPECs in the Emory, Clinch, and Tennessee Rivers: arsenic (the non-reference reaches); copper (the non-reference reaches except ER_C); lead (TR_A); manganese (CR_A; TR_Ref, TR_B, and TR_A); mercury (CR_Ref, CR_A, and CR_B; TR_B and TR_A); nickel (ER_B and ER_A; CR_A and CR_B; TR_B and TR_A); selenium (the non-reference reaches); and zinc (TR_A). Mean sediment HQs were less than 2, except for arsenic and mercury. The highest mean HQs for mercury were 16 at CR_Ref, 4 at CR_A, and 3 at CR_B. The highest mean HQs for arsenic were 3 at ER_B and 3 and ER_A. The generic sediment screening-level values are designed for the protection of benthic invertebrate toxicity and likely overestimate the potential for risks to turtles.
- Seasonally-exposed sediment concentrations of COPECs indicated low risks from direct contact sediment exposure at spatial scales that would be relevant to mobile turtles. Mean sediment HQs exceeded conservative sediment screening levels for the following COPECs in the Emory and Clinch Rivers: arsenic (ER_B and ER_A; CR_A and CR_B); copper (CR_A); lead (CR_A); mercury (CR_A and CR_B); and nickel (ER_B and ER_A; CR_A). Mean sediment HQs were less than 3. The generic

sediment screening-level values are designed for the protection of benthic invertebrate toxicity and likely overestimate the potential for risks to turtles.

11.1.8.1 Reptiles – Weight of Evidence

The WOE suggests that ash and ash-related COPECs pose a low risk to the reptile populations at the Site (Exhibit 10-2 and Exhibit 11-1). Turtle community surveys indicate that overall community structure is not altered and did not indicate any obvious anomalies or diseases. Chemical concentrations in turtle blood tissue indicate that concentrations of selenium and strontium were elevated; documenting increased exposure for this receptor, but no adverse effects have been reported in the literature and correlated with such concentrations. Surface water chemistry data indicate no toxicity to aquatic life or amphibians, and less susceptible turtle indicator species are not likely to be affected. Sediment chemistry data indicate no toxicity to amphibians, and less susceptible turtle indicator species are not likely to be affected.

11.1.8.2 Reptiles –Uncertainties

The assessment for reptiles was conducted using conservative exposure parameters to ensure, to the extent possible, that potential risks were not underestimated. The major sources of uncertainty associated with the assessment for reptiles are described in Section 10, summarized in Table 11-1, and listed below:

- Representativeness of Selected
- COPEC Exposure Estimation
- Bioavailability of COPECs
- Percent moisture
- Effects on Individual Organisms versus Populations
- Representativeness of Exposures
- Recently Collected Data

11.1.8.3 Reptiles – Conclusions

Reptile populations at the Site appear to be at low risk from exposure to ash-related COPECs. This conclusion is primarily based the results of the turtle community surveys, which indicate that overall community structure is not altered and did not identify any obvious anomalies or diseases. The risk evaluation for reptiles uses a combination of conservative and central tendency estimates for the exposure assessment and conservative estimates for the effects assessment. This approach very likely results in a significant overestimation of potential risks to reptiles. Even so, this evaluation demonstrates that COPECs at the Site do not pose substantial risks to reptiles. No risk management actions are recommended for the protection of the reptiles populations at the Site (Exhibit 11-1).

11.2 Risk Interpretations and Recommendations

The purpose of the risk assessment is to support the risk management decision-making process that occurs in Step 8 as part of the EE/CA (Figures 1-1 and 1-2). As noted above, risk management response actions for the residual ash in the river system range from monitored natural recovery to capping and/or ash removal. These actions include an obvious spatial component for each receptor identified as warranting risk management. Areas and recommendations for action for one receptor may overlap with other receptors. Therefore, the determinations of risk and recommendations for risk management are summarized by river reach in Exhibit 11-2. This depiction also helps portray patterns of risks and exposure pathways.

Exhibit 11-2. Receptor Risk Management Recommendations: Emory, Clinch, and Tennessee Rivers

Receptors	Emory River Reach C			Emory River Reach B			Emory River Reach A			Clinch River Reach B			Clinch River Reach A			Tennessee River Reach B			Tennessee River Reach A		
	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs	Risk	Risk Mgt?	COECs
Fish	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Benthic Invertebrates	◒	✓	As, Se, Ash	◒	✓	As, Se, Ash	◒	✓	As, Se, Ash	○			○			○			○		
Aquatic Vegetation	○			○			○			○			○			⊘			⊘		
Birds																					
Piscivore - Heron	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Piscivore - Osprey	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Insectivore - Killdeer	⊘			○	✓	Se	○	✓	As, Se	○	✓	Se	○	✓	Se	⊘			⊘		
Omnivore - Mallard	⊘			⊘			○			⊘			⊘			⊘			⊘		
Herbivore - Wood Duck	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Aerial Insectivore - Tree Swallow	⊘			○	✓	Se	○	✓	Se	○	✓	Se	○	✓	Se	⊘			⊘		
Mammals																					
Carnivore - Mink	⊘			○			⊘			○			○			⊘			○		
Omnivore - Raccoon	⊘			○			⊘			○			○			⊘			○		
Herbivore - Muskrat	⊘			⊘			⊘			⊘			⊘			⊘			⊘		
Aerial Insectivore - Gray Bat	⊘			○			○			○			○			⊘			⊘		
Amphibians	○			○			○			○			○			○			○		
Reptiles	○			○			○			○			○			○			○		

⊘ = risks are negligible; ○ = risks are low; ◒ = risks are moderate; ● = risks are high; ✓ = risk management is recommended.
COECs = Constituents of ecological concern; Mgt = Management

Ecological risks from exposure to ash or ash-related COECs are, at most, moderate in the Emory River and low or negligible elsewhere in the river system. None of the assessment endpoints are estimated to be at high potential risk in any of the river reaches. Ecological risks related to residual ash and ash-related COECs are primarily associated with:

- Direct exposures to ash, arsenic, and selenium in surface sediment
- Dietary exposures to arsenic and selenium via consumption of invertebrates that inhabit ash-impacted surface sediments.

These risks are most pronounced for the BIC, largely because they are the receptors most exposed to the residual ash. They have direct and prolonged contact with ash and ash-related COECs in surface sediments and the associated sediment pore waters. Arsenic and ash are most clearly associated with risks to benthic

invertebrates, based on the adverse effects observed in laboratory sediment toxicity tests with sensitive species of benthic invertebrates. Arsenic and selenium were bio-accumulated to higher levels in benthic invertebrates collected from the Site. Over time they may result in adverse effects on benthic invertebrates. Concentrations in mayfly nymphs are the primary evidence for an association between selenium and adverse effects on invertebrates, because selenium bioaccumulates and is typically difficult to measure in water and sediments at concentrations that are predictive of this exposure pathway. The WOE for risks to the BIC based on bioaccumulation and sediment toxicity test results is tempered by the results of community surveys that show no clear association between BIC structure and ash exposure. These surveys provide direct evidence that potential effects on individuals are not being manifested at the community level; however, this interpretation is complicated by spatial variations in habitat characteristics that also affect community structure.

Risks are also indicated for wildlife that consume primarily invertebrates that inhabit surface sediments for at least one life stage. These wildlife receptors consume invertebrates that are in the surface sediments habitat, or insects that developed in the surface sediments and then emerged as flying adults.

- Selenium is most clearly associated with risks to wildlife via this exposure pathway. Selenium is known to bioaccumulate from aquatic environments to concentrations in higher trophic levels that can be toxic to wildlife, with the largest increases in concentration occurring at lower trophic levels (e.g., from water to algae and invertebrates, rather than from fish to piscivores).
- Arsenic in benthic invertebrates is associated with risks to insectivorous birds that consume aquatic life stages of invertebrates, such as the killdeer. Trophic transfer of arsenic is not typically considered a major exposure pathway, which is consistent with the limited risks to wildlife identified in this assessment.

The spatial distribution of risks is consistent with an exposure gradient for residual ash, with the most pronounced ecological risks occurring in the Emory River, fewer risks indicated in the Clinch River, and only low risks indicated in the Tennessee River. This interpretation is based on the magnitudes of risks and the distribution of receptors for which risk management is recommended (Exhibits 11-1 and 11-2).

- Risks to BICs are estimated to be moderate in the three impacted reaches of the Emory River and low in the Clinch River and Tennessee River. Risks to insectivorous birds (killdeer and tree swallow) are low in the Emory River (ER-B and ER-A) and Clinch River, but negligible in the Tennessee River

- Risk management is recommended for three receptors in the Emory River (i.e., BIC, killdeer, tree swallow), two receptors in the Clinch River (i.e., killdeer and tree swallow), and no receptors in the Tennessee River.

Taken together, these patterns for estimated risks are consistent with the interpretation that risks related to residual ash and ash-related COECs are:

- Primarily attributable to arsenic and selenium associated with ash in sediment
- Diminished with increasing distance from the release area.

This information is considered along with other factors in the EE/CA in order to further support the risk management decision-making process.

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