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March 19, 2009

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4SD-FFB

Lisa Holstein
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SUBJ: Risk review comments for ecological aspects of the Final Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Former Choccolocco Corridor Ranges at Fort McClellan, Calhoun County, Alabama

Dear Ms. Holstein:

The Environmental Protection Agency (EPA) has reviewed the subject document and determined that the final report has addressed all EPA comments. The sample locations now include the maximum detected gamma-chlordane concentration in sediment. The text was revised to include a better description of sources and derivation of wildlife food-chain input parameters. Therefore, EPA approves the subject document. If you have any questions, please call me at (404) 562-8549.

Sincerely,

Doyle T. Brittain

Doyle T. Brittain
Senior Remedial Project Manager

cc: Michael Kelly, US Army AEC
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February 6, 2009

Mr. Scott Bolton
Site Manager
US Army Transition Force
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RE: ADEM Review and Concurrence: *Final Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Former Choccolocco Corridor Ranges;* received on September 5, 2008
Fort McClellan, Calhoun County, Alabama
Facility I.D. No. AL4 210 020 562

Dear Mr. Bolton:

The Alabama Department of Environmental Management (ADEM or the Department) has reviewed the Army's *Final Baseline Ecological Risk Assessment (BERA) Problem Formulation and Study Design (PF/SD) Report for Choccolocco Corridor Ranges.* On August 20, 2008, the Department submitted Evaluations to the Army's Responses to Comments on the Draft BERA PF/SD. General Comment 18 focused on the collection of riffle/run and pool macroinvertebrate samples. ADEM disagreed with the Army's response that these samples would be combined for analysis. In a December 11-12, 2008 meeting between the Army, USEPA, ADEM, U.S. Fish and Wildlife Services and their respective contractors, the Army revised their response and stated that the samples would be analyzed separately for each location per EPA guidance. ADEM concurs with this response and requests that the Army submit a slip page to revise the second paragraph on Page 9-8.

If you have any questions or concerns regarding this matter please contact Mrs. Brandi Little at 334-274-4226 or via email at blittle@adem.state.al.us.

Sincerely,

Stephen A. Cobb, Chief
Governmental Hazardous Waste Branch
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SAC/TPS/BCL/mal

cc: Mr. Doyle Brittain/EPA Region 4
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September 2, 2008

SHAW-MC-CK10-1142
Project No. 796887

Mr. Lee Coker
U.S. Army Corps of Engineers, Mobile District
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109 St. Joseph Street
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Contract: DACA21-96-D-0018, Task Order CK10
Fort McClellan, Alabama

Subject: Final Baseline Ecological Risk Assessment Problem Formulation and Study Design (PF/SD) for the Former Choccolocco Corridor Ranges

Dear Mr. Coker:

The subject document is enclosed for your records. The draft PF/SD was revised based on comments received from the U.S. Environmental Protection Agency (Region 4) and the Alabama Department of Environmental Management. Responses to regulator comments are included with this submittal. Upon completion of your review to ensure that the comments were adequately addressed, please provide a letter of concurrence on this submittal. As discussed during the August 12-14, 2008 BCT meeting, timely concurrence on this document is requested so that the fieldwork can commence in mid-September 2008.

At your request, I have distributed copies of this document as indicated below. If you have questions, or need further information, please contact me at (865) 694-7361.

Sincerely,

Stephen G. Moran

Stephen G. Moran, P.G.
Project Manager

Enclosure

Distribution: Lisa Holstein, Army TF (6 hardcopies; 2 CDs)
Brandi Little, ADEM (2 hardcopies; 2 CDs)
Doyle Brittain, EPA Region 4 (1 hardcopy; 2 CDs)
Richard H. Cumbie, Alabama Forestry Commission (1 hardcopy)

Final

Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Former Choccolocco Corridor Ranges

Fort McClellan
Calhoun County, Alabama

Prepared For:

U.S. Army Corps of Engineers
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Prepared By:

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Task Order CK10
Contract No. DACA21-96-D-0018
Shaw Project No. 796887

August 2008

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1.0 Introduction

A screening-level ecological risk assessment (SLERA) was conducted and presented in the report entitled *Remedial Investigation Report, Former Choccolocco Corridor Ranges, Parcels 94Q, 95Q, 96Q, 97Q, 131Q-X, 144Q-X, 145Q-X, 146Q, 147Q-X, and 148Q-X, Fort McClellan, Calhoun County, Alabama* (Shaw, 2005). The results of the SLERA indicated that several constituents in environmental media at the Choccolocco Corridor (CC) ranges have the potential to pose adverse ecological hazards. Therefore, a Baseline Ecological Risk Assessment (BERA) will be completed for the CC ranges in order to reduce the level of uncertainty inherent in the SLERA process and better define the potential for ecological risks. Per the U.S. Environmental Protection Agency's (EPA) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997), the first step in the BERA process is the "Problem Formulation." The Problem Formulation also constitutes "Step 3" of the EPA's eight-step process (EPA, 1997).

The Problem Formulation for the CC ranges utilizes the results of the SLERA presented in the remedial investigation report (Shaw, 2005) and on-site reconnaissance to identify the specific ecological values to be protected at the CC ranges, which are then used to establish assessment endpoints. The questions and issues that need to be addressed in the BERA are also defined in this Problem Formulation.

The Problem Formulation phase of the BERA addresses and expounds upon a number of issues described in the SLERA, including:

- Refinement of the constituents of potential ecological concern (COPEC) identified in the SLERA
- Description of the ecotoxicity of the COPECs
- Description of the fate and transport of the COPECs
- Description of the ecosystems potentially at risk
- Development and refinement of the site conceptual model
- Refinement of the complete exposure pathways
- Identification of the assessment endpoints.

Also included in this report is the BERA Study Design. The Study Design for the CC ranges utilizes the information gathered and presented in the Problem Formulation to establish measurement endpoints and to design studies that are appropriate to test the hypotheses concerning the assessment endpoints. Data quality objectives as well as statistical approaches are also presented in the BERA Study Design.

2.0 Identification of Constituents of Potential Ecological Concern

The SLERA for the CC ranges initially identified a number of COPECs in soil as well as the surface water and sediment in the ephemeral drainage features and perennial streams at the CC ranges. COPECs were initially identified by calculating screening-level hazard quotients, which were developed via a three-step process as follows:

- Comparison of maximum detected constituent concentrations to ecological screening values (ESV)
- Identification of essential macro-nutrients
- Comparison to naturally occurring background concentrations.

Constituents that were detected in environmental media at the CC ranges were evaluated against the ESVs by calculating a screening-level hazard quotient (HQ_{screen}) for each constituent in each environmental medium. ESVs are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000). A HQ_{screen} was calculated by dividing the maximum detected constituent concentration in each environmental medium by its corresponding ESV as follows:

$$HQ_{screen} = \frac{MDCC}{ESV}$$

where:

HQ_{screen} = screening-level hazard quotient
$MDCC$ = maximum detected constituent concentration
ESV = ecological screening value.

A calculated HQ_{screen} value of one or less than one indicated that the maximum detected constituent concentration (MDCC) was equal to or less than the chemical's conservative ESV, and was interpreted in the SLERA as a constituent that does not pose a potential for adverse ecological risk. Conversely, an HQ_{screen} value greater than one indicated that the MDCC was greater than the ESV and that the chemical might pose adverse ecological hazards to one or more receptors.

In order to better understand the potential risks posed by chemical constituents at the CC ranges, a mean hazard quotient was also calculated in the SLERA by comparing the arithmetic mean constituent concentration in each environmental medium to the corresponding ESV. The

calculated screening-level hazard quotients for surface soil, surface water, sediment, and groundwater at the CC ranges are presented in Tables 2-1 through 2-4.

The EPA recognizes several constituents in abiotic media that are necessary to maintain normal function in many organisms. These essential macronutrients are iron, magnesium, calcium, potassium, and sodium (EPA, 1989). Most organisms have mechanisms designed to regulate nutrient fluxes within their systems; therefore, these nutrients are generally only toxic at very high concentrations. Although iron is an essential nutrient and is regulated within many organisms, it may become increasingly bioavailable at lower soil pH values, thus increasing its potential to elicit adverse affects. Therefore, iron was not evaluated as an essential nutrient in the SLERA. Essential macronutrients were only considered COPECs if they were present in site samples at concentrations ten times the naturally occurring background concentration.

A study of the natural geochemical composition associated with Fort McClellan (FTMC) (Science Applications International Corporation [SAIC], 1998) determined the mean concentrations of 24 metals in surface soil, surface water, and sediment samples collected from presumably un-impacted areas. Per agreement with EPA Region 4, the background threshold value (BTV) for each metal was calculated as two times the mean background concentration for that metal. The BTV for each metal was used to represent the upper boundary of the range of natural background concentrations expected at FTMC, and was used as the basis for evaluating metal concentrations measured in site samples.

Thus, constituents were initially identified as COPECs within the SLERA if all of the following conditions were met:

- The maximum detected constituent concentration exceeded the ESV
- The maximum detected constituent concentration was 10-times the BTV if constituent is a macro-nutrient
- The maximum detected constituent concentration exceeded the BTV for inorganics.

If a constituent in a given environmental medium did not meet all of these conditions, then it was not considered a COPEC at the CC ranges and was not considered for further assessment. Again, identification of a constituent as a COPEC in the SLERA simply indicated that further assessment of that particular constituent in a given environmental medium was deemed

appropriate, and did not imply that a particular constituent posed a definite hazard to ecological receptors.

In order to focus future ecological risk assessment efforts on the constituents that are the most prevalent at the CC ranges and have the greatest potential to pose ecological hazard, additional lines of evidence were evaluated to refine the initial list of COPECs. These additional lines of evidence were scrutinized to aid in the decision process of whether or not to include a constituent as a COPEC in future ecological assessments at the CC ranges. Some of the additional lines of evidence used in the process of identifying COPECs include: 1) frequency of detection, 2) magnitude of the HQ_{screen} value, 3) statistical and geochemical background comparisons, 4) spatial distribution, 5) comparison to alternative ESVs; and 6) association of a chemical with known Army activities. These additional lines-of-evidence were used to further define the COPECs at the CC ranges and are discussed in the following sections.

2.1 COPECs in Surface Soil

The following constituents exceeded their respective ESVs, and naturally occurring levels in surface soil at the Former Choccolocco Corridor Ranges, and are not essential macro-nutrients (Table 2-1):

- antimony
- cadmium
- copper
- lead
- mercury
- nickel
- zinc
- MCPA
- MCPP
- 4,4'-DDD
- 4,4'-DDE
- 4,4'-DDT
- aldrin
- alpha-BHC
- beta-BHC
- gamma-BHC
- dieldrin
- endrin
- methoxychlor
- 2,4-DNT

Antimony was detected in 18 of 134 surface soil samples at concentrations that exceeded the ESV and naturally occurring levels. The HQ_{screen} value for antimony was calculated to be 14.4. Based on its known association with munitions and the frequency of detection, antimony was identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Cadmium was detected in two surface soil samples (HR-144Q-GP04 and HR-CCRI-GP22) out of 139 samples at elevated concentrations relative to ESVs and naturally occurring background levels. The HQ_{screen} value for cadmium was calculated to be 11.8. EPA (2005a) has developed Ecological Soil Screening Levels (Eco-SSL) for cadmium of 32 mg/kg (based on the protection

of terrestrial plants) and 140 mg/kg (based on the protection of soil invertebrates). All of the detected concentrations of cadmium are less than these Eco-SSLs. EPA (2005a) has also derived Eco-SSLs for the protection of avian and mammalian wildlife (0.77 mg/kg and 0.36 mg/kg, respectively). Although the maximum detected concentration of cadmium exceeds these values, the mean of the detected cadmium values is less than the avian Eco-SSL and only slightly greater than the mammalian Eco-SSL. Based on the relative infrequency of detection at elevated concentrations and the fact that the detected concentrations are mostly less than alternative ESVs, cadmium was not identified as a COPEC in soil at the Former Choccolocco Corridor Ranges.

Copper and lead were detected in numerous surface soil samples at elevated concentrations relative to ESVs and naturally occurring levels. The HQ_{screen} values for copper and lead were calculated to be 9.7 and 92.8, respectively. Based on the frequency of detection at elevated concentrations and the known association of these two metals with munitions, copper and lead were identified as COPECs in surface soil at the Former Choccolocco Corridor Ranges.

Mercury was detected in 16 surface soil samples out of 138 samples at concentrations that exceeded the ESV and naturally occurring levels. The calculated HQ_{screen} value for mercury was 11.8. Although mercury was detected relatively infrequently at elevated concentrations, there may be isolated locations with mercury contamination present in surface soil; therefore, mercury was identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Nickel was detected in 2 surface soil samples out of 126 samples at concentrations that exceeded the ESV and naturally occurring levels. The HQ_{screen} value for nickel was calculated to be 1.99. Based on the infrequency of detection at elevated concentrations and the low magnitude of the HQ_{screen} value, nickel was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Zinc was detected in 25 surface soil samples out of 139 samples at concentrations that exceeded the ESV and naturally occurring levels. The HQ_{screen} value for zinc was calculated to be 191. Based on the frequency of detection and the magnitude of the HQ_{screen} value, zinc was identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

MCPA and MCPP were detected in one surface soil sample each out of 53 samples at concentrations that exceeded their ESVs (MCPA at HR-CCRI-MW07 and MCPP at HR-97Q-GP04). The HQ_{screen} values for MCPA and MCPP were calculated to be 6.3 and 22, respectively. It should be noted that the analytical detection limits for MCPA and MCPP are very near or

above the ESVs for both MCPA and MCPP. Therefore, there is a certain amount of uncertainty regarding the frequency of detection above the ESVs. Because the detection limits are at or above the respective ESVs, it is possible for MCPA and/or MCPP to be present in a given soil sample at a concentration that exceeds the ESV, but is less than the detection limit. While acknowledging this uncertainty in the analytical method, it is also important to keep in mind that if MCPA or MCPP are present at concentrations that exceed their respective ESVs but are less than their detection limits, then they would be present at relatively low levels. MCPA and MCPP are relatively non-persistent in soil with half-lives ranging from one to two months. Based on their infrequency of detection, the magnitude of the difference between their detection limits and their ESVs, and their relatively short half-lives in relation to the historical nature of the use of these ranges, MCPA and MCPP were not identified as COPECs in surface soil at the Former Choccolocco Corridor Ranges.

4,4'-DDT and its metabolites (4,4'-DDD and 4,4'-DDE) were detected in 11, 2, and 3 surface soil samples out of 53 samples at concentrations that exceeded their respective ESVs. The HQ_{screen} values for 4,4'-DDT, 4,4'-DDD, and 4,4'-DDE were calculated as 23, 15, and 1.8, respectively. Although 4,4'-DDT and its metabolites were somewhat sporadically detected in surface soil, there may be isolated locations with surface soil contaminated with 4,4'-DDT and its metabolites. Therefore, based on their persistence in the environment and the magnitude of their HQ_{screen} values, 4,4'-DDT, 4,4'-DDD, and 4,4'-DDE were identified as COPECs in surface soil at the Former Choccolocco Corridor Ranges.

Aldrin was detected in two surface soil samples out of 53 samples at concentrations that exceeded the ESV (HR-97Q-GP04 and HR-CCRI-GP02). The HQ_{screen} value for aldrin was calculated to be 4.8. Based on the infrequency of detection at elevated concentrations and the relatively low HQ_{screen} value, aldrin was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Alpha-benzene hexachloride (BHC) was detected in two surface soil samples out of 53 samples at concentrations that exceeded the ESV (HR-145Q-DEP01 and HR-CCRI-DEP06). The HQ_{screen} value for alpha-BHC was calculated to be 3.5. Based on the infrequency of detection at elevated concentrations and the relatively low HQ_{screen} value, alpha-BHC was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Beta-BHC was detected in six surface soil samples out of 53 samples at concentrations that exceeded the ESV. The HQ_{screen} value for beta-BHC was calculated to be 8. If the USEPA Region 5 Ecological Screening Level (ESL) of 0.00398 mg/kg is considered as an alternative

screening value, four samples exhibit beta-BHC concentrations that exceed the alternative screening value. The arithmetic mean of the detected concentrations of beta-BHC (0.0036 mg/kg) is less than the alternative screening value. Based on the relative infrequency of detection, the relatively low concentrations of the detected beta-BHC, and the fact that the mean of the detected concentrations is less than the alternative screening value, beta-BHC was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Gamma-BHC was detected in five surface soil samples out of 53 samples at concentrations that exceeded the ESV. The HQ_{screen} value for gamma-BHC was calculated to be 120. If the USEPA Region 5 ESL (0.005 mg/kg) is used as an alternative screening value, only one soil sample exhibits a gamma-chlordane concentration that exceeds the alternative screening value. The HQ_{screen} using the alternative screening value is calculated to be 1.2. The arithmetic mean of the detected concentrations of gamma-BHC (0.000344 mg/kg) is less than the alternative screening value. Based on the relative infrequency of detection at elevated concentrations, the low HQ_{screen} value if the alternative screening value is used, and the fact that the mean of the detected concentrations is less than the alternative screening value, gamma-BHC was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Dieldrin was detected in six surface soil samples out of 53 samples at concentrations that exceeded the ESV. The HQ_{screen} value for dieldrin was calculated to be 16. EPA (2007) has developed Ecological Soil Screening Levels (Eco-SSL) for dieldrin of 0.022 mg/kg (based on the protection of avian wildlife) and 0.0049 mg/kg (based on the protection of mammalian wildlife). All of the detected concentrations of dieldrin are less than the Eco-SSL for the protection of avian wildlife. Two soil samples exhibited dieldrin concentrations greater than the Eco-SSL for the protection of mammalian wildlife. The HQ_{screen} value that was calculated using the Eco-SSL for the protection of mammalian wildlife is 1.6. The arithmetic mean of the detected dieldrin concentrations is less than the avian and mammalian Eco-SSLs. Based on the infrequency of detection, the low HQ_{screen} values calculated using the Eco-SSLs, and the fact that the arithmetic mean concentration is less than the Eco-SSLs, dieldrin was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

Endrin was detected in thirteen surface soil samples out of 53 samples at concentrations that exceeded the ESV. The HQ_{screen} value for endrin was calculated to be 34. Endrin was identified as a COPEC in surface soil due to the frequency of detection and the magnitude of the HQ_{screen} value.

Methoxychlor was detected in three surface soil samples out of 53 samples at concentrations that exceeded the ESV. The HQ_{screen} value for methoxychlor was calculated to be 1.3. Based on the relative infrequency of detection at elevated concentrations and the relatively low HQ_{screen} value, methoxychlor was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

2,4-Dinitrotoluene (2,4-DNT) was detected in one surface soil sample out of 53 samples at a concentration that exceeded the ESV. The HQ_{screen} value for 2,4-DNT was calculated to be 1.2. Based on the infrequency of detection at elevated concentrations and the relatively low HQ_{screen} value, 2,4-DNT was not identified as a COPEC in surface soil at the Former Choccolocco Corridor Ranges.

2.2 COPECs in Surface Water

Bis(2-ethylhexyl)phthalate (BEHP)was detected in two surface water samples (HR-CCRI-SW05 and HR-CCRI-SW09) out of 20 samples at concentrations that exceeded the ESV (Table 2-2). The HQ_{screen} value for BEHP was calculated to be 333. Although the magnitude of the HQ_{screen} value was relatively high, the fact that BEHP was only detected in two samples does not suggest that a source of BEHP is present at the CC ranges. Furthermore, BEHP is commonly used as a plasticizer in a wide variety of materials, and is also a common laboratory contaminant (National Library of Medicine [NLM], 1996; Agency for Toxic Substances and Disease Registry [ATSDR], 1993). Because of their many uses, phthalates are widespread in the environment and have been identified at low levels in the air, water, and soil. Because the screening value of 0.004 mg/L is an order of magnitude below the reporting limit typically achieved by laboratory analytical techniques (0.01 mg/L in this investigation), even very low detections of BEHP would result in HQ values greater than 1. Because the analytical detection limit for bis(2-ethylhexyl)phthalate is near or above the ESV, there is a certain amount of uncertainty regarding the frequency of detection of bis(2-ethylhexyl)phthalate above the ESV. It is possible for bis(2-ethylhexyl)phthalate to be present in a given surface water sample at a concentration that exceeds the ESV, but is less than the detection limit. However, due to the very low water solubility of BEHP, it is unlikely to be present in significant concentrations in surface water. BEHP is not a bioaccumulative chemical, and is unlikely to pose a hazard to populations of ecological receptors even if the detection in the stream is accurate. Therefore, although it was retained as a COPEC in the SLERA, due to the factors presented above, BEHP is not carried forward as a COPEC in surface water at the Former Choccolocco Corridor Ranges.

Lead was detected in three surface water samples out of 18 total samples at concentrations that exceeded the ESV. All surface water samples were analyzed for total recoverable metals;

therefore, the analytical results for lead are reported as total lead. Because the bioavailable portion of lead (and other metals) in surface water is more accurately represented by the dissolved fraction, the analytical results likely over-estimate the lead in surface water that is bioavailable. The HQ_{screen} value was calculated to be 8.26. If the National Recommended Water Quality Criterion (USEPA, 2002) is considered as the alternative ESV, then only one sample exhibits lead at a concentration that exceeds the alternative ESV. Statistical analysis (slippage test and Wilcoxon Rank Sum test) of the data indicate that the detected concentrations of lead in surface water are consistent with naturally occurring levels of lead in surface water. Based on these lines of evidence, lead was not identified as a COPEC in surface water at the Former Choccolocco Corridor Ranges in the SLERA. However, because lead was identified as a COPEC in soil at the Former Choccolocco Corridor Ranges and is a major component of small arms munitions, lead will be retained as a COPEC in surface water solely for the purpose of providing a means to develop a remedial action objective for lead in surface water. No site-specific testing of surface water will be required at the Former Choccolocco Corridor Ranges .

2.3 COPECs in Sediment

Several constituents were detected in sediment samples from the Former Choccolocco Corridor Ranges at concentrations that exceeded their respective ESVs, and naturally occurring levels. These constituents are the following (Table 2-3):

- lead
- 4,4'-DDT
- beta-BHC
- endrin
- gamma-chlordane
- heptachlor
- methoxychlor
- HMX
- acetone

Lead was detected in seven sediment samples out of 23 samples at concentrations that exceeded the ESV. The HQ_{screen} value for lead was calculated to be 13.05. Although lead was not selected as a COPEC in sediment in the SLERA because geochemical analysis indicated that the detected concentrations of lead in sediment were consistent with naturally occurring levels (Shaw, 2005), the frequency of detection at elevated concentrations and the fact that lead is a major component of munitions lead to the conclusion that lead should be retained as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

4,4'-DDT and beta-BHC were each detected in one sediment sample out of 21 samples at concentrations that exceeded their respective ESVs. The HQ_{screen} values for 4,4'-DDT and beta-BHC were calculated to be 1.6 and 2.2, respectively. Based on the infrequency of detection at

elevated concentrations and the relatively low HQ_{screen} values, 4,4'-DDT and beta-BHC were not identified as COPECs in sediment at the Former Choccolocco Corridor Ranges.

Endrin was detected in two sediment samples out of 21 samples at concentrations that exceeded the ESV. The HQ_{screen} value for endrin was calculated to be 1.4. Based on the infrequency of detection at elevated concentrations and the low magnitude of the HQ_{screen} value, endrin was not identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

Gamma-chlordane was detected in four sediment samples out of 21 samples at concentrations that exceeded the ESV. The HQ_{screen} value for gamma-chlordane was calculated to be 8.8. Due to the frequency of detection and the magnitude of the HQ_{screen} value, gamma-chlordane was identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

Heptachlor was detected in three sediment samples out of 21 samples at concentrations that exceeded the ESV. The HQ_{screen} value for heptachlor was calculated to be 4.3. Jones, et al. (1997) have derived an alternative screening value for heptachlor of 0.068 mg/kg based on equilibrium partitioning. If this alternative screening value is considered, all of the detected concentrations of heptachlor are less than the alternative screening value. Based on the infrequency of detection, the relatively low HQ_{screen} value, and the fact that none of the detected concentrations were greater than the alternative screening value, heptachlor was not identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

Methoxychlor was detected in two sediment samples out of 21 samples at concentrations that exceeded the ESV. The HQ_{screen} value for methoxychlor was calculated to be 1.5. It should be noted that the analytical detection limit for methoxychlor is very near or above the ESV. Therefore, there is a certain amount of uncertainty regarding the frequency of detection above the ESV. Because the detection limit is at or above the ESV, it is possible for methoxychlor to be present in a given sediment sample at a concentration that exceeds the ESV, but is less than the detection limit. While acknowledging this uncertainty in the analytical method, it is also important to keep in mind that if methoxychlor is present at concentrations that exceed the ESV but are less than the detection limit, then it would be present at relatively low levels. Based on the relative infrequency of detection at elevated concentrations, the low magnitude of the difference between the detection limit and the ESV, and the low magnitude of the HQ_{screen} value, methoxychlor was not identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

HMX was detected in one sediment sample out of 23 samples at a concentration that exceeded the ESV. The HQ_{screen} value for HMX was calculated to be 121. It should be noted that the detection limit for HMX exceeds the ESV. Therefore, it is possible for HMX to be present is a sediment sample at a concentration that exceeds the ESV but is reported as non-detect. This uncertainty in the analytical method for HMX in sediment is expressed in the frequency of detection; therefore, there is an unknown amount of uncertainty in the frequency of detection of HMX in sediment.

Research by Steevens et al. (2002) suggests that exposure of midge and amphipods to 400 mg/kg HMX in sediment did not have adverse effects on survival or growth. In fact, growth was significantly increased in both the amphipods and midges when exposed to 400 mg/kg HMX in sediment. The single detected concentration of HMX in sediment was significantly less than 400 mg/kg. Additionally, HMX was not detected in any other environmental media associated with the CC ranges. Due to the low frequency of detection, the fact that HMX was not detected in any other environmental media at the CC ranges, and alternative screening values suggest the detected concentration would likely not elicit adverse effects, HMX was not identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

Acetone was detected in one sediment sample out of 16 samples at a concentration that exceeded the ESV. The HQ_{screen} value for acetone was calculated to be 1.6. Based on the infrequency of detection and the low magnitude of the HQ_{screen} value, acetone was not identified as a COPEC in sediment at the Former Choccolocco Corridor Ranges.

2.4 COPECs in Groundwater

The rationale for assessing groundwater at the CC ranges using surface water ESVs was to determine the potential for impacts to aquatic organisms from groundwater intrusion to the ephemeral drainage features and perennial stream at the CC ranges. It is important to note that surface water ESVs are not intended to be applied to groundwater data; therefore, a great deal of uncertainty is introduced into this assessment by doing so. Additionally, if groundwater intrusion into surface water bodies is occurring at the CC ranges, the surface water samples would incorporate the groundwater that may be introduced to the surface water bodies. As such, screening the surface water samples using the surface water ESVs includes any introduction of groundwater constituents that may be taking place. However, in order to comply with a request from EPA Region 4, a comparison of groundwater data to surface water ESVs was completed.

Heptachlor was detected in one groundwater sample at a concentration that exceeded the surface water ESV, and was the only chemical initially selected as a COPEC in groundwater (Table 2-4).

However, exposure pathways involving groundwater at the Former Choccolocco Corridor Ranges are not complete. As stated previously, depth to groundwater at the Former Choccolocco Corridor Ranges is between 10 and 65 feet bgs. As such, groundwater discharge to surface water bodies (e.g. the ephemeral drainage features) is unlikely. Without groundwater discharge to a surface water body, ecological exposures are not complete. Therefore, no COPECs were identified in groundwater at the Former Choccolocco Corridor Ranges.

2.5 Summary of COPECs

In order to focus on the constituents that are most prevalent at the CC ranges and that have the greatest potential to pose adverse ecological effects to local ecological communities and populations, the initial list of COPECs was scrutinized using additional lines of evidence. These additional lines of evidence included frequency of detection, magnitude of the HQ_{screen} value, comparison to alternative screening values, association with Army activities, bioaccumulation and toxicity potential. Based on these additional lines of evidence, the COPECs that have been identified at the CC ranges are summarized below and presented in Table 2-5:

- ***Surface Soil –*** antimony, copper, lead, mercury, zinc, 4,4'-DDD, 4,4'DDE, 4,4'-DDT, and endrin;
- ***Surface Water –*** lead;
- ***Sediment –*** lead and gamma-chlordane; and
- ***Groundwater –*** none.

3.0 Ecotoxicity

The ecotoxicological properties of the COPECs identified at the CC ranges dictate which receptors have the greatest potential ecological risk and the pathways by which those receptors have the greatest potential for exposure. Factors such as the propensity to bioaccumulate or biomagnify, as well as their acute and/or chronic toxicity to immature or adult receptors are important factors in the consideration of a constituent's ecotoxicity and also in the development of assessment and measurement endpoints.

In order for a constituent to exhibit toxicity or to bioaccumulate, it first must be bioavailable. In general, there are three microbial processes affecting the bioavailability of metals (Connell and Miller, 1984). The first is biodegradation of organic matter into lower molecular weight compounds, which are more capable of complexing metal ions than the higher molecular weight organic molecules. The second is alterations to physico-chemical properties of metals by microbial metabolic activities (i.e., oxidation-reduction potential and pH conditions). Finally, the process of bacterial methylation, specifically of lead, may greatly enhance the bioavailability of certain inorganic compounds.

The actual uptake of bioavailable metals by terrestrial and aquatic organisms is through three main routes: 1) uptake across respiratory surfaces (lungs or gills), 2) adsorption from soil, sediment or water onto body surfaces, and 3) ingestion of food, water or incidental particles. Given the state of science relative to bio-uptake dynamics, the ingestion route is the most quantifiable uptake route at this time. Metal uptake from dietary sources, in comparison to direct adsorption, is also considered the primary uptake route in small terrestrial and aquatic receptors.

Although ecological receptors can readily absorb metals from food/water ingestion, their ability to regulate elevated concentrations of metals dictates their tolerance and is a critical factor in survival. Once the upper limit, or threshold, of metal sequestration and excretion is reached, sub-lethal effects such as inhibited reproduction and growth potentials may be exhibited, followed by lethality. Temporary metal storage is generally by binding to proteins, such as metallotheioneins, polysaccharides, and amino acids (Connell and Miller, 1984). Storage within liver and kidney tissues as well as bone, feathers, and fur also provide a useful means for sequestering metals such as lead.

Considerable inter- and intra-species differences exist in bioaccumulation potential of individual metals. In addition, according to Phillips (1980), different chemical forms of any one metal may

be absorbed and excreted at widely differing rates. Many studies support the premise that inorganic metals do not have a high propensity to biomagnify up through food chains.

The sections that follow highlight key toxicological properties of the COPECs that have been identified at the CC ranges (antimony, copper, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, beta-BHC, gamma-BHC, gamma-chlordane, dieldrin, endrin, and heptachlor).

3.1 Antimony

Antimony binds to soil and particulates (especially those containing iron, manganese, or aluminum) and is oxidized by bacteria in soil. Exposure routes for mammals include ingestion and inhalation. Antimony does not tend to biomagnify in terrestrial food chains (Ainsworth, 1988), and is not significantly metabolized and excreted in the urine and feces. Antimony at elevated levels has the potential to cause reproductive, pulmonary, and hepatic effects in mammals (EPA, 1999a).

Plants. Antimony is considered a non-essential element and is easily taken up by plants if available in the soil in soluble forms (Kabata-Pendias and Pendias, 1992). A screening level of 5.0 mg/kg has been proposed by Kabata-Pendias and Pendias (1992) based on a report of unspecified phytotoxic responses by plants grown in soil amended with antimony.

Terrestrial Invertebrates. EPA (2005b) has developed an ecological soil screening level (eco-SSL) for antimony of 78 mg/kg. The eco-SSL for antimony is the geometric mean of three EC_{20} values reported in the literature. Kuperman et al. (2002) reported an EC_{20} (concentration of a constituent that elicits an adverse effect to 20% of a population) value using enchytraeids (*Enchytraeus crypticus*) of 194 mg/kg; Phillips et al. (2002) reported an EC_{20} value using springtails (*Folsomia candida*) of 81 mg/kg; and Simini et al. (2002) reported an EC_{20} value using earthworms (*Eisenia fetida*) of 30 mg/kg.

Mammals. Female mice exposed to 5.0 milligrams per liter (mg/L) antimony as antimony potassium tartrate in their drinking water showed a reduction in their lifespan. This dose was equivalent to a lowest-observed-adverse-effects-level (LOAEL) of 1.25 mg/kg/ per day [mg/kg/day]), which can be converted to a no-observed-adverse-effects-level (NOAEL) of 0.125 mg/kg/day (Integrated Risk Information Service, 2001).

Laboratory data on antimony toxicity (as antimony potassium tartrate) in laboratory mice through drinking water ingestion were used to estimate a chronic NOAEL value of 0.125 mg/kg/day (Schroeder et al., 1968). Lifespan and longevity were the endpoints tested.

EPA (2005b) has derived an Eco-SSL for mammalian wildlife species of 0.27 mg/kg antimony in soil. This mammalian Eco-SSL is the lowest calculated value based on reproduction, growth, and survival of ground insectivores (shrew).

Birds. No information was found regarding the potential toxicity of antimony to birds.

Aquatic Life. The available data for antimony indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 9.0 and 1.6 mg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae can occur at concentrations as low as 0.61 mg/L.

Effects from antimony exposure on benthic community composition have been detected at levels between 3.2 and 150 mg/kg (Long and Morgan, 1990). Data on antimony suggest an effects range-low (ER-L) of 2 mg/kg and an effects range-medium (ER-M) of 25 mg/kg.

3.2 Copper

Copper is ubiquitously distributed in nature in the free state and in sulfides, arsenides, chlorides, and carbonates. Several copper containing proteins have been identified in biological systems as oxygen binding hemocyanin, cytochrome oxidase, tyrosinase, and lactase. Copper has also been identified with the development of metalloproteins employed in the sequestering and cellular detoxification of metals. Most organisms are able to regulate copper levels within their systems. Copper may accumulate in the tissues of certain organisms, but it does not tend to accumulate or magnify in higher trophic levels.

Copper has been known to sorb rapidly to sediment. The rate of sorption is of course dependent upon factors such as the sediment grain size, organic fraction, pH, competing cations, and the presence of ligands. In industrialized freshwater environments around the world total copper levels within sediments can range from 7 to 2,350 parts per million (ppm) (Moore and Ramamoorthy, 1984).

Plants. Copper is an essential nutrient for the growth of plants. Background concentrations of copper in grasses and clovers collected in the United States averaged 9.6 mg/kg and 16.2 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Copper is one of the least mobile heavy metals in soil, and its availability to plants is highly dependent on the molecular weight of soluble copper complexes (Kabata-Pendias and Pendias, 1992).

According to Rhodes et al. (1989), elevated copper concentrations in plant tissues do not serve as conclusive evidence of copper toxicity in species of plants such as tomatoes, because some species are able to tolerate higher concentrations of copper than others. The pH of soil may also influence the availability and toxicity of copper in soils to plants (Rhodes et al., 1989). In a study with tomato plants, Rhodes et al. (1989) found a reduction in plant growth when plants were grown in soils containing greater than 150 mg/kg of copper at a pH of less than 6.5. At pH values greater than 6.5, soil copper concentrations of greater than 330 mg/kg were required to reduce plant growth.

Concentrations of copper in leaf tissue that are excessive or toxic to various plant species range from 20 to 100 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 100 mg/kg has been proposed by Efroymson et al., (1997a) as a benchmark screening value for copper phytotoxicity in soil. General symptoms of copper toxicity in plants include the presence of dark green leaves followed by induced iron chlorosis; thick, short, or barbed wire roots; and depressed tillering (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Beyer (1982) and others have reported that copper concentrations in earthworms have been observed to be correlated with copper concentrations in soil. Further studies by Beyer (1990) indicate that copper can be more toxic to bioturbative earthworms than most metals. Research by Phillips (1980) suggests that copper and other metal accumulation within terrestrial invertebrates may vary significantly depending on soil conditions and other physical/chemical properties, and bioconcentration factors can approach 10,000. The EPA (2000a) has derived a soil screening level (SSL) for copper of 61 mg/kg. This invertebrate SSL was based on reproductive and growth data from studies conducted with natural soils under conditions of high or very high bioavailability. The tests were conducted with highly soluble salts and neither aging nor weathering, which would lower bioavailability, was included in the experimental designs.

Mammals. Copper is an essential trace element to animals as well as plants (Callahan et al., 1979), but becomes toxic at concentrations only slightly higher than essential levels (EPA, 1985). Copper is an essential element for hemoglobin synthesis and oxidative enzymes in animals, and is absorbed by mammals following ingestion, inhalation, and dermal exposure. Once absorbed, copper is distributed to the liver, and is not metabolized (Marceau et al., 1970). No evidence of bioaccumulation was obtained in a study of pollutant concentrations in the muscles and livers of 10 species of herbivorous, omnivorous, and carnivorous animals in Donana National Park in Spain (Hernandez et al., 1985). Copper concentrations in small mammals collected from various uncontaminated sites ranged from 8.3 to 13.4 mg/kg (whole-body

concentrations) (Talmage and Walton, 1991). Highest concentrations of copper tend to be in hair, followed in decreasing concentration by liver, kidney, and whole body (Hunter and Johnson, 1982). Among the small mammals collected, Hunter and Johnson (1982) found shrews (*Sorex araneus*) to contain the highest concentrations of copper. Mice were found to contain the lowest copper concentrations. Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg-day of copper as copper sulfate (Lecyk, 1980). Increased mortality rates in mink offspring have been observed at levels above 3.21 mg/kg-day (Aulerich et al., 1982).

Laboratory toxicity data for mink exposed to copper sulfate in their diet were used to estimate a NOAEL value of 11.7 mg/kg/day (Aulerich et al., 1982). Reproduction was the endpoint studied. Symptoms of acute copper poisoning in mammals include vomiting, hypotension, melena, coma, jaundice, and death (Klaassen et al., 1991). Selenium can act as an antidote for copper poisoning.

Birds. Laboratory toxicity data for one-day old chicks exposed to copper oxide in their diets were used to estimate a NOAEL value of 47 mg/kg/day (Mehring et al., 1960). Growth and mortality were the endpoints studied.

Aquatic Life. Invertebrates inhabiting "polluted" freshwaters worldwide have been known to have tissue residues of copper ranging from 5 to 200 ppm (Moore and Ramamoorthy, 1984). Field studies have shown that there is virtually no accumulation of this metal through the food chain (Fuller and Averett, 1975). Studies by Kosalwat and Knight (1987) indicated that copper present in the substrate or sediment was significantly less toxic to chironomid species than overlying water column levels. The substrate copper concentration at which chironomid larval growth was reduced 50 percent (EC_{50}) was 1,602 mg/kg. These researchers found that deformities in larval mouth parts were observed in elevated concentrations, and adult emergence was inhibited when the sediment concentration exceeded 1,800 mg/kg. Carins et al. (1984) reported copper toxicity in sediment for several chironomus midges and cladocerans with LC_{50}s (lethal concentration to 50% of population) ranging from 681 to 2,296 mg/kg.

Moore and Ramamoorthy (1984) reported that copper can be highly toxic to many aquatic plants and algae. Inhibition of growth can occur at levels as low as 0.1 mg/L. In some algal species, copper may inhibit electron transport during photosynthesis. In general, since low pH increases the proportion of free ions in solution, acidic waters may exhibit greater copper toxicity. However, Stokes (1975) reported the observance of algal adaptation to copper-tainted waters with certain species able to tolerate and flourish within highly copper-contaminated waters.

Moore and Ramamoorthy reported LC_{50} in fresh water fish ranging from 0.017 to 1.0 mg/L. Copper is similar to other metals in that its toxicity to fin fish is often greater within fresh water environments versus marine environments because of the lack of complexing agents within fresh water.

3.3 Lead

Global production of lead from both smelter and mining operations has been high throughout the past century. Lead is commonly used in storage batteries as well as in ammunition, solder, and casting materials. In addition, tetraethyl lead was a principal additive to gasolines as an anti-knock agent and was commonly used as an additive in paints. In short, lead is one of the most ubiquitous pollutants in the civilized world.

Lead is strongly sorbed in sediments, and the rate is strongly correlated with grain size and organic content. In the absence of soluble complexing species, lead is almost totally adsorbed to clay particles at pHs greater than 6 (Moore and Ramamoorthy, 1984).

Plants. Although lead is not an essential nutrient for plant growth, it is detected in plant tissues due to the prevalence of lead in the environment. The bioavailability to plants of lead in soil is limited. Bioavailability may be enhanced by a reduction in soil pH, a reduction in the content of organic matter and inorganic colloids in soil, a reduction in iron oxide and phosphorous content, and increased amounts of lead in soil (National Research Council of Canada [NRCC], 1973). Plants can absorb lead from soil and air. Aerial deposition of lead can also contribute significantly to the concentration of lead in above-ground plant parts. Lead is believed to be the metal of least bioavailability and the most highly accumulated metal in root tissue (Kabata-Pendias and Pendias, 1992). Lead tends not to accumulate into plants from soil unless concentrations are very high (i.e., percentage levels). The tips of some trees, such as pine and fir, can accumulate lead from contaminated soil when contamination levels are high. Such conditions often occur at mining sites (NLM, 1996). Lead inhibits plant growth, reduces photosynthesis, and reduces mitosis and water absorption.

Mean background concentrations of lead in grasses and clovers have been reported to range from 2.1 to 2.5 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Adverse effects of lead on terrestrial plants occur only at total concentrations of several hundred mg/kg of soil (Eisler, 1988). This is explained by the fact that, in most cases, lead is tightly bound to soils, and substantial amounts must accumulate before it can affect the growth of higher plants (Boggess, 1977).

The Eco-SSL for plants, as derived by EPA (2005c), is 110 mg/kg. The plant Eco-SSL is the geometric mean of the maximum acceptable toxicant concentration (MATC) for four test species (loblolly pine, red maple, Berseem clover, and ryegrass) under three different test conditions.

Terrestrial Invertebrates. Lead has been shown to accumulate in the tissues of lower trophic level organisms, including terrestrial invertebrates, but is not effectively transferred to higher trophic level organisms through the food web. Centipedes (*Lithobius variegatus*) that ate woodlice hepatopancreas did not assimilate lead even though the food contained concentrations that were many times greater than normally encountered. However, survival and reproduction were reduced in woodlice (*Porcellio scaber*) fed soil litter treated with 12,800 mg/kg lead (Beyer and Anderson, 1985). This concentration of lead is similar to the amount of lead reportedly associated with reductions in natural populations of decomposers, such as fungi, earthworms, and arthropods.

EPA (2005c) has derived an Eco-SSL based on soil invertebrate toxicity of 1,700 mg/kg lead in soil. The Eco-SSL for terrestrial invertebrates is the geometric mean of the MATC values for one test species (*Folsomia candida*) under three different test conditions.

Mammals. As with plants, lead is not considered an essential nutrient for mammalian life. Ingestion is the major route of exposure for wildlife. Lead tends to accumulate in bone, hair, and teeth. Biomagnification of lead is negligible (Eisler, 1988). Jenkins (1981) also reported that soil conditions of low alkalinity and low pH can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower animals, and higher plants. Reduced survival was reported at acute oral doses as low as 5 mg/kg body weight in rats, at a chronic dose of 0.3 mg/kg body weight in dogs, and at a dietary level of 1.7 mg/kg body weight in horses (Eisler, 1988). Laboratory data from studies of rats fed lead acetate in their diets were used to estimate a NOAEL value of 8.0 mg/kg-day (Azar et al., 1973). Reproduction was the endpoint for this study. Symptoms of lead poisoning in mammals are diverse and depend on the form of lead ingested, the concentration, and the species and its age. These symptoms may include reproductive impairment, decreased body weight, vomiting, uncoordinated body movements, visual impairment, reduced life span, renal disorders, and abnormal social behavior (Eisler, 1988).

In laboratory studies, breeding mice exposed to low doses of lead in drinking water (25 ppm) resulted in loss of the strain in two generations with many abnormalities (Schroeder and Mitchener, 1971). Exposure of rats in this same experiment resulted in many early deaths and runts. Blood δ-aminolevulinic acid dehydratase activity associated with exposure to lead was

reduced in white-footed mice living near a metal smelter (Beyer et al., 1985). Amounts of whole-body lead content and feeding habits of roadside rodents have been correlated with highest body burdens in insectivores such as shrews, intermediate in herbivores, and lowest in granivores (Boggess, 1977; Getz et al., 1977).

EPA (2005c) has derived an Eco-SSL for lead in soil of 56 mg/kg for the protection of mammalian species. This mammalian Eco-SSL for lead is based on the NOAEL for reproduction, growth and survival in a number of mammalian species.

Birds. Most of the information on the effects of lead to terrestrial vertebrates is concerned with acute poisoning of waterfowl by lead shot. Apparent symptoms include loss of appetite and mobility, avoidance of other birds, lethargy, weakness, emaciation, tremors, dropped wings, green feces, impaired locomotion, loss of balance and depth perception, nervous system damage, inhibition of heme synthesis, damage to kidneys and liver, and death (Eisler, 1988; Mudge, 1983). Anemia, kidney disease, testicular and liver lesions, and neurological disorders have been associated with high brain lead concentrations in mourning doves (*Zenaida macroura*) (Kendall, 1992). Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively more tolerant to moderate lead exposure, including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2,000 ppm (Hoffman et al., 1985).

Toxicity of lead to birds is dependent upon the form of lead, the route of exposure and exposure duration, and the species and age of the bird. Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively tolerant to moderate lead exposure (Eisler, 1988). Laboratory toxicity data for American kestrels fed metallic lead in their diet were used to estimate a NOAEL value of 3.85 mg/kg-day (Pattee, 1984). Reproduction was the endpoint for this study.

An avian Eco-SSL for lead has been derived by EPA (2005c) to be 11 mg/kg lead in soil. This avian Eco-SSL for lead is based on the NOAEL for reproduction, growth and survival in a number of avian species.

Aquatic Life. All life stages are sensitive to the toxic effects of lead; however, embryos are more sensitive to lead than are later juvenile stages (Davies et al., 1976). Lead uptake depends on exposure time, aqueous concentration, pH, temperature, salinity, diet, and other factors. For example, gill, liver, kidney, and erythrocytes accumulate lead from aqueous sources in proportion to exposure time and concentration (Holcombe et al., 1976). Direct erythrocyte injury is considered the first and most important sign of lead poisoning in catfish (Dawson, 1935). Respiratory distress occurs in fish living in rivers receiving lead mining wastes in England

(Carpenter, 1924, 1925, 1926). Fish are thought to be asphyxiated as a result of a mucous coating over the gills (National Academy of Sciences [NAS], 1972).

No significant biomagnification of lead occurs in aquatic ecosystems (Boggess, 1977). Background concentrations of lead in fish tend to be less than 1 mg/kg (dry weight) (Eisler, 1988). The EPA's National Recommended Water Quality Criteria for lead in freshwater are 65 µg/L for acute exposure and 2.5 µg/L for chronic exposure (EPA, 1999b). In general, dissolved lead is more toxic than total lead, and organic forms of lead are more toxic than inorganic forms. Soluble lead in the water column becomes less bioavailable as water hardness increases. Chronic exposure of fish to lead may result in signs of lead poisoning such as spinal curvature, anemia, darkening of the dorsal tail region, destruction of spinal neurons, difficulties in swimming, growth inhibition, changes in blood chemistry, retarded sexual development, and death (Eisler, 1988).

Physicochemical conditions within the water may also affect lead uptake and toxicity. Under conditions of low alkalinity (less than 50 microequivalents per liter) and low pH, lead can accumulate in fish, algae, mollusks, and benthic invertebrates (Wiener and Stokes, 1990). Irwin (1988) reported significant accumulations of lead in the Trinity River within mosquitofish, turtles, bullhead minnows, and crayfish. Nevertheless, lead concentrations were not higher in top-of-the-food-chain predators like gar than they were in mosquitofish, suggesting minimal biomagnification of lead.

The majority of benthic invertebrates do not bioconcentrate lead from water or abiotic sediment particles. There is some evidence of bioaccumulation through the food web of organic forms of lead, such as tetraethyl lead. Anderson, et al. (1980) reported a lead LC_{50} of 258 ppm for the chironomid and that growth of this organism was not reduced above this level in freshwater sediments. In addition, Suter and Tsao (1996) reported effect levels in the water flea (*Daphnia magna*) to be in the 12 parts per billion (ppb) range, while Khangrot and Ray (1989) reported a *D. magna* LC_{50} of 4.89 ppm.

3.4 Mercury

Mercury is a toxic compound with no known natural biological function. Mercury exists in three valence states: mercuric (Hg^{2+}), mercurous (Hg^{1+}), and elemental (Hg^{0+}) mercury. It is present in the environment in inorganic and organic forms. Inorganic mercury compounds are less toxic than organo-mercury compounds; however, the inorganic forms are readily converted to organic forms by bacteria commonly present in the environment. The organo-mercury compound of greatest concern is methylmercury (EPA, 1999a).

Mercury sorbs strongly to soil and sediment. Elemental mercury is highly volatile. In aquatic and terrestrial receptors, some forms of mercury, especially organo-mercury compounds, bioaccumulate significantly and biomagnify in the food chain. In all receptors, the target organs are the kidney, and central nervous system. However, mercury causes numerous other effects including teratogenicity and mutagenicity (EPA, 1999a).

Plants. Mercury is not required for plant growth. Background concentrations of mercury in plants usually range from 0.0026 to 0.086 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Pine needles have been reported to be good biomonitors of mercury-contaminated environments (Kabata-Pendias and Pendias, 1992). In general, the concentration of mercury in plants will be elevated when mercury concentrations in soils are high. Mercury concentrations in plants, however, generally do not exceed those in associated soils (Lisk, 1972). Methyl mercury is more available to plants than either phenyl- or sulfide-mercury. In addition to mercury uptake from the soil, plants can also absorb mercury vapor (Browne and Fang, 1978).

Concentrations of mercury in leaf tissue that are excessive or toxic to various plant species range from 1 to 3 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 0.3 mg/kg has been proposed by Efroymson et al., (1997a) as a benchmark screening value for mercury phytotoxicity. General symptoms of mercury toxicity in plants include severe stunting of seedlings and roots and leaf chlorosis and browning of leaf points (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Abbasi and Soni (1983) exposed earthworms (*Octochaetus pattoni*) to mercury (as HgCl) to assess the effect on reproduction and growth. Survival and cocoon production were reduced at 0.5 ppm mercury, the lowest concentration tested. The number of juveniles produced was not affected. Based on these test data, a benchmark value for mercury in soil of 0.1 mg/kg has been proposed by Efroymson et al., (1997b).

Mammals. Mercury is not an essential element for animal life. Background mercury concentrations in wildlife tend to be less than 1.0 mg/kg (wet weight) (Eisler, 1987). Biomonitoring studies have shown that mercury concentrations in mammals are highest in hair, followed by kidney and liver tissues (Bull et al., 1977; Klaassen et al., 1991; Wren, 1986). Mercury is bioaccumulated and biomagnified in terrestrial food chains (Eisler, 1987; Talmage and Walton, 1993). Talmage (1989) has shown the insectivorous shorttail shrew (*Blarina brevicauda*) to be a better monitor of environmental mercury contamination than the granivorous white-footed mouse (*Peromyscus leucopus*). Mink (*Mustela vison*) and river otter (*Lutra*

canadensis) have been shown to be good monitors of mercury contamination within river environments due to their consumption of contaminated fish (Kucera, 1983).

Organic mercury compounds, especially methyl mercury, are more toxic to mammals than inorganic forms of mercury. Selenium has been shown to have a protective effect against mercury poisoning (Ganther et al., 1972). Based on laboratory data for methylmercury fed to rats and mink in their diets, a NOAEL value of 0.015 has been derived. This NOAEL is based on mortality, weight loss, reproduction, and ataxia as endpoints (Wobeser et al., 1976).

Mercury has been shown to be teratogenic, mutagenic, and carcinogenic in animal studies (Eisler, 1987). Signs of mercury poisoning that have been observed in mink include anorexia, weight loss, ataxia and splaying of hind legs, irregular vocalization, salivation, and convulsions (Wren, 1986).

Birds. Concentrations of mercury acutely toxic to birds following oral exposure range from 2.2 to 31 mg/kg body weight (Eisler, 1987). Mercury concentrations in the livers of methylmercury-poisoned birds ranged from 17 to 70 mg/kg (dry weight) (Solonen and Lodenius, 1984). Methylmercury is more toxic to avian species than inorganic mercury (Hill, 1981). In addition to the form of mercury to which the bird is exposed, the species, gender, age, and health of the individual may also influence the toxic response (Fimreite, 1979). Physical signs of mercury poisoning in birds include muscular incoordination, falling, slowness, fluffed feathers, calmness, withdrawal, hyporeactivity, and eyelid drooping (Eisler, 1987).

Japanese quail were fed mercury in their diet for one year to study the effects on reproduction. Egg production increased with increasing mercury dose, while fertility and hatchability decreased. Adverse effects of mercury exposure were evident at the 8 mg/kg dose level. Based on the results of this study a NOAEL value of 0.45 mg/kg/day has been derived (Hill and Schaffner, 1976). Mallard ducks fed methyl mercury dicyandiamide in their diets produced fewer eggs and fewer ducklings at exposure levels as low as 0.5 mg/kg. A NOAEL value of 0.0064 mg/kg/day was derived from these data, with reproduction the endpoint studied (Heinz, 1979).

Aquatic Life. Concentrations of mercury in freshwater fish collected from 12 monitoring stations in the United States from 1978 to 1981 ranged from 0.1 to 1.1 mg/kg (wet weight), with an average of 0.11 mg/kg (Lowe et al., 1985). Elevated concentrations of mercury in fish have often been associated with low pH, low calcium concentrations in the water, and low water hardness (Eisler, 1987). Methylating bacteria in sediments actively convert inorganic mercury

into methylmercury. This results in an increase in the bioavailability of mercury. Fish absorb methylmercury more easily than inorganic mercury from the water column (Huckabee, et al., 1979). Because exposure of fish to methylmercury can occur via ingestion of contaminated prey, methylmercury concentrations are usually highest in organisms near the top of the food chain, such as carnivorous fish (Huckabee, et al., 1979).

Exposure of aquatic organisms to elevated mercury concentrations can result in reduced growth and reproduction (Eisler, 1987). The National Recommended Water Quality Criteria for acute and chronic exposure to mercury in freshwater systems are 1.4 and 0.77 μg/L, respectively (EPA, 2002). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} value for methylmercury is less than 0.03 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids has been determined to be 0.87 μg/L (Suter and Tsao, 1996).

Physical signs of acute mercury poisoning in fish include the flaring of gills, an increase in the frequency of respiratory movements, loss of equilibrium, and sluggishness (Armstrong, 1979).

3.5 Zinc

Zinc is a naturally occurring element that may be found in both organic and inorganic forms and, as such, is commonly found in the environment. In general, zinc is concentrated in the sediments of water bodies. The NAS (1979) has reported that zinc will probably be detected in 75 percent of all water bodies examined for the compound at various locations. The fate of zinc in soils appears to have a pH basis. Studies have shown that a pH of less than 7 often favors zinc desorption (EPA, 1984).

Plants. Background concentrations of zinc in terrestrial plants range from 25 to 150 mg/kg (dry weight) (NAS, 1979). The deficiency content of zinc in plants is between 10 and 20 ppm (dry weight). Roots often contain the highest concentrations of zinc (Kabata-Pendias and Pendias, 1992).

Certain species of plants, particularly those from the families Caryophyllaceae, Cyperaceae, and Plumbaginaceae, and some tree species are extremely tolerant to elevated zinc concentrations (Kabata-Pendias and Pendias, 1992). Concentrations of zinc in these plants may reach 1 percent

(dry weight) in the plant. Concentrations in leaf tissue that are excessive or toxic to various plant species range from 100 to 400 mg/kg. Concentrations of 100 to 500 mg/kg are expected to result in a 10 percent loss in crop yield (Kabata-Pendias and Pendias, 1992). General symptoms of zinc toxicity in plants include the presence of chlorotic and necrotic leaf tips, interveinal chlorosis in new leaves, retarded growth of the entire plant, and injured roots that resemble barbed wire (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. The EPA (2000a) has developed an ecological soil screening level (SSL) for zinc in soil of 120 mg/kg. This SSL was based on reproduction and population effects in experiments conducted with natural soils under conditions of high or very high zinc bioavailability. It is also important to note that in studies conducted with mixtures of cadmium, copper, and zinc, it was concluded that the three metals acted antagonistically. It has also been shown that a decrease in pH and/or organic matter in the soil tends to decrease the concentration of zinc in soil at which toxic effects are observed (Spurgeon and Hopkin, 1996). Zinc has been shown to accumulate in earthworm species (Beyer et al., 1982), but generally is not biomagnified through the food web.

Mammals. Zinc is an essential trace element for normal fetal growth and development. However, exposure to high levels of zinc in the diet has been associated with reduced fetal weights, altered concentrations of fetal iron and copper, and reduced growth in offspring (Cox et al., 1969). Poisoning has been observed in ferrets and mink from chewing corroded galvanized cages (Clark et al., 1981). Symptoms of zinc toxicity are lassitude, slower tendon reflexes, bloody enteritis, diarrhea, lowered leukocyte count, depression of the central nervous system, and paralysis of the extremities (Venugopal and Luckey, 1978). A study by Kinnamon (1963) showed a NOAEL for oral exposure to a zinc compound over a period of 73 days to be 250 mg/kg body weight, and mice given 500 mg/L of zinc as zinc sulfate in drinking-water have shown hypertrophy of the adrenal cortex and pancreas. Young animals are much more susceptible to poisoning by zinc than are mature animals (Clark et al., 1981).

Animals are quite tolerant of high concentrations of zinc in the diet. Levels 100 times that required in the diet usually do not cause detectable symptoms of toxicosis (NAS, 1979). Laboratory data for rats exposed to zinc oxide in their diet were used to estimate a NOAEL value of 160 mg/kg-day (Schlicker and Cox, 1968). Reproduction was the endpoint studied. Symptoms of zinc poisoning in mammals include lameness, acute diarrhea, and vomiting (Eisler, 1993).

Birds. Dietary zinc concentrations of greater than 2,000 mg/kg diet are known to result in reduced growth of domestic poultry and wild birds (Eisler, 1993). Reduced survival has been documented at zinc concentrations greater than 3,000 mg/kg diet or at a single dose of greater than 742 mg/kg body weight (Eisler, 1993). Laboratory data for white leghorn hens exposed to zinc sulfate in their diet were used to estimate a NOAEL value of 14.5 mg/kg-day (Stahl et al., 1990). Reproduction was the endpoint for this study. A value of 51 mg/L has been calculated as the NOAEL for chronic exposure of birds to zinc carbonate in drinking water (Sample et al., 1996).

Aquatic Life. Zinc residues in freshwater and marine fish are generally much lower than those found in algae and invertebrates. Thus there is little evidence for bioaccumulation (Moore and Ramamoorthy, 1984). Rainbow trout (*Oncorhyncus mykiss*) have the ability to detect and avoid areas of water containing 5.6 ppb zinc (Sprague, 1968). Cairns and Scheier (1968) reported 96-hour LC_{50}s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC_{50}s ranging from 2.86 to 3.78 ppm in soft water. These results demonstrate that water hardness affects the toxicity of zinc to fish. Chronic toxicity tests have been conducted with five species of freshwater fish. Chronic values ranged from 47 µg/L for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinallis*) (EPA, 1984).

Acute toxicity to freshwater invertebrates is relatively low and, as with other metals, increasing water hardness decreases the toxicity of zinc (Moore and Ramamoorthy, 1984). As reported by Baudouin and Scoppa (1974), the 48-hour LC_{50} for the cladaceran *Daphnia hyalina* was 0.055 mg/L, and 5.5 mg/L for the copepod *Cyclops abyssorum*. Four chronic toxicity tests are reported for *Daphnia magna*, with chronic values ranging from 47 µg/L to 136 µg/L (EPA, 1984). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1984).

3.6 DDT and Metabolites

DDT is a chlorinated pesticide that has been banned in the United States since 1972. DDE and DDD are metabolites of DDT. DDT and its metabolites will adsorb very strongly to soil and are subject to evaporation and photodegradation at the soil surface. DDT and related compounds are very persistent in soils (NLM, 1996).

Plants. DDT can be taken up by plant roots and translocated to the above-ground plant parts. Concentrations of DDT and its metabolites are usually greatest in the roots of the plants (Voerman and Besemer, 1975). The effects of DDT, DDD, and DDE on plant growth and reproduction are not well documented.

Terrestrial Invertebrates. No information was found regarding the toxicity of DDT and its metabolites to terrestrial invertebrates.

Mammals. As in plants, DDT and its major metabolites DDE and DDD are ubiquitous in wild mammals and birds. Because DDT can be biomagnified through food chains, insectivorous shrews generally contain higher concentrations than herbivorous small mammals collected from the same site. DDT residues were found to be higher in juveniles than adults and increased with increased body fat content (Talmage and Walton, 1991).

The most significant exposure route for vertebrates to DDT and its metabolites is oral exposure. Dermal exposure is believed to be very limited, and inhaled DDT and associated particulates are believed to be deposited in the upper respiratory tract and eventually swallowed. The toxicity of DDT and its metabolites to mammalian and avian wildlife is dependent on the fat content within the animal. In both mammals and birds the storage of DDT and DDE in fat is protective, because it decreases the amount of chemical in circulation that may reach the brain, which is the site of toxic action (NLM, 1996). Signs of acute DDT poisoning in animals include paresthesia of the tongue, lips and face; apprehension; dizziness; tremor; disturbed equilibrium; and convulsions (Klaassen et al., 1991).

Birds. DDE concentrations in eggs have been negatively correlated with eggshell thickness in bald eagles. Studies have shown reduced reproductive success at DDE egg residue concentrations of greater than 3 mg/kg for the white-faced ibis, 5 mg/kg for the snowy egret, and 8 mg/kg for the black-crowned night heron (Henny et al., 1985). Data from a study of brown pelicans fed DDT in their diets for 5 years were used to derive a NOAEL value of 0.0028 mg/kg/day (Anderson et al., 1975). Reproduction was the critical endpoint in this study. Signs of DDT poisoning in birds include ataxia, wing-drop, jerkiness in gait, continuous whole-body tremors, falling, and convulsions (Hudson et al., 1984).

Aquatic Life. DDT in freshwater environments partitions primarily into sediment. Biodegradation of DDT in sediment may be significant. Bioconcentration factors between 51,000 and 100,000 have been reported for fathead minnows exposed to DDT (NLM, 1996). The Federal Water Quality Criteria for the protection of aquatic life for acute and chronic exposures to DDT and its metabolites in freshwater systems are 1.1 and 0.001 μg/L, respectively (EPA, 2002).

3.7 Gamma-Chlordane

Chlordane is a chlorinated pesticide whose commercial use has been banned in the United States since 1988 (EPA, 1988). Technical chlordane is a mixture of some 50 different compounds; the major constituents being alpha-, beta-, and gamma-chlordane (Howard, 1991). Chlordane is generally immobile in soil; however, movement to groundwater can occur (Howard, 1991). The compound is biodegraded at a very slow rate (Howard, 1991). In aquatic systems, chlordane is not expected to undergo significant hydrolysis, oxidation, or direct photolysis (Howard, 1991). Because of its widespread use in the past and its low biodegradation rate, chlordane is commonly measured in low concentrations in environmental samples (Eisler, 1990). Chlordane, with an organic carbon-sediment partition coefficient (K_{oc}) of between 15,500 and 24,600, strongly adsorbs to sediment (Howard, 1991).

Plants. Information on the toxicity of chlordane in plants is very limited. Chlordane concentrations of greater than 100 µg/L were found to inhibit the growth of blue-green algae (Glooschenko et al., 1979), whereas concentrations of greater than 1,000 µg/L inhibited the growth of green algae (Glooschenko and Lott, 1977). A 1971 nationwide survey of chlordane concentrations in corn (*Zea mays*) and sorghum (*Sorgum halepense*) indicated 0.48 mg/kg (dry weight) in corn kernels, 1.26 mg/kg (dry weight) in corn stalk, and 0.42 mg/kg (dry weight) in sorghum (Carey et al., 1978).

Terrestrial Invertebrates. No information was found regarding the potential toxicity of chlordane to terrestrial invertebrates.

Mammals. Absorption of chlordane can occur through the skin, diet, and inhalation (Eisler, 1990). Studies have shown the metabolites of technical grade chlordane to be present in bat milk (Clark, et al., 1978). Oxychlordane and heptachlor epoxide are metabolites of the components of technical grade chlordane (Eisler, 1990). Oxychlordane is more toxic and persistent than any of the parent compounds.

Acute oral LD_{50} values for exposure of sensitive mammalian species to technical grade chlordane range from 25 to 50 mg/kg body weight. Chlordane fed to rats at 2.5 mg/kg caused slight liver damage (National Research Council [NRC], 1977). Growth retardation and liver damage were observed in rats fed 150 and 300 mg/kg chlordane in their diets over a two-year period (Clayton and Clayton, 1982). No measurable effects were found in Cynomolgus monkeys (*Macaca spp.*) exposed to technical grade chlordane concentrations of 10 µg/L in air for 90 days (Khasawinah et al., 1989). Based on a study of mice fed technical grade chlordane in their diets,

a NOAEL value of 4.6 mg/kg/day has been derived (Keplinger et al., 1968). The critical endpoint for this study was reproduction.

Symptoms of acute chlordane poisoning in mammals include diarrhea, avoidance of food and water, hair loss, hunched appearance, abdominal distension, labored respiration, salivation, muscle tremors, incoordination, convulsions, and death in some cases (Eisler, 1990). Chlordane has been reported to be carcinogenic in mice (Eisler, 1990) and teratogenic in rats (NRC, 1977). Chlordane potentiates aldrin, endrin, and parathion toxicity in mice (Jones et al., 1977).

Birds. Chlordane and related metabolites are commonly detected in wild birds throughout the United States. A 1982 survey revealed 45 percent of the captured European starlings (*Sturnus vulgaris*) contained detectable concentrations of oxychlordane (Eisler, 1990). Avian studies have shown older birds and raptors to contain relatively high concentrations of chlordane and its metabolites (Eisler, 1990). Lethal concentrations of oxychlordane and heptachlor have been measured in raptors that consumed either poisoned bait or prey (Blus et al., 1983). As in mammals, chlordane concentrations are greatest in tissues high in lipid content (Eisler, 1990). Biological half-lives of chlordane isomers in birds range from 11.2 to 35.4 years (Eisler, 1990).

Among the birds tested, California quail (*Callipepla californica*) are the most sensitive to chlordane, with an acute oral LD_{50} of 14.1 mg/kg body weight (Hudson et al., 1984). Toxic responses to technical chlordane are primarily attributed to the metabolite oxychlordane (Stickel, et al., 1983). Oxychlordane concentrations in brain tissue greater than 5 mg/kg (wet weight) are considered lethal to birds (Stickel et al., 1979). Based on exposure of red-winged blackbirds to chlordane in their diets for 84 days, a chronic NOAEL value of 2.14 mg/kg/day has been derived (Stickel, et al., 1983). The critical endpoint in this study was mortality. Signs of chlordane poisoning in birds include sluggishness, drooped eyelids, fluffed feathers, altered resting patterns, reduced appetite, weight loss, quivering and panting, neck arched over back, and arched back (Stickel et al., 1983).

Aquatic Life. Because of the widespread historical use of chlordane, it is commonly detected in aquatic environments. Veith et al. (1979) reported 36 percent of all fish samples collected in domestic watershed throughout the United States in 1976 to contain chlordane. Of the components of technical grade chlordane detected in fish, cis-chlordane was the most abundant, followed by trans-nonachlor, trans-chlordane, and cis-nonachlor (Ribick and Zajicek, 1983). Bioconcentration factors for aquatic organisms range from 108 for frogs (*Xenopus laevis*) to 38,000 for fathead minnows (*Pimephales promelas*) (Howard, 1991). There is some evidence of biomagnification of chlordane in freshwater fish (Eisler, 1990).

Variations in the response of aquatic organisms to chlordane are related to both biological factors of the biota and physical factors of chlordane and the aquatic environment. Younger life-stages are more sensitive to chlordane toxicity than older life-stages (Eisler, 1990). In addition, the health and lipid content of the organism can have an impact on the response (Eisler, 1990). Water temperature, salinity, and sediment loading are abiotic factors that influence the bioavailability of chlordane (NRCC, 1975). Cis-chlordane appears to be more toxic than trans-chlordane, and the photoisomers seem to be more toxic than the parent compounds (Eisler, 1990).

The EPA's National Ambient Water Quality Criteria for chlordane is 2.4 μg/L for acute exposure and 0.0043 μg/L for chronic exposure of aquatic life (EPA, 2002). The water quality criteria for chlordane represent the sum of alpha-chlordane, gamma-chlordane, alpha-chlordene, gamma-chlordene, alpha-nonachlor, gamma-nonachlor, and oxy-chlordane. The lowest chronic toxicity values for chlordane reported in the literature for fish and daphnids are 1.6 and 16 μg/L, respectively (Suter and Tsao, 1996).

The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of chlordane causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The fish EC_{20} value for fish is less than 0.25 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration of chlordane causing less than 20 percent reduction in the growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids is 12.1 μg/L (Suter and Tsao, 1996). Signs of chlordane poisoning in fish include hyperexcitability, increased respiration rate, erratic swimming, loss of equilibrium, convulsions, and death (NRCC, 1975).

3.8 Endrin

Endrin was first used as an insecticide, rodenticide, and avicide beginning in 1951 to control cutworms, voles, grasshoppers, borers, and other pests on cotton, sugarcane, tobacco, apple orchards, and grain (EPA, 1979). Endrin tends to persist in the environment mainly in forms sorbed to sediments and soil particles. Endrin is extremely persistent when released to soil. A conservative estimate of its half-life in sandy loam soils is approximately 14 years (Nash and Woolson, 1967).

Plants. No information was found regarding the toxicity of endrin to plants.

Terrestrial Invertebrates. No information was found regarding the toxicity of endrin to terrestrial invertebrates.

Mammals. Studies have demonstrated that the nervous system is the primary target for endrin toxicity (ATSDR, 1996). Exposure of animals to endrin causes central nervous system effects, particularly convulsions (Deichmann et al., 1970). Nonspecific degeneration of the liver, kidney, and brain was observed in animals exposed to lethal doses of endrin (Treon et al., 1955). Endrin appears to be well absorbed orally, and distribution is primarily to fat and skin (ATSDR, 1996).

Rats, mice, and guinea pigs administered 4 mg/kg endrin and sacrificed 24 hours later exhibited moderate hepatic necrosis, fatty degeneration, and inflammation (Hassan et al., 1991). Neurological effects are commonly observed in animals exposed to endrin. Hyperirritability to stimuli, tremors, convulsions, and ataxia occurred in three species of animals (dog, rat, and rabbit) administered endrin for acute, intermediate, and chronic durations (Treon et al., 1955).

Data from a study of mice fed endrin in their diets for 120 days were used to derive a NOAEL value of 0.092 mg/kg/day (Good and Ware, 1969). Reduced parental survival, reduced litter size, and reduced number of young were the critical effects determined in this study.

Birds. Mallard ducks fed endrin in their diets for over 200 days were studied for the effects on reproduction (Spann et al., 1986). No significant differences were observed in reproduction at the highest dose level (3 ppm). Based on these data, a NOAEL value of 0.3 mg/kg/day was derived. Screech owls were studied for reproductive effects after exposure to endrin in their diets for over 83 days (Fleming et al., 1982). Egg production and hatching success were reduced among owls fed 0.75 ppm endrin. Based on these data, a NOAEL value of 0.01 mg/kg/day was derived.

Aquatic Life. When released to water, endrin strongly adsorbs to sediment and bioconcentrates significantly in aquatic organisms (ATSDR, 1996). Endrin appears to be biomagnified only slightly through various levels of the food chain (Metcalf et al., 1973). Bioconcentration factors for endrin in aquatic organisms range from 80 for fathead minnows (*Pimephales promelas*) to 49,000 for snails (*Physa sp.*) (ATSDR, 1996). The estimated half-life of endrin in water is more than 4 years (ATSDR, 1996). The National Recommended Water Quality Criteria for endrin in freshwater are 0.086 and 0.036 μg/L for acute and chronic exposures, respectively (EPA, 2002). The Ontario Ministry of the Environment (1992) has identified a lowest effect level in sediment of 3 μg/kg and a severe effect level in sediment of 1,300 μg/kg (Persaud et al., 1993).

Acute toxicity (LC_{50}) for various freshwater fish species ranged from 0.5 μg/L for *Oncorhynchus kisutch* to 314 μg/L for mosquito fish (Verschueren, 1983). Acute toxicity (LC_{50}) for various aquatic insects ranged from 0.03 μg/L for *Acroneuria pacifica* to 2.4 μg/L *for Pteronarcys californica.*

Tissue residues as low as 0.0115 μg/g in largemouth bass (*Micopterus salmoides*) have been shown to reduce survival by 40 percent (Jarvinen and Ankley, 1999).

4.0 Fate and Transport

The environmental fate and transport of contaminants in surface soil, surface water, sediment, and groundwater at the Former Choccolocco Corridor Ranges will govern the potential for exposures to ecological receptors. In general, contaminants in environmental media may be available for direct exposure (e.g., plants exposed to surface soil), and they may also have the potential to migrate to other environmental media or areas of the site. This section discusses the mechanisms by which contaminants can be transported and the chemical properties that determine their transport.

4.1 Fate and Transport in Soil

Contaminants in surface soil at the Former Choccolocco Corridor Ranges have the potential to be transported from their source areas to off-site locations by a number of mechanisms, including volatilization, dust entrainment, surface runoff, and infiltration to subsurface soil/groundwater.

The inorganic and organic constituents detected in surface soil at the Former Choccolocco Corridor Ranges are generally closely associated with the soil particles themselves and are not expected to be transported great distances from their source. Volatilization of inorganic constituents is expected to be insignificant. Fugitive dust generation and entrainment by the wind with subsequent dispersion by atmospheric mixing could transport particulate-associated contaminants to other parts of the study area and to off-site locations. Fugitive dust generation in the forested areas is expected to be minimal due to continuous ground cover; however, in the areas that have been clear-cut or denuded of vegetation, fugitive dust generation could be a significant transport mechanism for surface soil contaminants. Since volatile organic compounds (VOCs) were infrequently detected and at relatively low concentrations, volatilization and subsequent transport of these compounds is expected to be insignificant with respect to other transport mechanisms potentially active at these sites.

The transport of surface soil-associated contaminants by surface runoff is another potential transport mechanism. Several small ephemeral and perennial drainage features drain these sites. These drainage features collect surface runoff from the sites and transport it off-site to the east. As such, surface runoff via these ephemeral and perennial drainage features has the potential for significant constituent transport off-site during periods of significant precipitation.

Contaminants in surface soil at the Former Choccolocco Corridor Ranges may be transported vertically to subsurface soils and groundwater via solubilization in rainwater and infiltration.

Migration in this manner is dependent upon contaminant solubility and frequency of rainfall. The soil type (rough, stony land) in the vicinity of the Former Choccolocco Corridor Ranges does not promote rapid infiltration, but rather is more conducive to the promotion of surface runoff. Based on the constituents detected in surface soil and the soil type found at these sites, vertical migration of surface soil constituents is expected to be minimal at the Former Choccolocco Corridor Ranges.

The transfer of contaminants in surface soil to terrestrial plants through root uptake and to terrestrial animals through ingestion and other pathways are potentially significant transfer mechanisms. Many metals are readily absorbed from soil by plants, but they are not biomagnified to a great extent through the food web. Volatile organic compounds and semi-volatile organic compounds do not bioaccumulate to any significant extent (Shugart et al., 1990); therefore, food web transfer of these constituents is expected to be minimal. Herbicides and pesticides in soil have the potential to accumulate and magnify through terrestrial food chains; therefore, food chain transfer could potentially be a significant transport mechanism for herbicides and pesticides. Nitroaromatic compounds generally do not accumulate or biomagnify through food web interactions; therefore, food web transfer of these compounds is expected to be minimal.

VOCs in the surface soil at the Former Choccolocco Corridor Ranges are expected to volatilize and/or photolyze relatively rapidly when exposed to sunlight (half-lives of 3 hours to 5 days) (Burrows et al., 1989).

4.2 Fate and Transport in Surface Water

Although the majority of the drainage features at the Former Choccolocco Corridor Ranges are ephemeral, they represent a potentially significant transport mechanism during periods of significant precipitation. Constituents transported to these ephemeral drainage features and the perennial streams via surface runoff could be transported off-site via surface water flow through these drainage features. Groundwater from the Former Choccolocco Corridor Ranges is unlikely to infiltrate the drainage features at the Former Choccolocco Corridor Ranges since the depth to groundwater ranges from approximately 10 feet to 65 feet below ground surface (bgs) with an average depth of approximately 28 feet bgs.

4.3 Fate and Transport in Sediment

As is the case with surface water, the transport of sediment in the ephemeral drainage features and perennial streams at these sites could represent a transport pathway for soil-related constituents to off-site areas during periods of significant precipitation. Sediment in the

ephemeral drainage features perennial streams could become entrained in the runoff conducted in these drainage features and could potentially be transported off-site.

4.4 Constituent-Specific Fate and Transport Properties

The following sub-sections describe the fate and transport properties of each of the COPECs identified at the CC ranges.

4.4.1 Antimony

Little is known of the adsorptive behavior of antimony, its compounds, and ions. The binding of antimony to soil is determined by the nature of the soil and the form of antimony deposited on the soil. Some forms of antimony may bind to inorganic ligands. On the other hand, a mineral form would be unavailable for binding. Since antimony has an anionic character, it is expected to have little affinity for organic carbon (ATSDR, 1992). Antimony binds to soil, particularly to particles containing iron, manganese, or aluminum (ATSDR, 1992). Bodek et al. (1988) indicate that antimony oxides are highly soluble, which suggests environmental mobility. However, Callahan et al. (1979) indicate that antimony may have an affinity for clay and other mineral surfaces. Some studies suggest that antimony is fairly mobile under diverse environmental conditions (Rai and Zachara, 1984), while others suggest that it is strongly adsorbed to soil (Ainsworth, 1988; Foster, 1989; King, 1988). There are no data available regarding the partitioning of the various forms of antimony to different solvents or environmental media. Therefore, it can be concluded that the fate of antimony in soil is somewhat uncertain, and dependent upon many inter-related environmental factors. Antimony is also oxidized by bacteria in the soil. In water, antimony is oxidized when exposed to atmospheric oxygen.

Antimony does not appear to bioconcentrate appreciably in fish and aquatic organisms. No detectable bioconcentration occurred during a 28-day test using bluegills (EPA, 1980). Bioconcentration factors (BCFs) for antimony ranged from 0.15 to 390 (Callahan, 1978). Uptake of antimony from soil by plants is minor and appears to be correlated with the amount of antimony that is soluble (Ainsworth, 1988). Antimony is not significantly metabolized and is excreted in the urine and feces. It does not biomagnify in terrestrial food chains, but can bioconcentrate to a slight degree in aquatic organisms. Antimony bioconcentration was measured in voles, shrews, rabbits, and invertebrates around a smelter. Analysis of antimony in organs of the small mammals, compared with estimates of their antimony intake from food, showed that, although the amount of antimony in the organs was elevated, it was low compared to the amount ingested. The results suggest that antimony does not biomagnify from lower to higher trophic levels in the food chain (ATSDR, 1992). It should also be noted that antimony is associated with ammunition, being present in lead alloys in bullets and in materials used as

primers. Antimony can be present in both the +3 and +5 valence states, depending on pH, oxidation-reduction potential, and several other chemical properties of the environmental medium in which it is found. Antimony can methylate via chemical and/or biological reactions into an organic form under reducing conditions such as those commonly found within highly organic fine sediments and hydric soils (ATSDR, 1992).

4.4.2 Copper

In general, adsorption is probably the most important controlling mechanism in determining copper mobility in the environment. Copper's movement in soil is determined by a host of physical and chemical interactions with the soil components. In general, copper will adsorb to organic matter, carbonate minerals, clay minerals, or hydrous iron and manganese oxides. Sandy soils with low pHs have the greatest potential for leaching. When the amount of organic matter is low, the mineral content or Fe, Mn, and Al oxides become important in determining the adsorption of copper. Copper binds to soil much more strongly than other divalent cations, and the distribution of copper in the soil solution is less affected by pH than other metals (ATSDR, 2004). The solubility of copper in soil tends to increase as the pH decreases. Because the soils at the Former Choccolocco Corridor Ranges exhibit a neutral to somewhat acidic pH, it would be reasonable to assume that the copper in soil would be subject to leaching and somewhat mobile. However, the iron and manganese content of the soil tends to form copper complexes and the copper tends to be fairly immobile at these ranges, as evidenced by the lack of copper contamination in subsurface soils and/or groundwater. There are no data available regarding the partitioning of the various forms of copper to different solvents or environmental media.

Copper binds primarily to organic matter in sediment, unless the sediment is organically poor. It also binds to iron oxides. The solubility of copper in sediments tends to increase as the pH of the sediment decreases.

The BCF of copper in fish obtained in field studies ranges from 10 to 667, indicating a low potential for bioconcentration. The BCF is higher in mollusks, where it may reach 30,000 (Perwak, et al., 1980). This may be due to the fact that many mollusks are filter feeders, and copper concentrations are higher in particulates than in water. There is abundant evidence, however, that there is no biomagnification of copper in the food chain. No evidence of bioaccumulation in herbivorous, omnivorous, and carnivorous mammals was obtained during a study of 10 mammal species in Donana National Park in Spain (Hernandez et al., 1985). A study of metals in cottontail rabbits showed that while the concentration of copper in surface soil was 130 percent higher than in control areas, the concentration of copper in foliar samples was insignificant. No significant increase in copper was observed in rabbit muscle, femur, kidney, or

liver, indicating that copper was not bioaccumulating in the food chain. Even at the lowest levels of the food chain, there is little evidence of copper bioaccumulation. In a study of earthworms and soil from 20 different sites, copper concentrations in earthworms poorly correlated with copper in soil (ATSDR, 2004).

At the pH values and carbonate concentrations characteristic of natural waters, most dissolved copper exists as carbonate complexes rather than as free (hydrated) cupric ions. The concentration of dissolved copper depends on factors such as pH, oxidation-reduction potential, and the presence of competing cations (Ca^{2+}, Fe^{2+}, Mg^{2+}, etc.), anions of insoluble cupric salts (OH^-, S^{2-}, PO_4^{3-}, etc.), and organic and inorganic complexing agents. Allard (1995) reported that copper can exist in the form of freely-dissolved divalent copper cation at a pH of less than 6. Complexation of copper with humic acids can increase the mobility of copper in groundwater and/or surface water but will also reduce the bioavailability to biota. The most significant precipitate formed in natural waters is malachite [$Cu_2(OH)_2CO_3$]. As a result of the aforementioned physico-chemical processes, copper in water may be dissolved or associated with colloidal or particulate matter. Copper complexed in colloidal or particulate forms is generally non-mobile. The combined processes of complexation, adsorption, and precipitation control the level of free copper. The chemical conditions in most natural waters are such that, even at relatively large copper concentrations, these processes will reduce the free copper concentration to extremely low values (ATSDR, 2004).

Between pH 5 and 6, adsorption is the principal process for removing copper from water; above pH 6, precipitation becomes more dominant. Copper binding in soil is correlated with pH, cation exchange capacity, organic content of the soil, and presence of iron oxides. Copper may also be incorporated into mineral lattices where it is unlikely to have ecological significance. In soils with high organic carbon content, copper will be tightly bound to organic matter (ATSDR, 2004). The soil/water partition coefficient for copper has been measured to be >64 for mineral soils and >273 for organic soils, indicating a relatively strong affinity for copper to remain adsorbed to soil (ATSDR, 2004). In sediment, copper is generally associated with mineral matter or tightly bound to organic material (Kennish, 1998).

4.4.3 Lead

The chemistry of lead in aqueous solution is highly complex because this element can be found in a multiplicity of forms. The form of lead at any given site is very important since its bioavailability and uptake dynamics are generally dictated by its form. For example, lead fumes, as from a smelter or gasses generated from the discharge of artillery or bullets, are more bioavailable than mining wastes or intact pieces of lead fragments. The difference is therefore

not only the size of the particles but its chemical form. It should also be noted that lead in soil can slowly undergo speciation to more insoluble sulfate, sulfide, oxide, and phosphate salts (NLM, 1996). Lead has a tendency to form compounds of low solubility with the major anions of natural water. In the natural environment, the divalent form is the stable ionic species of lead. Hydroxide, carbonate, sulfide and sulfate may act as solubility controls in precipitating lead from water. The amount of lead that remains in solution depends upon the pH of the water and the dissolved salt content. Lead is more soluble in softer water and low pH water (ATSDR, 2005a). Complexation of lead with humic acids can increase the mobility of lead in groundwater and/or surface water but will also reduce the bioavailability to biota.

A significant fraction of lead carried by surface water is expected to be in an undissolved form, which can consist of colloidal particles or lead compounds incorporated in other components of surface particulate matter from runoff. Lead may occur as sorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living or nonliving organic matter in water. The ratio of lead in suspended solids to lead in dissolved form ranges from 4:1 to 27:1 (ATSDR, 2005a).

Most lead in soil is retained there and very little is transported into surface water or groundwater (ATSDR, 2005a). The fate of lead in soil is affected by the adsorption at mineral surfaces, the precipitation of sparingly soluble solid forms of the compound, and the formation of relatively stable organic-metal complexes or chelates with soil organic matter. The mobility of lead increases in environments having low pH due to the enhanced solubility of lead under acidic conditions (ATSDR, 2005a). Lead may be immobilized by ion exchange with hydrous oxides or clays or by chelation with humic or fulvic acids in soil (Olson and Skogerboe, 1975). The downward movement of elemental lead and inorganic lead compounds from soil to groundwater by leaching is very slow under most natural conditions except for highly acidic situations (Boggess, 1977). The conditions that induce leaching are the presence of lead in soil at concentrations that either approach or exceed the cation exchange capacity of the soil, the presence of materials in soil that are capable of forming soluble chelates with lead, and a decrease in the pH of the leaching solution (e.g., acid rain) (Boggess, 1977).

The mobility of lead increases in environments having low pH due to the enhanced solubility of lead under acidic conditions. Because precipitation in northeastern Alabama is expected to be acidic in nature, it could be concluded that the mobility of lead in soil would be enhanced. However, elevated concentrations of lead in subsurface soil samples and groundwater at the Former Choccolocco Corridor Ranges are not prevalent, indicating that lead is not mobile in the environment at the Former Choccolocco Corridor Ranges. ATSDR (2005) also suggests that the

fate of lead in soil is affected by the adsorption at mineral surfaces, the precipitation of sparingly soluble solid forms of the compound, and the formation of relatively stable organic-metal complexes or chelates with soil organic matter. Due to the relatively low organic content of the soils at the Former Choccolocco Corridor Ranges, the formation of organic-metal complexes with soil organic matter is likely a minor fate process at the Former Choccolocco Corridor Ranges. The most likely processes affecting the fate of lead in soil are the adsorption of lead at mineral surfaces and the speciation of lead to more insoluble sulfate, sulfide, oxide, and phosphate salts.

Plants and animals may bioconcentrate lead, but biomagnification is not expected. Although the bioavailability of lead in soil to plants is limited because of the strong adsorption of lead to soil organic matter, the bioavailability increases as the pH and the organic matter content of the soil are reduced. Lead may be taken up in edible plants from the soil via the root system, by direct foliar uptake and translocation within the plant, and by surface deposition of particulate matter. The amount of lead in soil that is bioavailable to most plants depends on factors such as cation exchange capacity, pH, amount of organic matter present, soil moisture content, and type of amendments added to the soil (ATSDR, 2005a). Low alkalinity and low pH conditions in soils can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower trophic level animals, and plants (Jenkins, 1981).

Most lead does not appear to significantly bioaccumulate in most fish. However, bioaccumulation of tetraethyl lead can occur in aquatic organisms (ATSDR, 2005a). Plants commonly take up lead from soil and, therefore, may return it upon decomposition. Because the bioavailability of lead is dependent upon site-specific conditions, the accuracy of the ecological assessment of lead depends heavily on site-specific tests of bioavailability and subsequent toxicity and accumulation.

4.4.4 Mercury

Mercury is a metal that occurs in three forms: elemental, cationic, and complexed in organic compounds (ATSDR, 1999a). Natural mercury occurs combined with other constituents in various minerals, although some ores contain elemental (uncombined) mercury (NLM, 2007). Elemental or metallic mercury has a valence of 0 and occurs as a liquid at room temperature. This form of mercury has a tendency to volatilize from environmental media. Inorganic compounds of mercury exist as salts in which mercury exists as a cation with a valence of +1 or +2. Both elemental mercury and mercury salts are inorganic forms of mercury. Methylmercury forms naturally in water from the bioconversion of inorganic forms of mercury (ATSDR, 1999a; NLM, 2007). Methylmercury occurs as a cation with a valence of +1 (+Hg-CH3), which may

associate with any of several common anions in environmental media. Methylmercury has a low tendency to volatilize from environmental media.

Mercury participates in a global bio-geochemical cycle that includes natural as well as anthropogenic sources (ATSDR, 1999a; NLM, 2007). Briefly, the cycle can be considered to start with natural emissions of elemental mercury vapor from soil and surface water, as well as fugitives and emissions from anthropogenic sources, to the atmosphere. Mercury is then deposited from the air to soil and surface water, generally in the form of elemental mercury or cationic mercury (ATSDR, 1999a; NLM, 2007). Some of the deposited elemental mercury volatilizes again to the atmosphere; some is oxidized by microbes in the soil or water to the cationic form. Most of the cationic mercury in soil and surface water remains in that form. Very small amounts of cationic mercury in soil may be converted by microbes to organic mercury (methylmercury). A somewhat larger amount of cationic mercury in surface water may undergo microbial conversion to various organic forms, of which methylmercury predominates and is best understood. Simultaneously, competing microbial activity reduces methylmercury and cationic mercury in soil and surface water to elemental mercury, which is freely emitted to the atmosphere. Whether production of methylmercury or reduction to elemental mercury predominates depends on factors that regulate microbial activity, including ambient temperature, concentration of organic matter, concentration of sulfur, redox potential and pH.

Mercury compounds in moist soil environments may dissociate depending upon their solubility. Less soluble compounds bind strongly to humic matter and are practically immobile in soil. Mercuric sulfide has been found to act somewhat as a sink for mercury in soil and sediment. Acidification of soil strengthens the sorption of mercury compounds to humic constituents in soil. Mercury compounds are not expected to volatilize from moist or dry soil surfaces. Inorganic mercury compounds can be methylated by microorganisms indigenous to soil under both aerobic and anaerobic conditions; however, the methylation rate is generally considered to be quite low (EPA, 2005d) and the process is balanced by microbial processes that reduce inorganic cationic mercury and methylmercury to elemental mercury, which is free to volatilize from soil.

Volatilization is very unlikely to be a significant process of removing mercury compounds from water (ATSDR, 1999a; NLM, 2007). The solubility of mercury compounds in water varies greatly. Elevated chlorine concentrations in water (i.e., ocean water and other saline conditions) greatly increase the solubility of mercury compounds. Mercury compounds tend to bind strongly to organic matter in water and may settle to the bottom or be transported in runoff water to other surface waters and soils. Cationic mercury can be methylated by microorganisms indigenous to

the aqueous environment to methylmercury or, to a lesser extent to other organic compounds including dimethylmercury. Methylmercury binds strongly to organic matter in the dissolved or finely divided particulate form. It enters the foodchain at the zooplankton stage and bioaccumulates and biomagnifies as it moves up the trophic levels of the aquatic food chain. Inorganic mercury does not bioaccumulate or biomagnify.

The source of mercury at the Former Choccolocco Corridor Ranges is unknown. One possible source of mercury could be switches used to control electronic targets at the Former Choccolocco Corridor Ranges. However, there is no documentation to verify the presence of electronic-controlled targets at the Former Choccolocco Corridor Ranges. The analyses conducted on soil samples from the Former Choccolocco Corridor Ranges were for total recoverable mercury; therefore, the form of mercury present is not known. However, in soils, mercury can exist in the mercuric (Hg+2) and mercurous (Hg+1) states as a number of complex ions with varying water solubilities (ATSDR, 1999). The dominant process controlling the distribution of mercury compounds in the environment appears to be the sorption of nonvolatile forms to soil and sediment particulates (Bryan and Langston, 1992). Adsorption of mercury in soil is decreased with increasing pH and/or chloride ion concentrations (Schuster, 1991). Mercury is sorbed to soil with high iron and aluminum content up to a maximum loading capacity of 15,000 ppm (Ahmad and Qureshi, 1989). Inorganic mercury sorbed to particulate material is not readily desorbed. Thus, freshwater and marine sediments are important repositories for inorganic forms of mercury, and leaching is a relatively insignificant transport process in soils. Metallic mercury may move through the top 3-4 cm of dry soil at atmospheric pressure; however, it is unlikely that further penetration would occur (Eicholz, et al., 1988).

If mercury were ever present at the Former Choccolocco Corridor Ranges in the metallic form, it would have volatilized relatively rapidly, been transported via vapor transport, and deposited in association with dry deposition of particulate mater. Due to the historic nature of the soil contamination found at the Former Choccolocco Corridor ranges, volatilization of mercury is presently not an important transport process at these ranges.

It is well known that methylmercury is capable of substantial biomagnification in aquatic systems, but the same does not appear to hold for inorganic mercury in terrestrial food chains (ATSDR, 1999a). Mushrooms have been shown to bioconcentrate mercury, but higher plants such as peas, corn or wheat do not, even when the soil is amended with sewage sludge containing high levels of mercury. Although the concentration of mercury in the roots may reflect that of the surrounding soil, the above-ground parts of the plants showed no elevation in mercury levels.

Recent evaluations (ATSDR, 1999a) indicate that upwards of 99 percent of the mercury detected in fish tissue exists as methylmercury; inorganic forms generally are not detected. This suggests that methylmercury is bioaccumulated or biomagnified far more efficiently than inorganic forms, and that the inorganic forms contribute insignificantly to mercury levels in fish. Therefore, of the various forms of mercury potentially present in surface water, only methylmercury is evaluated for the indirect pathway involving uptake in fish followed by human ingestion of fish tissue.

Empirical data suggest great variability in the BCF for methylmercury (ATSDR, 1999a; NLM, 2007). Generally, higher BCFs are obtained for piscivorous fish at the higher trophic levels in the food chain, reflecting the biomagnification mentioned above.

4.4.5 Zinc

Zinc occurs in the environment mainly in the +2 oxidation state. Sorption is the dominant reaction, resulting in the enrichment of zinc in suspended and bed sediments. Zinc in aerobic waters is partitioned into sediment through sorption onto hydrous iron and manganese oxides, clay minerals, and organic material. The efficiency of these materials in removing zinc from solution varies according to their concentrations, pH, redox potential, nature and concentration of complexing ligands, cation exchange capacity, and the concentration of zinc (ATSDR, 2005b). Similar to copper, zinc is complexed at high pHs and can exist as freely-dissolved divalent cations at lower pHs, thus enhancing its bioavailability. Therefore, as the pH of the water decreases, the concentration of zinc ions in the water phase increases at the same rate as that of the release of zinc from the sediment. In anaerobic environments and in the presence of sulfide ions, precipitation of zinc sulfide limits the mobility of zinc. In most waters, zinc exists primarily as the hydrated form of the divalent cation. However, the metal often forms complexes with a variety of organic and inorganic ligands (ATSDR, 2005b). In aquatic environments, zinc partitions to sediments or suspended solids in surface waters through sorption onto hydrous iron and manganese oxides, clay minerals, and organic material.

In general, zinc sorbs strongly onto soil particles. The mobility of zinc in soil depends on the solubility of the speciated forms of the element and on soil properties such as cation exchange capacity, pH, redox potential, and chemical species present in the soil; under anaerobic conditions, zinc sulfide is the controlling species (Kalbasi et al., 1978). Since zinc sulfide is insoluble, the mobility of zinc in anaerobic soil is low. The mobility of zinc in soil increases at lower soil pH under oxidizing conditions and at lower cation exchange capacity of soil (Tyler and McBride, 1982). Distribution constants for zinc in soil range widely from 0.1 to 8,000 L/kg

(Baes and Sharp, 1983). Zinc in soluble form (e.g. zinc sulfate) is moderately mobile in most soils; however, the mobility is limited by a slow rate of dissolution. Consequently, movement towards groundwater is expected to be slow unless the zinc in the soil is in the soluble form or is accompanied by corrosive substances (e.g. mine tailings). Zinc in soil at the CC Ranges is not in the soluble form or accompanied by corrosive substances; therefore, zinc primarily remains in recalcitrant, immobile forms at the CC Ranges.

Zinc is an essential nutrient that is present in all organisms. Although biota appears to be a minor reservoir of zinc relative to soils and sediments, microbial decomposition of biota in water can produce ligands, such as humic acids, that can affect the mobility of zinc in the aquatic environment through zinc precipitation and adsorption (ATSDR, 2005b). Zinc can accumulate in freshwater animals at 51 to 1,130 times the concentration present in water (EPA, 1987a). In general, zinc does not biomagnify through food chains. Furthermore, although zinc bioaccumulates to some degree in aquatic systems, biota appears to represent a relatively minor sink compared to sediments. Steady-state zinc BCF for 12 aquatic species range from 4 to 24,000, with most being less than 100 (EPA, 1987a). With respect to bioconcentration from soil by terrestrial plants, invertebrates, and mammals, BCFs of 0.4, 8, and 0.6, have been reported, respectively. In general, plants do not concentrate zinc above levels present in the soil (ATSDR, 2005b).

4.4.6 4,4'-DDD, -DDE, and DDT

DDT, DDD and DDE released to soil are subject to volatilization, strong absorption to soil particles, chemical transformation, and uptake by vegetation (ATSDR, 2002b; NLM, 2003). Organic carbon partition coefficients (K_{oc}) of 1.5×10^5, 5×10^4, and 1.5×10^5 reported for *p,p'*-DDT, *p,p'*-DDE, and *p,p'*-DDD, respectively, suggest that these compounds adsorb strongly to soil. These chemicals are only slightly soluble in water, with water solubilities of 0.025, 0.12, and 0.09 mg/L for *p,p'*-DDT, *p,p'*-DDE, and *p,p'*-DDD, respectively (Howard and Meylan, 1997). Since these compounds are bound strongly to soil, DDT and its metabolites would remain in the surface layers of the soil and not leach into groundwater. The potential for volatilization will be seriously attenuated by adsorption. Photo-oxidation accounts for some destruction of DDT residues on the surface of soil, but hydrolysis is not expected to occur. Biodegradation of DDT yields DDE under aerobic conditions and DDD under anaerobic conditions. DDE is more persistent than DDT, so that the ratio of DDE to DDT in soil increases with time. Strong adsorption is expected to prevent leaching to groundwater; however, DDT residues have been identified in groundwater, possibly because of the co-solvent effects of other organic compounds.

Despite being strongly bound to soil, at least a portion of DDT, DDE, and DDD is bioavailable to plants and soil invertebrates. Verma and Pillai (1991) reported that grain, maize, and rice plants accumulate DDT adsorbed to soil. Most of the residues were found in the roots of the plant, and the lowest concentration of DDT residues was found in the shoots, indicating low translocation of DDT. This low translocation characteristic would tend to minimize exposures to herbivorous and omnivorous animals that feed on above-ground portions of plants.

Sorption to suspended sediment is probably the most important removal process for DDT residues released to water (ATSDR, 2002b; NLM, 2003). Volatilization of DDT, DDE, and DDD is known to account for considerable losses of these compounds from soil surfaces and water. Their tendency to volatilize from water can be predicted by their respective Henry's Law constants, which for the respective *p,p'*- and *o,p'*- isomers are 8.3×10^{-6}, 2.1×10^{-5}, 4.0×10^{-6}, 5.9×10^{-7}, 1.8×10^{-5}, and 8.2×10^{-6} atm-m^3/mol (Howard and Meylan, 1997). Sorption greatly reduces the potential for volatilization. DDT undergoes base-catalyzed hydrolysis with a half-life of approximately 81 days at pH 9; the product formed in the hydrolysis reaction is DDE. Hydrolysis of DDE and DDD is not a significant environmental fate process. The hydrolysis rate of DDT under acidic conditions is very slow, with a reported half-life of 12 years. DDT residues are subject to photodegradation, but the process is likely to be very slow for all residues except DDE (ATSDR, 2002b). Biodegradation in water and sediment may be significant over time, but the process is expected to be slow.

DDT, DDE and DDD are among the pesticides and pesticide residues that bioaccumulate in food chain pathways. DDT, DDE and DDD are highly lipid soluble, as reflected by their log octanol-water partition coefficients (log K_{ow}) of 6.91, 6.51, and 6.02, respectively for the *p,p'*- isomers and 6.79, 6.0, and 5.87, respectively, for the *o,p'*- isomers (Howard and Meylan, 1997). This lipophilic property, combined with an extremely long half-life is responsible for its high bioconcentration potential in aquatic organisms (ATSDR, 2002b).

4.4.7 Gamma-Chlordane

The physical and chemical properties of technical chlordane are difficult to specify since there are many components in the technical mixture, one of which is gamma-chlordane. The estimated octanol-water partition coefficient (log K_{ow}) for pure chlordane is 5.54 (ATSDR, 1994), indicating a strong tendency to sorb to fatty tissues in organisms that are exposed to chlordane. The estimated organic carbon partition coefficient (log K_{oc}) for chlordane ranges from 3.49 to 4.64 (ATSDR, 1994), indicating a relatively strong affinity to sorb to organic materials in soil and surface water. In soil, chlordane adsorbs to the organic matter and volatilizes slowly over time. Chlordane does not leach significantly. In general, chlordane remains in the top 20 cm of

most soils and for some soils, it stays at this level for over 20 years (Beeman and Matsumura, 1981). Chlordane in soil may persist for long periods of time (ATSDR, 1994; NLM, 2003). One review of chlordane soil persistence reported that the mean degradation rate of chlordane in soil under field conditions ranges from 4.05 to 28.33 percent per year. Another literature review reported the mean half-life of chlordane under field conditions to be 3.3 years. A third study found chlordane levels in the top 20 cm of most soils to remain largely unchanged for 20 years. In general, sandy soils and soils with small amounts of organic matter retain chlordane less than soils with high clay and/or organic content (Wiese and Basson, 1966). Soil moisture, however, is the most important factor, with chlordane volatilizing mush faster from moist soils than from dry soils. Chlordane is expected to be generally immobile or only slightly mobile in soil. However, the detection of chlordane in groundwater at several locations indicates that leaching can occur. The presence of other organic solvents, such as used in vehicles for pesticide application, can greatly increase the mobility of chlordane in soil. Chlordane can volatilize significantly from soil surfaces on which it has been sprayed, particularly moist soil surfaces; however, shallow incorporation into soil greatly restricts volatilization losses. Sufficient data are not available to predict the biodegradation rate of chlordane in soil; however, it is likely that biodegradation is very slow and is probably not a significant removal mechanism.

Chlordane released to water is not expected to undergo significant hydrolysis, oxidation, or direct photolysis (ATSDR, 1994; NLM, 2003). Chlordane is expected to volatilize significantly from the water column to the atmosphere. Adsorption to sediment, however, significantly attenuates volatilization. Adsorption to sediment is expected to be the most significant fate process for chlordane in surface water. Adsorption to suspended sediment results in widespread dispersion. Chlordane is very persistent in benthic sediment.

Chlordane is among a number of pesticides and pesticide residues that bioaccumulate in food chain pathways and are of special concern for biomagnification from sediment in benthic fish.

4.4.8 Endrin

Endrin tends to persist in the environment mainly in forms sorbed to sediments and soil particles. Endrin released to soil is expected to be essentially immobile. Endrin is extremely persistent when released to the soil. It adsorbs strongly to soil particles and tends to be immobile, based on an estimated organic carbon partition coefficient (K_{oc}) of 34,000 (Kenaga, 1980). Volatilization from moist soil surfaces is expected to be the most important removal process for endrin; however, adsorption may attenuate this process. In spite of its low vapor pressure, endrin has been found to volatilize significantly (20 – 30 %) from soils within days after application (Nash, 1983). The volatilization half-life of endrin from a sugarcane soil was reported to be 63 days.

Endrin appears to be resistant to biodegradation in most soils. The half-life of endrin in soil has been reported to range from 4 to 8 years. Degradation of endrin in soils under field conditions is not a significant fate process with a half-life on the order of 14 years (Nash and Woolson, 1967). Since endrin in solid form is hydrophobic and sorbs strongly to soil particles, migration into groundwater would not generally be expected (ATSDR, 1996). The presence of significant concentrations of endrin transformation products (including endrin ketone, endrin alsdehyde, and endrin alcohol) in a variety of plants grown in soil treated with endrin for periods as long as 16 years prior to planting indicates that there may be significant uptake of endrin and/or its transformation products by plants from endrin-treated soil (Beall, et al., 1972).

Endrin is expected to adsorb to suspended solids and sediment in water (NLM, 2004). When released to water, endrin strongly adsorbs to sediment (Kenaga, 1980) as indicated by its high log K_{oc} values (4.53) and log K_{ow} values (5.34 – 5.6), and bioconcentrates significantly in aquatic organisms. Typical bioconcentration factors (BCFs) for freshwater and marine organisms range from 80 to 49,000. Endrin appears to be biomagnified only slightly through various levels of the food chain (Metcalf, et al., 1973). Volatilization from water surfaces could potentially be a significant removal process for endrin; however, volatilization is expected to be severely attenuated by adsorption to suspended solids and sediment in the water column. Based on its very small calculated Henry's Law constant of $4.0 \times 10^{-7} - 5.4 \times 10^{-7}$ atm-m^3/mol, and its strong adsorption to sediment particles, endrin would be expected to partition very little from water into air (Thomas, 1990). The hydrolysis rate of endrin is very slow, with an estimated half-life of 12.8 years at pH 7. Less than 20 percent degradation of endrin was observed when it was incubated in water obtained from a drainage canal in Ontario, Canada over 16 weeks, suggesting that biodegradation in water is very slow. Under real world conditions, endrin released to surface waters would not be expected to biodegrade or hydrolyze to any significant extent (Eichelberger and Lichtenberg, 1971). Photoisomerization of endrin to endrin ketone may be a major transformation process (ATSDR, 1996).

Although bioaccumulation into terrestrial food chains has been observed, endrin is rapidly metabolized and eliminated by mammals, greatly reducing the potential for accumulation (ATSDR, 1996; EPA, 1987b). Biomagnification upwards through aquatic trophic levels is unlikely (ATSDR, 1996). EPA (1995) identifies endrin as a chemical that is not of concern for bioaccumulation in aquatic systems.

5.0 Ecosystems Potentially at Risk

The Former Choccolocco Corridor Ranges consist of the following four sets of ranges and associated parcels:

- Former Range 40 (Parcel 94Q); and Range, Choccolocco Corridor (Parcel 146Q);
- Former Range 41 (Parcel 95Q); and Impact Area (Parcel 131Q-X);
- Former Range 42 (Parcel 96Q); Range, Choccolocco Corridor (Parcel 145Q-X); and Impact Area, Choccolocco Corridor (Parcel 148Q-X); and
- Former Range 43 (Parcel 97Q); Range, Choccolocco Corridor (Parcel 144Q-X); and Impact Area, Choccolocco Corridor (Parcel 147Q-X).

The Choccolocco Corridor Ranges, Parcels 94Q, 95Q, 96Q, 97Q, and Associated Parcels are located west of the Choccolocco Mountains in the Choccolocco Corridor near the eastern boundary of the FTMC Main Post and were part of the Range 40 complex. The total area of all sites in the CC range complex is approximately 230 acres. The location of the Choccolocco Corridor ranges is shown in Figure 5-1.

Former Range 40 (Parcel 94Q) and Range, Choccolocco Corridor (Parcel 146Q) overlap and comprise an area of approximately 125 acres. These two ranges slope gently from west to east with the elevation of these two parcels ranging from approximately 1,000 feet above mean sea level (msl) in the western-most area to approximately 775 feet above msl in the eastern-most area. There are two small ephemeral drainage features that run generally west-to-east through Parcel 146Q. These drainage features are narrow (less than 3 feet wide) and shallow (less than 1 foot deep). They exhibit vegetation characteristic of the upland vegetation of the surrounding forest, indicating they are dry for the majority of the time and water is only present during short periods of high precipitation. The habitat is best described as mixed deciduous/coniferous forest. A significant portion of the forest in the central part of Parcel 94Q was clear-cut by the Alabama State Forestry Commission due to southern pine beetle infestation.

Former Range 41 (Parcel 95Q) and Impact Area (Parcel 131Q-X) comprise an area of approximately 12 acres. These two areas slope gently from west to east with the elevation of these two parcels ranging from approximately 850 feet above mean sea level (msl) in the western-most area to approximately 750 feet above msl in the eastern-most area. The most distinguishing feature of Former Range 41 is the lack of vegetation over most of its area. Longleaf pine and loblolly pine saplings have begun to colonize this area, but large potions of

this former range remain almost entirely denuded. The eastern-most portion of Parcel 95Q and the majority of Parcel 131Q-X are characterized by a relatively immature mixed deciduous/coniferous forest. A small perennial stream runs west-to-east along the northern boundary of Parcels 95Q and 131Q-X. The stream is less than 6 inches deep in most places and less than 3 feet wide. This perennial stream is situated in a 6-8 feet deep erosional channel with a substrate of mostly cobbles with small areas of sand, gravel, and leaf litter.

The Former Range 42 (Parcel 96Q) is approximately 23.8 acres in size. Range, Choccolocco Corridor (Parcel 145Q-X) is approximately 44 acres in size and overlaps the Former Range 42. Impact Area, Choccolocco Corridor (Parcel 148Q-X) lies within Parcels 96Q and 145Q-X and occupies approximately 5.8 acres. These parcels slope from west to east with the elevations ranging from approximately 1,200 feet above mean sea level (msl) in the western-most area to approximately 750 feet above msl in the eastern-most area. The same perennial stream that runs adjacent to the northern boundary of Parcel 95Q-X runs in a northwest–to-southeast direction across the western portion of Parcels 145Q-X and 96Q. This stream is less than 6 inches deep in most places and less than 3 feet wide with a substrate of mostly cobbles and small areas of sand, gravel, and leaf litter. The habitat at these parcels is best characterized as mixed deciduous/coniferous forest. However, large portions of this area have been clear-cut by the Alabama State Forestry Commission due to southern pine beetle infestation. The areas that were clear-cut are now in an oldfield successional state.

The study area for the Former Range 43 (Parcel 97Q) is approximately 9 acres. Range, Choccolocco Corridor (Parcel 144Q-X) is approximately 19 acres in size and encompasses about two-thirds of Parcel 97Q and most of Impact Area, Choccolocco Corridor (Parcel 147Q-X). Impact Area, Choccolocco Corridor (Parcel 147Q-X) has been identified as an impact area and is approximately 3 acres in size. These parcels slope from west to east with the elevations ranging from approximately 1,075 feet above mean sea level (msl) in the western-most area to approximately 750 feet above msl in the eastern-most area. Two small ephemeral drainage features run in a west-to-east direction across these parcels. These drainage features are narrow (less than 3 feet wide) and shallow (less than 1 foot deep). They exhibit vegetation characteristic of the upland vegetation of the surrounding forest, indicating they are dry for the majority of the time and water is only present during short periods of high precipitation. In fact at the time of the site visit, these drainage features contained significant leaf litter and exhibited signs that indicated they had not conducted water for several years. The habitat is best described as a relatively mature mixed deciduous/coniferous forest.

The terrestrial habitat at the Choccolocco Corridor Ranges generally consists of oldfield and mixed deciduous / coniferous forest. The areas of oldfield habitat are present in areas that have been clear-cut by the Alabama State Forestry Commission to address southern pine beetle infestation. Since the clear-cutting was performed, pioneer species have colonized these areas. Typically, the pioneer species most likely to colonize these types of areas are the "weed" species that tend to be vigorous pioneer plants that grow and spread rapidly. The first of the pioneer species to invade these clear-cut areas are the grasses and other herbaceous species. These clear-cut areas are classified as being in an early oldfield successional state. Over time, the grass and other herbaceous species will be followed by shrubs, vines, and small trees. The early oldfield successional areas at the Choccolocco Corridor Ranges are dominated by various grasses and herbs, including *Rumex spp.* (dock), *Trifolium spp.* (clover), *Astragalus spp.* (vetch), *Ascelepias spp.* (milkweed), *Galium spp.* (bed straw), *Chrysanthemum leucanthemum* (ox-eye daisy), and *Sorghum halepense* (Johnson grass). Other oldfield herbaceous species occurring at the Choccolocco Corridor Ranges are *Rubus occidentalis* (black raspberry), *Toxicodendron radicans* (poison ivy), *Rubus glabra* (smooth sumac), *Smilax rotundiflora* (green brier), *Lonicera japonica* (Japanese honeysuckle), *Vitus labrusca* (fox grape), and *Rosa multiflora* (multiflora rose). Scrub pine (*Pinus virginiana*) and loblolly pine (*Pinus taeda*) saplings have also begun to encroach on these cleared areas.

The portions of the Choccolocco Corridor Ranges that have not been clear-cut are characteristic of a mixed deciduous/coniferous forest. The canopy species typically found in this forest type at FTMC include shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), longleaf pine (*Pinus palustris*), yellow poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), white oak (*Quercus alba*), and northern red oak *(Quercus rubra*). The dominant understory species of this forest type are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by chokeberry (*Aronia arbutifolia*), mountain laurel (*Kalmia latifolia*), Piedmont azalea (*Rhododendron canescens*), southern low blueberry *(Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), and poison ivy (*Toxicodendron radicans*). The herbaceous layer consists mainly of foxglove (*Aureolaria pectinata*), little bluestem (*Schizachyrium scoparium*), narrow-leaved sensitive brier (*Schrankia microphylla*), poison oak (*Toxicodendron toxicarium*), goat's rue (*Tephrosia virginiana*), and pencil flower (*Stylosanthes biflora*).

Typical terrestrial species that may inhabit the Choccolocco Corridor Ranges include opossum, shorttail shrew, raccoon, white-tail deer, red fox, coyote, gray squirrel, striped skunk, a number of species of mice and rats (e.g., white-footed mouse, eastern harvest mouse, cotton mouse, eastern woodrat, and hispid cotton rat), and eastern cottontail. Approximately 200 avian species reside at FTMC at least part of the year (USACE, 1998). Common species expected to occur in the vicinity of the Choccolocco Corridor Ranges include northern cardinal (*Cardinalis cardinalis*), northern mockingbird (*Mimus polyglottus*), warblers *(Dendroica spp.*), indigo bunting (*Passerina cyanea*), red-eyed vireo (*Vireo olivaceus*), American crow (*Corvus brachyrhynchos*), bluejay (*Cyanocitta cristata*), several species of woodpeckers (*Melanerpes spp., Picoices spp.*), and Carolina chickadee (*Parus carolinensis*). Game birds present in the vicinity of the Choccolocco Corridor Ranges may include northern bobwhite (*Colinus virginianus*), mourning dove (*Zenaida macroura*), and eastern wild turkey (*Meleagris gallopavo*). A variety of woodland hawks (e.g., sharp-shinned hawk) and other raptors (e.g., red-tailed hawk, barred owl, and great horned owl) are expected to use this area for a hunting and/or nesting area.

Several small, ephemeral drainage features conduct surface runoff from the Choccolocco Corridor Ranges generally in a west-to-east direction. One perennial stream drains in the same west-to-east direction across Parcel 145Q-X, Parcel 96Q, Parcel 95Q, and Parcel 131Q-X. When water is present in these small drainage features, they are narrow (2 to 3 feet wide) and shallow (3 to 6 inches deep). The substrate is mostly cobbles and gravel with small depositional areas of sand and leaf litter, interspersed throughout. The small size and ephemeral nature of most of these drainage features preclude the presence of most fish species and other animals that might prey on fish (piscivores); however, semi-aquatic species (amphibians) could occur in these drainage features and drought-tolerant fish species could occur during periods of high precipitation. The one perennial stream could potentially support small fish species and other small aquatic species. The small size of the perennial stream would preclude the presence of larger fish species. Bullfrog *(Rana catesbeiana*) and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in the small ephemeral and perennial streams. Fish species that may be found in the perennial stream at the CC ranges include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*), and various darters (*Etheostoma spp.*). Larger fish species are not expected to inhabit these drainage features due to habitat restrictions.

In general, the terrain at FTMC supports large numbers of amphibians and reptiles. Jacksonville State University has prepared a report titled *Amphibians and Reptiles of Fort McClellan, Calhoun County, Alabama* (Cline and Adams, 1997). The report indicated that surveys in 1997

found 16 species of toads and frogs, 12 species of salamanders, 5 species of lizards, 7 species of turtles, and 17 species of snakes. Typical inhabitants of the area surrounding the CC Ranges are copperhead (*Agkistrodon contortix*), king snake (*Lampropeltis getulus*), black racer (*Coluber constrictor*), fence lizard (*Sceloporour undulatus*), and six-lined racerunner (*Cnemidophorous sexlineatus*).

The Federally listed species that have the potential to occur in the vicinity of the Former Choccolocco Corridor Ranges are the gray bat and the blue shiner (Garland, 1996). The drainage features at the CC ranges have not been classified for their potential to support foraging habitat for the gray bat (Garland, 1996). Additionally, gray bat surveys have not been conducted on the small drainage features at the CC ranges. However, mist net surveys conducted along Choccolocco Creek have resulted in the capture of gray bats (Garland, 1996). Therefore, the potential exists for gray bats to use the drainage features at the CC ranges for foraging; however, the ephemeral nature of most of these drainage features reduces the potential for gray bat foraging to a minimal level.

Within Alabama, the blue shiner (*Cyprinella caerules*) is currently restricted to Weogufka and Choccolocco Creeks, and the lower reaches of the Little River. Within Choccolocco Creek, the blue shiner is limited to about 22.6 km of the main channel and the lower reaches of Shoal Creek. Habitat preference on Choccolocco Creek appears to be slack to slow current over sand or a mixture of boulders, cobbles, and sand. Lateral pools away from the main run of the stream and backwaters with sand substrate were ideal habitat for blue shiners (Pierson and Krotzer, 1987). Blue shiners appear to be intolerant of high turbidity and are probably a mid-depth feeder competitively dependent on high visibility (Mount, 1986). The main channel of Choccolocco Creek provides optimal habitat for the blue shiner (Pierson and Krotzer, 1987). However, the small drainage features in the vicinity of the Former Choccolocco Corridor Ranges are small and mostly ephemeral; therefore, they likely do not provide suitable habitat for blue shiner.

6.0 Complete Exposure Pathways

For exposure to occur, a complete exposure pathway must exist between the contaminant and the receptor. A complete exposure pathway requires the following four components:

- A source mechanism for contaminant release;
- A transport mechanism;
- A point of environmental contact; and
- A route of uptake at the exposure point (EPA, 1989).

If any of these four components is absent, then a pathway is generally considered incomplete. Potentially complete and incomplete exposure pathways at the Former Choccolocco Corridor Ranges are depicted in the conceptual site model (CSM) shown on Figure 6-1.

Ecological receptors may be exposed to constituents in soils via direct and/or secondary exposure pathways. Direct exposure pathways include soil ingestion, dermal absorption, and inhalation of volatile constituents and constituents adsorbed to fugitive dust. Significant exposure via dermal contact is limited to organic constituents that are lipophilic and can penetrate epidermal barriers. Mammals are less susceptible to exposure via dermal contact with soils because their fur prevents skin from coming into direct contact with soil. However, soil ingestion may occur while grooming, preening, burrowing, or consuming plants, insects, or invertebrates resident in soil.

Exposures to site-related constituents in surface water and sediment are expected to be minimal due to the ephemeral nature of most of the drainage features present at the Former Choccolocco Corridor Ranges. However, the one perennial stream that drains the Former Range 41 (Parcel 95Q), Impact Area (Parcel 131Q-X), and Former Range 42 (Parcel 96Q) at the Former Choccolocco Corridor Ranges does provide a significant potential for exposure to surface water and sediment. Ecological receptors could be exposed to constituents in surface water via direct contact or through consumption of water. Constituents present in sediment may result from erosion or adsorption of water-borne constituents onto sediment particles. If sediment is present in an area that is periodically inundated with water, then previously described exposure pathways for soils would be applicable during dry periods. Water overlying sediments prevents constituents from being carried by wind erosion. Because the majority of the constituents detected in sediment are not prone to volatilization because they are inorganic compounds or organic compounds that tend to bind tightly to sediment, volatilization from sediments is not an important fate mechanism at this site. Volatile compounds were detected in sediment samples

albeit at very low concentrations. Therefore, inhalation of constituents originating from the sediment is not a significant exposure pathway. Exposure via dermal contact may occur, especially for benthic organisms and wading birds or other animals that may use the perennial stream as a feeding area. Some aquatic organisms consume sediment and ingest organic material from the sediment. Inadvertent ingestion of sediments may also occur as the result of feeding on benthic organisms and plants.

Exposure via inhalation of fugitive dust is limited to contaminants present in surface soils at areas that are devoid of vegetation or other cover material. The amount of vegetative cover, the inherent moisture content of the soil, and the frequency of soil disturbance play important roles in the amount of fugitive dust generated at a particular site. A large majority of the area of the Former Choccolocco Corridor Ranges is forested and covered with vegetation; therefore, the amount of fugitive dust generated in these vegetated areas is expected to be minimal. The areas that have been clear-cut due to southern pine beetle infestation (Former Ranges 40 and 42) have been re-vegetated with pioneer species and are currently best categorized as oldfield/ successional habitats. The high degree of vegetation cover on the ground greatly reduces the potential for dust generation. Therefore, inhalation of fugitive dust is not considered a significant pathway at these sites.

Secondary exposure pathways involve constituents that are transferred through different trophic levels of the food chain and may be bioaccumulated and/or bioconcentrated. This may include constituents bioaccumulated from soil into plant tissues or into terrestrial species ingesting soils. These plants or animals may, in turn, be consumed by animals at higher trophic levels.

In general, the metals and semi-volatile organic compounds detected in surface soil at the Former Choccolocco Corridor Ranges may bioaccumulate in lower trophic level organisms (i.e. terrestrial invertebrates may bioaccumulate inorganic compounds detected in soil); however, they will not bioconcentrate through the food chain. The pesticides and herbicides detected in surface soil and sediment have the potential to bioaccumulate and/or bioconcentrate. Therefore, food chain exposures for these constituents at the Former Choccolocco Corridor Ranges could be significant.

Potential ecological receptors at the CC ranges fall into two general categories: terrestrial and aquatic. Within these two general categories there are several major feeding guilds that could be expected to occur at the CC ranges: herbivores, invertivores, omnivores, and carnivores. All of these feeding guilds have the potential to be directly exposed to various combinations of surface soil at the CC ranges and sediment in the perennial and ephemeral drainage features via various

activities (e.g., feeding, drinking, grooming, bathing, etc.). These feeding guilds may also be exposed to site-related chemicals via food web transfers. Since no COPECs were identified in groundwater, exposures to this environmental medium at the CC ranges is considered incomplete.

As discussed above, ingestion of COPECs in soil and sediment are the pathways that pose the greatest potential for exposure for ecological receptors at the CC ranges. Dermal absorption and inhalation exposures are expected to be insignificant. Food web transfers of COPECs are also possible exposure pathways for ecological receptors at the CC ranges, particularly for the pesticide COPECs.

Potentially complete exposure pathways are depicted in the CSM as presented in Figure 6-1 and are described in the following sections for the various feeding guilds.

6.1 Herbivorous Feeding Guild

The major route of exposure for herbivores is through ingestion of plants that may have accumulated contaminants from the soil. Since terrestrial herbivores by definition are grazers and browsers, they could be exposed to chemicals that have accumulated in the vegetative tissues of the plants at the site. Terrestrial herbivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while grazing, grooming, or other activities. Herbivores could also be exposed to surface water through ingestion of water in the drainage features at the Former Choccolocco Corridor Ranges. Herbivores could also be exposed to sediment while foraging in the drainage features at the site.

Typical herbivorous species that could be expected to occur at the Former Choccolocco Corridor Ranges and are commonly used as sentinel species in ecological risk assessment include eastern cottontail (*Sylvilagus floridanus*), eastern gray squirrel (*Sciurus carolinensis*), pine vole (*Pitymys pinetorum*), and whitetail deer (*Odocoileus virginianus*).

Aquatic herbivores, such as muskrat (*Ondatra zibethica*) and mallard (*Anas platyrhynchos*) are not expected to occur with any regularity at the Former Choccolocco Corridor Ranges due to the ephemeral nature of most of the drainage features and the restricted aquatic habitat at the Former Choccolocco Corridor Ranges.

6.2 Invertivorous Feeding Guild

Invertivores specialize in eating insects and other invertebrates. As such, they may be exposed to site-related chemicals that may have accumulated in insects and other invertebrates.

Invertivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while probing for insects, grooming, or other activities. Ingestion of soil while feeding is a potential exposure pathway for invertivores since much of their food (i.e., earthworms and other invertebrates) lives on or below the soil surface. Invertivores could be exposed to surface water through ingestion of water in the drainage features at the Former Choccolocco Corridor Ranges. Invertivores could also be exposed to sediment while foraging in the drainage features at the site.

Typical invertivorous species that could be expected to occur at the Former Choccolocco Corridor Ranges and are commonly used as sentinel species in ecological risk assessment include American woodcock (*Philohela minor*), carolina wren (*Thryothorus ludovicianus*), shorttail shrew (*Blarina brevicauda*), and eastern mole (*Scalopus aquaticus*). Aquatic invertivores such as the wood duck (*Aix sponsa*) and blacknose dace (*Rhinichthys atratulus*) are not expected to occur with any regularity at the Former Choccolocco Corridor Ranges due to the ephemeral nature of most of the drainage features at the site and the restricted aquatic habitat provided by the one perennial stream at the Former Choccolocco Corridor Ranges.

6.3 Omnivorous Feeding Guild

Omnivores consume both plant and animal material in their diet, depending upon availability. Therefore, they could be exposed to chemicals that have accumulated in the vegetative tissues of plants at the site and also chemicals that may have accumulated in smaller animal tissues that the omnivores prey upon. Omnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities. Omnivores could be exposed to surface water through ingestion of water in the drainage features at the Former Choccolocco Corridor Ranges. Omnivores could also be exposed to sediment while foraging in the drainage features at the site.

Typical omnivorous species expected to occur at the Former Choccolocco Corridor Ranges and are commonly used as sentinel species in ecological risk assessment include red fox (*Vulpes vulpes*), white-footed mouse (*Peromyscus leucopus*), wild turkey (*Meleagris gallopavo*), and American robin (*Turdus migratorius*). Aquatic omnivores, such as the creek chub (*Semotilus atromaculatus*) are not expected to occur with any regularity at the Former Choccolocco Corridor Ranges due to the ephemeral nature of most of the drainage features at the site and the restricted aquatic habitat provided by the one perennial stream at the Former Choccolocco Corridor Ranges.

6.4 Carnivorous Feeding Guild

Carnivores are meat-eating animals and are; therefore, potentially exposed to site-related chemicals through consumption of prey animals that may have accumulated contaminants in their tissues. Carnivores are quite often top predators in a local food web and are often subject to exposure to contaminants that have bioaccumulated in lower trophic-level organisms or biomagnified through the food web. Food web exposures for carnivores are based on the consumption of prey animals that have accumulated COPECs from various means. Smaller herbivores, omnivores, invertivores, and other carnivores may consume soil, plant, and animal material as food and accumulate COPECs in their tissues. Subsequent ingestion of these prey animals by carnivorous animals would result in indirect food chain exposure to COPECs. Food chain exposures to metals, volatile organic compounds, and semi-volatile organic compounds in soil are expected to be minimal at the Former Choccolocco Corridor Ranges because these compounds are not accumulated in animal tissues to any great extent (Shugart, 1991; U.S. Army Environmental Hygiene Agency, 1994). Exposure of carnivorous animals to herbicides and pesticides through the food chain could be significant since these chemicals may bioaccumulate and/or bioconcentrate.

Carnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities. Carnivores could be exposed to surface water through ingestion of water in the drainage features at the Former Choccolocco Corridor Ranges. Carnivores could also be exposed to sediment while foraging in the drainage features at the site.

Typical carnivorous species that may occur at the Former Choccolocco Corridor Ranges and commonly used as sentinel species in ecological risk assessment include red-tailed hawk (*Buteo jamaicensis*), black vulture (*Coragyps atratus*), and bobcat (*Lynx rufus*).

6.5 Piscivorous Feeding Guild

Piscivores are specialists that feed almost exclusively on fish. Therefore, they may be exposed to site-related chemicals that have accumulated in small fish that may inhabit the perennial stream at the Former Choccolocco Corridor Ranges. They may also be exposed to surface water and sediment in these drainage features through ingestion of drinking water and during feeding. Because many of the drainage features at the Former Choccolocco Corridor Ranges are ephemeral in nature, they would only provide habitat and a drinking water supply during periods of significant precipitation. Piscivores are unlikely to inhabit the CC ranges due to the lack of a continuous food source..

Food web exposures for piscivores are based on the consumption of fish that have accumulated COPECs from surface water and sediment. Forage fish may consume surface water, sediment, benthic invertebrates, aquatic plants, and planktonic material as food and accumulate COPECs in their tissues. Subsequent ingestion of these forage fish by piscivorous animals would expose them to COPECs. However, most inorganic compounds are not accumulated in fish tissues to any great extent. Therefore, food web exposures to these chemicals are expected to be minimal. Semi-volatile and volatile organic compounds are readily metabolized by most fish species and are not accumulated to any extent. Thus the piscivorous feeding guild is not expected to have significant exposure to semi-volatile or volatile organic compounds at the Former Choccolocco Corridor Ranges through the food web. Chlorinated herbicides and pesticides have the potential to bioaccumulate and biomagnify through the food chain; therefore, there is the potential for significant exposure to these classes of chemicals by piscivores, if they are present at the Former Choccolocco Corridor Ranges.

Typical piscivorous species that could occur at the Former Choccolocco Corridor Ranges and are commonly used as sentinel species in ecological risk assessment include great blue heron (*Ardea herodias*) and belted kingfisher (*Megaceryle alcyon*). Larger piscivorous fish species (e.g., smallmouth bass, spotted gar, etc.) are not expected to occur in the single perennial stream or ephemeral drainage features at the Former Choccolocco Corridor Ranges due to the habitat limitations of these small creeks and drainage features. Piscivorous birds and mammals (e.g. mink) may visit these drainage features at the Former Choccolocco Corridor Ranges on occasion, but populations of piscivorous receptors are not likely to regularly utilize the CC ranges due to habitat restrictions (small stream size, lack of sizeable fish populations, ephemeral nature of most of the streams, etc.).

6.6 Threatened and Endangered Species

Four species listed as threatened or endangered by the U.S. Fish and Wildlife Service (USFWS) have been recorded at FTMC. These threatened and endangered species are as follows:

- Gray Bat (*Myotis grisescens*)
- Blue Shiner (*Cyprinella caerules*)
- Mohr's Barbara Buttons (*Marshallia mohrii*)
- Tennessee Yellow-Eyed Grass (*Xyris tennesseensis*)

An additional endangered species, the red-cockaded woodpecker (*Picoides borealis*), historically has inhabited the installation.

The Federally listed species that have the potential to occur in the vicinity of the Former Choccolocco Corridor Ranges are the gray bat and the blue shiner (Garland, 1996). The drainage features at the Former Choccolocco Corridor Ranges have not been classified for their potential to support foraging habitat for the gray bat (Garland, 1996). However, mist net surveys conducted along Choccolocco Creek have resulted in the capture of gray bats (Garland, 1996). These preliminary data suggest that the major stream corridors at FTMC may provide at least a minimum foraging habitat for gray bats. However, gray bat surveys have not been conducted on the small drainage features in the vicinity of the Former Choccolocco Corridor Ranges. The small size and ephemeral nature of the drainage features at these sites are expected to limit the potential for gray bats to utilize these drainage features for foraging.

Within Alabama, the blue shiner (*Cyprinella caerules*) is currently restricted to Weogufka and Choccolocco Creeks, and the lower reaches of the Little River. Within Choccolocco Creek, the blue shiner is limited to about 22.6 km of the main channel and the lower reaches of Shoal Creek. Habitat preference on Choccolocco Creek appears to be slack to slow current over sand or a mixture of boulders, cobbles, and sand. Lateral pools away from the main run of the stream and backwaters with sand substrate were ideal habitat for blue shiners (Pierson and Krotzer, 1987). Blue shiners appear to be intolerant of high turbidity and are probably a mid-depth feeder competitively dependent on high visibility (Mount, 1986). The main channel of Choccolocco Creek provides optimal habitat for the blue shiner (Pierson and Krotzer, 1987). However, the small drainage features in the vicinity of the Former Choccolocco Corridor Ranges are small and mostly ephemeral; therefore, they likely do not provide suitable habitat for blue shiner. Because of the habitat restrictions, the blue shiner is unlikely to occur at the Former Choccolocco Corridor Ranges and is not considered further in this Problem Formulation.

Although historical records indicate the presence of red cockaded woodpeckers at FTMC, the last remaining active cluster of red cockaded woodpeckers at FTMC was recorded in 1968. Subsequent surveys in 1972, 1982, and 1985 failed to find any red cockaded woodpeckers at FTMC. Thus, it can be concluded that red cockaded woodpeckers no longer exist at FTMC.

7.0 Selection of Assessment and Measurement Endpoints

Assessment and measurement endpoints are the basis of the Study Design phase of the BERA and define the ecological values that require protection and the methodologies by which those ecological values are measured, respectively. The following sections describe the assessment endpoints that have been identified for the CC ranges, the risk hypotheses, and the corresponding measurement endpoints.

7.1 Assessment Endpoints

An assessment endpoint is "an explicit expression of the environmental value that is to be protected" (EPA, 1992). Assessment endpoints focus the risk assessment on particular valuable components of the ecosystem(s) that could be adversely affected by contaminants at a site. Individual assessment endpoints usually encompass a group of species or populations with some common characteristic, such as a specific exposure route or contaminant sensitivity.

Assessment endpoints for the BERA for the CC ranges were selected based on the ecosystems, communities, and species present in the vicinity of the sites. Selection of the assessment endpoints was dependent upon the following factors:

- The COPECs, their characteristics, and their concentrations at the CC ranges
- The mechanisms of toxicity of the COPECs to different groups of organisms
- Ecologically relevant receptors that are potentially sensitive or highly exposed to the COPECs
- The presence of complete exposure pathways contributing to potential risk.

The potential for toxic effects to individual receptors can have consequences at the population, community, and ecosystem level. Population level effects may determine the nature of changes in community structure and function, such as reduction in species diversity, simplification of food webs, and shifts in competitive advantages among species sharing a limited resource. Ecosystem function may also be affected by contaminants, which can cause changes in productivity or disruption of key processes.

Population level assessment endpoints are generally recognized in ecological risk assessments because of their role in maintaining biological diversity, ecological integrity, and productivity in ecosystems.

Based on the fact that the COPECs in surface soil at the CC ranges are a mixture of metals that are expected to have a range of tendencies to bioaccumulate, and organic chemicals such as DDD that may biomagnify in the food chain, the terrestrial ecological receptors with the potential for the greatest exposure to soil COPECs at the CC ranges were determined to be omnivorous and invertivorous small mammals and birds. Herbivores were considered to have a lower exposure potential to soil COPECs because few, if any, of the COPECs accumulate appreciably in plant tissues, the herbivores' main food source. Carnivores were expected to have moderate exposure potential to soil COPECs because several of the COPECs may bioconcentrate or biomagnify in prey animal tissues, but carnivores have larger home ranges which would tend to minimize their exposures to COPECs at the CC ranges. Therefore, the terrestrial assessment endpoints for the CC ranges focus on the protection of the terrestrial omnivorous and invertivorous feeding guilds present at the CC ranges.

The aquatic assessment endpoints for the CC ranges focus on potential exposures to sediments in the ephemeral and perennial drainage features at the CC ranges. As such, the assessment endpoints focus on the protection of the benthic invertebrate communities present in the sediment of the perennial and ephemeral drainage features at the CC ranges. The protection of riparian insectivorous mammals and birds is also an assessment endpoint for the CC ranges because these receptors consume benthic invertebrates that spend a significant portion of their life in sediment. Piscivorous species were not considered as assessment endpoints because they have lower exposure potential to sediment COPECs. Although some of the sediment COPECs tend to bioconcentrate or biomagnify in fish tissue, sustainable fish populations large enough to represent a stable and reliable food source for piscivorous receptors are not readily found in the drainage features at the CC ranges. Additionally, although lead was identified as a COPEC in surface water at the CC ranges, it was identified solely for the purpose of identifying a remedial action objective for lead in surface water. No assessment or measurement endpoints will be identified specifically for surface water constituents. COPECs were not identified in groundwater; therefore, no assessment endpoints are identified specifically for groundwater.

7.1.1 Terrestrial Assessment Endpoints

Given the overall goal of protecting the integrity and quality of the terrestrial old field and upland forest ecosystems at the CC ranges, the terrestrial assessment endpoints focus on critical community niches within these habitat types. As discussed above, the ecological receptors with the potential for the greatest exposure to soil COPECs at the CC ranges were determined to be invertivorous and omnivorous small mammals and birds. Additionally, the terrestrial plant and invertebrate communities have the potential for significant exposure to COPECs. These ecological communities form the basis for the assessment endpoints described herein.

The terrestrial plant community has the potential to be significantly exposed to COPECs in surface soil and constitutes a critical food source for herbivorous and omnivorous birds and mammals. Terrestrial plants may also accumulate COPECs in their tissues and act as a conduit for the transfer of COPECs to higher trophic level organisms in the food chain. For these reasons, the terrestrial plant community was identified as an important ecological resource at the CC ranges. The assessment endpoint that has been identified with respect to the terrestrial plant community is the following:

- Survival and growth of the terrestrial plant communities at the CC ranges.

The terrestrial invertebrate community forms a critical link in many terrestrial food webs and constitutes a food source for many omnivorous and invertivorous birds and mammals. Terrestrial invertebrates also perform an important function in the degradation of organic matter in soil through their bioturbative activities. Terrestrial invertebrates may also accumulate COPECs in their tissues and act as a conduit for the transfer of COPECs to higher trophic level organisms in the food chain. For these reasons, the terrestrial invertebrate community was identified as an important ecological resource at the CC ranges. The assessment endpoint that has been identified with respect to the terrestrial invertebrate community is the following:

- Survival and growth of the terrestrial invertebrate community at the CC ranges.

Invertivorous mammals and birds were identified as having significant potential for exposure to COPECs at the CC ranges, mainly through ingestion of terrestrial invertebrates that may have accumulated COPECs in their tissues. In addition to the fact that this feeding guild has the potential to be maximally exposed to COPECs due to their feeding habits, these species also form an important food group for higher trophic level organisms. Carnivorous mammals and/or birds may prey on small invertivorous mammals and birds and thus become exposed to COPECs through ingestion of COPECs that have become incorporated into the prey species' tissues. For these reasons, invertivorous mammals and birds were identified as being an important ecological resource at the CC ranges. It is important to assess the survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the CC ranges for the protection of these species themselves, and potentially more importantly, because these species constitute an important food source and a possible conduit for COPECs to upper trophic level organisms. The assessment endpoint that has been identified with respect to the terrestrial invertivorous mammal and bird feeding guilds is the following:

- Survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the CC ranges.

Omnivorous mammals and birds were identified as having significant potential for exposure to COPECs at the CC ranges, mainly because a portion of their diet includes terrestrial plants and terrestrial invertebrates that may have accumulated COPECs in their tissues. In addition to the fact that this feeding guild has the potential to be maximally exposed to COPECs due to their feeding habits, these species also form an important food group for higher trophic level organisms. Carnivorous mammals and/or birds may prey on small omnivorous mammals and birds and thus become exposed to COPECs through ingestion of COPECs that have become incorporated into the prey species' tissues. For these reasons, omnivorous mammals and birds were identified as being an important ecological resource at the CC ranges. The assessment endpoint that has been identified with respect to the terrestrial omnivorous mammal and bird feeding guilds is the following:

- Survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the CC ranges.

Because these terrestrial assessment endpoints are highly dependent upon the bioavailability of the COPECs in soil, a study of the binding capacity of soils commonly found at FTMC was conducted and the results presented in the BERA for the Iron Mountain Road (IMR) and Bains Gap Road (BGR) ranges (Shaw, 2004). In summary, a total of eight surface soil samples from the IMR ranges (Parcels 69Q, 70Q, 71Q, and 75Q) and the BGR ranges (Parcels 77Q, 78Q, 80Q, and 85Q) were collected from five soil mapping units (U.S. Department of Agriculture, 1961): Anniston and Allen gravelly loams; Anniston and Allen stony loams; Stony rough land, sandstone; Jefferson stony fine sandy loam; and Jefferson gravelly fine sandy loam. The results of the binding capacity study showed that the soils at the IMR and BGR ranges could be classified as "low", "medium", or "high" with regard to their potential metal–binding capacity. However, the terrestrial invertebrate toxicity testing and bioaccumulation testing conducted as part of the BERA for the IMR and BGR ranges showed no significant differences in toxicity or bioaccumulation potential between the "high", "medium", or "low" binding capacity soils. Therefore, it was assumed that all of the soils at the IMR and BGR ranges exhibited similar metal-binding capacities.

The vast majority of the soils at the CC ranges are mapped as Anniston and Allen gravelly loams; Stony Rough Land, sandstone; Anniston and Allen stony loam; and Jefferson gravelly fine sandy loam. Because the soil mapping units at the CC ranges are the same or similar to the soil mapping units at the IMR and BGR ranges, and all the soil mapping units at the IMR and

BGR ranges were found to have similar metal binding capacities, it is assumed for this BERA that the binding capacities for the soils at the CC ranges are all similar and no differentiation will be made between soil mapping units.

The assessment endpoints that have been identified for the terrestrial ecosystems at the CC ranges are summarized in Table 7-1.

7.1.2 Aquatic Assessment Endpoints

The overall goal of the aquatic assessment endpoints is the protection of the integrity and quality of the aquatic ecosystem in the perennial and ephemeral streams at the CC ranges. The aquatic assessment endpoints focus on critical community niches within the sediment of these drainage features at the CC ranges. The ecological receptors with the potential for the greatest exposure to COPECs in the sediment of the drainage features at the CC ranges are those populations and communities that live in direct contact with the sediment within the drainage features, and those feeding guilds that utilize the drainage features as a major food source. These ecological communities formed the basis for the aquatic assessment endpoints described herein.

The benthic invertebrate community forms a critical link in many aquatic food webs and constitutes a food source for many aquatic and riparian omnivorous and invertivorous birds and mammals. Aquatic benthic invertebrates also perform an important function in the degradation of organic material in sediment. Aquatic benthic invertebrates may also accumulate COPECs in their tissues and act as a conduit for the transfer of COPECs to higher trophic level organisms in the food chain. For these reasons, the aquatic benthic invertebrate community was identified as an important ecological resource at the CC ranges. The assessment endpoint that has been identified with respect to the aquatic benthic invertebrate community is the following:

- Survival, growth, and reproduction of aquatic benthic invertebrates in the perennial and ephemeral drainage features at the CC ranges.

Riparian invertivorous mammals and birds were identified as having significant potential for exposure to COPECs at the CC ranges, mainly through ingestion of aquatic benthic invertebrates that may have accumulated COPECs in their tissues. In order to differentiate the invertivores that feed mainly on terrestrial invertebrates from those that feed mainly on aquatic invertebrates, this latter group is termed "riparian invertivores" for this assessment. In addition to the fact that this feeding guild has the potential to be maximally exposed to COPECs in sediment due to their feeding habits, these species also form an important food group for higher trophic level organisms such as raptors. Raptors may prey on flying invertivorous mammals (e.g., bats) and

invertivorous birds (e.g., swallows, wrens) and thus become exposed to COPECs through ingestion of COPECs that have become incorporated into the prey species' tissues. For these reasons, riparian invertivorous mammals and birds were identified as being an important ecological resource at the CC ranges. The assessment endpoint that has been identified with respect to the riparian invertivorous mammal and bird feeding guild is the following:

- Survival, growth, and reproduction of riparian invertivorous small mammals and birds at the CC ranges.

The assessment endpoints that have been identified for the aquatic/riparian ecosystems at the CC ranges are summarized in Table 7-1.

7.2 Risk Hypotheses

The risk hypotheses in a BERA are questions about the relationships among the assessment endpoints and the predicted responses at a given site. The risk hypotheses described in the following sections may be more accurately described as "test hypotheses" as they may not actually describe the probability (or risk) that a receptor will develop a toxicological endpoint. Rather, the hypotheses described herein are actually statements of a testing framework, and provide a basis for developing the Study Design for the assessment endpoints. The most basic question applicable to most sites is whether site-related contaminants are causing, or have the potential to cause, adverse effects on the assessment endpoints. Using this basic premise, risk hypotheses were developed for the assessment endpoints identified in the previous section.

7.2.1 Terrestrial Risk Hypothesis

Two test hypotheses were identified as being appropriate to address the assessment endpoint of "survival and growth of the terrestrial plant communities" at the CC ranges. These test hypotheses were the following:

- Are concentrations of COPECs in surface soil at the CC ranges greater than ecological screening values for the survival or growth of terrestrial plants?

- Is the survival and growth of terrestrial plants exposed to surface soil from the CC ranges significantly lower than that for terrestrial plants exposed to soil from reference sites?

The test hypothesis regarding ecological screening values will aid in the interpretation of the toxicity tests results and may help in the identification of the most likely causative agent(s) in the terrestrial plant toxicity tests. The test hypothesis relative to the terrestrial plant toxicity tests

will identify differences in terrestrial plant survivability and growth when exposed to on-site soils and off-site reference soils in laboratory toxicity tests.

Two test hypotheses were also identified as being appropriate to address the assessment endpoint of "survival and growth of the terrestrial invertebrate community" at the CC ranges. These test hypotheses were the following:

- Are concentrations of COPECs in surface soil at the CC ranges greater than ecological screening values for the survival or growth of terrestrial invertebrates?
- Is the survival and growth of terrestrial invertebrates exposed to surface soil from the CC ranges significantly lower than that for terrestrial invertebrates exposed to soil from reference sites?

The test hypothesis regarding ecological screening values will aid in the interpretation of the toxicity test results and may help in the identification of the most likely causative agent(s) in the terrestrial invertebrate toxicity tests. The test hypothesis relative to the terrestrial invertebrate toxicity tests will identify differences in terrestrial invertebrate survivability and growth when exposed to on-site soils and off-site reference soils in laboratory toxicity tests.

The test hypothesis that was identified as being appropriate to address the assessment endpoint of "survival, growth, and reproduction of terrestrial invertivorous small mammals and birds" was determined to be the following:

- Does the daily dose of COPECs received by terrestrial invertivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the toxicity reference values (TRV) for survival, reproduction, or growth?

This test hypothesis will determine whether calculated daily doses of COPECs exceed feeding guild-specific toxicity reference values.

The test hypothesis that was identified as being appropriate to address the assessment endpoint of "survival, growth, and reproduction of terrestrial omnivorous small mammals and birds" was determined to be the following:

- Does the daily dose of COPECs received by terrestrial omnivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the TRVs for survival, reproduction, or growth?

This test hypothesis will determine whether calculated daily doses of COPECs exceed feeding guild-specific toxicity reference values.

Table 7-1 presents risk/test hypotheses for each of the terrestrial assessment endpoints. It is important to note that the hypotheses are expressed as a positive response in order to minimize the likelihood of Type II statistical errors (i.e., a false negative decision) at a standard confidence level of $p = 0.05$.

Daily doses of COPECs for terrestrial invertivorous and omnivorous small mammals and birds will be calculated using standard exposure algorithms. These algorithms will incorporate species-specific natural history parameters (i.e., feeding rates, water ingestion rates, dietary composition, etc.) and will also utilize site-specific area use factors (AUF). Measured COPEC concentrations in earthworms will be used as input to the exposure algorithm as the concentration in the invertebrate portion of the food of the terrestrial invertivorous and omnivorous small mammals and birds. Additionally, site-specific soil-to-earthworm bioaccumulation factors will be used to estimate COPEC concentrations in the terrestrial invertebrate portions of invertivorous and omnivorous species' diets. Site-specific soil-to-plant bioaccumulation factors will be used to estimate COPEC concentrations in the terrestrial vegetation portions of the receptor species' diets and measured COPEC concentrations in terrestrial plants sampled from on-site locations will also be used in the terrestrial food web model.

In order to calculate COPEC exposures, indicator species that represent the feeding guilds of interest must be identified. For this risk assessment, the small terrestrial invertivorous mammal will be represented by the shorttail shrew (*Blarina brevicauda*) and the terrestrial invertivorous bird will be represented by the American woodcock (*Philohela minor*). The small terrestrial omnivorous mammal will be represented by the white-footed mouse (*Peromyscus leucopus*) and the terrestrial omnivorous bird will be represented by the American robin (*Turdus migratorius*). Natural history parameters for these indicator species (Table 7-2) will be used in combination with site-specific exposure parameters to estimate exposures (total daily doses) to terrestrial invertivorous and omnivorous small mammals and birds at the CC ranges.

The algorithm that will be used to estimate exposures to COPECs by terrestrial invertivorous and omnivorous small mammals and birds is the following:

$$TDD_{wildlife} = \left[\left(IR_{food} \times f_{worm} \times C_{worm}\right) + \left(IR_{food} \times f_{veg} \times C_{veg}\right) + \left(IR_{food} \times f_{soil} \times \left\{1 - M_{diet}\right\} \times C_{soil}\right)\right] \times AUF$$

where:

$TDD_{wildlife}$	=	total daily dose of COPEC received by omnivorous or invertivorous mammals or birds through ingestion (mg/kg/day);
IR_{food}	=	ingestion rate of food by receptor species (kg/kg/day);
f_{worm}	=	fraction of daily diet comprised of invertebrates (percent);
C_{worm}	=	concentration of COPEC in invertebrate tissue (mg/kg);
f_{veg}	=	fraction of daily diet comprised of vegetation (percent);
C_{veg}	=	concentration of COPEC in terrestrial vegetation (mg/kg);
f_{soil}	=	fraction of daily diet comprised of soil (percent);
M_{diet}	=	weighted average moisture content of diet (percent);
C_{soil}	=	concentration of COPEC in soil (mg/kg); and
AUF	=	area use factor (fraction of site used by receptor species (percent).

7.2.1.1 Terrestrial Omnivorous Mammal Model Parameters

The surrogate species used in the terrestrial food web model to assess omnivorous mammals was the white-footed mouse (*Peromyscus leucopus*). The home range for white-footed mice ranges from one-half to one and one-half acres in size (Burt and Grossenheider, 1976). For the terrestrial food web model, the mean (one acre) of this range was used as the foraging area. Body weights for white-footed mice range from 14 to 31 grams (Burt and Grossenheider, 1976). The mean of this range (22.5 grams) was used as the representative body weight for white-footed mice in the terrestrial food web model. USEPA (1993) reports a water ingestion rate of 0.19 g/g/day for deer mice (*Peromyscus maniculatus*) based on two studies conducted in laboratories. Since water ingestion rates for white-footed mice were not readily available and the body weights for white-footed mice and deer mice are very similar, the water ingestion rate for deer mice (0.19 g/g/day) was used in the terrestrial food web model. Similarly, the food ingestion rate for deer mice was used in the terrestrial food web model. The food ingestion rates reported by USEPA (1993) for adult male and female non-breeding and lactating deer mice ranged from 0.18 to 0.45 g/g/day. The mean of this range is 0.2683 g/g/day (wet weight). The weighted average moisture content of the white-footed mouse diet (invertebrates and vegetation) has been estimated to be 53.9 percent. Taking this moisture content into account, the food ingestion rate for the white-footed mouse was estimated to be 0.1237 g/g/day (dry weight). The estimated percent of soil in the white-footed mouse diet is less than two percent (USEPA, 1993). For the terrestrial food web model, it was assumed that two percent (0.00247 g/g/day, dry weight) of the white-footed mouse diet was made up of soil. These input parameters are summarized in Table 7-2 and were used to estimate total daily exposures to COPECs for omnivorous mammals.

7.2.1.2 Terrestrial Omnivorous Bird Model Parameters

The surrogate species used in the terrestrial food web model to assess omnivorous birds was the American robin (*Turdus migratorius*). The territory size for adult male and female robins is

reported to range from 0.11 to 0.42 hectares (USEPA, 1993). The mean of this range (0.61 acres) was used as the foraging area for American robins in the terrestrial food web model. Body weights for adult male and female robins in New York and Pennsylvania was reported to range from 77.3 grams to 86.2 grams (USEPA, 1993). The mean of this range (81 grams) was used to represent the body weight of American robins in the terrestrial food web model. USEPA (1993) reports an estimated water ingestion rate of 0.14 g/g/day for adult male and female American robins. This estimated water ingestion rate was used in the terrestrial food web model. The food ingestion rate for free-living adult male and female American robins in California is reported to be 0.89 g/g/day (wet weight) (USEPA, 1993). The weighted average moisture content of the American robin diet (invertebrates and vegetation) has been estimated to be 79.6 percent. Taking this moisture content into account, the food ingestion rate for the American robin was estimated to be 0.1816 g/g/day (dry weight). Soil ingestion rates for American robins were not readily available; therefore, the soil ingestion rate of 2 percent of the diet reported for other birds (USEPA, 1993) was assumed to be representative of American robins. Assuming two percent of the robin's diet is made up of soil, the soil ingestion rate was estimated to be 0.00363 g/g/day (dry weight). These input parameters are summarized in Table 7-2 and were used to estimate total daily exposures to COPECs for omnivorous birds.

7.2.1.3 Terrestrial Invertivorous Mammal Model Parameters

The surrogate species used in the terrestrial food web model to assess invertivorous mammals was the short-tailed shrew (*Blarina brevicauda*). The home range for adult male and female short-tailed shrew in a Manitoba bog was reported to be 0.964 acres (USEPA, 1993). This home range was used for the short-tailed shrew in the terrestrial food web model. Body weights for adult male and female short-tailed shrews in New Hampshire and Pennsylvania were reported to range from 15 to 19.2 grams. The mean of this range (16.8 grams) was used as the body weight for short-tailed shrews in the terrestrial food web model. The water ingestion rate used in the terrestrial food web model for short-tailed shrews (0.223 g/g/day) was the water ingestion rate reported by USEPA (1993) for adult male and female short-tailed shrews in an Illinois laboratory. Food ingestion rates for adult male and female short-tailed shrews in a Wisconsin laboratory were reported to range from 0.49 to 0.62 g/g/day (wet weight), with a mean value of 0.555 g/g/day (wet weight). The weighted average moisture content of the short-tailed shrew's diet has been estimated to be 83.8 percent. Taking the moisture content of the shrew's diet into consideration, the food ingestion rate for the short-tailed shrew that was used in the terrestrial food web model was 0.0899 g/g/day (dry weight). Soil ingestion rates for short-tailed shrews were not readily available; therefore, the soil ingestion rate of 2.4 percent of the diet reported for meadow voles (USEPA, 1993) was assumed to be representative of short-tailed shrews. Assuming 2.4 percent of the shrew's diet is made up of soil, the soil ingestion rate was estimated

to be 0.00216 g/g/day (dry weight). These input parameters are summarized in Table 7-2 and were used to estimate total daily exposures to COPECs for invertivorous mammals.

7.2.1.4 Terrestrial Invertivorous Bird Model Parameters

The surrogate species used in the terrestrial food web model to assess invertivorous birds was the American woodcock (*Scolopax minor*). The home range for adult male and female American woodcocks in Pennsylvania and Wisconsin ranges from 7.7 to 182 acres (USEPA, 1993). The mean of this range is 61.3 acres and is the home range for American woodcocks used in the terrestrial food web model. Body weights for adult male and female American woodcocks range from 133 to 218 grams (USEPA, 1993). The mean of this range is 169.4 grams and is the value used to estimate the body weight of American woodcock in the terrestrial food web model. The water ingestion rate for adult male and female American woodcock estimated by USEPA (1993) is 0.1 g/g/day. This value was used as the water ingestion rate for American woodcock in the terrestrial food web model. The food ingestion rate for captive adult male and female American woodcocks in Louisiana fed earthworms was 0.77 g/g/day (wet weight) (USEPA, 1993). The weighted average moisture content of the American woodcock's diet has been estimated to be 80.3 percent. Taking the moisture content of the woodcock's diet into account, the food ingestion rate for the American woodcock that was used in the terrestrial food web model was 0.1517 g/g/day (dry weight). USEPA (1993) reports an estimated percent of soil in a woodcock's diet of 10.4 percent. Assuming 10.4 percent of a woodcock's diet is made up of soil, the soil ingestion rate was estimated to be 0.0158 g/g/day (dry weight). These input parameters are summarized in Table 7-2 and were used to estimate total daily exposures to COPECs for invertivorous birds.

COPEC concentrations in terrestrial invertebrate tissues will need to be estimated in order to calculate a total COPEC dose. The COPEC concentrations in terrestrial invertebrate tissues will be estimated using site-specific soil-to-invertebrate bioaccumulation factors ($BAF_{soil\text{-}to\text{-}worm}$) determined through the analysis of earthworm tissue samples and the soil samples used in the earthworm toxicity/bioaccumulation studies. These $BAF_{soil\text{-}to\text{-}worm}$ will be applied to the soil concentrations of COPECs to estimate concentrations of COPECs in terrestrial invertebrate food material in the following manner:

$$C_{worm} = C_{soil} \times BAF_{soil-to-worm}$$

where:

C_{worm} = COPEC concentration in terrestrial invertebrates (mg/kg-dry weight);

C_{soil}	=	COPEC concentration in soil (mg/kg-dry weight); and
$BAF_{soil\text{-}to\text{-}worm}$	=	soil-to-worm bioaccumulation factor (unitless).

Measured earthworm tissue concentrations of COPECs determined during the earthworm bioaccumulation studies will also be used as input to the terrestrial food web model.

Because portions of the receptor species' diets consist of vegetative material, COPEC concentrations in terrestrial plant matter will need to be estimated in order to calculate a total COPEC dose. The COPEC concentrations in terrestrial plant matter will be estimated using site-specific soil-to-plant BAFs determined through the collection and analysis of co-located soil and terrestrial plant tissue samples. These soil-to-plant BAFs will be applied to the soil concentrations of COPECs to estimate concentrations of COPECs in terrestrial vegetative food material in the following manner:

$$C_{plant} = C_{soil} \times BAF_{soil-to-plant}$$

where:

C_{plant}	=	COPEC concentration in terrestrial plants (mg/kg-dry weight);
C_{soil}	=	COPEC concentration in soil (mg/kg-dry weight); and
$BAF_{soil\text{-}to\text{-}plant}$	=	soil-to-plant bioaccumulation factor (unitless).

Measured plant tissue concentrations from on-site terrestrial plant samples will also be used in the terrestrial food web model.

The soil ingestion rate for the receptor species is most often represented as a percentage of a receptor species' diet. In order to account for the methodology used in the estimation of the soil ingestion rates, the moisture content of the receptor species' diets must be accounted for. The relationship used to estimate the soil ingestion rates for the terrestrial invertivorous and omnivorous small mammals and birds that have been identified as receptors in this ecological risk assessment is as follows:

$$IR_{soil} = IR_{food} \times Diet_{soil}$$

where:

IR_{soil}	=	ingestion rate of soil (kg/kg/day, dry weight);
IR_{food}	=	ingestion rate of food (kg/kg/day, wet weight); and
$Diet_{soil}$	=	portion of diet that is soil (percent).

The moisture contents of the invertebrate and vegetative material in the receptor species' diets were referenced from the EPA's *Wildlife Exposure Factors Handbook* (EPA, 1993) and are as follows:

- Earthworms - 84%
- Fruit - 77%
- Roots / young grass - 82%
- Seeds - 9.3%
- Fruit / young grass - 78%

The weighted-average moisture contents of the diets of the receptor species of interest are as follows:

	Percent Moisture	Weighted-Average Moisture Content
White-footed mouse:		
invertebrates =	84%	53.9%
vegetation =	43.6%	
American robin:		
invertebrates =	84%	79.6%
vegetation =	77%	
Shorttail shrew:		
invertebrates =	84%	83.8%
vegetation =	82%	
American woodcock:		
invertebrates =	84%	80.3%
vegetation =	9.3%	

It was also assumed that if a receptor species' diet contained multiple vegetative components, then the percentage of each vegetative component would be equal. For instance, the vegetative component of the shorttail shrew's diet was assumed to be comprised of 50 percent roots and 50 percent young grass.

Dietary composition of the indicator species will be simplified for modeling purposes but will incorporate the major food types for the different feeding guilds. It will be assumed that food intake for invertivores is comprised almost entirely of terrestrial invertebrates (i.e., earthworms). It will also be assumed that omnivores consume both plant and animal material, a portion of which will consist of terrestrial invertebrates.

The AUFs for each of the indicator species will take into account the home range and habitat requirements for each species and the size of the contaminated areas and viable habitat at the CC ranges.

7.2.2 Aquatic Risk Hypothesis

Three test hypotheses were identified as being appropriate to address the assessment endpoint of "survival, growth, and reproduction of aquatic benthic invertebrates in the perennial and ephemeral drainage features at the CC ranges." The first test hypothesis relative to benthic invertebrates in the drainage features was the following:

- Are the concentrations of COPECs in sediment samples from the perennial and ephemeral drainage features at the CC ranges greater than ecological screening values for the survival, growth, and reproduction of aquatic invertebrates?

This test hypothesis will aid in the interpretation of the toxicity test results and may help in the identification of the most likely causative agent(s) in the aquatic invertebrate toxicity tests.

The second test hypothesis relative to benthic invertebrates in the drainage features was the following:

- Is the survival and growth of aquatic benthic invertebrates exposed to sediment from the perennial and ephemeral drainage features at the CC ranges significantly lower than that for aquatic benthic invertebrates exposed to sediment from reference sites?

This test hypothesis will identify differences in aquatic benthic invertebrate survivability and growth when exposed to on-site sediments from the perennial and ephemeral drainage features, and off-site reference sediments in laboratory toxicity tests.

The third test hypothesis relative to benthic invertebrates in the drainage features was the following:

- Is the benthic community structure (using Rapid Bioassessment Protocol [RBP] II) significantly different in reaches of the perennial and ephemeral drainage features at the CC ranges compared to benthic communities in a non-impacted reference stream?

This test hypothesis will identify differences in aquatic benthic invertebrate community structure in reaches of the perennial and ephemeral drainage features when compared to the benthic invertebrate community structure in a non-impacted stream using in-situ RBP II assessment techniques.

The test hypothesis that was identified as being appropriate to address the assessment endpoint of "survival, growth, and reproduction of riparian invertivorous small mammals and birds" was determined to be the following:

- Does the daily dose of COPECs received by riparian invertivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the TRVs for survival, reproduction, or growth?

This test hypothesis will determine whether calculated daily doses of COPECs exceed feeding guild-specific toxicity reference values and will determine if COPECs in sediment have the potential to be transferred through the riparian food chain via aquatic insects.

Table 7-1 presents risk/test hypotheses for each of the aquatic assessment endpoints. It is important to note that the hypotheses are expressed as a positive response in order to minimize the likelihood of Type II statistical errors (i.e., a false negative decision) at a standard confidence level of $p = 0.05$.

Daily doses of COPECs for riparian invertivorous mammals and birds will be calculated using standard exposure algorithms. These algorithms will incorporate species-specific natural history parameters (i.e., feeding rates, dietary composition, etc.) and will also use site-specific AUFs. Laboratory-derived bioaccumulation factors will be used to estimate COPEC concentrations in the aquatic insect portions of the receptor species' diets. In addition, measured concentrations of COPECs in chironomid tissues will also be used as input to the riparian food web model to calculate dosages of COPECs potentially received by the riparian receptor species.

In order to calculate COPEC exposures, indicator species that represent the feeding guilds of interest must be identified. For this risk assessment, the riparian invertivorous mammal will be represented by the little brown bat (*Myotis lucifugus*) and the riparian invertivorous bird will be represented by the marsh wren (*Cistothorus palustris*). Natural history parameters for these indicator species (Table 7-3) will be used in combination with site-specific exposure parameters to estimate exposures to riparian invertivorous mammals and birds at the CC ranges.

The following algorithm will be used to estimate exposures to COPECs by riparian invertivorous mammals and birds:

$$TDD_{wildlife} = \left[\left(IR_{food} \times f_{invert} \times \left\langle C_{sed} \times BAF_{sed-to-invert} \right\rangle\right) + \left(IR_{water} \times C_{water}\right)\right] \times AUF$$

where:

$TDD_{wildlife}$	=	total daily dose of COPEC received by riparian invertivorous mammals or birds through ingestion (mg/kg/day);
IR_{food}	=	ingestion rate of food by receptor species (kg/kg/day);
f_{invert}	=	fraction of daily diet comprised of benthic invertebrates (percent);
C_{sed}	=	concentration of COPEC in sediment (mg/kg-dry wt.);
$BAF_{sed\text{-}to\text{-}invert}$	=	sediment-to-benthic invertebrate bioaccumulation factor;
IR_{water}	=	ingestion rate of water (L/kg/day);
C_{water}	=	concentration of COPEC in surface water (mg/L); and
AUF	=	area use factor (fraction of site used by receptor species) (percent).

7.2.2.1 Riparian Invertivorous Mammal Model Parameters

The surrogate species used in the riparian food web model to assess invertivorous mammals was the little brown bat (*Myotis lucifugus*). Information regarding the foraging area of the little brown bat was not available in the literature. However, according to LaVal, et al. (1977), gray bats (a cogeneric species) may travel as far as 12 kilometers from roost caves to foraging areas. This value was used as the foraging range for the little brown bat in the riparian food web model. Based on this foraging range and the total length of the perennial and ephemeral drainage features within the CC ranges (2.7 km), the area use factor (AUF) for the little brown bat was estimated to be 0.22. The body weights of adult male and female little brown bats range from 7 to 9 grams, with a mean of 8 grams. The mean body weight of 8 grams was used as the body weight for the little brown bat in the riparian food web model. The water ingestion rate for little brown bats was estimated using the allometric equation presented in Sample, et al. (1997). Assuming a mean body weight of 8 grams, the water ingestion rate for little brown bats was estimated to be 0.16 L/kg/day. The food ingestion rates for pregnant, lactating, and juvenile little brown bats in New Hampshire were reported by Anthony and Kunz (1977) to be 0.23, 0.48, and 0.29 g/g/day, respectively, with a mean value of 0.333 g/g/day. The average moisture content of aquatic isopods, amphipods, caldocerans, and insect larvae is 79 percent (USEPA, 1993). Taking the moisture content of the little brown bat's diet into consideration, the food ingestion rate for the little brown bat that was used in the riparian food web model was 0.0699 g/g/day (dry weight). Since little brown bats are assumed to feed exclusively on emergent benthic invertebrates (aerial insectivore), their potential exposures to sediment is expected to be negligible. These input parameters are summarized in Table 7-3 and were used to estimate total daily exposures to COPECs for invertivorous mammals.

7.2.2.2 Riparian Invertivorous Bird Model Parameters

The surrogate species used in the riparian food web model to assess invertivorous birds was the marsh wren (*Cistothorus palustris*). The size of an adult male marsh wren's territory ranges

from 0.015 acres to 0.42 acres, with a mean value of 0.13 acres (USEPA, 1993). This value was used as the foraging area for the marsh wren in the riparian food web model. This foraging area results in an area use factor for the marsh wren of 1.0. Body weights for adult male and female marsh wrens range from 9.4 to 11.9 grams (USEPA, 1993), with a mean value of 10.38 grams. This mean value was used as the marsh wren's body weight in the riparian food web model. USEPA (1993) has estimated the water ingestion rate for marsh wrens to be 0.26 g/g/day for adult males and 0.28 g/g/day for adult females. The mean value (0.27 g/g/day) was used for the water ingestion rate for marsh wrens in the riparian food web model. Food ingestion rates for adult male and female free-living marsh wrens range from 0.67 to 0.99 g/g/day (wet weight) (USEPA, 1993), with a mean value of 0.873 g/g/day. The average moisture content of aquatic isopods, amphipods, cladocerans, and insect larvae is 79 percent (USEPA, 1993). Taking the moisture content of the marsh wren's diet into consideration, the food ingestion rate for the marsh wren that was used in the riparian food web model was 0.1833 g/g/day (dry weight). Since marsh wrens are assumed to feed exclusively on emergent benthic invertebrates (aerial insectivore), their potential exposures to sediment is expected to be negligible. These input parameters are summarized in Table 7-3 and were used to estimate total daily exposures to COPECs for invertivorous birds.

It will be assumed that the receptor species' diets consist entirely of emergent benthic invertebrates; therefore, COPEC concentrations in benthic invertebrate tissues will need to be estimated in order to calculate a total COPEC dose. The COPEC concentrations in emergent benthic invertebrate tissues will be estimated using site-specific sediment-to-invertebrate bioaccumulation factors ($BAF_{sed\text{-}to\text{-}invert}$) determined through the analysis of chironomid tissue samples and the sediment samples used in the chironomid toxicity/bioaccumulation studies. These $BAF_{sed\text{-}to\text{-}invert}$ will be applied to the sediment concentrations of COPECs to estimate concentrations of COPECs in emergent benthic invertebrate food material in the following manner:

$$C_{invert} = C_{sed} \times BAF_{sed-to-invert}$$

where:

C_{invert}	=	COPEC concentration in emergent benthic invertebrates (mg/kg-dry weight);
C_{sed}	=	COPEC concentration in sediment (mg/kg-dry weight); and
$BAF_{sed\text{-}to\text{-}invert}$	=	sediment-to-invertebrate bioaccumulation factor (unitless).

Measured COPEC concentrations in chironomid tissues from the bioaccumulation tests will also be used as input to the riparian food web model. The total daily doses of COPECs received by the riparian invertivorous mammals and birds will not include the ingestion of soil or sediment as the receptors' diets are assumed to consist solely of emergent aquatic insects and the potential for exposure to site-related soil or sediment is minimal for these receptors. The moisture content of the receptor species' diets (aquatic benthic invertebrates) will be assumed to be 79 percent (EPA, 1993).

The AUFs for each of the indicator species will take into account the home range and habitat requirements for each species and the size of the contaminated areas and viable habitat at the CC ranges.

7.3 Selection of Measurement Endpoints

A measurement endpoint is "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint" and is a measure of biological effects (e.g., mortality, reproduction, growth) (EPA, 1992). Measurement endpoints are frequently numerical expressions of observations (e.g., toxicity test results, community diversity measures) that can be compared statistically to a control or reference site to detect adverse responses to site contaminants.

7.3.1 Terrestrial Measurement Endpoints

The terrestrial measurement endpoints described herein have been designed such that the information garnered from them can adequately address the assessment endpoints identified previously.

In order to facilitate the interpretation of the plant toxicity tests results and aid in the identification of the most likely causative agent(s) in the terrestrial plant toxicity tests, the following measurement endpoint has been identified:

- Comparison of COPEC concentrations in surface soil at the CC ranges to ecological screening values for the survival or growth of terrestrial plants.

The measurement endpoint that has been identified to address the assessment endpoint of "survival and growth of the terrestrial plant communities at the CC ranges" is the following:

- Statistical comparison of perennial ryegrass seed germination success, plant height, above ground biomass, root length, and root biomass between plants grown

in on-site soils from the CC ranges to plants grown in soils from a reference location.

In order to facilitate the interpretation of the terrestrial invertebrate toxicity test results and aid in the identification of the most likely causative agent(s) in the terrestrial invertebrate toxicity tests, the following measurement endpoint has been identified:

- Comparison of COPEC concentrations in surface soil at the CC ranges to ecological screening values for the survival or growth of terrestrial invertebrates.

The measurement endpoint that has been identified to address the assessment endpoint of "survival and growth of terrestrial invertebrate community at the CC ranges" is the following:

- Statistical comparison of earthworm survival rates and body weights between earthworms exposed to on-site soils from the CC ranges to earthworms exposed to soils from a reference location.

The measurement endpoint that has been identified to address the assessment endpoint of "survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the CC ranges" is the following:

- Comparison of calculated total daily doses of COPECs for terrestrial invertivorous small mammal (shorttail shrew) and invertivorous bird (American woodcock) to TRVs.

The measurement endpoint that has been identified to address the assessment endpoint of "survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the CC ranges" is the following:

- Comparison of calculated total daily doses of COPECs for terrestrial omnivorous small mammal (white-footed mouse) and omnivorous bird (American robin) to TRVs.

In order to estimate the bioavailability of the COPECs in soil at the CC ranges, and to provide data for the other assessment endpoints, a second measurement endpoint has been established to address the assessment endpoints of "survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the CC ranges" and "survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the CC ranges". This measurement endpoint is the following:

- Quantification of COPEC concentrations in tissues of earthworms exposed to soils from the CC ranges and tissues of earthworms exposed to soils from a reference location.

In order to provide site-specific information regarding the potential for COPEC accumulation in plant tissues, and its effect on the food web interactions of herbivores and omnivores at the CC ranges, the following measurement endpoint has been identified:

- Quantification of COPEC concentrations in above-ground plant tissues exposed to soils from the CC ranges and above-ground plant tissues exposed to soils from a reference location.

These measurement endpoints will provide the necessary data to answer the risk/test hypotheses for the terrestrial ecosystems at the CC Ranges presented in previous sections of this report. An important factor in assessing these measurement endpoints is an understanding of the degree of impairment to a biological attribute that is understood to be biologically or ecologically significant. Statistically significant differences in population survivability, growth, reproduction, or hazard quotient values that cannot be related to biological or ecological significance should not be interpreted as indicating a population or community is at risk or that a remedy is necessary (Tannenbaum, 2005). Therefore, ecological and biological significance will be considered within the context of these measurement endpoints.

Table 7-1 presents the measurement endpoints corresponding to each assessment endpoint and risk/test hypothesis. The methodologies used to collect the necessary data and how the data will be used to answer the risk/test hypotheses are presented in the following chapters.

7.3.2 Aquatic Measurement Endpoints

The aquatic measurement endpoints described herein have been designed such that the information garnered from them can adequately address the assessment endpoints identified previously.

In order to facilitate the interpretation of the aquatic benthic invertebrate toxicity test results and aid in the identification of the most likely causative agent(s) in the benthic invertebrate toxicity tests, the following measurement endpoint has been identified:

- Comparison of COPEC concentrations in sediment from the perennial and ephemeral drainage features at the CC ranges to ecological screening values for the survival or growth of aquatic benthic invertebrates.

The measurement endpoints that have been identified to address the assessment endpoint of "survival, growth, and reproduction of aquatic benthic invertebrates in the perennial and ephemeral drainage features at the CC ranges" are the following:

- Comparison of survival and growth of the benthic amphipod *Chironomus tentans* exposed to "on-site" sediment to survival and growth of *Chironomus tentans* exposed to sediment from a reference stream.

- Comparison of the benthic community assemblage in the perennial and ephemeral drainage features at the CC ranges with the benthic community assemblage in a reference stream using RBP II methodology.

The measurement endpoint that has been identified to address the assessment endpoint of "survival, growth, and reproduction of riparian invertivorous small mammals and birds at the CC ranges" is the following:

- Comparison of calculated total daily doses of COPECs for riparian invertivorous mammal (little brown bat) and invertivorous bird (marsh wren) to TRVs.

In order to provide site-specific information regarding the potential for COPEC accumulation in benthic invertebrate tissues, and its effect on the food web interactions of riparian invertivorous mammals and birds, the following measurement endpoint has been identified:

- Quantification of COPEC concentrations in tissues of chironomids exposed to sediment from the perennial and ephemeral drainage features at the CC ranges and from a reference location.

These measurement endpoints will provide the necessary data to answer the risk/test hypotheses for the aquatic ecosystems at the CC ranges presented in previous sections of this report. An important factor in assessing these measurement endpoints is an understanding of the degree of impairment to a biological attribute that is understood to be biologically or ecologically significant. Statistically significant differences in population survivability, growth, reproduction, or hazard quotient values that cannot be related to biological or ecological significance should not be interpreted as indicating a population or community is at risk or that a remedy is necessary (Tannenbaum, 2005). Therefore, ecological and biological significance will be considered within the context of these measurement endpoints.

Another important factor to recognize while interpreting the results of the toxicity tests is the fact that the organisms used in the laboratory toxicity tests may not be indigenous to the Fort McClellan area. As such, the laboratory species may be more or less sensitive to the COPECs

than indigenous organisms. Therefore, the results of the toxicity tests and the conclusions rendered from these tests will consider these uncertainties.

Table 7-1 presents the measurement endpoints corresponding to each assessment endpoint and risk/test hypothesis. The methodologies used to collect the necessary data and how the data will be used to answer the risk/test hypotheses are presented in the following chapters.

8.0 Data Quality Objectives

Data quality objectives (DQO) are "qualitative and quantitative statements that clarify study objectives, define the appropriate type of data, and specify tolerable levels of potential decision errors that will be used as the basis for establishing the quality and quantity of data needed to support decisions" (EPA, 2000b). The DQO process enables investigators to define performance criteria and limit the likelihood of committing Type I or Type II decision errors. EPA's DQO process is a seven-step process for the development of acceptance criteria. The initial five steps of the process are focused on identifying qualitative criteria, while the sixth and seventh steps define quantitative criteria and a data collection design, respectively. The seven steps are addressed below in Sections 8.1 through 8.7.

8.1 Problem Statement

The SLERA conducted at the CC ranges (Shaw, 2005) identified five inorganic compounds (antimony, copper, lead, mercury, and zinc) and four pesticides (DDD, DDE, DDT, and endrin) as COPECs in surface soil. In addition, lead and gamma-chlordane were identified as COPECs in sediment in the drainage features at the CC ranges. Lead was identified as a COPEC in surface water solely for the purpose of identifying a remedial action objective for surface water, and no site-specific investigation or testing will be conducted for surface water at the CC ranges.

The CC ranges Problem Formulation and conceptual site model (Chapters 1.0 – 7.0) suggest that exposure pathways for the inorganic and organic constituents identified as COPECs to terrestrial and aquatic receptors do exist and, therefore, require further study. The Problem Formulation process further identified the need for additional information to address questions related to constituent bioavailability, bioaccumulation potential, and site-specific toxicity.

Based on the findings of the SLERA and Problem Formulation, the objectives of the BERA for the CC ranges include the following:

- Collect site-specific data to address bioavailability and bioaccumulation potentials in lower trophic level organisms that form the basis of the terrestrial and aquatic food webs at the CC ranges.
- Collect site-specific data to address the existence and level of site-specific toxicity to terrestrial and aquatic receptors resulting from exposure to the COPECs.
- Determine the concentrations of the COPECs within the surface soils, and sediment at the CC ranges at which ecological hazards could occur.

- Provide data of sufficient quality to develop a technically defensible characterization of the potential ecological hazards at the CC ranges for use by risk managers in their acceptance or rejection of present and future ecological hazards posed by the COPECs in surface soil and sediment and, if necessary, develop ecologically-based cleanup criteria.

8.2 Decision Identification

The following decisions require site-specific data in order to address the issues identified in the Problem Statement presented in the previous section.

- Determine if the COPECs at the CC ranges are available for bio-uptake (i.e., bioavailable) in terrestrial or aquatic systems

- Determine what levels of COPECs in soil and sediment promote acute or chronic toxicity to terrestrial and aquatic receptors

- Determine if the COPECs bioaccumulate in the tissues of terrestrial invertebrates (e.g., earthworms), terrestrial plants, or benthic invertebrates, and if so, to what extent

- Determine whether the tissue burdens of COPECs in terrestrial invertebrates have the potential to pose adverse effects to higher trophic level organisms that utilize terrestrial invertebrates as a major food source

- Determine whether the tissue burdens of COPECs in terrestrial plants have the potential to pose adverse effects to higher trophic level organisms that utilize terrestrial plants as a major food source

- Determine whether benthic communities within the perennial and ephemeral drainage features at the CC ranges are adversely affected by exposure to COPECs in sediment

- Determine whether the concentrations of COPECs in emergent benthic invertebrates have the potential to pose adverse effects to higher trophic level organisms that utilize emergent benthic invertebrates as a major food source

- Develop constituent-specific cleanup goals for soil or sediment if the BERA concludes that there is the potential for unacceptable ecological hazard.

8.3 Decision Inputs

This step identifies the information required to support the decisions identified above. The information that will be required include the following:

- Surface soil concentrations of the 9 soil COPECs within the CC ranges soil

- Earthworm mortality based on earthworm no-observed-adverse-effect-level (NOAEL) and lowest-observed-adverse-effect-level (LOAEL) data for the CC ranges soil

- Earthworm growth based on total tissue weight measured at the termination of the toxicity test

- Bio-uptake and accumulation potential of soil COPECs in terrestrial invertebrates based on the ratio of soil COPEC concentrations to earthworm tissue concentrations

- Terrestrial plant growth and reproduction based on seed germination success, plant height, above-ground biomass, root length, and below-ground biomass

- Bio-uptake and accumulation potential in terrestrial plants based on the ratio of soil COPEC concentrations to plant tissue concentrations

- Total daily dose estimates of the soil COPECs in the terrestrial invertivorous shorttail shrew and American woodcock, as well as the omnivorous American robin and white-footed mouse (mg COPEC per unit of body mass per day)

- Estimated levels of concern to the invertivorous shorttail shrew and American woodcock as well as the omnivorous American robin and white-footed mouse based on modeled hazard quotient (HQ) values (estimated total daily dose/literature-based effect value)

- Sediment concentrations of lead and gamma-chlordane

- *Chironomus tentans* mortality based on exposure to various COPEC concentrations in sediment and derivation of sediment NOAEL and LOAEL values

- *Chironomus tentans* growth based on total tissue weight measured at the termination of the toxicity test

- Bio-uptake and accumulation potential based on the ratio of sediment COPEC concentrations in the perennial and ephemeral drainage features to *Chironomus sp.* tissue concentrations

- Total daily dose estimates of the COPECs in the riparian invertivorous little brown bat and marsh wren (mg COPEC per unit of body mass per day)

- Estimated levels of concern to the riparian invertivorous little brown bat and marsh wren based on modeled HQ values (estimated total daily dose/literature-based effect value)

- Benthic invertebrate community structure as determined by rapid bioassessment measurements.

These data will be used to help determine whether COPECs in surface soil or sediment at the CC ranges present (or might present) the potential to pose harm to ecological receptors. If potential hazards to ecological receptors are predicted using the information presented above, then this information may also be used to determine concentrations of COPECs in surface soil or sediment that may be more protective of the terrestrial and aquatic receptors at the CC ranges. The Uncertainty Analysis of the BERA will describe the limitations associated with various assessment techniques and in establishing protective COPEC concentrations.

8.4 Study Boundaries

Study boundaries define the spatial scale of the assessment at the CC ranges. In order to conduct a useful BERA, it is imperative to define the geographic and temporal boundaries of the potential hazard and to identify the target populations of interest. The CC ranges consist of four sets of ranges and associated parcels. The CC ranges SLERA identified the old field/upland forest terrestrial ecosystems at the CC ranges and the aquatic habitats associated with the perennial and ephemeral drainage features as the habitats with greatest potential risk given their quality, level of contamination, and receptors likely to be exposed to the COPECs. Therefore, the CC ranges BERA will focus on the old field/upland forest terrestrial habitats and aquatic ecosystems associated with these ranges.

Additionally, based on the historical nature of the contamination at the CC ranges, and the physical/chemical properties of the COPECs themselves, the concentrations of the COPECs in surface soil and sediment are not likely to change over time due to natural processes. Therefore, temporal variability of COPEC concentrations is not considered an important variable for these relatively static upland and riparian habitats.

The target populations for the BERA are the resident aquatic and terrestrial invertebrate communities and the wildlife feeding guilds that may be present within the bounds of the CC ranges. The target populations of greatest concern are the lower trophic level organisms (e.g., earthworms and benthic invertebrates) and the wildlife receptors that feed on them.

8.5 Decision Rule

The objective in developing specific decision rules is to construct theoretical "if…then…" statements relative to the ecological habitats, populations and COPECs. These statements can

then be used by risk managers in deciding whether to accept or reject the characterized ecological hazard and, if necessary, in generating ecological based cleanup goals. The decision rules proposed for the CC ranges BERA include the following:

- If COPECs in soils from the CC ranges cause terrestrial plant toxicity (e.g., reduced seed germination, reduced plant height, reduced above-ground biomass, reduced root length, reduced root biomass) which is statistically greater than plant toxicity in soils from a reference site, then there is the potential for unacceptable hazards to terrestrial plants at the CC ranges.

- If terrestrial plants exposed to soils from the CC ranges demonstrate statistically higher tissue concentrations of COPECs than terrestrial plants exposed to reference soils, then there is the potential for significant COPEC accumulation in terrestrial plant tissues.

- If COPECs in soils from the CC ranges cause acute earthworm toxicity which is statistically greater than earthworm toxicity in soils from a "reference" site, then there is the potential for unacceptable hazards to terrestrial invertebrate receptors at the CC ranges.

- If earthworms exposed to soils from the CC ranges demonstrate statistically higher tissue concentrations of COPECs than earthworms exposed to "reference" soils, then there is the potential for significant COPEC accumulation in terrestrial invertebrate tissues.

- If calculated doses of COPECs for terrestrial invertivorous mammals or birds are greater than literature-derived toxicity reference values, then there is the potential for hazard to terrestrial invertivorous mammals or birds at the CC ranges.

- If calculated doses of COPECs for terrestrial omnivorous mammals or birds are greater than literature-derived toxicity reference values, then there is the potential for hazard to terrestrial invertivorous mammals or birds at the CC ranges.

- If, based on the collective evaluation of the lines-of-evidence, COPECs are thought to pose hazards to terrestrial receptors at the CC ranges, then remedial goals for soil will be developed using the data collected during the BERA.

- If COPECs within the sediments of the perennial and ephemeral drainage features cause acute toxicity to the benthic invertebrate *Chironomus sp.*, which is statistically greater than toxicity from reference sediments, then there is the potential for hazard to emergent benthic invertebrates at the CC ranges.

- If chironomids exposed to sediment from the perennial and ephemeral drainage features demonstrate statistically higher tissue concentrations of COPECs than chironomids exposed to "reference" sediment, then there is the potential for significant COPEC accumulation in benthic invertebrate tissue.

- If the benthic community assemblage in the perennial and ephemeral drainage features at the CC ranges is significantly different than the benthic community assemblage in a non-impacted reference stream, then there is the potential for hazard to the benthic ecosystem in the on-site drainage features.

- If calculated doses of COPECs for riparian invertivorous mammals or birds are greater than literature-derived toxicity reference values, then there is the potential for hazard to riparian invertivorous mammals or birds at the CC ranges.

It is important to consider the role of background concentrations of COPECs when developing specific decision rules. It is possible that naturally-occurring concentrations of certain inorganic constituents in environmental media could result in a determination of unacceptable risk. Therefore, background will be considered within the context of each of the aforementioned decision rules.

It is also important to consider the effects that physical disturbance of the ecosystems at many of the CC ranges may have on the ecology. Routine maintenance activities at many of these ranges (e.g., grading of soil, removal of trees, continuous mowing of grass) have altered the ecosystems greatly from their "native" state and it may take many years for the "native" ecosystems to re-establish themselves. For instance, the clear-cutting of the forest at some of the CC ranges by the Alabama State Forestry Commission due to southern pine beetle infestation has greatly altered the forest ecology in large areas of the CC ranges and the former forest ecology is unlikely to regenerate in the near future. Therefore, physical disturbance of a site will also be considered when interpreting the results within the established decision rules.

8.6 Tolerable Limits on Decision Errors

Chemical and biological data collected as part of the BERA process will be collected in a manner such that they are representative of the abiotic media and biotic communities at the CC ranges. Since the collected data are only small sub-populations of the entire CC ranges, they can only be used to predict responses that may actually occur at these ranges under natural conditions. As such, these data must be interpreted with a level of confidence or probability that will be less than 100 percent error free. The objective in establishing tolerable probability limits is to generate the proper quantity and quality of data to meet the targeted limit. The decision data employed in the BERA will be of sufficient quantity and quality as to result in a decision confidence level of 95 percent. The tolerable limit will be made on statistical probabilities of less than 95 percent.

8.7 Design Optimization

The objective in design optimization is to develop a "resource-effective" sampling and analysis plan for generating data. The sampling and analysis plans presented in Appendices A and B have been optimized to ensure that the tolerable limits on decision errors will be met.

9.0 Site Investigation Tasks

The BERA for the CC ranges will focus on characterizing ecological hazards associated with the COPECs in surface soil within the old field/upland forest terrestrial habitats, as well as sediments within the perennial and ephemeral drainage features at the CC ranges. The site investigation tasks are directly linked to the assessment and measurement endpoints described within Section 7.0.

The principal objective of this site investigation is to outline a laboratory- and field-based approach to reduce uncertainty associated with the SLERA process and to provide risk managers with information to incorporate into site remedial decisions. It is important to note that the study outlined in this section is designed to provide a number of lines of evidence relative to present and future risks to terrestrial and aquatic receptors.

9.1 Terrestrial Receptor Study Design

The tasks involved in the study design for terrestrial ecosystems at the CC ranges are described in the following sections.

9.1.1 Soil Collection for Chemical Analysis

It is important to note that surface soils collected from the CC ranges are being used jointly in order to characterize toxicity and bioaccumulation potential of the soil COPECs. Surface soil (0 to 0.5 feet below ground surface) will be collected from ten (10) locations within the study areas of the CC ranges. Figure 9-1 presents the sample locations for each of the surface soil samples at the CC ranges. These ten locations represent the lead concentration gradient most likely to elicit adverse effects on terrestrial plants and terrestrial invertebrates at the CC ranges. Table 9-1 presents the proposed surface soil sampling locations and the historical COPEC concentrations detected at these locations. Locations for soil sampling will be identified using *in-situ* x-ray fluorescence technology. These surface soil samples will be analyzed for target analyte list (TAL) metals, volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, total organic carbon (TOC), pH, and grain size. Additionally, one soil sample will be collected from an area that is un-impacted by FTMC activities. This soil sample will represent the reference soil.

Analytical data from these surface soil samples will be used in conjunction with the perennial rye grass and earthworm toxicity test results to derive lowest-observed-adverse-effect-levels

(LOAEL) and no-observed-adverse-effect-levels (NOAEL) for terrestrial plants and earthworms. These soil data will also be used in the food web models to calculate total COPEC doses for the terrestrial invertivorous and omnivorous birds and mammals. Additionally, these soil data will be used in conjunction with the measured tissue concentrations of COPECs in terrestrial plants and earthworms to estimate soil-to-plant bioaccumulation factors ($BAF_{soil\text{-}to\text{-}plant}$) , and soil-to earthworm bioaccumulation factors ($BAF_{soil\text{-}to\text{-}worm}$).

Details of the collection methods, decontamination procedures, quality assurance/quality control, and other sampling procedures are presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002) and are summarized in Appendix A of this report.

9.1.2 Soil Collection for Earthworm Toxicity Tests

Assessment of the terrestrial invertebrate community associated with CC ranges surface soil will be based on quantitative laboratory testing using the detritivorous earthworm. The earthworm's bioturbative feeding habits, its ability to bioaccumulate the identified COPECs, and its critical position in terrestrial food webs make it an ideal surrogate to represent the terrestrial invertebrate community.

Earthworm toxicity testing will use surface soil from the ten locations identified in the previous section. As is the case with the soil samples for chemical analysis, soil samples for earthworm toxicity testing will be collected from ten locations within the study areas of the CC ranges and one location (reference location) outside of the influence of the CC ranges.

Quantification of possible adverse effects to terrestrial invertebrates and the potential for constituent transfer up the food chain to higher trophic level feeding guilds will be accomplished with the use of earthworm toxicity tests and tissue burden analysis. As a soil-boring detritivore, the common earthworm *(Eisenia fetida)* is an excellent sentinel not only to assess surface soil toxicity, but also to approximate food chain bioaccumulation potential. The 28-day earthworm growth and survival test recommended by the EPA (1996) and the American Society for Testing and Materials (ASTM) (1998) will be employed to assess the potential for hazards to members of this critical ecological trophic level. Earthworms have been shown to be acutely sensitive to soil-bound metal toxicity, and they represent a key prey item for mammalian and avian omnivores and invertivores.

Appendix A provides details relative to the EPA-recommended earthworm test. In brief, the 28-day static earthworm test will consist of exposing 10 worms per test chamber to 100 percent

undiluted soil from the CC ranges, soil from a reference location, and laboratory control soil. At the halfway point of the 28-day exposure period (14 days), mortality will be assessed within each test chamber/tray and dead worms will be removed. The 14-day mortality rate will be noted, along with soil pH and temperature. All live worms will be carefully re-introduced into the test soils. This same procedure will be followed upon completion of the test (28 days), with the exception that all living worms will be preserved in separate containers for COPEC whole-body burden analysis.

Since surface soil lead concentration gradients will be used as exposure gradients, as described in the sampling and analysis plan (Appendix A), data from earthworm tests will consist of 14-day NOAELs and LOAELs as well as 28-day NOAELs and LOAELs. In addition, whole-body tissue burdens for each of the COPECs will be determined.

Details of the collection methods, decontamination procedures, quality assurance/quality control, and other sampling procedures are presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002) and are summarized in Appendix A of this report.

9.1.3 Soil Collection for Plant Toxicity Tests

Assessment of the terrestrial plant community associated with CC ranges surface soils will be based on quantitative laboratory testing using the perennial ryegrass (*Lolium perenne*). Perennial ryegrass is a commonly used species for toxicity testing and is an ideal surrogate to represent a number of terrestrial plant species that could colonize the CC ranges.

Perennial ryegrass toxicity testing will use surface soil from the same ten site locations identified in the previous section for earthworm toxicity testing. As is the case with the soil samples for chemical analysis and earthworm toxicity testing, soil samples for perennial ryegrass toxicity testing will be collected from ten locations within the study areas of the CC ranges and one location (reference site) outside the influence of the CC ranges.

Quantification of possible adverse effects to terrestrial plants will be accomplished with the use of perennial ryegrass toxicity tests. As a terrestrial perennial plant, the perennial ryegrass *(Lolium perenne)* is an excellent sentinel to assess surface soil toxicity to terrestrial plant species. The perennial ryegrass toxicity test recommended by the ASTM (1998) will be employed to assess the potential for risk to members of this critical ecological trophic level (primary producers). Perennial ryegrass has been shown to be acutely sensitive to soil-bound metal toxicity, and it represents a key food item for herbivores and omnivores.

Appendix A provides details relative to the ASTM-recommended terrestrial plant toxicity tests. The 15-day perennial ryegrass toxicity test will consist of exposing 10 seeds per container to 100 percent undiluted soil from the CC ranges, soil from a reference location, and laboratory control soil. Five replicates for each of the ten surface soil locations, reference station, and laboratory controls, will be used in the perennial ryegrass toxicity tests. At the termination of the tests, the number of seedlings out of the total number of seeds planted that emerge above the soil will be counted and the germination success calculated. Additional measurements that will be taken include above-ground plant height, total above-ground biomass, below-ground root length, and total below-ground biomass. Additionally, abnormal patterns in growth and development, or abnormal plant morphology as compared to laboratory controls will be noted.

Since surface soil lead concentration gradients will be used as exposure gradients, as described in the sampling and analysis plan (Appendix A), data from perennial ryegrass tests will consist of 15-day NOAELs and LOAELs.

Details of the collection methods, decontamination procedures, quality assurance/quality control, and other sampling procedures are presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002) and are summarized in Appendix A of this report.

9.1.4 Plant Collection for COPEC Chemical Analysis

The potential for COPEC accumulation in terrestrial plant tissues will be assessed through the collection and chemical analysis of terrestrial plants. Herbivorous and omnivorous animals using the CC ranges as feeding grounds could potentially be exposed to COPECs via plant uptake and accumulation from the soil, with subsequent ingestion by animals. In order to assess this potential uptake and accumulation, terrestrial plants will be collected from the same locations identified in the previous section for soil sampling. As is the case with the soil samples for chemical analysis, terrestrial plant samples will be collected from ten locations within the study areas of the CC ranges and one location (reference site) outside the influence of the CC ranges.

The target plant species for collection at the CC ranges will be grass species since the perennial ryegrass (*Lolium perenne*) is the species that will be used in the laboratory toxicity tests. A grass species will be collected so that inferences can be made between the COPEC accumulation measured in the grass species from the CC ranges and the results of the terrestrial plant toxicity tests conducted in the laboratory using perennial ryegrass. An attempt will be made to collect

the same plant species at each sampling location. Only the above-ground portion of each plant will be sampled as most herbivorous and omnivorous species only consume the above-ground portions of most plants.

Plant tissue samples will be analyzed for the COPECs identified in surface soil at the CC ranges (TAL metals and pesticides) so that relationships can be drawn between concentrations of COPECs in surface soil and concentrations of COPECs in co-located terrestrial plant tissues. Details of the collection methods, decontamination procedures, quality assurance/quality control, and other sampling procedures are presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002) and are summarized in Appendix A of this report.

9.2 Aquatic Study Design

The freshwater stream habitat addressed in this quantitative BERA is the benthic habitat provided by the sediments in the perennial and ephemeral drainage features at the CC ranges. Sediments are critical to the viability of the overall stream ecosystem. Although lead was identified as a COPEC in surface water, it was identified as a COPEC solely for the purpose of identifying a remedial action objective for surface water and no site-specific studies for surface water are proposed for this BERA.

The aquatic study design is designed to address exposure and potential effects to receptors within and around the perennial and ephemeral drainage features as they flow through the CC ranges. Elevated levels of COPECs may or may not pose a risk to aquatic receptors depending upon their availability for uptake (bioavailability) from the underlying sediments. The study is therefore designed to assess bioavailability of the COPECs as well as the potential for acute or chronic toxicity and bioaccumulation to lower trophic level organisms that rely on the sediment for food and shelter.

9.2.1 Sediment Collection for Chemical Analysis

The sediment assessment will focus on characterization of potential hazards to benthic invertebrates as well as the upper trophic level organisms that may feed on them. In many ways, sediments represent a more definite assessment of potential hazards to aquatic systems because the receptors are generally less mobile and the COPECs can accumulate within depositional zones. Sediment samples will be collected from eight (8) locations within the perennial and ephemeral drainage features representing eight different lead concentrations detected in previous investigations at the CC ranges. Lead will be used as the indicator of COPEC concentrations because it has been detected in sediment at all of the CC ranges and has been used as an indicator

of contamination resulting from small arms range activity at other range complexes at FTMC. Figure 9-1 presents the proposed locations for sediment samples that will be collected to represent the range of COPEC concentrations in sediment at the CC ranges. These sediment samples will be used in the toxicity and bioaccumulation tests described in the following sections. Additionally, one sediment sample will be collected from an ephemeral drainage feature and one sediment sample will be collected from the perennial stream with similar substrate characteristics as the perennial and ephemeral drainage features but outside the influence of the CC ranges. These two sediment samples will be used as the reference sediments. Table 9-2 presents a summary of the proposed sediment sample locations and the historical COPEC concentrations detected in sediment at these locations. All sediment samples will be analyzed for TAL metals, VOCs, SVOCs, chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, total organic carbon, pH, and grain size. The COPEC concentrations in sediment samples from the perennial and ephemeral drainage features will also be utilized in the food web models that will be used to calculate the total dose of COPECs potentially received by riparian invertivorous mammals and birds.

Details of the collection methods, decontamination procedures, quality assurance/quality control, and other sampling procedures are presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002) and are summarized in Appendix B of this report.

9.2.2 Sediment Collection for Benthic Invertebrate Studies

In order to evaluate potential toxicity to benthic invertebrates, the standard 21-day *Chironomus tentans* survival and growth test will be conducted using eight (8) sediment samples collected from the perennial and ephemeral drainage features representative of the range of lead detected in sediment samples during previous investigations at the CC ranges. Test procedures for the sediment toxicity tests will be in accordance with the guidance set forth by EPA (2000c) in *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants With Freshwater Invertebrates* and ASTM (2000) in *Standard Guide for Determination of the Bioaccumulation of Sediment-Associated Contaminants by Benthic Invertebrates*. Appendix B references the test protocol for *Chironomus tentans* survival and growth tests. Five replicates for each of the eight sediment locations, reference stations, and laboratory controls, will be used for measurements of lethality and growth. All test organisms will be laboratory reared and less than 24-hours old at test initiation. At the termination of the test, all living chironomids will be preserved in separate containers for COPEC whole-body burden analysis.

Although collected sediments will not be "cut" with reference or laboratory grade sediments to generate a concentration series, the eight sediment sample locations will represent a gradient of lead concentrations previously detected at the CC ranges. This field-collected concentration gradient will allow investigators to generate sediment NOAELs and LOAELs based on mortality and growth. Additionally, the various sediment concentrations and corresponding chironomid tissue concentrations of COPECs will provide data for the calculation of sediment-to-invertebrate bioaccumulation factors ($BAF_{sed\text{-}to\text{-}invert}$).

In addition to laboratory-based sediment toxicity testing, direct in-field measurements of benthic invertebrate community structure using rapid bioassessment protocol (RBP II) will be conducted. Direct measurement of biological condition is considered the most effective means of evaluating cumulative impacts of non-point source contamination patterns such as those that may exist within the CC ranges. The presence or absence of habitat degradation assists in evaluating the present level of hazard or impact to existing receptors. When combined with laboratory toxicity testing, direct field measurements reduce uncertainty and strengthen the line-of-evidence relative to potential hazard levels.

For the perennial and ephemeral drainage features at the CC ranges, benthic macroinvertebrate surveys will be conducted within riffle and pool zones of the streams as they flow through the CC ranges. The riffle/run sample and the pool zone sample will be combined prior to analysis to form a single benthic macroinvertebrate sample for each sample location. It is important to note the need to carefully compare benthic communities in the areas of concern with comparable communities present in reference areas. For example, due to the ephemeral nature of some of the drainage features, a diverse and well-established in-faunal community may not be present. A similar reference area will be located in order to properly compare the benthic assemblage in the perennial and ephemeral drainage features to a similar stream un-impacted by small arms range activities. Great care will be taken in establishing off-site reference locations to ensure that the sediment grain size, TOC and stream bank makeup are comparable.

The advantages of employing benthic macroinvertebrates as a measure of risk to stream communities include the following (EPA, 1997):

- Macroinvertebrate assemblages are good indicators of localized conditions. Because many benthic macroinvertebrates have limited migration patterns or a sessile mode of life, they are particularly well-suited for assessing site-specific impacts (upstream-downstream studies).

- Macroinvertebrates integrate the effects of short-term environmental variations. Most species have a complex life cycle of approximately one year or more. Sensitive life stages will respond quickly to stress; the overall community will respond more slowly.

- Degraded conditions can often be detected by an experienced biologist with only a cursory examination of the benthic assemblage. Macroinvertebrates are relatively easy to identify to family; many "intolerant" taxa can be identified to lower taxonomic levels with ease.

- Benthic macroinvertebrate assemblages are made up of species that constitute a broad range of trophic levels and pollution tolerances, thus providing strong information for interpreting cumulative effects.

- Sampling is relatively easy, requires few people and inexpensive gear, and has no detrimental effect on the resident biota.

- Benthic macroinvertebrates serve as a primary food source for many recreationally and commercially important fish.

- Benthic macroinvertebrates are abundant in most streams. Many small streams (i.e., the perennial stream at the CC ranges), may support a diverse macroinvertebrate fauna, but only support a limited fish community.

Observations that will be made during the RBP benthic invertebrate survey will include substrate type, surrounding land use, evidence of erosion and pollutant sources, vegetative stream canopy, and other relevant data. In addition to benthic sampling, which will consist of one kick net sample and one course particulate organic matter (CPOM) sample, *in situ* water quality parameters (temperature, conductivity, dissolved oxygen, pH) will be measured with the use of a Horiba U-10, or similar, water quality instrument. Measurements will be taken at mid-stream at approximately mid-depth. Water quality parameters will be obtained prior to any sampling activities in the stream.

Two macroinvertebrate samples will be collected at each sampling station; the riffle/run sample will be collected with a kick net and the CPOM sample will be collected by hand. The kick net sample and the CPOM sample will be combined prior to analysis to form one benthic macroinvertebrate sample for each sample location. All macroinvertebrate samples will be transported to an appropriate laboratory for identification and analysis.

The kick net sample provides data as to the abundance of the scraper and filtering functional feeding groups and is generally collected in a riffle and a run area of the stream. The riffle and

the run sample will be composited in the field for processing as one sample per location. The kick net consists of a 0.9 mm mesh bag attached to a rectangular 8- by 18-inch frame mounted on a handle. The use of the sampler is described as follows:

1. The sampler is positioned securely on the substrate with the opening of the net facing upstream.

2. An area of one square-meter immediately upstream of the sampler is disturbed by overturning and scraping rocks and large stones by shifting the feet to dislodge clinging or attached organisms. Any rocks or other large items that have been swept into the net are examined to ensure that organism removal is complete.

3. The remaining sediment is agitated with the feet to dislodge epibenthic and burrowing organisms.

All organisms and debris such as sticks and leaves will be removed from the kick net bag and placed into a container with 95 percent ethanol to preserve the organisms.

One CPOM sample will be collected at each location from depositional areas of little or no current velocity in the stream. The CPOM sample, which provides data as to the abundance of the shredder feeding group, will be collected by hand including a composite variety of leaves, twigs, bark and other fragments. The collected material and organisms will be placed into a sample container with 95 percent ethanol.

Organisms will be identified in the laboratory to Family level or to the lowest practical taxon. Identification of organisms will be made using published keys such as those developed by Merritt and Cummins (1984), Peckarsky et al. (1990), and Pennak (1989 and 1978). Each family of organisms identified at each location will be placed into separate vials containing ethanol as a preservative in order to assemble a reference collection for the project.

Eight metrics will be calculated from the benthic macroinvertebrate data obtained at each sampling station in accordance with the procedures outlined in EPA's Rapid Bioassessment Protocol II (EPA, 1999c). Each metric result will be given a score based on percent comparability to a reference station. Scores will be totaled, and a Biological Condition Category will be assigned based on percent comparability with the reference station score. The following metrics will be calculated:

Metric 1: Taxa Richness. Taxa richness will be calculated by counting the number of taxa present in the sample. In general, taxa richness increase with increasing water quality.

Metric 2: Modified Family Biotic Index. This index, developed by Hilsenhoff (1988), summarizes the tolerances of the benthic arthropod community to organic pollutants with a single value. Tolerance values used in the calculation of the Family Biotic Index (FBI) were obtained from Hilsenhoff (1988) and Bodek et al. (1988). The FBI is calculated by multiplying the number of organisms in each taxon by the tolerance value for that taxon, summing the products, and dividing by the total number of organisms in the sample for which an index will be calculated. Values for the FBI range from 0.00 to 10.00 with higher values corresponding to greater levels of organic pollution as shown in the following:

Family Biotic Index	Water Quality	Degree of Organic Pollution
3.5	Excellent	Organic pollution unlikely
3.51-4.5	Very good	Possible slight organic pollution
4.51-5.5	Good	Some organic pollution probable
5.51-6.5	Fair	Fairly substantial pollution likely
6.51-7.5	Fairly poor	Substantial pollution likely
7.51-8.5	Poor	Very substantial pollution likely
8.51-10	Very poor	Severe organic pollution likely

Metric 3: Ratio of Scraper and Filtering Collector Functional Feeding Groups. The relative abundance of scrapers and filtering collectors in the riffle/run habitat is an indicator of the food sources available. Functional feeding group designations for the taxa identified will be obtained from Merritt and Cummins (1984) and Barbour et al., (1999). This metric is calculated by dividing the relative abundance of scrapers by the relative abundance of filter feeding organisms.

Metric 4: Ratio of EPT and Chironomidae Abundances. The ratio of Ephemeroptera, Plecoptera and Trichoptera (EPT) and chironomidae abundance will be calculated by dividing the relative abundance of EPT taxa by the relative abundance of chironomidae. The ratio of EPT to chironomidae will indicate if there is an even distribution between the pollution sensitive EPT taxa and more pollution tolerant chironomidae.

Metric 5: Percent Contribution of Dominant Taxon. The percent contribution of the dominant taxon will be calculated by dividing the abundance of the taxon which is numerically

dominant by the total number of organisms in the sample. A low percent contribution of the dominant family indicates a balanced community. Factors influencing this percentage include environmental stress, habitat quality, and life histories of the organisms collected in the sample.

Metric 6: EPT Index. This result of the EPT index is determined by counting the number of distinct taxa within the groups Ephemeroptera, Plecoptera, and Trichoptera. The EPT index usually increases with increasing water quality as EPT taxa are generally considered pollution sensitive.

Metric 7: Community Similarity Index. This index evaluates the benthic populations at specific locations relative to populations present at a "reference" location. For this metric, the upstream and/or downstream stations (where pesticide related effects are not expected) will be used as the reference stations for each of the IRP sits. The community loss (CL) index is calculated by subtracting the number of taxa common to both locations (B) from the number of taxa present at the reference location R divided by the number of taxa present at the potential impact location (I), as follows:

$$CL = \frac{R - B}{I}$$

Metric 8: Ratio of Shredder Functional Feeding Group and Total Number of Individuals Collected. The ratio of the relative abundance of shredders to the abundance of all other functional feeding groups will be calculated by dividing the relative abundance of shredders by the total number of organisms in the sample. The abundance of shredders in comparison to other functional feeding groups can be influenced by climate, seasonality, and vegetation within the riparian zone, as well as levels of toxicants adsorbed to CPOM while in the riparian zone, or adsorption of toxicants to the CPOM while it is in the water.

10.0 Data Analysis, Validation, and Interpretation

Data usefulness is paramount relative to the BERA and related testing and analysis. The principal objective in the Study Design is to ensure that the hypotheses are effectively tested and rejected or accepted with a high degree of confidence. A summary of the statistical methods is provided below, and a discussion of hypothetical results follows. These hypothetical results should assist the reader in better understanding the usefulness of the collected data as they relate to characterizing risk to terrestrial receptors within and around the CC ranges, as well as the aquatic data associated with the perennial and ephemeral drainage features.

10.1 Data Analysis and Validation

Figure 9-1 presents the sample locations for ten surface soil samples and six sediment samples that will be collected at the CC ranges. The hypothetical results for the earthworm toxicity tests presented in Table 10-1 would indicate that the NOAEL for lead in soil may be as low as 800 mg/kg lead. Results such as these could be useful to risk managers in setting possible soil cleanup goals.

The hypothetical results for the perennial ryegrass toxicity tests presented in Table 10-2 indicate that the NOAEL for lead in soil for terrestrial plants may be as low as 300 mg/kg lead. Results such as these could be useful to risk managers in setting possible soil cleanup goals.

The sediment chironomid toxicity tests in conjunction with the RBP will provide lines of evidence regarding potential sediment toxicity (e.g., NOAEL and LOAEL values for sediment) and also quantitative comparisons of the benthic invertebrate assemblages in the perennial and ephemeral drainage features with benthic invertebrate assemblages from an un-impacted reference stream. The results of these tests will be used, along with other lines of evidence to develop ecological sediment cleanup goals, if deemed appropriate.

The overall objective in conducting the field- and laboratory-based studies is to test the null hypotheses stated in Chapter 7.0. Each hypothesis will be accepted or rejected based on findings from the relevant toxicity test or field measurement. Acceptance or rejection of each hypothesis will be instrumental in characterizing ecological risks associated with the surface soil and sediment at the CC ranges.

NOAEL and LOAEL values for soil and sediment COPECs will be derived using Dunnett's procedure or Steel's Many-One Rank Test. Dunnett's procedure is a parametric test that assumes that observations within treatments are independent and normally distributed and that the variance of the observations is homogenous across all toxicant concentrations. The Shapiro-Wilk's test will be used to test for normality in order to decide whether to use parametric (Dunnett's) or nonparametric (Steel's Many-One Rank) analyses. In order to test the variances of the data obtained from each toxicant concentration and the control, Bartlett's test for variance will be employed.

It is important to note that the field-collected soil samples will not be cut or diluted into a dilution series but will be tested as 100 percent "un-cut" samples. Derivation of toxicity response curves in the form of NOAELs and LOAELs will be done via the lead concentration gradient. By collecting soil samples with ten different concentrations of lead, a gradient series will be present and appropriate toxicity response curves can be computed. Therefore, Dunnet's Procedure (for parametric distributions) or Steel's Many-One Rank Test (for nonparametric distributions) can be applied.

In addition to deriving toxicant dose-response curves (i.e., NOAEL, LOAEL), it is critical to apply Analysis of Variance (ANOVA) tests to determine if soil and sediment samples differ from off-site reference samples, thus dictating whether null hypotheses are accepted or rejected. A significance level of $p = 0.05$ will be adopted as a probability of committing a Type I or Type II error. In comparing toxicity or biomeasurement results, single and nested ANOVAs will be conducted coincident with appropriate normality and variance testing.

Data validation will be conducted in accordance with the Installation Wide Sampling and Analysis Plan (IT, 2002).

10.2 Data Interpretation

Interpretation of bioassay results is dependent upon bracketing a response or effect level and a no effect level. Effects will be measured via toxicity responses within a specified exposure period, depending on the exposure medium and test species. At a confidence level of 95 percent ($p \leq 0.05$), test responses consisting of acute toxicity will be compared to reference soil or sediment responses. Test chambers that are statistically different from reference chambers will be characterized as "effect concentrations," while those exhibiting no significant difference will be listed as "no-effect concentrations." The highest no-effect concentration and the lowest effect concentration will be reported as the NOAEL and LOAEL, respectively.

A second use of the data relates to COPEC concentrations measured within tissues following completion of exposure periods. Organisms from each replicate chamber will be tested as separate and distinct composite samples. The mean concentration and 95 percent upper confidence limit (UCL) for each exposure concentration will be used to derive body burden concentrations which will then be used as input values for the food chain models as described in Section 5.0. These models, representing the various terrestrial and riparian trophic levels, will then be employed for HQ derivations.

11.0 Data Management Plan

The primary data management activities for the CC ranges BERA will include:

- Data transfer from field and laboratory activities to a project filing system
- Data management to ensure that data are stored and output in a manner that continues the chain of custody
- Review of requirements to ensure that plans for data collection were fulfilled
- Validation of analytical data that will report data to be used for treatment interpretation activities
- Evaluation of analytical and field data resulting in a report of guidance to be followed for using project data in treatment interpretation
- Reporting functions, which may include outputting data for report tables, statistical analysis, interpretation of data, and electronic transfer.

The FTMC ShawView^{Tm} database will be used for data management. A series of programs allows electronic reporting of data. The laboratory is responsible for reporting data in both hard copy and electronic data deliverable formats.

11.1 Records Control

All project documentation and original reports will be maintained in a central file for the project.

11.2 Document Filing and Access

At least two copies of all data forms and deliverables will be generated during the project and sorted at different locations. Wherever practical, original forms will be archived at the Shaw's office in Knoxville, Tennessee, and the laboratory and field personnel will retain copies. Analytical data, hard copy, and electronic files will be archived at least seven years by the laboratory.

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ATTACHMENT 1

LIST OF ABBREVIATIONS AND ACRONYMS

List of Abbreviations and Acronyms

2-ADNT	2-amino-4,6-dinitrotoluene
4-ADNT	4-amino-2,6-dinitrotoluene
2,4-D	2,4-dichlorophenoxyacetic acid
2,4,5-T	2,4,5-trichlorophenoxyacetic acid
2,4,5-TP	2,4,5-trichlorophenoxypropionic acid
3D	3D International Environmental Group
AB	ambient blank
AbB3	Anniston gravelly clay loam, 2 to 6 percent slopes, severely eroded
AbC3	Anniston gravelly clay loam, 6 to 10 percent slopes, severely eroded
AbD3	Anniston and Allen gravelly clay loams, 10 to 15 percent slopes, eroded
ABLM	adult blood lead model
Abs	skin absorption
ABS	dermal absorption factor
AC	hydrogen cyanide
ACAD	AutoCadd
AcB2	Anniston and Allen gravelly loams, 2 to 6 percent slopes, eroded
AcC2	Anniston and Allen gravelly loams, 6 to 10 percent slopes, eroded
AcD2	Anniston and Allen gravelly loams, 10 to 15 percent slopes, eroded
AcE2	Anniston and Allen gravelly loams, 15 to 25 percent slopes, eroded
ACGIH	American Conference of Governmental Industrial Hygienists
AdE	Anniston and Allen stony loam, 10 to 25 percent slope
ADEM	Alabama Department of Environmental Management
ADPH	Alabama Department of Public Health
AEC	U.S. Army Environmental Center
AEDA	ammunition, explosives, and other dangerous articles
AEL	airborne exposure limit
AET	adverse effect threshold; apparent effects threshold
AF	soil-to-skin adherence factor
AHA	ammunition holding area
AL	Alabama
ALARNG	Alabama Army National Guard
ALAD	δ-aminolevulinic acid dehydratase
ALDOT	Alabama Department of Transportation
amb.	amber
amsl	above mean sea level
ANAD	Anniston Army Depot
ANOVA	Analysis of Variance
AOC	area of concern
AOI	area of investigation
AP	armor piercing
APEC	areas of potential ecological concern
APT	armor-piercing tracer
AR	analysis request
ARAR	applicable or relevant and appropriate requirement
AREE	area requiring environmental evaluation
AS/SVE	air sparging/soil vapor extraction
ASP	Ammunition Supply Point
ASR	Archives Search Report
AST	aboveground storage tank
ASTM	American Society for Testing and Materials
AT	averaging time
atm-m^3/mol	atmospheres per cubic meter per mole
ATSDR	Agency for Toxic Substances and Disease Registry
ATV	all-terrain vehicle
AUF	area use factor
AWARE	Associated Water and Air Resources Engineers, Inc.
AWQC	ambient water quality criteria
AWWSB	Anniston Water Works and Sewer Board
'B'	Analyte detected in laboratory or field blank at concentration greater than the reporting limit (and greater than zero)
BAF	bioaccumulation factor
BBGR	Baby Bains Gap Road
BCF	blank correction factor; bioconcentration factor
BCT	BRAC Cleanup Team
BERA	baseline ecological risk assessment
BEHP	bis(2-ethylhexyl)phthalate
BFB	bromofluorobenzene
BFE	base flood elevation
BFM	bonded fiber matrix
BG	Bacillus globigii
BGR	Bains Gap Road
bgs	below ground surface
BHC	hexachlorocyclohexane
BHHRA	baseline human health risk assessment
BIRTC	Branch Immaterial Replacement Training Center
bkg	background
bls	below land surface
BOD	biological oxygen demand
Bp	soil-to-plant biotransfer factors
BRAC	Base Realignment and Closure
Braun	Braun Intertec Corporation
BSAF	biota-to-sediment accumulation factors
BSC	background screening criterion
BSV	background screening values
BTAG	Biological Technical Assistance Group
BTEX	benzene, toluene, ethyl benzene, and xylenes
BTOC	below top of casing
BTV	background threshold value
BW	biological warfare; body weight
BZ	breathing zone; 3-quinuclidinyl benzilate
C	ceiling limit value
Ca	carcinogen
$CaCO_3$	calcium carbonate
CAA	Clean Air Act
CAB	chemical warfare agent breakdown products
CACM	Chemical Agent Contaminated Media
CAIS	chemical agent identification set
CAMU	corrective action management unit
CBR	chemical, biological, and radiological
CCAL	continuing calibration
CCB	continuing calibration blank
CCV	continuing calibration verification
CD	compact disc
CDTF	Chemical Defense Training Facility
CEHNC	U.S. Army Engineering and Support Center, Huntsville
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CERFA	Community Environmental Response Facilitation Act
CESAS	Corps of Engineers South Atlantic Savannah
CF	chloroform
CF	conversion factor
CFC	chlorofluorocarbon
CFDP	Center for Domestic Preparedness
CFR	Code of Federal Regulations
CG	phosgene (carbonyl chloride)
CGI	combustible gas indicator
ch	inorganic clays of high plasticity
CHPPM	U.S. Army Center for Health Promotion and Preventive Medicine
CIH	Certified Industrial Hygienist
CK	cyanogen chloride
cl	inorganic clays of low to medium plasticity
Cl	chlorinated
CLP	Contract Laboratory Program
cm	centimeter
CN	chloroacetophenone
CNB	chloroacetophenone, benzene, and carbon tetrachloride
CNS	chloroacetophenone, chloropicrin, and chloroform
CO	carbon monoxide
CO_2	carbon dioxide
Co-60	cobalt-60
CoA	Code of Alabama
COC	chain of custody; chemical of concern
COE	Corps of Engineers
Con	skin or eye contact
COPC	chemical of potential concern
COPEC	constituent of potential ecological concern
CPOM	coarse particulate organic matter
CPSS	chemicals present in site samples
CQCSM	Contract Quality Control System Manager
CRDL	contract-required detection limit
CRL	certified reporting limit
CRQL	contract-required quantitation limit
CRZ	contamination reduction zone
Cs-137	cesium-137
CS	ortho-chlorobenzylidene-malononitrile

CSEM	conceptual site exposure model
CSM	conceptual site model
CT	central tendency
CT	carbon tetrachloride
ctr.	container
CWA	chemical warfare agent; Clean Water Act
CWM	chemical warfare materiel; clear, wide mouth
CX	dichloroformoxime
'D'	duplicate; dilution
D&I	detection and identification
DAAMS	depot area agent monitoring station
DAF	dilution-attenuation factor
DANC	decontamination agent, non-corrosive
ºC	degrees Celsius
ºF	degrees Fahrenheit
DCA	dichloroethane
DCE	dichloroethene
DD	Defense Department
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
DEH	Directorate of Engineering and Housing
DEHP	di(2-ethylhexyl)phthalate
DEP	depositional soil
DFTPP	decafluorotriphenylphosphine
DI	deionized
DID	data item description
DIMP	di-isopropylmethylphosphonate
DM	dry matter; adamsite
DMBA	dimethylbenz(a)anthracene
DMMP	dimethylmethylphosphonate
DNAPL	dense nonaqueous-phase liquid
DNT	dinitrotoluene
DO	dissolved oxygen
DOD	U.S. Department of Defense
DOJ	U.S. Department of Justice
DOT	U.S. Department of Transportation
DP	direct-push
DPDO	Defense Property Disposal Office
DPT	direct-push technology
DQO	data quality objective
DRMO	Defense Reutilization and Marketing Office
DRO	diesel range organics
DS	deep (subsurface) soil
DS2	Decontamination Solution Number 2
DSERTS	Defense Site Environmental Restoration Tracking System
DWEL	drinking water equivalent level
E&E	Ecology and Environment, Inc.
EB	equipment blank
EBC	Eastern Bypass Corridor
EBS	environmental baseline survey
EBV	EBV Explosives Environmental Co.
EC_{20}	effects concentration for 20 percent of a test population
EC_{50}	effects concentration for 50 percent of a test population
ECBC	Edgewood Chemical Biological Center
ED	exposure duration
EDD	electronic data deliverable
EF	exposure frequency
EDQL	ecological data quality level
EE/CA	engineering evaluation and cost analysis
Eh	oxidation-reduction potential
Elev.	elevation
EM	electromagnetic
EMI	Environmental Management Inc.
EM31	Geonics Limited EM31 Terrain Conductivity Meter
EM61	Geonics Limited EM61 High-Resolution Metal Detector
EOD	explosive ordnance disposal
EODT	explosive ordnance disposal team
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
EPIC	Environmental Photographic Interpretation Center
EPRI	Electrical Power Research Institute
EPT	Ephemeroptera, Plecoptera, Trichoptera
ER	equipment rinsate
ERA	ecological risk assessment
ER-L	effects range-low
ER-M	effects range-medium
ESE	Environmental Science and Engineering, Inc.
ESL	ecological screening level
ESMP	Endangered Species Management Plan
ESN	Environmental Services Network, Inc.
ESV	ecological screening value
ET	exposure time
EU	exposure unit
Exp.	Explosives
EXTOXNET	Extension Toxicology Network
E-W	east to west
EZ	exclusion zone
FAR	Federal Acquisition Regulations
FB	field blank
FBI	Family Biotic Index
FD	field duplicate
FDC	Former Decontamination Complex
FDA	U.S. Food and Drug Administration
Fe^{+3}	ferric iron
Fe^{+2}	ferrous iron
FedEx	Federal Express, Inc.
FEMA	Federal Emergency Management Agency
FFCA	Federal Facilities Compliance Act
FFE	field flame expedient
FFS	focused feasibility study
FI	fraction of exposure
Fil	filtered
Flt	filtered
FMDC	Fort McClellan Development Commission
FML	flexible membrane liner
f_{oc}	fraction organic carbon
FOMRA	Former Ordnance Motor Repair Area
FOST	Finding of Suitability to Transfer
Foster Wheeler	Foster Wheeler Environmental Corporation
FR	Federal Register
Frtn	fraction
FS	field split; feasibility study; fuming sulfuric acid
FSP	field sampling plan
ft	feet
ft/day	feet per day
ft/ft	feet per foot
ft/yr	feet per year
FTA	Fire Training Area
FTMC	Fort McClellan
FTRRA	FTMC Reuse & Redevelopment Authority
g	gram
g/m^3	gram per cubic meter
G-856	Geometrics, Inc. G-856 magnetometer
G-858G	Geometrics, Inc. G-858G magnetic gradiometer
GAF	gastrointestinal absorption factor
gal	gallon
gal/min	gallons per minute
GB	sarin (isopropyl methylphosphonofluoridate)
gc	clay gravels; gravel-sand-clay mixtures
GC	gas chromatograph
GCL	geosynthetic clay liner
GC/MS	gas chromatograph/mass spectrometer
GCR	geosynthetic clay liner
GFAA	graphite furnace atomic absorption
GIS	Geographic Information System
gm	silty gravels; gravel-sand-silt mixtures
gp	poorly graded gravels; gravel-sand mixtures
gpm	gallons per minute
GPR	ground-penetrating radar
GPS	global positioning system
GRA	general response action
GS	ground scar
GSA	General Services Administration; Geologic Survey of Alabama

GSBP	Ground Scar Boiler Plant
GSSI	Geophysical Survey Systems, Inc.
GST	ground stain
GW	groundwater
gw	well-graded gravels; gravel-sand mixtures
H&S	health and safety
HA	hand auger
HC	mixture of hexachloroethane, aluminum powder, and zinc oxide (smoke producer)
HCl	hydrochloric acid
HD	distilled mustard (bis-[dichloroethyl]sulfide)
HDPE	high-density polyethylene
HE	high explosive
HEAST	Health Effects Assessment Summary Tables
Herb.	herbicides
HHRA	human health risk assessment
HI	hazard index
HN	hydrogen mustard
H_2O_2	hydrogen peroxide
HPLC	high-performance liquid chromatography
HNO_3	nitric acid
HQ	hazard quotient
HQ_{screen}	screening-level hazard quotient
hr	hour
HRC	hydrogen releasing compound
HSA	hollow-stem auger
HSDB	Hazardous Substance Data Bank
HTRW	hazardous, toxic, and radioactive waste
'I'	out of control, data rejected due to low recovery
IASPOW	Impact Area South of POW Training Facility
IATA	International Air Transport Authority
ICAL	initial calibration
ICB	initial calibration blank
ICP	inductively-coupled plasma
ICRP	International Commission on Radiological Protection
ICS	interference check sample
ID	inside diameter
IDL	instrument detection limit
IDLH	immediately dangerous to life or health
IDM	investigative-derived media
IDW	investigation-derived waste
IEUBK	Integrated Exposure Uptake Biokinetic
IF	ingestion factor; inhalation factor
ILCR	incremental lifetime cancer risk
IMPA	isopropylmethyl phosphonic acid
IMR	Iron Mountain Road
in.	inch
Ing	ingestion
Inh	inhalation
IP	ionization potential
IPS	International Pipe Standard
IR	ingestion rate
IRDMIS	Installation Restoration Data Management Information System
IRIS	Integrated Risk Information Service
IRP	Installation Restoration Program
IS	internal standard
ISCP	Installation Spill Contingency Plan
IT	IT Corporation
ITEMS	IT Environmental Management System™
ITRC	Interstate Trade and Regulatory Council
IWWP	installation-wide work plan
'J'	estimated concentration
JeB2	Jefferson gravelly fine sandy loam, 2 to 6 percent slopes, eroded
JeC2	Jefferson gravelly fine sandy loam, 6 to 10 percent slopes, eroded
JfB	Jefferson stony fine sandy loam, 0 to 10 percent slopes have strong slopes
JPA	Joint Powers Authority
K	conductivity
K_d	soil-water distribution coefficient
kg	kilogram
KeV	kilo electron volt
K_{oc}	organic carbon partioning coefficient
K_{ow}	octonal-water partition coefficient
$KMnO_4$	potassium permanganate
L	liter; Lewisite (dichloro-[2-chloroethyl]sulfide)
L/kg/day	liters per kilogram per day
l	liter
LAW	light anti-tank weapon
lb	pound
LBP	lead-based paint
LC	liquid chromatography
LCS	laboratory control sample
LC_{50}	lethal concentration for 50 percent population tested
LD_{50}	lethal dose for 50 percent population tested
LEL	lower explosive limit
LOAEL	lowest-observed-advserse-effects-level
LOEC	lowest-observable-effect-concentration
LRA	land redevelopment authority
LT	less than the certified reporting limit
LUC	land-use control
LUCAP	land-use control assurance plan
LUCIP	land-use control implementation plan
max	maximum
MB	method blank
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MCPA	4-chloro-2-methylphenoxyacetic acid
MCPP	2-(2-methyl-4-chlorophenoxy)propionic acid
MCS	media cleanup standard
MD	matrix duplicate
MDC	maximum detected concentration
MDCC	maximum detected constituent concentration
MDL	method detection limit
m	meter
mg	milligrams
mg/kg	milligrams per kilogram
mg/kg/day	milligram per kilogram per day
mg/kgbw/day	milligrams per kilogram of body weight per day
mg/L	milligrams per liter
mg/m^3	milligrams per cubic meter
mh	inorganic silts, micaceous or diatomaceous fine, sandy or silt soils
MHz	megahertz
m/yr	meters per year
µg/g	micrograms per gram
µg/kg	micrograms per kilogram
µg/L	micrograms per liter
µmhos/cm	micromhos per centimeter
MEC	munitions and explosives of concern
MeV	mega electron volt
min	minimum
MINICAMS	miniature continuous air monitoring system
ml	inorganic silts and very fine sands
mL	milliliter
mm	millimeter
MM	mounded material
MMBtu/hr	million Btu per hour
MNA	monitored natural attenuation
MnO_4-	permanganate ion
MOA	Memorandum of Agreement
MOGAS	motor vehicle gasoline
MOUT	Military Operations in Urban Terrain
MP	Military Police
MPA	methyl phosphonic acid
MPC	maximum permissible concentration
MPM	most probable munition
MQL	method quantitation limit
MR	molasses residue
MRL	method reporting limit
MS	matrix spike
mS/cm	millisiemens per centimeter
mS/m	millisiemens per meter
MSD	matrix spike duplicate; minimum separation distance
MTBE	methyl tertiary butyl ether
msl	mean sea level
MtD3	Montevallo shaly, silty clay loam, 10 to 40 percent slopes , severely eroded
mV	millivolts

MW	monitoring well
MWI&MP	Monitoring Well Installation and Management Plan
Na	sodium
NA	not applicable; not available
NAD	North American Datum
NAD83	North American Datum of 1983
$NaMnO_4$	sodium permanganate
NAVD88	North American Vertical Datum of 1988
NAS	National Academy of Sciences
NCEA	National Center for Environmental Assessment
NCP	National Contingency Plan
NCRP	National Council on Radiation Protection and Measurements
ND	not detected
NE	no evidence; northeast
ne	not evaluated
NEW	net explosive weight
NFA	No Further Action
NG	National Guard
NGP	National Guardsperson
ng/L	nanograms per liter
NGVD	National Geodetic Vertical Datum
Ni	nickel
NIC	notice of intended change
NIOSH	National Institute for Occupational Safety and Health
NIST	National Institute of Standards and Technology
NLM	National Library of Medicine
NO_3^-	nitrate
NOEC	no-observable-effect-concentration
NPDES	National Pollutant Discharge Elimination System
NPW	net present worth
No.	number
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effects-level
NR	not requested; not recorded; no risk
NRC	National Research Council
NRCC	National Research Council of Canada
NRHP	National Register of Historic Places
NRT	near real time
ns	nanosecond
N-S	north to south
NS	not surveyed
NSA	New South Associates, Inc.
nT	nanotesla
nT/m	nanoteslas per meter
NTU	nephelometric turbidity unit
nv	not validated
O_2	oxygen
O_3	ozone
O&G	oil and grease
O&M	operation and maintenance
OB/OD	open burning/open detonation
OD	outside diameter
OE	ordnance and explosives
oh	organic clays of medium to high plasticity
OH•	hydroxyl radical
ol	organic silts and organic silty clays of low plasticity
OP	organophosphorus
ORC	Oxygen Releasing Compound
ORP	oxidation-reduction potential
OSHA	Occupational Safety and Health Administration
OSWER	Office of Solid Waste and Emergency Response
OVM-PID/FID	organic vapor meter-photoionization detector/flame ionization detector
OWS	oil/water separator
oz	ounce
PA	preliminary assessment
PAH	polynuclear aromatic hydrocarbon
PARCCS	precision, accuracy, representativeness, comparability, completeness, and sensitivity
Parsons	Parsons Engineering Science, Inc.
Pb	lead
PBMS	performance-based measurement system
PC	permeability coefficient
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-p-dioxins
PCDF	polychlorinated dibenzofurans
PCE	perchloroethene
PCP	pentachlorophenol
PDS	Personnel Decontamination Station
PEF	particulate emission factor
PEL	permissible exposure limit
PERA	preliminary ecological risk assessment
PERC	perchloroethene
PES	potential explosive site
Pest.	pesticides
PETN	pentaerythritoltetranitrate
PFT	portable flamethrower
PG	professional geologist
PID	photoionization detector
PkA	Philo and Stendal soils local alluvium, 0 to 2 percent slopes
PM	project manager
POC	point of contact
POL	petroleum, oils, and lubricants
POTW	publicly owned treatment works
POW	prisoner of war
PP	peristaltic pump; Proposed Plan
ppb	parts per billion
ppbv	parts per billion by volume
PPE	personal protective equipment
ppm	parts per million
PPMP	Print Plant Motor Pool
ppt	parts per thousand
PR	potential risk
PRA	preliminary risk assessment
PRG	preliminary remediation goal
PS	chloropicrin
PSSC	potential site-specific chemical
pt	peat or other highly organic silts
PVC	polyvinyl chloride
QA	quality assurance
QA/QC	quality assurance/quality control
QAM	quality assurance manual
QAO	quality assurance officer
QAP	installation-wide quality assurance plan
QC	quality control
QST	QST Environmental, Inc.
qty	quantity
Qual	qualifier
QuickSilver	QuickSilver Analytics, Inc.
R	rejected data; resample; retardation factor
R^2	coefficient of determination
R&A	relevant and appropriate
RA	remedial action
RAO	remedial action objective
RBC	risk-based concentration; red blood cell
RBP	Rapid Bioassessment Protocol
RBRG	risk-based remedial goal
RCRA	Resource Conservation and Recovery Act
RCWM	Recovered Chemical Warfare Material
RD	remedial design
RDX	cyclotrimethylenetrinitramine
ReB3	Rarden silty clay loams
REG	regular field sample
REL	recommended exposure limit
RFA	request for analysis
RfC	reference concentration
RfD	reference dose
RGO	remedial goal option
RI	remedial investigation
RL	reporting limit
RME	reasonable maximum exposure
ROD	Record of Decision
RPD	relative percent difference
RR	range residue
RRF	relative response factor

RRSE	Relative Risk Site Evaluation
RSD	relative standard deviation
RTC	Recruiting Training Center
RTECS	Registry of Toxic Effects of Chemical Substances
RTK	real-time kinematic
RWIMR	Ranges West of Iron Mountain Road
SA	exposed skin surface area
SAD	South Atlantic Division
SAE	Society of Automotive Engineers
SAIC	Science Applications International Corporation
SAP	installation-wide sampling and analysis plan
SARA	Superfund Amendments and Reauthorization Act
sc	clayey sands; sand-clay mixtures
Sch.	schedule
SCM	site conceptual model
SD	sediment
SDG	sample delivery group
SDWA	Safe Drinking Water Act
SDZ	safe distance zone; surface danger zone
SEMS	Southern Environmental Management & Specialties, Inc.
SF	cancer slope factor
SFSP	site-specific field sampling plan
SGF	standard grade fuels
Shaw	Shaw Environmental, Inc.
SHP	installation-wide safety and health plan
SI	site investigation
SINA	Special Interest Natural Area
SL	standing liquid
SLERA	screening-level ecological risk assessment
sm	silty sands; sand-silt mixtures
SM	Serratia marcescens
SMDP	Scientific Management Decision Point
s/n	signal-to-noise ratio
SO_4^{-2}	sulfate
SOD	soil oxidant demand
SOP	standard operating procedure
SOPQAM	U.S. EPA's *Standard Operating Procedure/Quality Assurance Manual*
sp	poorly graded sands; gravelly sands
SP	submersible pump
SPCC	system performance calibration compound
SPCS	State Plane Coordinate System
SPM	sample planning module
SQRT	screening quick reference tables
Sr-90	strontium-90
SRA	streamlined human health risk assessment
SRI	supplemental remedial investigation
SRM	standard reference material
Ss	stony rough land, sandstone series
SS	surface soil
SSC	site-specific chemical
SSHO	site safety and health officer
SSHP	site-specific safety and health plan
SSL	soil screening level
SSSL	site-specific screening level
SSSSL	site-specific soil screening level
STB	supertropical bleach
STC	source-term concentration
STD	standard deviation
STEL	short-term exposure limit
STL	Severn-Trent Laboratories
STOLS	Surface Towed Ordnance Locator System®
Std. units	standard units
SU	standard unit
SUXOS	senior UXO supervisor
SVOC	semivolatile organic compound
SW	surface water
SW-846	U.S. EPA's *Test Methods for Evaluating Solid Waste: Physical/Chemical Methods*
SWMU	solid waste management unit
SWPP	storm water pollution prevention plan
SZ	support zone
TAL	target analyte list
TAT	turn around time
TB	trip blank
TBC	to be considered
TCA	trichloroethane
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TCDF	tetrachlorodibenzofurans
TCE	trichloroethene
TCL	target compound list
TCLP	toxicity characteristic leaching procedure
TDEC	Tennessee Department of Environment and Conservation
TDGCL	thiodiglycol
TDGCLA	thiodiglycol chloroacetic acid
TEA	triethylaluminum
TeCA	1,1,2,2-tetrachloroethane
Tetryl	trinitrophenylmethylnitramine
TERC	Total Environmental Restoration Contract
TEU	Technical Escort Unit
THI	target hazard index
TIC	tentatively identified compound
TLV	threshold limit value
TN	Tennessee
TNB	trinitrobenzene
TNT	trinitrotoluene
TOC	top of casing; total organic carbon
TPH	total petroleum hydrocarbons
TR	target cancer risk
TRADOC	U.S. Army Training and Doctrine Command
TRPH	total recoverable petroleum hydrocarbons
TRV	toxicity reference value
TSCA	Toxic Substances Control Act
TSDF	treatment, storage, and disposal facility
TSS	total suspended solids
TWA	time-weighted average
UCL	upper confidence limit
UCR	upper certified range
'U'	not detected above reporting limit
UIC	underground injection control
UF	uncertainty factor
URF	unit risk factor
USACE	U.S. Army Corps of Engineers
USACHPPM	U.S. Army Center for Health Promotion and Preventive Medicine
USAEC	U.S. Army Environmental Center
USAEHA	U.S. Army Environmental Hygiene Agency
USACMLS	U.S. Army Chemical School
USAMPS	U.S. Army Military Police School
USATCES	U.S. Army Technical Center for Explosive Safety
USATEU	U.S. Army Technical Escort Unit
USATHAMA	U.S. Army Toxic and Hazardous Material Agency
USC	United States Code
USCS	Unified Soil Classification System
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
UST	underground storage tank
UTL	upper tolerance level; upper tolerance limit
UXO	unexploded ordnance
UXOQCS	UXO Quality Control Supervisor
UXOSO	UXO safety officer
V	vanadium
VC	vinyl chloride
VOA	volatile organic analyte
VOC	volatile organic compound
VOH	volatile organic hydrocarbon
VQlfr	validation qualifier
VQual	validation qualifier
VX	nerve agent (O-ethyl-S-[diisopropylaminoethyl]-methylphosphonothiolate)
WAC	Women's Army Corps
Weston	Roy F. Weston, Inc.
WP	white phosphorus
WRS	Wilcoxon rank sum
WS	watershed
WSA	Watershed Screening Assessment

WWI	World War I
WWII	World War II
XRF	x-ray fluorescence
yd^3	cubic yards
ZVI	zero-valent iron

TABLES

Table 2-1

Constituents of Potential Ecological Concern in Surface Soil[a] at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [b] (mg/kg)	Ecological Screening Value [c] (mg/kg)	Maximum Detected Conc. (mg/kg)	Minimum Detected Conc. (mg/kg)	Mean Detected Conc. (mg/kg)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Metals :									
Aluminum	1.63E+04	5.00E+01	3.62E+04	5.27E+03	1.56E+04	139 / 139	724	311	5
Antimony	1.99E+00	3.50E+00	5.03E+01	4.30E+00	3.59E+00	18 / 134	14.4	1.03	**YES**
Arsenic	1.37E+01	1.00E+01	1.38E+01	1.77E+00	5.07E+00	136 / 136	1.38	0.51	4
Barium	1.24E+02	1.65E+02	2.29E+03	2.89E+01	1.36E+02	139 / 139	13.9	0.82	5
Beryllium	8.00E-01	1.10E+00	2.66E+00	3.65E-01	7.44E-01	124 / 139	2.4	0.68	5
Cadmium	2.90E-01	1.60E+00	1.88E+01	8.38E-01	4.62E-01	5 / 139	11.8	0.29	**YES**[7]
Calcium	1.72E+03	NA	1.90E+04	5.35E+01	8.55E+02	139 / 139	ND	ND	2,5
Chromium	3.70E+01	4.00E-01	5.12E+01	3.80E+00	1.54E+01	139 / 139	128	38.42	5
Cobalt	1.52E+01	2.00E+01	2.14E+02	1.24E+00	1.03E+01	138 / 139	10.7	0.52	5
Copper	1.27E+01	4.00E+01	3.89E+02	4.86E+00	6.00E+01	139 / 139	9.73	1.50	**YES**
Iron	3.42E+04	2.00E+02	6.68E+04	5.85E+03	2.02E+04	139 / 139	334	101	5
Lead	4.01E+01	5.00E+01	4.64E+03	8.44E+00	3.86E+02	151 / 151	92.8	7.73	**YES**
Magnesium	1.03E+03	4.40E+05	1.16E+04	1.83E+02	7.47E+02	139 / 139	0.026	0.002	1,2,5
Manganese	1.58E+03	1.00E+02	1.05E+04	4.88E+01	1.14E+03	139 / 139	105	11.45	5
Mercury	8.00E-02	1.00E-01	1.18E+00	2.78E-02	7.87E-02	133 / 138	11.8	0.79	**YES**
Nickel	1.03E+01	3.00E+01	5.97E+01	2.16E+00	8.67E+00	126 / 126	1.99	0.29	**YES**[7]
Potassium	8.00E+02	NA	1.62E+03	2.94E+02	6.60E+02	107 / 107	ND	ND	2,5
Selenium	4.80E-01	8.10E-01	3.79E+00	5.50E-01	1.14E+00	73 / 97	4.68	1.41	5
Silver	3.60E-01	2.00E+00	1.51E+00	1.11E+00	6.54E-01	2 / 139	0.76	0.33	1,5
Sodium	6.34E+02	NA	1.28E+02	1.99E+01	3.66E+01	74 / 109	ND	ND	2,3
Thallium	3.43E+00	1.00E+00	5.68E+00	7.27E-01	5.02E-01	14 / 129	5.68	0.50	4
Vanadium	5.88E+01	2.00E+00	5.31E+01	5.66E+00	2.33E+01	139 / 139	26.55	11.63	3
Zinc	4.06E+01	5.00E+01	9.54E+03	9.74E+00	1.82E+02	139 / 139	191	3.63	**YES**
Herbicides :									
2,4-D	NA	1.00E-01	8.90E-03	8.90E-03	2.63E-03	1 / 53	0.089	0.026	1
MCPA	NA	1.00E-01	6.30E-01	6.30E-01	1.61E-01	1 / 53	6.30	1.61	**YES**[7]
MCPP	NA	1.00E-01	2.20E+00	2.20E+00	1.56E-01	1 / 53	22	1.56	**YES**[7]

Table 2-1

Constituents of Potential Ecological Concern in Surface Soil[a] at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [b] (mg/kg)	Ecological Screening Value [c] (mg/kg)	Maximum Detected Conc. (mg/kg)	Minimum Detected Conc. (mg/kg)	Mean Detected Conc. (mg/kg)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Pesticides :									
4,4'-DDD	NA	2.50E-03	3.80E-02	1.40E-03	1.46E-03	6 / 53	15.2	0.59	**YES**
4,4'-DDE	NA	2.50E-03	4.50E-03	1.40E-03	9.00E-04	11 / 53	1.8	0.36	**YES**
4,4'-DDT	NA	2.50E-03	5.80E-02	8.10E-04	2.62E-03	16 / 53	23.2	1.05	**YES**
Aldrin	NA	2.50E-03	1.20E-02	1.10E-03	8.01E-04	8 / 53	4.8	0.32	**YES**[7]
alpha-BHC	NA	2.50E-03	8.70E-03	3.10E-04	7.34E-04	30 / 53	3.5	0.29	**YES**[7]
alpha-Chlordane	NA	1.00E-01	1.70E-03	3.60E-04	1.66E-04	4 / 53	0.017	0.0017	1
beta-BHC	NA	1.00E-03	8.00E-03	3.30E-04	7.21E-04	8 / 53	8.0	0.72	**YES**[7]
delta-BHC	NA	9.94E+00	2.30E-03	6.00E-04	2.88E-04	5 / 53	0.00023	0.00003	1
Dieldrin	NA	5.00E-04	8.00E-03	9.60E-04	5.83E-04	6 / 53	16.0	1.17	**YES**[7]
Endosulfan I	NA	1.19E-01	8.30E-04	8.30E-04	4.25E-04	1 / 53	0.0070	0.0036	1
Endosulfan II	NA	1.19E-01	2.70E-03	1.70E-03	3.69E-04	2 / 53	0.0227	0.0031	1
Endosulfan sulfate	NA	3.58E-02	6.70E-04	6.70E-04	2.55E-04	1 / 53	0.019	0.0071	1
Endrin	NA	1.00E-03	3.40E-02	4.50E-04	1.78E-03	14 / 53	34	1.78	**YES**
Endrin aldehyde	NA	1.05E-02	3.30E-03	2.60E-03	3.11E-04	2 / 53	0.314	0.030	1
Endrin ketone	NA	1.05E-02	2.10E-03	2.10E-03	3.01E-04	1 / 53	0.20	0.029	1
gamma-BHC (Lindane)	NA	5.00E-05	6.00E-03	8.60E-04	3.44E-04	5 / 53	120	6.89	**YES**[7]
gamma-Chlordane	NA	1.00E-01	7.40E-04	7.40E-04	1.36E-04	1 / 53	0.0074	0.0014	1
Heptachlor	NA	1.00E-01	3.40E-03	3.40E-04	4.41E-04	14 / 53	0.034	0.0044	1
Heptachlor epoxide	NA	1.52E-01	1.40E-03	5.90E-04	1.52E-04	3 / 53	0.0092	0.0010	1
Methoxychlor	NA	1.99E-02	2.60E-02	3.30E-03	3.68E-03	8 / 53	1.31	0.185	**YES**[7]
Nitroaromatics :									
1,3-Dinitrobenzene	NA	6.55E-01	1.60E-01	1.60E-01	3.24E-02	1 / 139	0.244	0.050	1
2,4-Dinitrotoluene	NA	1.28E+00	1.50E+00	1.10E-01	5.59E-02	2 / 139	1.172	0.044	**YES**[7]
Semivolatile Organic Compounds :									
Bis(2-Ethylhexyl)phthalate	NA	9.26E-01	4.60E-01	8.30E-02	4.10E-02	2 / 53	0.497	0.044	1
Di-n-butyl phthalate	NA	2.00E+02	6.40E-01	6.40E-01	4.36E-02	1 / 53	0.0032	0.00022	1
N-Nitrosodiphenylamine	NA	2.00E+01	3.50E+00	3.50E+00	9.61E-02	1 / 53	0.175	0.0048	1

Table 2-1

Constituents of Potential Ecological Concern in Surface Soil[a] at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [b] (mg/kg)	Ecological Screening Value [c] (mg/kg)	Maximum Detected Conc. (mg/kg)	Minimum Detected Conc. (mg/kg)	Mean Detected Conc. (mg/kg)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Volatile Organic Compounds :									
2-Butanone	NA	8.96E+01	7.40E-02	7.10E-03	2.33E-02	41 / 45	0.0008	0.0003	1
Acetone	NA	2.50E+00	2.10E+00	4.40E-02	4.11E-01	48 / 50	0.840	0.16	1
Chloromethane	NA	1.00E-01	4.70E-03	4.70E-03	1.49E-03	1 / 53	0.047	0.015	1
p-Cymene	NA	NA	1.30E-02	6.00E-04	1.10E-03	15 / 53	ND	ND	6
Styrene	NA	1.00E-01	1.00E-03	1.00E-03	3.02E-04	1 / 53	0.010	0.0030	1
Toluene	NA	5.00E-02	2.60E-03	6.00E-04	4.60E-04	7 / 53	0.052	0.0092	1
Trichlorofluoromethane	NA	1.00E-01	4.60E-03	1.10E-03	7.00E-04	11 / 50	0.046	0.0070	1

[a] Surface soil at the Choccolocco Corridor Ranges is defined as the interval from 0 to 0.5 feet below ground surface.

[b] Background threshold value is two times (2x) the arithmetic mean of background metals (SAIC, 1998). For SVOCs, the BTV is the background screening value for soils adjacent to asphalt as given in IT Corporation (IT), 2000, *Final Human Health and Ecological Screening Values and PAH Background Summary Report, Fort McClellan, Calhoun County, Alabama*, July.

[c] Ecological Screening Values (ESV) are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000).

NA - Not available. ND - Not determined.

Rationale for inclusion / exclusion as a COPEC:

1 - Maximum detected concentration is less than ESV
2 - Essential macro-nutrient, only toxic at extremely high concentrations (i.e. 10-times naturally-occurring background concentrations).
3 - Maximum detected concentration is less than the background threshold value (BTV).
4 - Slippage Test and Wilcoxon Rank Sum Test indicate the concentration of this constituent is statistically similar to background concentrations.
5 - Geochemical evaluation of the data indicate that this constituent is naturally occurring.
6 - No ESV available; however, maximum detected concentration of this constituent is less than ESV for similar compounds.
7 - Additional lines of evidence indicate that this constituent may not be a COPEC (see text).

Table 2-2

Constituents of Potential Ecological Concern in Surface Water at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [a] (mg/L)	Ecological Screening Value [b] (mg/L)	Maximum Detected Conc. (mg/L)	Minimum Detected Conc. (mg/L)	Mean Detected Conc. (mg/L)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Metals :									
Aluminum	5.26E+00	8.70E-02	4.05E-01	5.67E-02	1.26E-01	10 / 15	4.66	1.45	3
Barium	7.54E-02	3.90E-03	3.46E-02	1.42E-02	2.19E-02	23 / 23	8.87	5.61	3
Calcium	2.52E+01	1.16E+02	9.13E-01	2.35E-01	3.70E-01	21 / 21	0.0079	0.0032	1,2,3
Cobalt	NA	3.00E-03	1.83E-02	1.83E-02	6.06E-03	1 / 23	6.10	2.02	4
Copper	1.27E-02	6.54E-03	6.90E-03	6.28E-03	2.86E-03	2 / 23	1.06	0.44	3
Iron	1.96E+01	1.00E+00	7.07E-01	2.02E-02	1.57E-01	22 / 23	0.71	0.16	1,3
Lead	8.67E-03	1.32E-03	1.09E-02	2.06E-03	1.39E-03	3 / 18	8.26	1.05	**YES**[4]
Magnesium	1.10E+01	8.20E+01	4.19E-01	2.56E-01	3.53E-01	23 / 23	0.0051	0.0043	1,2,3
Manganese	5.65E-01	8.00E-02	5.89E-02	3.36E-03	1.46E-02	15 / 17	0.74	0.18	1,3
Potassium	2.56E+00	5.30E+01	2.68E+00	1.32E+00	1.89E+00	7 / 8	0.051	0.036	1,2
Sodium	3.44E+00	6.80E+02	1.16E+00	8.06E-01	9.39E-01	20 / 20	0.0017	0.0014	1,2,3
Semivolatile Organic Compounds ;									
Bis(2-Ethylhexyl)phthalate	NA	3.00E-04	1.00E-01	1.30E-02	1.01E-02	2 / 20	333.33	33.72	**YES**[7]
Volatile Organic Compounds ;									
Methylene Chloride	NA	1.93E+00	4.00E-04	3.20E-04	1.00E-04	2 / 15	0.00021	0.00005	1

[a] Background threshold value is two times (2x) the arithmetic mean of background metals (SAIC, 1998).

[b] Ecological Screening Values (ESV) are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000).

NA - Not available. ND - Not determined.

Rationale for inclusion / exclusion as a COPEC:

1 - Maximum detected concentration is less than ESV

2 - Essential macro-nutrient, only toxic at extremely high concentrations (i.e. 10-times naturally-occurring background concentrations).

3 - Maximum detected concentration is less than the background threshold value (BTV).

4 - Slippage Test and Wilcoxon Rank Sum Test indicate the concentration of this constituent is statistically similar to background concentrations.

5 - Geochemical evaluation of the data indicate that this constituent is naturally occurring.

6 - No ESV available; however, maximum detected concentration of this constituent is less than ESV for similar compounds.

7 - Additional lines of evidence indicate that this constituent may not be a COPEC (see text).

Table 2-3

Constituents of Potential Ecological Concern in Sediment at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [b] (mg/kg)	Ecological Screening Value [c] (mg/kg)	Maximum Detected Conc. (mg/kg)	Minimum Detected Conc. (mg/kg)	Mean Detected Conc. (mg/kg)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Metals :									
Aluminum	8.59E+03	NA	2.94E+04	2.62E+03	1.22E+04	23 / 23	ND	ND	5
Arsenic	1.13E+01	7.24E+00	6.28E+00	1.31E+00	3.41E+00	19 / 19	0.87	0.47	1,3
Barium	9.89E+01	NA	3.04E+02	3.78E+01	1.17E+02	23 / 23	ND	ND	5
Beryllium	9.70E-01	NA	2.07E+00	4.86E-01	8.64E-01	21 / 23	ND	ND	5
Calcium	1.11E+03	NA	9.11E+02	1.29E+02	3.04E+02	23 / 23	ND	ND	2,3
Chromium	3.12E+01	5.23E+01	2.69E+01	2.59E+00	1.04E+01	23 / 23	0.51	0.20	1,3
Cobalt	1.10E+01	5.00E+01	1.93E+01	2.37E+00	6.61E+00	23 / 23	0.39	0.13	1,5
Copper	1.71E+01	1.87E+01	4.21E+01	3.36E+00	1.27E+01	23 / 23	2.25	0.68	5
Iron	3.53E+04	NA	3.40E+04	8.10E+03	1.75E+04	23 / 23	ND	ND	3
Lead	3.78E+01	3.02E+01	3.94E+02	8.53E+00	4.34E+01	23 / 23	13.05	1.44	**YES**[5]
Magnesium	9.06E+02	NA	1.31E+03	1.62E+02	5.35E+02	23 / 23	ND	ND	2,5
Manganese	7.12E+02	NA	1.49E+03	6.84E+01	5.54E+02	23 / 23	ND	ND	5
Mercury	1.10E-01	1.30E-01	5.78E-02	3.36E-02	2.74E-02	8 / 23	0.44	0.21	1,3
Nickel	1.30E+01	1.59E+01	1.30E+01	2.52E+00	7.18E+00	18 / 18	0.82	0.45	1,3
Potassium	1.01E+03	NA	2.03E+03	5.83E+02	1.19E+03	22 / 22	ND	ND	2,5
Selenium	7.20E-01	NA	9.19E-01	7.36E-01	3.70E-01	2 / 23	ND	ND	4
Sodium	6.92E+02	NA	1.86E+02	2.32E+01	7.21E+01	16 / 23	ND	ND	2,3
Vanadium	4.09E+01	NA	4.20E+01	5.47E+00	1.69E+01	23 / 23	ND	ND	4
Zinc	5.27E+01	1.24E+02	5.08E+01	7.04E+00	2.33E+01	23 / 23	0.41	0.19	1,3
Pesticides :									
4,4'-DDT	NA	3.30E-03	5.30E-03	2.40E-03	7.71E-04	2 / 21	1.61	0.23	**YES**[7]
alpha-BHC	NA	6.00E-03	2.20E-03	8.30E-04	4.48E-04	5 / 21	0.37	0.07	1
beta-BHC	NA	5.00E-03	1.10E-02	1.20E-03	8.73E-04	3 / 21	2.20	0.17	**YES**[7]
Endrin	NA	3.30E-03	4.80E-03	4.60E-03	6.60E-04	2 / 21	1.45	0.20	**YES**[7]
Endrin aldehyde	NA	3.20E+00	2.70E-03	2.70E-03	3.55E-04	1 / 21	0.00084	0.00011	1
gamma-Chlordane	NA	1.70E-03	1.50E-02	8.70E-04	1.59E-03	5 / 21	8.82	0.93	**YES**
Heptachlor	NA	6.00E-04	2.60E-03	5.50E-04	4.34E-04	4 / 21	4.33	0.72	**YES**[7]
Methoxychlor	NA	3.59E-03	5.40E-03	4.60E-03	2.65E-03	2 / 21	1.50	0.74	**YES**[7]

Table 2-3

Constituents of Potential Ecological Concern in Sediment at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [b] (mg/kg)	Ecological Screening Value [c] (mg/kg)	Maximum Detected Conc. (mg/kg)	Minimum Detected Conc. (mg/kg)	Mean Detected Conc. (mg/kg)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Nitroaromatics :									
HMX	NA	4.70E-03	5.70E-01	5.70E-01	1.06E-01	1 / 23	121.28	22.57	**YES**[7]
Volatile Organic Compounds :									
2-Butanone	NA	1.37E-01	6.90E-02	8.30E-03	1.75E-02	13 / 20	0.50	0.13	1
Acetone	NA	4.53E-01	7.20E-01	5.40E-02	2.56E-01	15 / 16	1.59	0.56	**YES**[7]
N-Butylbenzene	NA	NA	1.10E-02	1.10E-02	8.14E-04	1 / 21	ND	ND	6
p-Cymene	NA	NA	1.10E-01	6.50E-04	5.53E-03	2 / 21	ND	ND	6
sec-Butylbenzene	NA	NA	1.10E-02	1.10E-02	8.41E-04	1 / 21	ND	ND	6
Toluene	NA	6.70E-01	2.70E-03	7.70E-04	6.55E-04	4 / 21	0.0040	0.0010	1

[a] Background threshold value is two times (2x) the arithmetic mean of background metals (SAIC, 1998).

[b] Ecological Screening Values (ESV) are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000).

NA - Not available. ND - Not determined.

Rationale for inclusion / exclusion as a COPEC:

1 - Maximum detected concentration is less than ESV

2 - Essential macro-nutrient, only toxic at extremely high concentrations (i.e. 10-times naturally-occurring background concentrations).

3 - Maximum detected concentration is less than the background threshold value (BTV).

4 - Slippage Test and Wilcoxon Rank Sum Test indicate the concentration of this constituent is statistically similar to background concentrations.

5 - Geochemical evaluation of the data indicate that this constituent is naturally occurring.

6 - No ESV available; however, maximum detected concentration of this constituent is less than ESV for similar compounds.

7 - Additional lines of evidence indicate that this constituent may not be a COPEC (see text).

Table 2-4

Constituents of Potential Ecological Concern in Groundwater at the Former Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

Constituents	Background Threshold Value [a] (mg/L)	Ecological Screening Value [b] (mg/L)	Maximum Detected Conc. (mg/L)	Minimum Detected Conc. (mg/L)	Mean Detected Conc. (mg/L)	Frequency of Detection	Maximum Hazard Quotient	Mean Hazard Quotient	Constituent Of Potential Ecological Concern
Metals :									
Aluminum	2.34E+00	8.70E-02	3.27E+00	1.21E-01	5.92E-01	17 / 23	37.59	6.80	4
Arsenic	1.78E-02	1.90E-01	3.79E-03	3.79E-03	9.55E-04	1 / 28	0.020	0.005	1,3
Barium	1.27E-01	3.90E-03	3.27E-02	5.91E-03	1.66E-02	31 / 31	8.38	4.25	3
Calcium	5.65E+01	1.16E+02	3.50E+00	3.90E-01	1.28E+00	28 / 28	0.030	0.011	1,2,3
Cobalt	2.34E-02	3.00E-03	2.29E-02	1.86E-02	6.96E-03	3 / 31	7.63	2.32	3
Copper	2.55E-02	6.54E-03	1.39E-02	6.63E-03	3.02E-03	2 / 30	2.13	0.46	3
Iron	7.04E+00	1.00E+00	5.09E+00	1.05E-02	5.77E-01	28 / 28	5.09	0.58	3
Lead	8.00E-03	1.32E-03	2.84E-03	1.38E-03	7.84E-04	3 / 29	2.15	0.59	3
Magnesium	2.13E+01	8.20E+01	1.23E+00	1.82E-01	5.34E-01	31 / 31	0.015	0.007	1,2,3
Manganese	5.81E-01	8.00E-02	1.81E+00	6.03E-03	2.27E-01	30 / 30	22.63	2.84	4
Potassium	7.20E+00	5.30E+01	5.45E+00	1.37E+00	1.95E+00	11 / 16	0.10	0.037	1,2,3
Sodium	1.48E+01	6.80E+02	5.93E+00	6.27E-01	1.68E+00	28 / 28	0.0087	0.0025	1,2,3
Zinc	2.20E-01	5.89E-02	1.09E-01	1.09E-01	1.66E-02	1 / 31	1.85	0.28	3
Pesticides :									
Heptachlor	NA	3.80E-06	2.70E-05	2.70E-05	5.08E-06	1 / 22	7.11	1.34	**YES**[7]
Nitroaromatics :									
1,3,5-Trinitrobenzene	NA	1.10E-02	3.20E-04	3.20E-04	1.02E-04	1 / 31	0.029	0.009	1
2,4,6-Trinitrotoluene	NA	9.00E-02	6.80E-04	6.80E-04	8.97E-05	1 / 31	0.0076	0.0010	1

[a] Background threshold value is two times (2x) the arithmetic mean of background metals (SAIC, 1998).

[b] Ecological Screening Values (ESV) are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000).

NA - Not available. ND - Not determined.

Rationale for inclusion / exclusion as a COPEC:

1 - Maximum detected concentration is less than ESV
2 - Essential macro-nutrient, only toxic at extremely high concentrations (i.e. 10-times naturally-occurring background concentrations).
3 - Maximum detected concentration is less than the background threshold value (BTV).
4 - Slippage Test and Wilcoxon Rank Sum Test indicate the concentration of this constituent is statistically similar to background concentrations.
5 - Geochemical evaluation of the data indicate that this constituent is naturally occurring.
6 - No ESV available; however, maximum detected concentration of this constituent is less than ESV for similar compounds.
7 - Additional lines of evidence indicate that this constituent may not be a COPEC (see text).

Table 2-5

Summary of COPECs
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

COPECs	CC Ranges Surface Soil	Perennial and Ephemeral Streams Surface Water	Perennial and Ephemeral Streams Sediment	CC Ranges Groundwater
antimony	X			
cadmium	O			
copper	X			
lead	X	X	X	
mercury	X			
nickel	O			
zinc	X			
MCPA	O			
MCPP	O			
4,4'-DDD	X			
4,4'-DDE	X			
4,4'-DDT	X		O	
aldrin	O			
alpha-BHC	O			
beta-BHC	O		O	
dieldrin	O			
endrin	X		O	
gamma-BHC	O			
gamma-chlordane			X	
heptachlor			O	O
methoxychlor	O		O	
2,4-dinitrotoluene	O			
HMX			O	
bis(2-ethylhexyl)phthalate		O		
acetone			O	

O - HQ > 1.0; however, additional lines of evidence indicate that this constituent is not a COPEC
X - Constituent identified as a COPEC

Table 7-1

Assessment Endpoints, Risk Hypotheses, and Measurement Endpoints
Former Choccolocco Corridor Ranges
Fort McClellan, Alabama

Assessment Endpoint	Risk Hypothesis	Measurement Endpoint
Terrestrial Ecosystems:		
I. Survival and growth of the terrestrial plant communities at the CC ranges.	IA. Is the survival and growth of terrestrial plants exposed to surface soil from the CC ranges significantly lower than that for terrestrial plants exposed to soil from reference sites?	IA. Statistical comparison of perennial ryegrass seed germination success, plant height, above ground biomass, root length, and root biomass between plants grown in on-site soils from the CC ranges to plants grown in soils from a reference location.
	IB. Are concentrations of COPECs in surface soil at the CC ranges greater than ESVs for the survival or growth of terrestrial plants?	IB. Comparison of COPEC concentrations in surface soil at the CC ranges to ESVs for the survival and growth of terrestrial plants.
II. Survival and growth of the terrestrial invertebrate community at the CC ranges.	IIA. Is the survival and growth of terrestrial invertebrates exposed to surface soil from the CC ranges significantly lower than that for terrestrial invertebrates exposed to soil from reference sites?	IIA. Statistical comparison of earthworm survival rates and body weights between earthworms exposed to on-site soils from the CC ranges to earthworms exposed to soils from a reference location.
	IIB. Are concentrations of COPECs in surface soil at the CC ranges greater than ESVs for the survival or growth of terrestrial invertebrates?	IIB. Comparison of COPEC concentrations in surface soil at the CC ranges to ESVs for the survival and growth of terrestrial invertebrates.
III. Survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the CC ranges.	III. Does the daily dose of COPECs received by terrestrial invertivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the toxicity reference values (TRVs) for survival, reproduction, or growth?	IIIA. Comparison of calculated daily doses of COPECs for terrestrial invertivorous small mammal (shorttail shrew) and invertivorous bird (American woodcock) to TRVs.
		IIIB. Quantification of COPEC concentrations in tissues of earthworms exposed to soils from the CC ranges and earthworms exposed to soils from a reference location.

Table 7-1

Assessment Endpoints, Risk Hypotheses, and Measurement Endpoints
Former Choccolocco Corridor Ranges
Fort McClellan, Alabama

Assessment Endpoint	Risk Hypothesis	Measurement Endpoint
IV. Survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the CC ranges.	IV. Does the daily dose of COPECs received by terrestrial omnivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the toxicity reference values (TRVs) for survival, reproduction, or growth?	IVA. Comparison of calculated daily doses of COPECs for terrestrial omnivorous small mammal (white-footed mouse) and omnivorous bird (American robin) to TRVs.
		IVB. Quantification of COPEC concentrations in tissues of earthworms exposed to soils from the CC ranges and earthworms exposed to soils from a reference location.
		IVC. Quantification of COPEC concentrations in above-ground plant tissues exposed to soils from the CC ranges and above-ground plant tissues from a reference location.
Aquatic Ecosystems:		
I. Survival, growth, and reproduction of aquatic benthic invertebrates in the perennial and ephemeral drainage features at the CC ranges.	IA. Is the survival and growth of aquatic benthic invertebrates exposed to sediment from the perennial and ephemeral drainage features at the CC ranges significantly lower than that for aquatic benthic invertebrates exposed to sediment from reference sites?	IA. Comparison of survival and growth of the benthic amphipod *Chironomus tentans* exposed to "on-site" sediment to survival and growth of *Chironomus tentans* exposed to sediment from a reference stream.
	IB. Is the benthic community structure significantly different in reaches of the perennial and ephemeral drainage features at the CC ranges compared to benthic communities in reference streams?	IB. Comparison of benthic community assemblage in the perennial and ephemeral drainage features at the CC ranges with the benthic community assemblage in a reference stream using RBPII methodology.
	IC. Are the concentrations of COPECs in sediment from the perennial and ephemeral drainage features at the CC ranges greater than ESVs for the survival, growth, and reproduction of aquatic benthic invertebrates?	IC. Comparison of COPEC concentrations in sediment from the perennial and ephemeral drainage features at the CC ranges to ESVs for the survival and growth of aquatic benthic invertebrates.

Table 7-1

Assessment Endpoints, Risk Hypotheses, and Measurement Endpoints
Former Choccolocco Corridor Ranges
Fort McClellan, Alabama

Assessment Endpoint	Risk Hypothesis	Measurement Endpoint
II. Survival, growth, and reproduction of riparian invertivorous small mammals and birds at the CC ranges.	II. Does the daily dose of COPECs received by riparian invertivorous small mammals or birds via consumption of tissues of prey species and from other media at the CC ranges exceed the toxicity reference values (TRVs) for survival, reproduction, or growth?	IIA. Comparison of calculated total daily doses of COPECs for riparian invertivorous mammal (little brown bat) and invertivorous bird (marsh wren) to TRVs.
		IIB. Quantification of COPEC concentrations in tissues of chironomids exposed to sediment from the perennial and ephemeral drainage features at the CC ranges and from a reference location.

Table 7-2

Terrestrial Foodweb Model Input Parameters
Former Choccolocco Corridor (CC) Ranges
Fort McClellan, Calhoun County, Alabama

Common Name	Scientific Name	Feeding Guild	Foraging Area (acres)	Area Use Factor (unitless)	Body Weight (kg)	Water Ingestion Rate (L/kg/day)	Food Ingestion Rate (kg/kg/day-dry wt.)	Soil Ingestion Rate [f] (kg/kg/day-dry wt.)	Dietary Fraction (unitless)	Dietary Component
White-Footed Mouse	*Peromyscus leucopus*	Omnivorous Mammal	1.0 (b)	1.0	0.0225 (b)	0.19 (a)	0.1237 (a)	0.00247 (a)	0.254 0.746	Terrestrial Invertebrates Terrestrial Vegetation (seeds & young grass / fruit)
American Robin	*Turdus migratorius*	Omnivorous Bird	0.61 (a)	1.0	0.081 (a)	0.14 (a)	0.1816 (a)	0.00363 (c)	0.375 0.625	Terrestrial Invertebrates Terrestrial Vegetation (fruit)
Shorttail Shrew	*Blarina brevicauda*	Invertivorous Mammal	0.964 (a)	1.0	0.0168 (a)	0.223 (a)	0.0899 (a)	0.00216 (a)	0.887 0.113	Terrestrial Invertebrates Terrestrial Vegetation (roots / young grass)
American Woodcock	*Scolopax minor*	Invertivorous Bird	61.3 (a)	1.0	0.169 (a)	0.1 (a)	0.1517 (a)	0.0158 (a)	0.95 0.05	Terrestrial Invertebrates Terrestrial Vegetation (seeds)

Notes:

All of the values presented in this table represent arithmetic mean values if more than one value was presented in the referenced source.

a USEPA, 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187a

b Burt, W.H. and R.P. Grossenheider. *Mammals, Peterson Field Guide*.

c Assumed value based on soil ingestion values for other birds presented in USEPA (1993).

Soil ingestion rates (dry weight) were calculated using the following relationship: $IR_{soil} = IR_{food} \times Diet_{soil}$

where:

IR_{soil} = ingestion rate of soil (kg/kg/day, dry weight);
IR_{food} = food ingestion rate (kg/kg/day, wet weight); and
$Diet_{soil}$ = percentage of diet that is soil (percent).

Table 7-3

Riparian Foodweb Model Input Parameters
Former Choccolocco Corridor (CC) Ranges
Fort McClellan, Calhoun County, Alabama

Common Name	Scientific Name	Feeding Guild	Foraging Area (acres)	Area Use Factor (unitless)	Body Weight (kg)	Water Ingestion Rate (L/kg/day)	Food Ingestion Rate (kg/kg/day)	Sediment Ingestion Rate ° (kg/kg/day)	Dietary Fraction (unitless)	Dietary Component
Little Brown Bat	*Myotis lucifugus*	Invertivorous Mammal	12 Km (c)	0.22	0.008 (b)	0.16 (e)	0.0699 (d)	NA	1.0	Aquatic Emergent Invertebrates
Marsh Wren	*Cistothorus palustris*	Invertivorous Bird	0.13 (a)	1.0	0.01038 (a)	0.27 (a)	0.1833 (a)	NA	1.0	Aquatic Emergent Invertebrates

Notes:

All of the values presented in this table represent arithmetic mean values if more than one value was presented in the referenced source.

a USEPA, 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187a

b Burt, W.H. and R.P. Grossenheider. *Mammals, Peterson Field Guide*.

c LaVal, et al., 1977. Foraging Behavior and Nocturnal Activity Patterns of Missouri Bats, with Emphasis on the Endangered Species *Myotis grisescens* and *Myotis sodalis*.

d Anthony and Kunz, 1977. Feeding Strategies of the Little Brown bat, *Myotis lucifugus*, in Southern New Hampshire.

e Sample, et al., 1997. Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants.

Table 9-1

Surface Soil Sample Location Summary
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Range	Sample Location	Lead Concentration (mg/kg)	Copper Concentration (mg/kg)	Mercury Concentration (mg/kg)	Zinc Concentration (mg/kg)	Endrin Concentration (mg/kg)
Range 40 (Parcel 94Q)	HR-94Q-GP02	4,530	132	0.0873	69.1	ND
Range 43 (Parcel 97Q)	HR-CCRI-GP26	2,510	197	0.0702	66.4	ND
Impact Area, Choccolocco Corridor (Parcel 148Q)	HR-148Q-MW01	1,010	151	0.156	39	ND
Range 40 (Parcel 94Q)	HR-CCRI-MW03	793	132	0.0318	30.7	ND
Range 41 (Parcel 95Q)	HR-CCRI-GP13	674	259	0.033	29.5	0.00045
Impact Area, Choccolocco Corridor (Parcel 148Q)	HR-CCRI-GP10	515	80	0.0624	50.7	0.0017
Range 43 (Parcel 97Q)	HR-97Q-GP03	401	86.6	0.0695	46	ND
Range, Choccolocco Corridor (Parcel 144Q-X)	HR-CCRI-GP02	188	13.5	0.0563	366	ND
Impact Area, Choccolocco Corridor (Parcel 148Q)	HR-CCRI-GP20	106	20.7	0.0611	27.3	ND
Range, Choccolocco Corridor (Parcel 145Q-X)	HR-96Q-GP04	64.4	383	0.0781	56.4	0.0015
Reference	HR-146Q-MW01	21	9.67	0.045	19	ND

Table 9-2

Sediment Sample Location Summary
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Range	Sample Location	Lead Concentration (mg/kg)	gamma-Chlordane Concentration (mg/kg)
Range, Choccolocco Corridor (Parcel 144Q-X)	HR-CCRI-SD18	394	ND
Range, Choccolocco Corridor (Parcel 146Q)	HR-CCRI-SD02	124	ND
Range, Choccolocco Corridor (Parcel 146Q)	HR-CCRI-SD04	52	0.0023
Range, Choccolocco Corridor (Parcel 144Q-X)	HR-CCRI-SD17	45	ND
Range, Choccolocco Corridor (Parcel 146Q)	HR-CCR1-SD07	35	0.015
Range 41 (Parcel 95Q)	HR-95Q-SD01	34.7	ND
Range 41 Impact Area (Parcel 131Q-X)	HR-131Q-SD01	30.4	NA
Range, Choccolocco Corridor (Parcel 145Q-X)	HR-145Q-SD02	19.4	NA
Perennial Reference	HR-CCRI-SD12	8.53	ND
Ephemeral Reference	HR-CCRI-SD10	13.5	ND

NA - Not Analyzed
ND - Not Detected

Table 10-1

Hypothetical Earthworm Toxicity Test Results
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

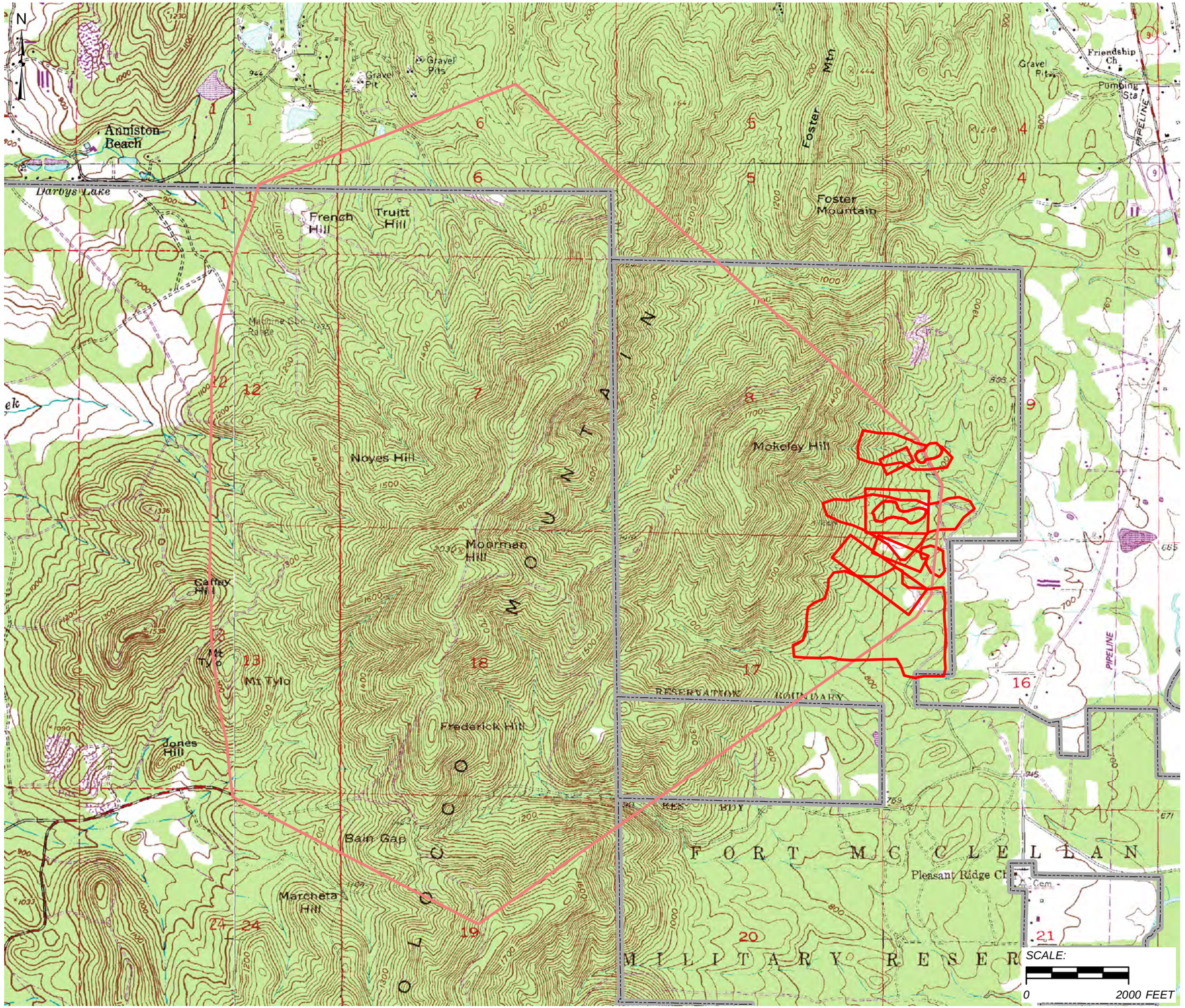
Lead Concentration (mg/kg)	Test	Endpoint	Significant Difference When Compared to Reference Location (p = 0.05)	NOAEL / LOAEL
147	Earthworm Toxicity	Survival	No	
		Growth	No	
300	Earthworm Toxicity	Survival	No	
		Growth	No	
800	Earthworm Toxicity	Survival	No	**NOAEL**
		Growth	No	
2,500	Earthworm Toxicity	Survival	No	**LOAEL**
		Growth	Yes	
4,500	Earthworm Toxicity	Survival	Yes	
		Growth	Yes	

Table 10-2

Hypothetical Perennial Ryegrass Toxicity Test Results
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Lead Concentration (mg/kg)	Test	Endpoint	Significant Difference When Compared to Reference Location (p = 0.05)	NOAEL / LOAEL
147	Perennial Ryegrass Toxicity	Germination Success	No	
		Plant Height	No	
		Above-Ground Biomass	No	
		Root Length	No	
		Below-Ground Biomass	No	
300	Perennial Ryegrass Toxicity	Germination Success	No	
		Plant Height	No	
		Above-Ground Biomass	No	**NOAEL**
		Root Length	No	
		Below-Ground Biomass	No	
800	Perennial Ryegrass Toxicity	Germination Success	No	
		Plant Height	Yes	
		Above-Ground Biomass	Yes	**LOAEL**
		Root Length	No	
		Below-Ground Biomass	No	
2,500	Perennial Ryegrass Toxicity	Germination Success	Yes	
		Plant Height	Yes	
		Above-Ground Biomass	Yes	
		Root Length	Yes	
		Below-Ground Biomass	Yes	
4,500	Perennial Ryegrass Toxicity	Germination Success	Yes	
		Plant Height	Yes	
		Above-Ground Biomass	Yes	
		Root Length	Yes	
		Below-Ground Biomass	Yes	

FIGURES



LEGEND

- Study Area Boundaries
- Range Fan
- Fort McClellan Boundary
- USGS 1:24,000 Topographic Quadrangles

FIGURE 5-1

SITE LOCATION AND TOPOGRAPHY MAP
FORMER CHOCCOLOCCO CORRIDOR
RANGES

Ecological Risk Assessment Problem Formulation
and Study Design Report
Choccolocco Corridor Ranges

FORT MCCLELLAN
ANNISTON, ALABAMA

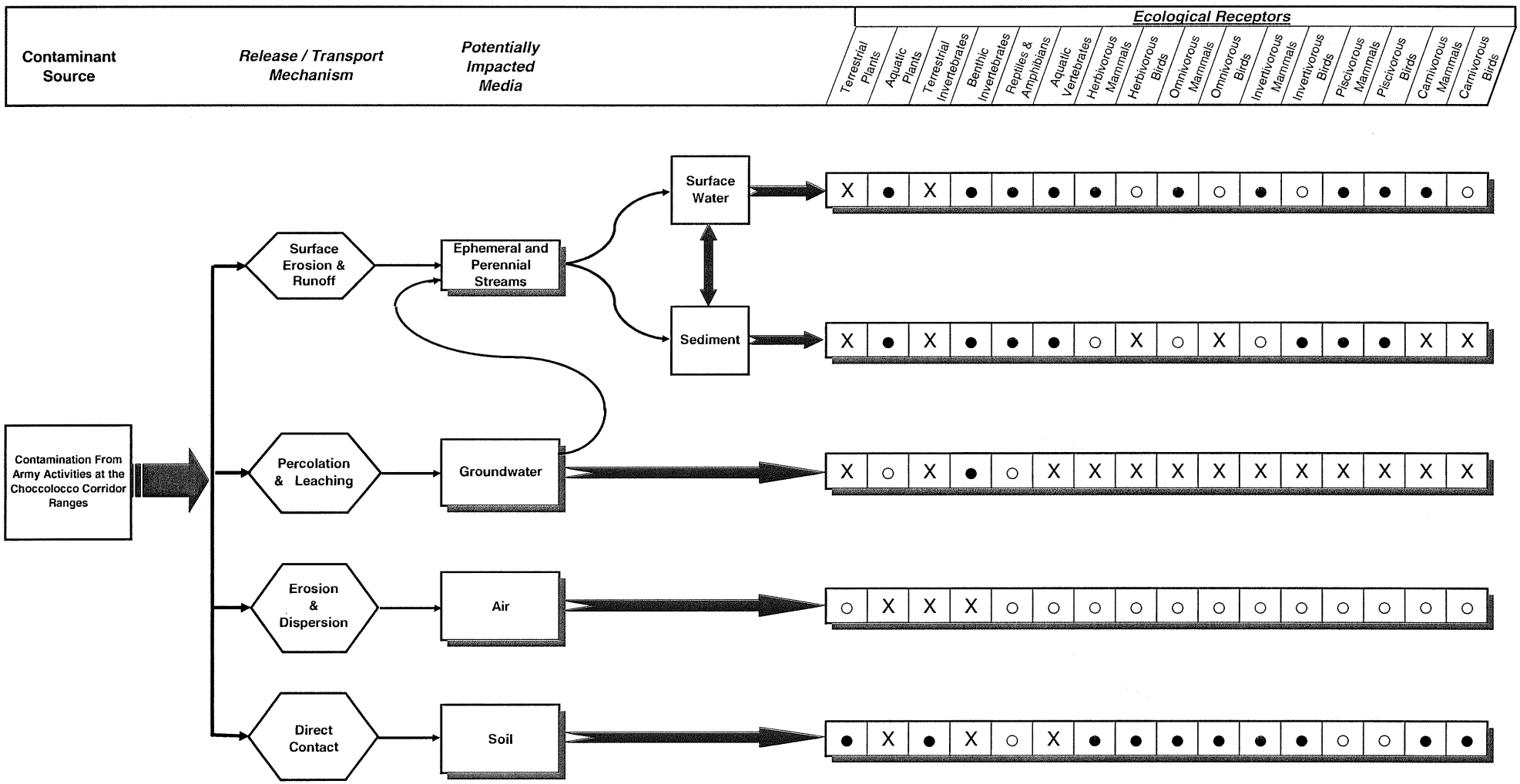
Shaw® Shaw Environmental, Inc.

Project Directory: X:\FTMC\GIS_Documents\map_documents\Chocco_Corridor\Eco_Risk_Assessment_Rpt_Aug2008\chocco_loc_map_topo.mxd

Creation Date: 8/12/08	Revision Date:	Author: J. Bond	Initiator: RP 8/1/08	Check: Bond 12/12/06	Tech Review:	Sr. Tech Review:

Figure 6-1

Site Conceptual Model
Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama



Key To Potential Exposure Routes

- ● - Potentially complete exposure pathway
- X - Incomplete exposure pathway
- ○ - Potentially complete exposure pathway but insignificant

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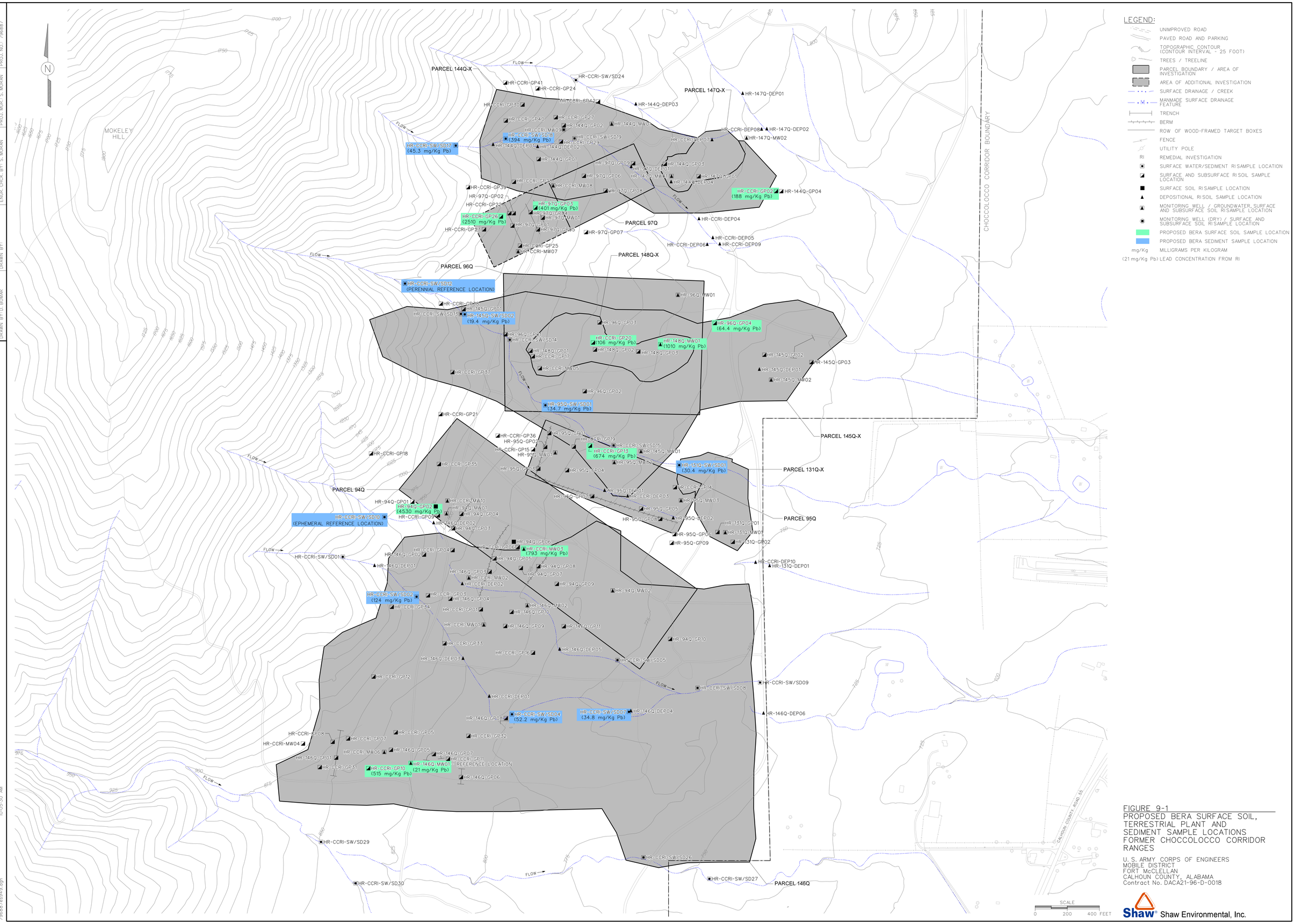

FIGURE 9-1
PROPOSED BERA SURFACE SOIL, TERRESTRIAL PLANT AND SEDIMENT SAMPLE LOCATIONS FORMER CHOCCOLOCCO CORRIDOR RANGES

U. S. ARMY CORPS OF ENGINEERS
MOBILE DISTRICT
FORT McCLELLAN
CALHOUN COUNTY, ALABAMA
Contract No. DACA21-96-D-0018

Shaw® Shaw Environmental, Inc.

STARTING DATE: 04/15/03	DATE LAST REV.:	DRAFT. CHECK BY:	INITIATOR: J. RAGSDALE
DRAWN BY: D. BOMAR	DRAWN BY:	ENGR. CHECK. BY: S. MORAN	PROJ. MGR.: S. MORAN
			DWG. NO.: 796887es949.dgn
			PROJ. NO.: 796887

8/26/2008 MAFB.tbl 10:05:30 AM

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30 X 42

APPENDIX A

FIELD SAMPLING AND ANALYSIS PLAN FOR SURFACE SOIL

Appendix A
Field Sampling and Analysis Plan For Surface Soil at the Choccolocco Corridor Ranges Fort McClellan, Calhoun County, Alabama

A.1.0 Introduction

The screening-level ecological risk assessment (SLERA) for the Choccolocco Corridor (CC) ranges (Shaw, 2005), identified five inorganic constituents (antimony, copper, lead, mercury, and zinc) and four pesticides (4,4'-DDD, 4,4-DDE, 4,4-DDT, and endrin) as chemicals of potential ecological concern (COPEC) in surface soil. To provide information for the baseline ecological risk assessment (BERA), surface soil samples will be collected from within the study areas and analyzed for target analyte list (TAL) metals, volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, pH, total organic carbon (TOC), and grain size. In addition, these same surface soil samples will be used to assess the toxicity to earthworms and terrestrial plants, and the bioaccumulation potential of the identified COPECs in surface soil to earthworm tissues and terrestrial plant tissues. Soil-to-worm bioaccumulation factors ($BAF_{soil\text{-}to\text{-}worm}$) and soil-to-plant bioaccumulation factors ($BAF_{soil\text{-}to\text{-}plant}$) will be developed for use in food chain modeling to higher trophic level receptors that reside within the CC ranges.

A.2.0 Selection of Sample Locations

Surface soil sample locations for the BERA are based on the range of lead concentrations in surface soil most likely to pose adverse effects in terrestrial invertebrates and terrestrial plants. Because lead has been identified as a COPEC in soil at all of the CC ranges and has been used as one of the indicators of potential contamination from Army activities at small arms ranges at Fort McClellan, ecological sample locations will be based upon lead concentrations.

Data in the scientific literature indicate that phytotoxicity can occur in soils with lead concentrations as low as 110 mg/kg (USEPA, 2005) and that terrestrial invertebrate toxicity can occur in soils with lead concentrations greater than 5,940 mg/kg (Neuhauser et al., 1985). Additionally, data collected for the Iron Mountain Road and Bains Gap Road Ranges BERA indicated terrestrial invertebrate toxicity due to lead in soil at concentrations as low as 779 mg/kg. Based on this broad range of lead in soil that has the potential to pose adverse effects on terrestrial plants and/or terrestrial invertebrates, a range of lead in soil was identified for

assessment in this BERA. The range of lead in soil that will be the focus of this BERA has been identified as between 100 mg/kg and 5,000 mg/kg, with the majority of the samples collected in the 100 to 800 mg/kg range. Ten surface soil samples will be collected from the CC ranges that represent this range of lead in soil. These soil sampling locations were summarized and presented in Table 9-1 of the BERA Problem Formulation and Study Design report and are also summarized below.

Sample Location	Range	Lead Concentration (mg/kg)
HR-94Q-GP02	Range 40	4,350
HR-CCRI-GP26	Range 43	2,510
HR-148Q-MW01	Impact Area, Choccolocco Corridor	1,010
HR-CCRI-MW03	Range 40	793
HR-CCRI-GP13	Range 41	674
HR-CCRI-GP10	Impact Area, Choccolocco Corridor	515
HR-97Q-GP03	Range 43	401
HR-CCRI-GP02	Range, Choccolocco Corridor	188
HR-CCRI-GP20	Impact Area, Choccolocco Corridor	106
HR-96Q-GP04	Range, Choccolocco Corridor	64.4
HR-146Q-MW01	Reference	21

In addition, one surface soil sample will be collected from a non-impacted area within or adjacent to the CC ranges and will represent the reference location. Figure 9-1 in the BERA Problem Formulation and Study Design report presents the approximate locations of the surface soil samples representative of the lead concentration gradient to be sampled at the CC ranges.

A.3.0 Sampling and Analysis Requirements

The following sections present the soil sampling and analysis requirements for the collection of soil for use in the earthworm toxicity and bioaccumulation studies and the terrestrial plant toxicity and bioaccumulation studies in conjunction with the BERA for the CC ranges.

A.3.1 Sample Confirmation

Prior to the collection of soil for analytical and toxicological testing, lead concentrations at the selected sample locations will be screened *in situ* using x-ray fluorescence (XRF) technology to verify that the selected locations are appropriate (i.e., lead concentration) for the intended lead gradient. XRF methodology will follow the procedures outlined in the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.3.2 Sample Collection Procedures

Once the lead concentrations have been confirmed using XRF, soil will be collected to a depth of 0.5 feet, using a stainless-steel hand auger or spoon, and homogenized in a stainless-steel bowl following the sampling procedures outlined in the Installation-Wide Sampling and Analysis Plan (IT, 2002). Soil samples will then be transferred to the appropriate sample containers. Visible bullets and lead fragments will be manually removed from the sample prior to being transferred to the sample containers. Samples for chemical analysis and toxicity testing will not be sieved.

A.3.3 Decontamination Procedures

All equipment used for collection, homogenization, and transfer of soil samples will be properly decontaminated prior to collecting samples and between sampling locations, as described in the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.3.4 Quality Assurance/Quality Control Samples

As established by the data quality objectives process, field and laboratory quality assurance/quality control indicator soil samples and analyses will be collected to provide information concerning the measured quality and usability of the field data. As presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002), the frequency of field duplicates, matrix spike/matrix spike duplicates, and equipment rinse blanks will be 1 in 10 (10 percent), 1 in 20 (5 percent), and once per sampling event, respectively.

As presented in the earthworm toxicity/bioaccumulation and plant toxicity test protocols (Section A.3.6), both a reference and laboratory control sample will be used to ensure the quality of biological testing.

A.3.5 Sample Labeling, Packaging, and Shipment

All prepared samples will be labeled, packaged, and shipped to the appropriate analytical or biological testing laboratory as presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.3.6 Analysis

Analysis of surface soil samples will consist of chemical analysis, earthworm toxicity tests, earthworm bioaccumulation tests, and terrestrial plant toxicity tests. These analyses are described below.

A.3.6.1 Chemical Analyses

As presented in Table A-1, chemical analyses of soils collected for earthworm toxicity, earthworm bioaccumulation, and plant toxicity testing will include TAL metals, VOCs, SVOCs, chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, total organic carbon, pH, and grain size. Chemical analyses of earthworm tissue after termination of the toxicity/bioaccumulation tests will include TAL metals and organochlorine pesticides (Table A-2). Chemical analyses of terrestrial plant tissues collected on-site will include TAL metals and organochlorine pesticides (Table A-3).

A.3.6.2 Earthworm Toxicity Testing

Surface soil used for earthworm toxicity and bioaccumulation testing will be "split" from the soil samples used for chemical analysis. The 28-day earthworm survival and growth tests, and the earthworm bioaccumulation tests will be conducted using the earthworm *Eisenia fetida*. Earthworm toxicity and bioaccumulation tests will be conducted in accordance with the guidelines set forth in *Standard Guide for Conducting Laboratory Soil Toxicity of Bioaccumulation With the Lumbricid Earthworm E. fetida* (ASTM, 1998a), and USEPA's *Ecological Effects Test Guidelines* (1996).

A.3.6.2.1 Test Initiation

Earthworm toxicity/bioaccumulation tests will begin within 10 days of test soil collection. Immediately prior to testing, the temperature of the test soils will be adjusted to 20 ± 2 degrees Celsius (°C). Test conditions are presented in Table A-4.

Test soils will be hydrated with deionized water to create a moist testing environment. The earthworm test soils will be hydrated to 75 percent of water holding capacity. Ten earthworms will be placed into each of five replicate containers each containing 200 grams (dry weight) of test soil. The earthworms will be placed on the surface of the test soil in a pint jar, capped, and secured. The tests will be incubated within an environmental chamber to give soil temperature of 20 ± 2°C under continuous light. Lighting will be at continuous ambient laboratory levels, which is approximately 540 to 1,080 lux, with no shading.

A.3.6.2.2 Termination of the Test

Mortality will be assessed after 14 and 28 days (termination of the test) by emptying the test soil onto a tray and sorting the worms from the soil. Dead worms will be separated from the live worms and live worms will be placed back into their test jars and placed on the surface of the soil. The numbers of live and dead worms in each test chamber will be recorded at 14 days and at the termination of the test (28 days). The total weight of living earthworms in each test chamber will also be recorded at the termination of the test.

Table A-1

Surface Soil Sample Designations and QA/QC Sample Quantities
Baseline Ecological Risk Assessment Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Sample Depth (ft)	QA/QC Samples			Analytical Suite
			Field Duplicates[a]	Field Splits	MS/MSD[a]	
HR-94Q-GP02	HR-94Q-GP02-SS-RW0001-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-CCRI-GP26	HR-CCRI-GP26-SS-RW0002-REG	0-0.5	HR-CCRI-GP26-SS-RW0003-FD			VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-148Q-MW01	HR-148Q-MW01-SS-RW0004-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-CCRI-MW03	HR-CCRI-MW03-SS-RW0005-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-CCRI-GP13	HR-CCRI-GP13-SS-RW0006-REG	0-0.5			HR-CCRI-GP13-SS-RW0006-MS/MSD	VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-CCRI-GP10	HR-CCRI-GP10-SS-RW0007-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-97Q-GP03	HR-97Q-GP03-SS-RW0008-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.

Table A-1

Surface Soil Sample Designations and QA/QC Sample Quantities
Baseline Ecological Risk Assessment Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Sample Depth (ft)	QA/QC Samples			Analytical Suite
			Field Duplicates[a]	Field Splits	MS/MSD[a]	
HR-CCRI-GP02	HR-CCRI-GP02-SS-RW0009-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-96Q-GP04	HR-96Q-GP04-SS-RW0010-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-CCRI-GP20	HR-CCRI-GP20-SS-RW0011-REG	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.
HR-146Q-MW01	HR-146Q-MW01-SS-RW0012-REF	0-0.5				VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, Chlorinated Herbicides by 8151A, and TAL Metals by SW6010B/ SW7471A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, 28-day Earthworm Survival and Growth Test, and 15-day Perennial Ryegrass Germination and Growth Test.

[a] Field duplicates and MS/MSDs are collected for chemical analysis only and not for biological testing.

FD - Field duplicate.
FS - Field split.
MS/MSD - Matrix spike/matrix spike duplicate.
QA/QC - Quality assurance/quality control.
REG - Field sample.
REF - Reference sample

VOCs - Volatile Organic Compounds.
SVOCs - Semivolatile Organic Compounds.
ASTM - American Society of Testing and Materials.
TAL - Target Analyte List.
TOC - Total Organic Carbon.

Table A-2
Earthworm Tissue Sample Designations
BERA Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Analytical Suite
HR-94Q-GP02	HR-94Q-GP02-WORM-RW4001-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-WORM-RW4002-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-WORM-RW4003-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-WORM-RW4004-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-WORM-RW4005-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP26	HR-CCRI-GP26-WORM-RW4006-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-WORM-RW4007-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-WORM-RW4008-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-WORM-RW4009-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-WORM-RW4010-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-148Q-MW01	HR-148Q-MW01-WORM-RW4011-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-WORM-RW4012-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-WORM-RW4013-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-WORM-RW4014-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-WORM-RW4015-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-MW03	HR-CCRI-MW03-WORM-RW4016-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-WORM-RW4017-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-WORM-RW4018-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-WORM-RW4019-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-WORM-RW4020-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP13	HR-CCRI-GP13-WORM-RW4021-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-WORM-RW4022-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-WORM-RW4023-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-WORM-RW4024-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-WORM-RW4025-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP10	HR-CCRI-GP10-WORM-RW4026-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-WORM-RW4027-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-WORM-RW4028-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-WORM-RW4029-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-WORM-RW4030-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-97Q-GP03	HR-97Q-GP03-WORM-RW4031-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-WORM-RW4032-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-WORM-RW4033-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-WORM-RW4034-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-WORM-RW4035-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A

Table A-2
Earthworm Tissue Sample Designations
BERA Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Analytical Suite
HR-CCRI-GP02	HR-CCRI-GP02-WORM-RW4036-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-WORM-RW4037-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-WORM-RW4038-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-WORM-RW4039-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-WORM-RW4040-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-96Q-GP04	HR-96Q-GP04-WORM-RW4041-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-WORM-RW4042-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-WORM-RW4043-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-WORM-RW4044-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-WORM-RW4045-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP20	HR-CCRI-GP20-WORM-RW4046-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-WORM-RW4047-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-WORM-RW4048-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-WORM-RW4049-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-WORM-RW4050-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-146Q-MW01	HR-146Q-MW01-WORM-RW4051-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-WORM-RW4052-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-WORM-RW4053-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-WORM-RW4054-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-WORM-RW4055-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A

REG - Field sample.
TAL - Target Analyte List.
REF - Reference sample

Table A-3
Terrestrial Plant Tissue Sample Designations
BERA Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	QA / QC Samples		Analytical Suite
		Field Duplicates	MS / MSD	
HR-94Q-GP02	HR-94Q-GP02-PLANT-RW3001-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-PLANT-RW3002-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-PLANT-RW3003-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-PLANT-RW3004-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-94Q-GP02-PLANT-RW3005-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP26	HR-CCRI-GP26-PLANT-RW3006-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-PLANT-RW3007-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-PLANT-RW3008-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-PLANT-RW3009-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP26-PLANT-RW3010-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-148Q-MW01	HR-148Q-MW01-PLANT-RW3011-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-PLANT-RW3012-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-PLANT-RW3013-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-PLANT-RW3014-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-148Q-MW01-PLANT-RW3015-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-MW03	HR-CCRI-MW03-PLANT-RW3016-REG	HR-CCRI-MW03-PLANT-RW3021-FD		TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-PANT-RW3017-REG	HR-CCRI-MW03-PLANT-RW3022-FD		TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-PLANT-RW3018-REG	HR-CCRI-MW03-PLANT-RW3023-FD		TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-PLANT-RW3019-REG	HR-CCRI-MW03-PLANT-RW3024-FD		TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-MW03-PLANT-RW3020-REG	HR-CCRI-MW03-PLANT-RW3025-FD		TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP13	HR-CCRI-GP13-PLANT-RW3026-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-PLANT-RW3027-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-PLANT-RW3028-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-PLANT-RW3029-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP13-PLANT-RW3030-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP10	HR-CCRI-GP10-PLANT-RW3031-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-PLANT-RW3032-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-PLANT-RW3033-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-PLANT-RW3034-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP10-PLANT-RW3035-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-97Q-GP03	HR-97Q-GP03-PLANT-RW3036-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-PLANT-RW3037-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-PLANT-RW3038-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-PLANT-RW3039-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-97Q-GP03-PLANT-RW3040-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP02	HR-CCRI-GP02-PLANT-RW3041-REG		HR-CCRI-GP02-PLANT-RW3041-MS/MSD	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-PLANT-RW3042-REG		HR-CCRI-GP02-PLANT-RW3042-MS/MSD	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-PLANT-RW3043-REG		HR-CCRI-GP02-PLANT-RW3043-MS/MSD	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-PLANT-RW3044-REG		HR-CCRI-GP02-PLANT-RW3044-MS/MSD	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP02-PLANT-RW3045-REG		HR-CCRI-GP02-PLANT-RW3045-MS/MSD	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A

Table A-3
Terrestrial Plant Tissue Sample Designations
BERA Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	QA / QC Samples		Analytical Suite
		Field Duplicates	MS / MSD	
HR-96Q-GP04	HR-96Q-GP04-PLANT-RW3046-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-PLANT-RW3047-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-PLANT-RW3048-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-PLANT-RW3049-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-96Q-GP04-PLANT-RW3050-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-GP20	HR-CCRI-GP20-PLANT-RW3051-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-PLANT-RW3052-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-PLANT-RW3053-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-PLANT-RW3054-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-CCRI-GP20-PLANT-RW3055-REG			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-146Q-MW01	HR-146Q-MW01-PLANT-RW3056-REF			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-PLANT-RW3057-REF			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-PLANT-RW3058-REF			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-PLANT-RW3059-REF			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
	HR-146Q-MW01-PLANT-RW3060-REF			TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A

REG - Field sample.
TAL - Target Analyte List.
REF - Reference sample
FD - Field Duplicate
MS/MSD - Matrix Spike / Matrix Spike Duplicate

Table A-4
Summary of Test Conditions For Earthworm Survival and Bioaccumulation Test Choccolocco Corridor Ranges BERA

Parameter	Condition
Test Type	Static
Soil Temperature	20 ± 2°C
Light Quality	Ambient laboratory light
Light Intensity	540 – 1,080 lux
Photoperiod	Continuous illumination
Test Vessel Type and Size	1 liter glass jar with screw-top lid and air hole
Test Soil Mass	200 grams
Test Soil pH	> 4 and < 10
Laboratory Control Soil	10% 2.36 mm screened sphagnum peat, 20% colloidal kaolinite clay, and 70 % grade 70 silica sand
Test Soil Moisture Content	75% water holding capacity
Test Soil Renewal	None
Test Organism Age	> 60 days
Number of Test Organisms per Chamber	10
Number of Replicate Chambers	5
Feeding Regime	No feeding, unless reference site mortality is 20% or greater, or if the total mean weight of worms in reference soil decreases 30% or more
Test Soil Dilution	None (100% undiluted site soil)
Test Duration	28 days
Effects Measured	Mortality, growth, and tissue analysis for COPEC concentrations

A.3.6.2.3 Acceptability of Test Results

For test results to be acceptable, mean survival in the laboratory control tests must be at least 90 percent.

A.3.6.2.4 Data Interpretation

The effects measured in the earthworm toxicity tests are mortality and growth, while the effect measured in the earthworm bioaccumulation test is COPEC concentrations within earthworm tissue. Results of the toxicity and bioaccumulation testing will be interpreted as described in Chapter 10 of the BERA Problem Formulation and Study Design report.

A.3.6.3 Perennial Ryegrass Toxicity Testing

Surface soil used for perennial ryegrass toxicity testing will be "split" from the soil samples used for chemical analysis. The 15-day perennial ryegrass toxicity tests will be conducted using the

perennial ryegrass *Lolium perenne* as described in *Standard Guide for Conducting Terrestrial Plant Toxicity Tests* (ASTM, 1998b).

A.3.6.3.1 Test Initiation

Perennial ryegrass toxicity tests will begin within 10 days of test soil collection. Immediately prior to testing, the temperature of the test soils will be adjusted to 20 ±2 degrees Celsius (°C). Test conditions are presented in Table A-5.

Table A-5
Summary of Test Conditions For Perennial Ryegrass Germinati on and Growth Test Choccolocco Corridor Ranges BERA

Parameter	Condition
Test Type	Static
Soil Temperature	20 ± 2°C
Light Quality	Ambient laboratory light
Light Intensity	540 – 1,080 lux
Photoperiod	16 hours light: 8 hours dark
Test Vessel Type and Size	Petri dish
Test Soil Mass	200 grams
Test Soil pH	> 4 and < 10
Laboratory Control Soil	10% 2.36 mm screened sphagnum peat, 20% colloidal kaolinite clay, and 70 % grade 70 silica sand
Test Soil Moisture Content	85% water holding capacity
Test Soil Renewal	None
Test Organism Age	Seeds
Number of Seeds per Chamber	10
Number of Replicate Chambers	5
Watering Regime	Daily in order to maintain 85% soil holding capacity
Test Soil Dilution	None (100% undiluted site soil)
Test Duration	15 days
Effects Measured	Percent germination, shoot length, above-ground biomass, root length, below-ground biomass

Test soils will be hydrated with deionized water to create a moist testing environment. The perennial ryegrass test soils will be hydrated to 85 percent of water holding capacity. Each soil sample will have five replicate containers each containing 200 grams (dry weight) of test soil. Ten seeds will be placed into the soil in each of the soil containers at a depth of two times the seed diameter. The tests will be incubated within an environmental chamber to give a soil

temperature of 20 ± 2°C with a photoperiod of sixteen hours of light and eight hours of dark. Lighting will be at ambient laboratory levels, which is approximately 540 to 1,080 lux.

A.3.6.3.2 Termination of the Test

The primary data collected from this study will be the number of seedlings out of the total number of seeds planted that emerge above the soil. All measurements will be taken after termination of the test (15 days post planting). Additional measurements that will be made at the termination of the test are plant height above the soil surface, above-ground biomass, root length, and below-ground biomass.

Shoot length measurements (height above soil surface) are made from the transition point between the hypocotyl and root to the tallest point on the shoot. Root length measurements are made from the transition point between the hypocotyl and root to the tip of the root. Above-ground biomass and below-ground biomass will be measured by separating the above-ground portion of the plant from the below-ground portion of the plant, placing each in a separate pre-weighed drying vessel, and placing in a drying oven set at 70 °C until constant weight is achieved (approximately 24 hours). The total weight of the above-ground biomass and below-ground biomass from each soil sample will them be recorded to the nearest 0.001 gram.

A.3.6.3.3 Acceptability of Test Results

For test results to be acceptable, mean germination success in the laboratory control tests must be at least 90 percent.

A.3.6.3.4 Data Interpretation

The effects measured in the perennial ryegrass toxicity tests are germination success and growth. Results of the perennial ryegrass toxicity testing will be interpreted as described in Section 10 of the BERA Problem Formulation and Study Design report.

A.3.6.4 Terrestrial Plant Collection and Analysis

Terrestrial plants will be collected and analyzed for TAL metals and chlorinated pesticides in order to assess the bioavailability of COPECs in soil and the potential for COPEC accumulation in terrestrial plant tissues. Plants will be collected from the same locations as the soil samples used for earthworm and plant toxicity testing (10 on-site samples and 1 off-site reference location). Collecting plants from the same locations as the soil samples will ensure that a broad spectrum of COPEC soil concentrations will be assessed and incorporated into the analysis. Only the above-ground portions of the plants will be sampled since most herbivores and omnivores only eat the above-ground portion of the plant. An attempt will be made to collect

grass species since the terrestrial plant toxicity testing will be conducted with a grass species, and collecting a similar species will allow for inferences to be made regarding plant bioaccumulation and potential toxic responses.

A.3.6.4.1 Plant Sample Collection

Plant samples will be collected by cutting the above-ground portion of the plant as close to the ground surface as possible with a pair of scissors. All loose dirt will be removed from the plant sample by physical action prior to placement into their respective sample containers. The plant samples will then be transferred to a plastic bag and placed in a cooler with ice.

A.3.6.4.2 Decontamination Procedures

All equipment used for collection of terrestrial plant samples will be properly decontaminated prior to collecting samples and between sampling locations, as described in the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.3.6.4.3 Sample Labeling, Packaging, and Shipment

All terrestrial plant tissue samples will be labeled, packaged, and shipped to the appropriate analytical laboratory as presented in the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.3.6.4.4 Plant Tissue Analysis

As presented in Table A-3, chemical analyses of terrestrial plant tissues will include TAL metals and chlorinated pesticides.

A.4.0 Safety and Health and Unexploded Ordnance Support

All work conducted in conjunction with the BERA for the CC ranges will be conducted in accordance with the August 2001, Site Specific Safety and Health Plan and Site-Unexploded Ordnance Safety Plan Attachments to the Site-Specific Field Sampling Plan for the Former Choccolocco Corridor Ranges. These attachments will be updated to be consistent with the Installation-Wide Sampling and Analysis Plan (IT, 2002).

A.5.0 References

American Society for Testing and Materials (ASTM), 1998a. ***Standard Guide for Conducting Laboratory Soil Toxicity or Bioaccumulation with the Lumbricid Earthworm E. fetida. ASTM,*** West Conshohocken, PA, ***E1676-97***

American Society for Testing and Materials (ASTM), 1998. S***tandard Guide for Conducting Terrestrial Plant Toxicity Tests***, ASTM, West Conshohocken, PA, E 1963-98.

Shaw Environmental, Inc., 2004, ***Remedial Investigation Report, Baby Bains Gap Road Ranges, Fort McClellan, Calhoun County, Alabama***, August.

IT Corporation (IT), 2002, ***Installation-Wide Sampling and Analysis Plan, Fort McClellan, Calhoun County, Alabama***, Revision 3, February.

Neuhauser, E.F., R.C. Loehr, D.L. Milligan, and M.R. Malecki, 1985, "Toxicity of Metals to the Earthworm *Eisenia fetida,*" ***Biol. Fertil. Soils***, 1: 149 - 152.

U.S. Environmental Protection Agency (EPA), 2005, ***Ecological Soil Screening Levels for Lead***, Office of Solid Waste and Emergency Response, Washington, D.C., OSWER Directive 9285.7-70.

U.S. Environmental Protection Agency (EPA), 1996. Ecological Effects Test Guidelines. OPPTS 850.6200, Earthworm Subchronic Toxicity Test. EPA/712/C-96/167.

APPENDIX B

FIELD SAMPLING AND ANALYSIS PLAN FOR SEDIMENT

Appendix B
Field Sampling and Analysis Plan
For Sediment at the Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

B.1.0 Introduction

As presented in the baseline ecological risk assessment (BERA) problem formulation for the Choccolocco Corridor (CC) ranges (Shaw Environmental Inc. [Shaw], 2005), one inorganic constituent (lead) and one pesticide (gamma chlordane) were identified as chemicals of potential ecological concern (COPEC) in sediment at the CC ranges. To provide information for the BERA, sediment samples will be collected and analyzed for target analyte list (TAL) metals, volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, pH, grain size, and total organic carbon (TOC). In addition, sediment samples will be analyzed for toxicity and bioaccumulation in chironomid larva. COPEC concentrations in sediments will also be used in food web models to predict the total daily doses of COPECs in invertivorous mammals and birds.

B.2.0 Selection of Sample Locations

Eight (8) sediment samples will be collected from locations representative of the full range of historical lead concentrations detected in sediment at the CC ranges. Sediment will also be collected from two streams (one ephemeral and one perennial) with similar substrate characteristics as those drainage features found within the CC ranges, but outside the influence of the CC ranges, to serve as the reference locations for the sediment toxicity tests. Sediment sample locations will be based on historical lead concentrations in sediment because lead has been identified as a COPEC in sediment at each of the CC ranges, and has been used as one of the indicators of potential contamination from Army activities at small arms ranges at FTMC. The sediment sample locations were summarized in Table 9-2 of the BERA Problem Formulation and Study Design report and are also summarized below.

Sample Location	Range	Lead Concentration (mg/kg)
HR-CCRI-SD18	Range, Choccolocco Corridor (Parcel 144Q-X)	394
HR-CCRI-SD02	Range, Choccolocco Corridor (Parcel 146Q)	124
HR-CCRI-SD04	Range, Choccolocco Corridor (Parcel 146Q)	52
HR-CCRI-SD17	Range, Choccolocco Corridor (Parcel 144Q-X)	45

Sample Location	Range	Lead Concentration (mg/kg)
HR-CCRI-SD07	Range, Choccolocco Corridor (Parcel 146Q)	35
HR-95Q-SD01	Range 41, Choccolocco Corridor (Parcel 145Q)	34.7
HR-131Q-SD01	Range 41 Impact Area (Parcel 131Q-X)	30.4
HR-145Q-SD02	Range, Choccolocco Corridor (Parcel 145Q-X)	19.4
HR-CCRI-SD12	Perennial Reference	8.53
HR-CCRI-SD10	Ephemeral Reference	13.5

Sediment will also be collected from both an ephemeral and perennial stream with similar substrate characteristics as the drainage features at the CC ranges but outside the influence of the CC ranges to serve as reference sediment for the toxicity tests. Additionally, the benthic invertebrate community will be analyzed using the rapid bioassessment protocol II (RBP II) at each of the on-site sediment sample locations and the reference sites. Sediment sample locations will be identified by analyzing sediments using x-ray fluorescence technology to verify that the selected locations exhibit the intended lead concentration range. X-ray fluorescence methodology will follow the procedures outlined in the installation-wide sampling and analysis plan (IT, 2002).

B.3.0 Sampling and Analysis Procedures

Unless otherwise specified, sample collection procedures will follow the *Installation-Wide Sampling and Analysis Plan* (IT, 2002). The benthic macroinvertebrate community (riffle/run and coarse particulate organic matter [CPOM] samples) will be sampled first at each sediment sampling location, followed by the collection of sediment samples for chemical and toxicity testing. Sediment samples will be collected from the farthest downstream location first and then proceed upstream.

B.3.1 Sediment Sampling

Prior to the collection of sediment samples, the following in-stream water quality measurements will be recorded:

- pH
- Conductivity
- Dissolved oxygen
- Temperature
- Oxidation reduction potential

Sediment samples will be collected from the zero to six-inch depth interval with stainless steel spoons or trowels and homogenized in a stainless steel bowl following the procedures outlined in the *Installation-Wide Sampling and Analysis Plan* (IT, 2002). A list of the sample containers and preservatives required for each analysis for sediment samples is also provided in the *Installation-Wide Sampling and Analysis Plan* (IT, 2002).

At each sediment sampling location, the water depth, stream width, substrate type, and approximate flow velocity will also be recorded. Other observations that will be recorded include weather conditions, surrounding vegetative cover, surrounding land-use, and evidence of erosion.

B.3.2 Decontamination Procedures

All equipment used for collection, homogenization, and transfer will be properly decontaminated prior to collecting samples and between sampling locations, as described in the *Installation-Wide Sampling and Analysis Plan* (IT, 2002).

B.3.3 Quality Assurance/Quality Control Samples

As established by the DQO process, field and laboratory QA/QC indicator sediment samples and analyses will be collected to provide information concerning the measured quality and usability of the field data. As presented in the *Installation-Wide Sampling and Analysis Plan* (IT, 2002), the frequency of field duplicates, MS/MSDs, and equipment rinse blanks will be 1 in 10 (10%), 1 in 20 (5%), and once per sampling event, respectively.

B.3.4 Sample Labeling, Packaging, and Shipment

All prepared samples will be labeled, packaged, and shipped to the appropriate analytical or biological testing laboratory as presented in the *Installation-Wide Sampling and Analysis Plan* (IT, 2002).

B.3.5 Chemical Analysis

As presented in Table B-1, chemical analyses of sediments collected for chironomid toxicity and bioaccumulation testing will include TAL metals, VOCs, SVOCs, chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, total organic carbon, pH, and grain size. Chemical analyses of chironomid tissue after termination of the toxicity/bioaccumulation tests will include TAL metals, and organochlorine pesticides (Table B-2).

Table B-1
Sediment Sample Designations and QA/QCSample Quantities
BERA Study Design for the Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Sample Depth (ft)	QA/QC Samples			Analytical Suite
			Field Duplicates[a]	Field Splits	MS/MSD[a]	
HR-CCRI-SD18	HR-CCRI-SD18-SD-RW2001-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD02	HR-CCRI-SD02-SD-RW2002-REG	0 - 0.5	HR-CCRI-SD02-SD-RW2003-FD			TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD04	HR-CCRI-SD04-SD-RW2004-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD17	HR-CCRI-SD17-SD-RW2005-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD07	HR-CCRI-SD07-SD-RW2006-REG	0 - 0.5			HR-CCRI-SD07-SD-RW2006-MS/MSD	TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-95Q-SD01	HR-95Q-SD01-SD-RW2007-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-131Q-SD01	HR-131Q-SD01-SD-RW2008-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.

Table B-1
Sediment Sample Designations and QA/QCSample Quantities
BERA Study Design for the Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Sample Depth (ft)	QA/QC Samples			Analytical Suite
			Field Duplicates[a]	Field Splits	MS/MSD[a]	
HR-145Q-SD02	HR-145Q-SD02-SD-RW2009-REG	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD12	HR-CCRI-SD12-SD-RW2010-REF	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.
HR-CCRI-SD10	HR-CCRI-SD10-SD-RW2011-REF	0 - 0.5				TAL Metals by SW6010B/SW7470A, VOCs by 8260B, SVOCs by 8270C, Chlorinated Pesticides by 8081A, Organophosphorus Pesticides by 8141A, and Chlorinated Herbicides by 8151A. TOC by Walkley Black, pH by SW9045C, Grain Size by ASTM 421/422, *Chironomus tentans* 21-day Survival and Growth Test by EPA 100.5.

[a] Field duplicates and MS/MSDs are collected for chemical analysis only and not for biological testing.

FD - Field duplicate.
FS - Field split.
MS/MSD - Matrix spike/matrix spike duplicate.
QA/QC - Quality assurance/quality control.
TOC - Total Organic Carbon.
REG - Field sample.
REF - Reference sample
TAL - Target analyte list.
VOCs - volatile organic compounds.
SVOCs - semivolatile organic compounds.

Table B-2
Chironomid Tissue Sample Designations
BERA Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Calhoun County, Alabama

Sample Location	Sample Designation	Analytical Suite
HR-CCRI-SD18	HR-CCRI-SD18-MIDGE-RW5001-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD02	HR-CCRI-SD02-MIDGE-RW5002-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD04	HR-CCRI-SD04-MIDGE-RW5003-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD17	HR-CCRI-SD17-MIDGE-RW5004-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD07	HR-CCRI-SD07-MIDGE-RW5005-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-95Q-SD01	HR-95Q-SD01-MIDGE-RW5006-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-131Q-SD01	HR-131Q-SD01-MIDGE-RW5007-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-145Q-SD02	HR-145Q-SD02-MIDGE-RW5008-REG	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD12	HR-CCRI-SD12-MIDGE-RW5009-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A
HR-CCRI-SD10	HR-CCRI-SD10-MIDGE-RW5010-REF	TAL Metals by SW6010B/SW7471A and Chlorinated Pesticides by 8081A

B.3.6 Chironomid Toxicity Testing

Biological testing of sediments collected at the CC ranges will consist of toxicity testing and a bioaccumulation study of the benthic invertebrate *Chironomus tentans*.

B.3.6.1 Test Objective

The direct toxicity of sediment-bound COPECs will be measured by exposing benthic invertebrates (*Chironomus tentans*) to streambed sediment. Use of chironomids to measure toxicity of sediment-associated contaminants is quite common and has been standardized by USEPA (USEPA, 2000).

Measuring growth as well as survival over the 21-day exposure period permits an evaluation of chronic (sub-lethal) endpoints in addition to acute toxicity. Adverse sub-lethal responses could affect the long-term viability of benthic invertebrate communities within impact zones and, therefore, affect the viability of the stream ecosystem. A summary of the test conditions is provided in Table B-3.

B.3.6.2 Test Sediment Dilution Series

Given the uncertainties and difficulties associated with laboratory dilution and subsequent mixing of sediments, test organisms will be exposed to 100 percent undiluted field collected sediment. Tests will be set up with exposure to laboratory-based synthetic control sediment, reference sediment, and on-site sediment representing the full range of lead concentrations detected in historical sediment samples from the CC ranges.

B.3.6.3 Test Initiation

Tests will be initiated within 10 days of sample collection, and the laboratory grade overlying test water will be maintained at 23 $\pm$ 1°C. Test chambers will consist of 300 milliliter high form lipless beakers containing 100 ml of sediment and 175 ml of overlying water. Ten second-to-third instar *C. tentans* midges (approximately 10 days old) will be used at test initiation. A total of 5 replicates will be employed for each parallel test.

Midges within each test chamber will be fed 1-5 ml of a 4-g/100 ml tetrafin suspension on a daily basis throughout the 21-day test period. Each replicate test chamber will receive two-volume additions/day of overlying water. Water renewals will be conducted in a manner that minimizes suspension of sediment. All testing will, therefore, be static daily renewals with careful monitoring of physico-chemical parameters within the overlying water. These parameters will include pH, temperature, ammonia, alkalinity, hardness, conductivity, and dissolved oxygen.

Table B-3
Summary of Chironomus tentans Survival and Growth Test

Parameter	Conditions
Test Type	Whole-sediment toxicity with renewal of overlying water
Temperature	23 ± 1°C
Light Quality	Wide-spectrum fluorescent lights
Illuminance	~100 – 1,000 lux
Photoperiod	16 hours light:8 hours dark
Test chamber	300-ml high form lipless beaker
Sediment volume	100 ml
Overlying water volume	175 ml
Renewal of overlying water	2 volume additions per day, either continuous or Intermittent
Age of organisms	Approx. 10 day old larvae at start of test
Number of organisms per chamber	10
Number of replicate chambers per treatment	5
Feeding	Tetrafin goldfish food, fed 1.5 ml daily to each test chamber starting day-1. If fungal or bacterial growth develops on sediment surface, feeding should be suspended for one or more days. If DO drops below 2.5 mg/L during the test, feeding should be suspended for the amount of time necessary to increase the DO. If feeding is suspended in one treatment, it is suspended in all treatments.
Aeration	None, unless DO in overlying water drops below 2.5 mg/L
Overlying water	Culture water, laboratory-grade freshwater, or reconstituted water
Test chamber cleaning	Gently brush outside of overflow screens if they become clogged
Overlying water quality	Hardness, alkalinity, conductivity, and ammonia at the beginning and at the end of the test (day 21). Temperature daily. DO and pH three times/week. Conductivity weekly. Concentrations of DO should be measured more often if DO has declined by more than 1 mg/L since previous measurement. Overlying water quality should be measured just prior to water renewals. Overlying water should be measured from about 1 to 2 cm above the sediment surface.
Test duration	Five replicates are ended at 21 days for survival and weight.
Endpoints	21-day survival and weight; COPEC concentrations in chironomid tissues
Test acceptability	Average size of *C. tentans* in control sediment at 21 days ≥ 0.6 mg/surviving organism as dry weight or 0.48 mg/surviving organisms as AFDW. Emergence ≥ 50%

B.3.6.4 Test Monitoring

All chambers will be checked daily and observations made to assess test organism behavior such as sediment avoidance.

B.3.6.5 Measurement of Overlying Water-Quality Characteristics

Conductivity, hardness, pH, alkalinity, and ammonia will be measured in all treatments at the beginning and end of the test. Overlying water will be sampled just before water renewal from about 1 to 2 cm above the sediment surface using a pipet.

B.3.6.6 Test Termination

At the termination of the toxicity test (day 21 of the exposure period), immobile organisms isolated from the sediment surface or from sieved material will be considered dead. A #40 sieve (425-µm mesh) will be used to remove midges from sediment. Surviving midges will be removed from the sediment and enumerated to determine survivability. All live organisms will then be pooled and weighed to determine weight change.

B.3.6.7 Test Data

Ash-free dry weight (AFDW) and survival will be the endpoints measured at the end of the 21-day sediment toxicity test.

For determination of AFDW, all living larvae in each replicate will be pooled and the sample will be dried to a constant weight (e.g., 60 °C for 24 hours). At the termination of the test and after determination of AFDW, each pooled sample will be analyzed for the COPECs as presented in Table B-2.

B.3.7 Rapid Bioassessment

A biological assessment of the benthic invertebrate community using the EPA Rapid Bioassessment Protocol II (RBP II) (Plafkin, et. al., 1989) will be performed at each sediment sampling location. RBP II will be used to determine whether on-site benthic invertebrate community structure is being adversely affected by sediment COPECs at the CC ranges

The locations for benthic invertebrate community analysis will be co-located with the sediment sample locations. The sampling locations will be located in areas similar in habitat so that the benthic community can be evaluated under similar environmental conditions.

Rapid Bioassessment Protocol (RBP) II as developed by EPA (Plafkin et. al., 1989) will be used to quantitatively assess the biotic health of the benthic community in the perennial and ephemeral drainage features at the CC ranges. RBPs were initially designed as a relatively inexpensive screening tool for use in determining if freshwater streams were capable of supporting designated aquatic life uses. However according to EPA, the bioassessment protocols have also been found useful in characterizing the existence and severity of use impairment within freshwater systems including full watersheds, as well as identifying sources and causes to the impairment. RBP II is well-suited for screening the streams within the CC ranges for biotic integrity.

At each sampling location, water quality measurements will be obtained. Habitat quality observations including substrate type, surrounding land use, evidence of erosion and pollutant sources, vegetative stream canopy, and other relevant data will be noted.

B.3.7.1 Macroinvertebrate Sampling

Two macroinvertebrate samples will be collected at each sampling station, the riffle/run sample will be collected with a kick net and the CPOM sample will be collected by hand.

B.3.7.1.1 Kick Net Samples

The kick net sample provides data as to the abundance of the scraper and filtering collector functional feeding groups and is generally collected in a riffle and a run area of the stream. The riffle and the run sample will be composited in the field for processing as one sample per location. The kick net consists of a 0.9 mm mesh bag attached to a rectangular 8 by 18-inch frame mounted on a handle. The use of the sampler is described as follows:

1. The sampler is positioned securely on the substrate with the opening of the net facing upstream.

2. An area of approximately one square meter immediately upstream of the sampler is disturbed by overturning and scraping rocks and large stones by shifting the feet to dislodge clinging or attached organisms. Any rocks or other large items that have been swept into the net are examined to ensure that organism removal is complete.

3. The remaining sediment is agitated with the feet to dislodge epibenthic and burrowing organisms.

All organisms and debris such as sticks and leaves will be removed from the kick net bag and placed into a container with 95% ethanol to preserve the organisms.

B.3.7.1.2 Coarse Particulate Organic Matter (CPOM) Samples

One CPOM sample will be collected at each location from depositional areas with low current velocity within the stream. The CPOM sample, which provides data as to the abundance of the shredder feeding group, will be collected by hand including a composite variety of leaves, twigs, bark and other fragments. The collected material and organisms will be placed into a sample container with 95% ethanol.

Organisms in the riffle/run sample and the CPOM sample will be identified and enumerated separately and then combined prior to metric analysis to form a single benthic macroinvertebrate sample for each sample location per USEPA guidance (Barbour, et al., 1999).

4.0 Health and Safety and Unexploded Ordnance Support

All work conducted during the BERA for the CC ranges will be conducted in accordance with the RI site-specific work plan for the Choccolocco Corridor Ranges (Shaw, 2003). These attachments will be updated to be consistent with the February 2002, *Draft Revision 3, Installation-Wide Sampling and Analysis Plan, Fort McClellan, Calhoun County, Alabama*, for the final BERA study design for the CC ranges.

5.0 References

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Shaw Environmental Inc. (Shaw), 2005, ***Remedial Investigation Report, Former Choccolocco Corridor Ranges, Parcels 94Q, 95Q, 96Q, 97Q, 131Q-X, 144Q-X, 145Q-X, 146Q, 147Q-X, and 148Q-X, Fort McClellan, Calhoun County, Alabama***, Draft, August.

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U.S. Environmental Protection Agency (EPA), 2000. ***Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants With Freshwater Invertebrates***. Office of Research and Development, Mid-Continent Ecology Division, Duluth, MN, EPA 600/R-99/064.

RESPONSE TO COMMENTS

ADEM

Response to ADEM Comments
Draft Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Former Choccolocco Corridor Ranges
Fort McClellan, Alabama

Comments from Stephen A. Cobb, Chief – Governmental Hazardous Waste Branch, Land Division, dated May 27, 2008.

General Comments

Comment 1: **A total of six sediment samples are proposed to be collected within three ephemeral drainage features within the Former CC Ranges. Additional samples should be collected from the perennial stream present within the central portion of the ranges as this stream represents the most important aquatic habitat present at the site and previous sediment sampling indicated the potential for adverse effects to the benthic community. A minimum of three samples should be collected from this perennial stream and undergo sediment toxicity testing (including bioaccumulation) and macroinvertebrate community assessment. Please address.**

Response 1: The studies proposed in the Baseline Ecological Risk Assessment Problem Formulation and Study Design are not designed for the purpose of characterizing the nature and extent of contamination at the Former Choccolocco Corridor Ranges. The nature and extent of contamination was determined and reported in the Remedial Investigation Report for the Former Choccolocco Corridor Ranges (Shaw, 2005). The sample locations proposed in the Problem Formulation and Study Design for the Former Choccolocco Corridor Ranges report represent the full range of COPEC concentrations in each environmental media as determined in the remedial investigation, such that the data provided by each of the proposed samples can be used to adequately fulfill the data quality objectives described in detail in Chapter 8 of the Problem Formulation and Study Design report.

In order to ensure the environmental condition of the perennial stream that flows through Parcels 145Q-X, 96Q, 95Q, and 131Q-X is assessed, three new sample locations have been proposed along this stream reach that will be analyzed for sediment COPEC concentrations, chironomid toxicity/bioaccumulation, and benthic macroinvertebrate community structure. It should be noted, however, that the concentrations of lead detected in sediment samples from this stream were all less than the background screening value (37.8 mg/kg) and gamma chlordane was not detected in any of the sediment samples.

Comment 2: **The discussion of fate and transport should include summaries of key contaminant properties, such as portioning coefficients and distribution coefficients (or ranges for the estimates for these parameters) to allow ready reference and comparison among the contaminants. In addition, the discussion should tie into the known or estimated properties of the site and discuss their relevance with regard to the specific site setting. Please address.**

Response 2: The fate and transport section will be revised to include key contaminant properties where available and known or estimated properties of the site as they relate to COPEC fate and transport.

Comment 3: **Several acronyms listed throughout the document are not identified as they are introduced. Please revise the text to identify BHC, EC_{20}, EC_{50}, LC_{50}, LD_{50}, VOC, BCF, CSM, TAL, SVOC, IT, TOC and UCL. Also, gamma-BHC is incorrectly identified on Page 8-1. Please revise.**

Response 3: The text will be revised to identify acronyms as they are introduced. Based on comments received by USEPA Region 4, several pesticides were screened out as soil (beta-BHC, dieldrin, and gamma-BHC) and sediment (heptachlor) COPECs.

Specific Comments

Comment 1: **Sections 2.1, 2.2 and 2.3. MCPA and MCPP were not identified as constituents of potential ecological concern (COPEC) in surface soil. Bis(2-ethylhexyl)phthalate was not identified as a COPEC in surface water. Methoxychlor, 4,4'-DDT, endrin and HMX were not identified as COPECs in sediment. One of the reasons for not including these compounds as COPECs was based on their infrequency of detection. However, as noted in ADEM's review of the August 2005 Remedial Investigation report, most or all of the remaining samples exhibited nondetect results with reporting limits above the ecological screening values (ESV). The rationale of infrequency of detection should not be used for these compounds as it cannot be definitively stated that these compounds were below the ESVs in the majority of samples. Please address.**

Response 1: The rationale for not selecting the aforementioned constituents as COPECs will be revised to include additional lines of evidence. Additionally, as discussed at the May 2008 BCT meeting, a brief discussion of detection limits compared to ESVs will be provided with respect to uncertainties inherent in the identification of COPECs.

Comment 2: **<u>Section 3.0</u>. Please discuss all of the ESVs used in COPEC screening, their sources and derivation *sensu* ecological receptor groups, including ecological soil screening levels (EcoSSL) and the lower of National (Environmental Protection Agency [EPA], 2006) or Alabama chronic ambient water quality criteria (AWQC), to complement the requested ESV summary table. Please explain all bioavailability-related adjustments made to the ESVs using site-specific data and indicate if any ESV conversions are being made using the EPA-prescribed conversion factors. Please discuss any site-specific sediment ESV adjustments using total organic carbon and/or considerations of metals bioavailability in relation to ratios of simultaneously extracted metals (SEM) to acid volatile sulfides (AVS), *if* such SEM:AVS data are available and considered relevant for site-derived sediments. Also, when discussing aquatic toxicity thresholds, please indicate whether the value is for total or dissolved metal concentrations.**

Response 2: All ESVs are described in detail in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000). As described in that report, all of the ESVs are values extracted from the scientific literature. No adjustments were made to these values for bioavailability or site-specificity.

ESV screening and COPEC identification are accomplished in the T-24A SLERA and are not part of the Problem Formulation and Study Design report. For clarity, a reference to the *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000) will be added to the beginning of Chapter 2.

A distinction between dissolved and total metal concentrations will be included in the discussion of surface water toxicity.

Comment 3: **<u>Section 3.2, Page 3-4</u>. In Paragraph 1, the text as written in Sentence 3 is an incomplete sentence. Please revise.**

Response 3: The sentence in question will be revised for grammatical correctness.

Comment 4: **<u>Section 3.3, Page 3-6</u>. Please insert a period at the end of Paragraph 3.**

Response 4: A period will be added to the sentence in question.

Comment 5: **<u>Section 4.0</u>. Figure 6-1 indicates some complete or potentially complete exposure pathways for groundwater. For completeness, please provide a separate section discussing the general fate and transport potential associated with groundwater, however limited.**

Response 5: Chapter 4 presents a discussion of the fate and transport mechanisms for the environmental media and the chemical-specific properties of the identified COPECs that are present at the Former Choccolocco Corridor Ranges. Because no COPECs were identified in groundwater at the Former Choccolocco Corridor Ranges, it is inappropriate to discuss groundwater fate and transport in this chapter.

There is a potential for complete exposure pathways for groundwater and ecological receptors if groundwater is expressed at the ground surface as seeps or wetlands (as presented in Figure 6-1); however, because no COPECs were identified in groundwater at the Former Choccolocco Corridor Ranges, these pathways were considered incomplete (as discussed in Chapter 6.

Comment 6: **<u>Section 4.4.1</u>. Please discuss the influence of pH on antimony mobility, the typical pH values in media where antimony was detected at the site and evaluate the potential impact on fate.**

Response 6: According to ATSDR (1992), the mobility of antimony under various environmental conditions is poorly understood, with confounding data reported as to its mobility under various conditions of pH, as presented in Section 4.4.1 of the Problem Formulation and Study Design report. This section will be revised to include additional information regarding the conflicting fate and transport data for antimony.

Comment 7: **<u>Section 4.4.3</u>. Please discuss the soil characteristics of the areas of lead deposition (e.g., pH, mineralogy, cation exchange capacity) and the potential influence on the mobility of lead at this site.**

Response 7: The specific soil characteristics that the commentor has requested have not been measured at the Former Choccolocco Corridor Ranges. However, some general conclusions can be drawn from the available data. As suggested by ATSDR (2005) and summarized in Section 4.4.3 of the Former Choccolocco Corridor Ranges Problem Formulation and Study Design report, "The mobility of lead increases in environments having low pH due to the enhanced solubility of lead under acidic conditions." Because precipitation in northeastern Alabama is expected to be acidic in nature, it could be concluded that the mobility of lead in soil would be enhanced. However, elevated concentrations of lead in subsurface soil samples and groundwater at the Former Choccolocco Corridor Ranges are not prevalent, indicating that lead is not mobile in the environment at the Former Choccolocco Corridor Ranges. ATSDR (2005) also suggests that "the fate of lead in soil is affected by the adsorption at mineral surfaces, the precipitation of sparingly soluble solid forms of the compound, and the formation of relatively stable organic-metal complexes or chelates with soil organic matter." Due to the relatively low organic content of the soils at the Former Choccolocco Corridor Ranges, the

formation of organic-metal complexes with soil organic matter is likely a minor fate process at the Former Choccolocco Corridor Ranges. The most likely processes affecting the fate of lead in soil are the adsorption of lead at mineral surfaces and the speciation of lead to more insoluble sulfate, sulfide, oxide, and phosphate salts.

All of this information is presented in Section 4.4.3 of the Former Choccolocco Corridor Ranges Problem Formulation and Study Design report; however, it will be re-formatted as presented above in order to draw site-specific conclusions.

Comment 8: **<u>Section 4.4.4</u>. Please clarify or discuss the following:**

- **The probable forms of mercury contamination found at the site (e.g., elemental, compound) based on site "process knowledge" and the potential impact on mobility;**
- **The conditions at the site and whether they present the potential for production of methylmercury;**
- **The existence of processes at the site that would promote volatilization of mercury,**
- **The relevance of discussing the impact of chlorine concentrations in water on the volatility of mercury for a site with freshwater bodies; and**
- **The basis for assuming 15 percent of the mercury present in surface water exists as methylmercury.**

Response 8: The source of mercury at the Former Choccolocco Corridor Ranges is unknown. One possible source of mercury could be switches used to control electronic targets at the Former Choccolocco Corridor Ranges. However, there is no documentation to verify the presence of electronic-controlled targets at the Former Choccolocco Corridir ranges. The analyses conducted on soil samples from the Former Choccolocco Corridor Ranges were for total recoverable mercury; therefore, the form of mercury present is not known. However, in soils, mercury can exist in the mercuric (Hg+2) and mercurous (Hg+1) states as a number of complex ions with varying water solubilities (ATSDR, 1999). The dominant process controlling the distribution of mercury compounds in the environment appears to be the sorption of nonvolatile forms to soil and sediment particulates (Bryan and Langston, 1992). Adsorption of mercury in soil is decreased with increasing pH and/or chloride ion concentrations (Schuster, 1991). Mercury is sorbed to soil with high iron and aluminum content up to a maximum loading capacity of 15,000 ppm (Ahmad and Qureshi, 1989). Inorganic mercury sorbed to particulate material is not readily desorbed. Thus, freshwater and marine sediments are important repositories for inorganic forms of mercury, and leaching is a relatively insignificant transport process in soils. Metallic mercury may move through the top 3-4 cm of dry soil at atmospheric pressure; however, it is

unlikely that further penetration would occur (Eicholz, et al., 1988). This discussion will be added to Section 4.4.4.

If mercury were ever present at the Former Choccolocco Corridor Ranges in the metallic form, it would have volatilized relatively rapidly, been transported via vapor transport, and deposited in association with dry deposition of particulate mater. Due to the historic nature of the soil contamination found at the Former Choccolocco Corridor ranges, volatilization of mercury is presently not an important transport process at these ranges.

The last sentence on page 4-8 referring to 15% of mercury in surface water being present as methyl mercury will be deleted.

Comment 9: **Section 4.4.6. Please discuss the probability for co-solvent effects at this site and potential for impact, if any, on contaminant mobility. Please also discuss the potential impact of carrier solvents from pesticide application.**

Response 9: The probability for co-solvent effects at the Former Choccolocco Corridor Ranges is low. Volatile and semi-volatile compounds were detected infrequently and at low concentrations in soil, indicating that these compounds are not prevalent in soil and do not significantly impact contaminant mobility. Furthermore, if co-solvent effects were prevalent at the Former Choccolocco Corridor Ranges, one would expect to find significant contamination in surface water and/or groundwater due to increased contaminant mobility. No COPECs were identified in either surface water or groundwater at the Former Choccolocco Corridor Ranges. Co-solvent effects are not a significant transport mechanism at the Former Choccolocco Corridor Ranges and do not warrant further discussion.

Comment 10: **Sections 4.4.7, 4.4.8, 4.4.9 and 4.4.10. Please discuss the significance of volatilization from water surfaces for benzene hexachloride (BHC) compounds, chlordane compounds, dieldrin and endrin at this site.**

Response 10: None of these compounds were identified as COPECs in surface water at the Former Choccolocco Corridor Ranges. Of the aforementioned compounds, only gamma-chlordane was identified as a COPEC in sediment, and it is closely associated with particulates in the sediment. Therefore, volatilization of these compounds is not a significant transport process at the Former Choccolocco Corridor Ranges.

Comment 11: **Section 5.0. Please provide a U.S. Geological Survey (USGS) topographic locus map illustrating the location and limits of the BERA study area along with a mapping overlay of major habitats, sediment depositional reaches of stream channels and any jurisdictional boundaries on the site sampling plan.**

Response 11: A USGS topographic map illustrating the location of the Former Choccolocco Corridor Ranges will be included in the Former Choccolocco Corridor Ranges Problem Formulation and Study Design Report. A map illustrating the major habitat types and approximate locations of wetland boundaries (if any exist) will be included in the Former Choccolocco Corridor Ranges BERA report as these data are to be collected as part of the BERA effort.

Comment 12: **<u>Section 7.3.1</u>. The proposed study to quantify COPEC concentrations within earthworms does not represent a measurement endpoint for evaluating the assessment endpoint of survival/growth of the terrestrial invertebrate community. If the measure levels of COPECs within the earthworm tissues are compared to invertebrate tissue thresholds associated with adverse effects or no adverse effects then it would represent a measurement endpoint. Otherwise, it simply represents a method to estimate exposure to the assessment endpoint of survival, growth and reproduction of invertivorous small mammals and birds. Please revise the text to reflect this distinction.**

Response 12: Referring to the measurement of COPEC concentrations in earthworm tissues as a "measurement endpoint" or simply a "measurement" is a matter of semantics. As presented in USEPA (1997), measurement endpoints can include measures of exposure as well as measures of effect. USEPA (1997) defines a measurement endpoint as "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint." Bioaccumulation of soil COPECs in terrestrial invertebrate tissues can certainly be considered a "measurable ecological characteristic" and the bioaccumulation potential of a COPEC can be "related to the valued characteristic chosen as the assessment endpoint" via its use as a site-specific input parameter in a food web model. There is a direct correlation between the COPEC concentration in terrestrial invertebrate tissues (measurement endpoint) and the potential impact on survival, growth, and reproduction of terrestrial invertivorous and omnivorous small mammals and birds (assessment endpoints) via food web interactions. Therefore, the "quantification of COPEC concentrations in tissues of earthworms exposed to soils from the CC Ranges and tissues of earthworms exposed to soils from a non-impacted reference location" is a valid measurement endpoint and no revisions are necessary.

Comment 13: **<u>Section 7.3.1</u>. The proposed study to quantify COPEC concentrations within above-ground portions of plants does not represent a measurement endpoint for evaluating the assessment endpoint of survival/growth of the terrestrial plant community. If the measure levels of COPECs within the plant tissues are compared to plant tissue thresholds associated with adverse effects or no adverse effects then it**

would represent a measurement endpoint. Otherwise, it simply represents a method to estimate exposure to the assessment endpoint of survival, growth and reproduction of omnivorous small mammals and birds. Please revise the text to reflect this distinction.

Response 13: Please see Response to Specific Comment 12.

Comment 14: **<u>Section 7.3.2</u>. The proposed study to quantify COPEC concentrations within chironomids does not represent a measurement endpoint for evaluating the assessment endpoint of survival/growth of the benthic terrestrial invertebrate community. If the measure levels of COPECs within the chironomid tissues are compared to invertebrate tissue thresholds associated with adverse effects or no adverse effects then it would represent a measurement endpoint. Otherwise, it simply represents a method to estimate exposure to the assessment endpoint of survival, growth and reproduction of insectivorous small mammals and birds. Please revise the text to reflect this distinction.**

Response 14: Please see Response to Specific Comment 12.

Comment 15: **<u>Section 8.5</u>. Decision rules for both terrestrial and aquatic invertebrates should include adverse effects other than mortality. Measurement endpoints for both terrestrial and aquatic invertebrates also include growth of the respective organisms during the test exposure period. Both mortality and growth statistical differences from the reference site indicate potential for unacceptable hazards to these receptors. Please address.**

Response 15: As presented in Section 8.3, Decision Inputs, measurement endpoints for terrestrial invertebrates include both mortality and growth. As presented in numerous sections including 7.2.2, 7.3.2, 9.2.2, and Appendix B of the Problem Formulation and Study Design report, chironomid growth has also been proposed as a measurement endpoint; however, it was inadvertently omitted from Section 8.3. The following Decision Input will be included in Section 8.3 of the Problem Formulation and Study Design report:

- Chironomus tentans growth based on total tissue weight measured at the termination of the toxicity test

Comment 16: **<u>Section 9.2.2</u>. The report states that six sediment samples are proposed to be collected from the perennial stream and ephemeral drainage features present within the site. However, Figure 9-1 indicates that the six proposed sample locations are within three ephemeral drainage features and none will be located within the perennial stream which represents the most significant aquatic habitat present at the site. Several sediment**

samples previously collected from this perennial stream contain concentrations of lead that exceed the ecological benchmark indicating possible adverse effects to the benthic community are present. It is recommended that a minimum of three samples be collected from this perennial stream and that both sediment toxicity testing and rapid bioassessment protocol (RBP II) analyses be conducted at each location. Sampling locations can include previous sample locations HR-95Q-SD01, either HR-131Q-SD-01 or HR-CCRI-SD14 and either HR-CCRI-SD15 or HR-145Q-SD02. Please address.

Response 16: Please see response to General Comment 1.

Comment 17: **Section 9.2.2. A stream reference community is proposed to be located at a similar off-site stream in order to compare benthic assemblages with samples collected from the on-site streams. Please provide an assessment for the potential for locating the reference community within an upgradient portion of the perennial stream (e.g., northwest of Parcel 145Q-X). Similarly, the proposed reference sediment sampling location (HR-CCRI-SD30) should also be evaluated for its potential to serve as a reference community for the on-site ephemeral streams. Please address.**

Response 17: An additional reference location will be located up-gradient of Parcel 145Q-X on the perennial stream at sample location HR-CCRI-SW/SD12 which will be used to collect sediment for chemical analysis, toxicity/bioaccumulation studies, and benthic community analyses. This additional reference location will serve as a reference for the perennial stream.

Comment 18: **Section 9.2.2. The Study Design states that riffle/run and pool macroinvertebrate samples will be collected from each sampling station and eight metrics will be calculated from the benthic macroinvertebrate data collected at each sampling station. Please clarify whether the riffle/run and pool samples will be combined for this analysis or evaluated separately for the eight metrics.**

Response 18: Organisms in the riffle/run benthic macroinvertebrate sample and the coarse particulate organic matter (CPOM) macroinvertebrate sample will be identified and enumerated independently and then combined to form a single macroinvertebrate sample for each sampling location prior to metric analysis per USEPA (1999) guidance. The second paragraph on page 9-8 will be revised accordingly.

Comment 19: **Table 9-1 and Appendix A, Section A2.0. Two discrepancies were noted between the lists of surface soil sample locations/identifications in these sections. Please clarify the following discrepancies:**

- **HR-CCRI-GP26 was listed in Table 9-1, but listed as HR-CCRI-GP28 in Appendix A, and**
- **Range 40 was listed in Table 9-1 as the range for sample HR-CCRI-MW03, but listed as Range 41 in Appendix A.**

Response 19: A number of revisions have been made to Table 9-1 and Appendix A as a result of EPA comments. Chapter 9 and Appendices A and B will be reviewed for consistency and revised as necessary.

EPA

Response to EPA Comments on the
Draft Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Former Choccolocco Ranges at Fort McClellan, Calhoun County, Alabama

Comments from Doyle Brittain T. Brittain, EPA Remedial Project Manager, dated November 1, 2007.

General Comments

Comment 1: **Samples HR-CCRI-SW/SD27, HR-CCRI-SW/SD29, and HR-CCRI-SW/SD30 were taken south of Parcel 146Q-X in Choccolocco Corridor (CC) to determine the extent of sediment contamination (Table 2-1 of Draft RI Report). Low levels of gamma-chlordane and methoxychlor were detected in some of these sediment samples outside the boundaries of the CC characterization area. Results for pesticides in sediments suggest that pesticides might be ubiquitous at the CC Ranges. If pesticides are associated with the ranges, the footprint of pesticide contamination might extend beyond the footprint of metals contamination. Spraying might have occurred over a wide area or drift may have occurred. Data might suggest low-level pesticide contamination is ubiquitous at Fort McClellan; however, data are insufficient at the CC Ranges to make this determination. Pesticides are also found in soils. Endrin is one of the more frequently detected pesticides in soils. Endrin has a maximum concentration of 3.4E-02 mg/kg in soil sample HR-CCRI-MW01, located in the central portion of Parcel 146Q. Concentrations of endrin in soils above the screening value occur at the boundaries of the characterization area. Please discuss how uncertainties regarding pesticides will be addressed.**

Response 1: All of the soil and sediment samples collected as part of the BERA for the CC Ranges will be analyzed for pesticides. Additionally, earthworm tissues, terrestrial plant tissues, and chironomid tissues will be analyzed for pesticides after exposure to soils and sediments, respectively. The ranges at FTMC have shown remarkable consistency with regard to types and patterns of contamination relative to Army activities at these ranges. If low levels of pesticides are found to be ubiquitous in these environmental samples, but inconsistent with the pattern of metals that have been found to be consistent with Army activities at these ranges, then the conclusion will be that the detected pesticides are the result of routine maintenance activities at these ranges and not the result of any spill or release. The multiple lines of evidence collected in the BERA will take into account the possibility of low-level anthropogenic background levels of pesticides. Based on the frequency and magnitude of the detected concentrations of pesticides, and the results of the other lines of evidence gathered in the BERA, a conclusion will be drawn as to whether pesticides at the CC Ranges have the potential to pose ecological hazards.

Comment 2: **The document identified several pesticides in soils and sediments as COPECs. The proposed soil and sediment samples will be analyzed for pesticides; however, the study is designed primarily to investigate potential risks associated with gradients of lead contamination. If the data to support the BERA will only tangentially investigate pesticide risk, insufficient data might be collected to screen out the risks from pesticides.**

Response 2: The ranges at FTMC have shown remarkable consistency with regard to types and patterns of contamination relative to Army activities that these ranges. If low levels of pesticides are found to be ubiquitous in these environmental samples, but inconsistent with the pattern of metals that have been found to be consistent with Army activities at these ranges, then the conclusion will be that the detected pesticides are the result of routine maintenance activities at these ranges and not the result of any spill or release. The multiple lines of evidence collected in the BERA will take into account the possibility of low-level anthropogenic background levels of pesticides.

Specific Comments

Comment 1: **Section 2.1. COPECs in Surface Soil, Pages 2-3 through 2-6. Mercury in surface soil had a maximum detected concentration of 1.18 mg/kg and an average concentration of 0.078 mg/kg. This comment examines the potential risks at these levels to plants, earthworms, and wildlife. Mercury at 1.18 mg/kg will be unlikely to have unacceptable risk to plants. The benchmark for mercury toxicity to plants was 0.3 mg/kg from Efroymson et al. (1997a) and was based on a NOEC of 34 mg/kg with an uncertainty factor of 100. Mercury can be toxic to plants but at much higher concentrations than in soil at the CC Ranges. Mercury can be toxic to earthworms, but a common screening value for mercury toxicity to earthworms is 0.1 mg/kg (Efroymson et al. 1997b). It was based on reduced cocoon production at 0.5 mg/kg with a safety factor of 5 applied because 40 % and 65 % reduction was observed. Mean concentrations are below these literature benchmarks. Mercury at CC Ranges is isolated to a few stations, where it slightly exceeds the literature benchmarks. For a credible risk to wildlife, the mean mercury concentration in soil would need to be higher than the Region 4 ESV of 0.1 mg/kg. Mercury was probably used in electronic switches for targets. It is possible that mercury spills took place. Mercury concentrations should be measured in soils tested for toxicity to plants and earthworms to prevent a potential hot spot from confounding test results.**

Response 1: As presented in the CC Ranges Problem Formulation and Study Design, mercury will be retained as a COPEC in soil. No revisions to the subject report are necessary.

Comment 2: **<u>Section 2.1. COPECs in Surface Soil, Pages 2-3 through 2-6</u>. Dieldrin in soil can screen out as a COPEC. The average concentration of dieldrin in soil (5.8E-04 mg/kg) is less than the recently revised Eco-SSL for dieldrin for protection of mammals of 4.9E-03 mg/kg.**

Response 2: Agreed. Dieldrin will be removed from the list of soil COPECs. The CC Ranges Problem Formulation and Study Design report will be revised accordingly.

Comment 3: **<u>Section 2.1. COPECs in Surface Soil, Pages 2-3 through 2-6</u>. Endrin was detected in surface soil at a maximum concentration of 3.4E-02 mg/kg. The average detected concentration of endrin in soil was 1.78E-03 mg/kg. It had a maximum HQ of 34 using the Region 4 screening value of 1E-03 mg/kg. An alternative screening value to consider is the Region 5 RCRA value of 1E-02 mg/kg. Because an alternative screening value is used, multiple lines of evidence would be required to screen endrin out, such as frequency of detection. Endrin was detected at 14 of 53 locations or in 26 percent of the samples. One way to handle it would be to carry endrin through to the BERA for site-specific foodweb modeling, because it is a bioaccumulative chemical.**

Response 3: As presented in the CC Ranges Problem Formulation and Study Design report, endrin will be retained as a COPEC in soil at the CC Ranges. No revisions to the subject report are necessary.

Comment 4: **<u>Section 2.1. COPECs in Surface Soil, Pages 2-3 through 2-6</u>. The chemicals beta-BHC and gamma-BHC (lindane) can possibly screen out as COPECs in soil based on Region 5 soil screening values, which are 0.00398 mg/kg for beta-BHC and 0.005 mg/kg for gamma-BHC. Given the low frequency of detection, low exceedance of the alternative screening values, and mean concentrations below alternative screening values, it might be possible to screen out the BHC as COPECs in soil.**

Response 4: Agreed. Beta-BHC and gamma-BHC will be eliminated from the list of soil COPECs. The CC Ranges Problem Formulation and Study Design report will be revised accordingly.

Comment 5: **<u>Section 2.2, COPECs in Surface Water, Pages 2-6 through 2-7</u>. Lead in surface water was screened out based on the Slippage Test and Wilcoxon Rank Sum Test. Lead in surface water had a maximum HQ of 8.26 in sample HR-CCRI-SW/SD18 in the western portion of Choccolocco Corridor, Parcel 144Q-X. The concentration of lead in the highest surface water sample was 1.09E-02 mg/L relative to the ESV of 1.32E-03 mg/L. A sample just downstream at HR-CCRI-SW/SD19 had a concentration of 2.06E-03J mg/L of lead and was also above the screening value. The third sample with detected lead in surface water above the**

screening value (HR-145Q-SW/SD02) had 2.35E-03J mg/L. The third sample was located in the western portion of Parcel 145Q-X. The pattern of exceedance suggests that creeks might exceed State water quality standards for lead in localized areas of high lead contamination in soils and sediments. The maximum detected lead in surface water was detected at the same location as the maximum detected lead in sediment (394 mg/kg), station HR-CCRI-SW/SD18. This comment is asking that lead be retained as a COPEC in surface water, not to require site-specific toxicity testing, but to eventually be expressed as a remedial action objective. The exact wording for the remedial action objective would appear in the FS; but it could express a goal to prevent migration of lead in surface water and a goal to meet applicable State surface water standards.

Response 5: Agreed. Lead will be retained as a COPEC in surface water at the CC Ranges. No site-specific toxicity testing will be conducted for the BERA, but a remedial action objective for lead in surface water will be recommended for inclusion in the feasibility study for the CC Ranges.

Comment 6: **<u>Section 2.3, COPECs in Sediment, Pages 2-7 through 2-8</u>. Heptachlor has an alternative screening value for sediments from Jones et al. (1997) of 0.068 mg/kg. The maximum concentration of heptachlor detected in sediment (0.015 mg/kg) is below the alternative benchmark. The benchmark was calculated from the secondary chronic value by the equilibrium partitioning method. Heptachlor can be screened out as a COPEC.**

Response 6: Agreed. Heptachlor will be eliminated from the list of COPECs in sediment. The CC Ranges Problem Formulation and Study Design report will be revised accordingly.

Comment 7: **<u>Section 2.3, COPECs in Sediment, Pages 2-7 through 2-8</u>. The pesticide gamma-chlordane was detected above the screening values in CC Parcel 146-Q. The BERA for Baby Bains Gap conducted site-specific tests on gamma-chlordane and did not detect significant bioaccumulation into Chironomids. Low levels of gamma-chlordane are difficult to screen out with the Region 4 screening value of 1.7E-03 mg/kg. Other screening values to try are the McDonald et al. (2000) consensus value of 3.24E-03 mg/kg or the CCME (2002) probable effects level (PEL) of 8.87E-03 mg/kg. The highest concentration of gamma-chlordane at CC Ranges was 1.5E-02 mg/kg. This level was only slightly above the highest concentration tested at BBGR Ranges (9.0E-03 mg/kg) where no toxicity to Chironomids or bioaccumulation was observed. It can be difficult to detect slight biological effects that might be expected at low levels. Low levels of pesticides appear to be ubiquitous in the CC Ranges sediments. The pesticides, especially gamma-chlordane, might confound toxicity tests**

on sediments and benthic community analyses. Pesticides such as gamma-chlordane will probably not be final COPECs in the conclusions to the BERA; however, they should be measured in sediments planned for toxicity testing even if not all are COPECs. (This is the case in this study design.) Concentrations of pesticide COPECs should be factored into the interpretation of toxicity test results and benthic community assessment results. This comment is recommending gamma-chlordane remain a COPEC in sediment. Unless the highest concentration of gamma-chlordane in sediments is targeted for toxicity testing, it may not be possible to draw any conclusions from the tests. Careful choice of sample locations is recommended.

Response 7: As presented in the CC Ranges Problem Formulation and Study Design, gamma-chlordane has been identified as a COPEC in sediment at the CC Ranges. As is the case with all of the identified COPECs, the pesticide COPECs will be factored into the interpretation of the toxicity tests and benthic community analyses conducted at the CC Ranges. The location of the highest detected concentration of gamma-chlordane has been identified as a sediment sampling location for the CC Ranges BERA for additional chemical analysis, chironomid toxicity testing, chironomid bioaccumulation, and benthic community analysis.

Comment 8: **<u>Table 7-3, Riparian Foodweb Model Input Parameters</u>. The model input parameters such as the food ingestion rate in Table 7-3 are citing the same sources used in the BBGR Ranges and BGR and IMR Ranges. The input parameters for ingestion rate are different here than those used in the BBGR Ranges or in the BGR and IMR Ranges BERA reports. This comment is recommending addition of a section in the text for each foodweb model receptor explaining the sources of information and specific information used to come up with the input parameters. This comment is also recommending the parameters to be reviewed for consistency.**

Response 8: The ingestion rates presented in Table 7-3 are the ingestion rates cited in the references provided in the table, adjusted for the moisture content of the food item (benthic invertebrates). A section will be included in the revised CC Ranges Problem Formulation and Study Design report that cites the sources for the input parameters used in the terrestrial and riparian food web models and the specific methods used to derive certain input parameters if they are not directly cited from a literature source.

Comment 9: **<u>Section 9.1.1. Soil Collection for Chemical Analysis, Page 9-1</u>. The gradient for lead on Figure 9-1 is good for lead, but should consider the copper, zinc and endrin also to ensure the results have a broad range of applicability in the BERA risk characterization. The gradient for lead in**

soil is okay; but, unless the maximum detected concentration is sampled, a possible finding of no toxicity could not apply to the entire site.

Response 9: The full range of detected concentrations of lead in soil is proposed for sampling in the BERA. Additionally, several different (new) soil sample locations are proposed in order to provide a broader range of concentrations for copper and zinc in soil. The concentration range for lead in the proposed BERA soil samples is 4,530 mg/kg to 21 mg/kg. The concentration range for copper is 383 mg/kg to 9.67 mg/kg and for zinc is 366 mg/kg to 19 mg/kg in the proposed BERA soil samples. These changes will be incorporated into the revised CC Ranges Problem Formulation and Study Design report.

Comment 10: **<u>Section 9.2, Aquatic Study Design, Page 9-5</u>. No surface water sampling is planned to support the BERA. Five of the 23 surface water samples taken in the RI had blank contamination and could not be used to support the evaluation of potential migration of lead from the site. This comment is suggesting the Army examine the uncertainty in existing data and to possibly collect surface water metals at stations where toxicity tests and benthic community analyses are planned if valid surface water data is not available at these locations.**

Response 10: As with any environmental data set, there is a certain level of uncertainty associated with the surface water data for the CC Ranges. However, the uncertainty is minimal and does not adversely affect the conclusions of the SLERA or the testing procedures proposed in the CC Ranges Problem Formulation and Study Design report. As the commentor noted, 5 of the 23 surface water samples collected at the CC Ranges had blank contamination and were not used in the SLERA. All of the excluded data points exhibited lead concentrations less than the BTV for lead, indicating the "detected" concentrations of lead in the excluded data points are naturally occurring and not the result of Army activities. However, if these blank-contaminated data were included in the data set for the SLERA, the maximum HQ_{screen} value would not change and the mean HQ_{screen} value would change from 1.05 to 1.13. This change is considered insignificant for the purposes of a SLERA. Both the original and the revised mean HQ_{screen} values indicate the concentrations of lead in surface water are very close to the conservative screening level. All of the detected concentrations of lead in surface water (both the 18 data points used in the SLERA and the 5 blank-contaminated data points) were less than the BTV for lead in surface water except for a single sample. These data do not indicate a level of concern that would warrant additional costly sampling and analysis.

Comment 11: **<u>Section 9.2.1, Sediment Collection for Chemical Analysis, Page 9-5</u>. Station HR-CCRI-SW/SD30 is proposed as a reference station. This station was not considered a reference station in the RI Report (Table 2-1 in draft RI). The pesticide gamma-chlordane was detected at HR-CCRI-**

SW/SD30 at a level above the sediment screening value. It is unclear whether this is the best choice for a reference station if gamma-chlordane is a COPEC in sediment. This location could be a control station for the influence of low-levels of gamma-chlordane with another station chosen as the reference station.

Response 11: Agreed. A reference sample location more clearly outside the influence of the CC Ranges will be identified and utilized in the CC Ranges BERA.

References

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Acronyms

BBGR	Baby Bains Gap Road
BGR	Bains Gap Road
BERA	Baseline Ecological Risk Assessment
CC	Choccolocco Corridor
COPEC	Constituent of Potential Ecological Concern

Eco-SSL	Ecological Soil Screening Level
ESV	Ecological Screening Value
HQ	Hazard Quotient
IMR	Iron Mountain Road
NOEC	No Observable Effect Concentration
PEL	Probable Effect Level