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Draft

Baseline Ecological Risk Assessment for Baby Bains Gap Road Ranges

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Table of Contents

Page

List of Tables v
List of Figures ix
1.0 Introduction 1-1
2.0 Screening-Level Ecological Risk Assessment Summary 2-1
2.1 Environmental Setting at the Baby Bains Gap Road Ranges 2-1
2.1.1 Range 18 Habitat 2-3
2.1.2 Range 20 Habitat 2-4
2.1.3 Range 23 Habitat 2-5
2.1.4 Range 25 Habitat 2-7
2.1.5 Range 25-East Habitat 2-8
2.1.6 Range 26 Habitat 2-9
2.1.7 Ranges South of Range 25 Habitat 2-9
2.1.8 Ingram Creek Tributaries Habitat 2-10
2.1.9 South Branch of Cane Creek Tributaries Habitat 2-11
2.2 Potential Receptors at the Baby Bains Gap Road Ranges 2-13
2.2.1 Herbivorous Feeding Guild 2-14
2.2.2 Invertivorous Feeding Guild 2-15
2.2.3 Omnivorous Feeding Guild 2-15
2.2.4 Carnivorous Feeding Guild 2-16
2.2.5 Piscivorous Feeding Guild 2-16
2.2.6 Threatened and Endangered Species 2-17
2.3 Constituents of Potential Ecological Concern 2-18
2.3.1 COPECs in Surface Soil 2-22
2.3.2 COPECs in Surface Water 2-22
2.3.3 COPECs in Sediment 2-22
2.3.4 COPECs in Groundwater 2-22
2.3.5 Summary of COPECs 2-23
3.0 Assessment and Measurement Endpoints 3-1
3.1 Assessment Endpoints 3-1
3.1.1 Terrestrial Assessment Endpoints 3-1
3.1.2 Aquatic Assessment Endpoints 3-3
3.2 Risk Hypotheses 3-4
3.2.1 Terrestrial Risk Hypothesis 3-4

Table of Contents *(Continued)*

Page

3.2.2 Aquatic Risk Hypotheses 3-6
3.3 Measurement Endpoints 3-7
3.3.1 Terrestrial Measurement Endpoints 3-8
3.3.2 Aquatic Measurement Endpoints 3-10
4.0 Data Quality Objectives 4-1
5.0 Site Investigation Results 5-1
5.1 Results of Abiotic Sampling and Analysis 5-1
5.1.1 Surface Soil Sampling and Results 5-1
5.1.2 Sediment Sampling and Results 5-2
5.2 Results of Terrestrial Plant Chemical Analysis 5-3
5.3 Results of Perennial Rye Grass Toxicity Tests 5-4
5.4 Results of Earthworm Toxicity and Bioaccumulation Tests 5-6
5.5 Results of Midge Survival, Growth, and Bioaccumulation Tests 5-7
5.6 Results of Benthic Community Analysis 5-10
5.6.1 Benthic Invertebrate Habitat Assessment 5-10
5.6.2 Benthic Invertebrate Community Assessment 5-11
6.0 Risk Characterization 6-1
6.1 Terrestrial Ecosystems 6-2
6.1.1 Terrestrial Plant Germination and Growth 6-2
6.1.2 Terrestrial Plant Bioaccumulation 6-6
6.1.3 Earthworm Survival and Growth 6-7
6.1.4 Earthworm Bioaccumulation 6-8
6.1.5 Terrestrial Good Web Exposures 6-9
6.2 Aquatic Ecosystems 6-15
6.2.1 Chironomid Survival and Growth 6-16
6.2.2 Chironomid Bioaccumulation 6-17
6.2.3 Benthic Invertebrate Community Assemblages 6-18
6.2.3.1 Taxa Richness 6-19
6.2.3.2 Modified Family Biotic Index 6-20
6.2.3.3 Ratio of Scraper and Filtering Collector Functional Feeding Group 6-20
6.2.3.4 Ratio of EPT and Chironomidae Abundances 6-21

Table of Contents *(Continued)*

Page

6.2.3.5 Percent Contribution of Dominant Taxon 6-21
6.2.3.6 EPT Index 6-22
6.2.3.7 Community Loss Index 6-22
6.2.3.8 Ration of Shredder Functional Feeding Group and Total Number of Individuals Collected 6-22
6.2.3.9 Total Metric Scores and Bioassessment 6-23
6.2.4 Riparian Food Web Exposures 6-24
7.0 Uncertainty Analysis 7-1
7.1 Uncertainties in Earthworm Survival and Growth Tests 7-1
7.2 Uncertainties in Earthworm Bioaccumulation Test 7-2
7.3 Uncertainties in the Perennial Rye Grass Germination and Growth Tests 7-2
7.4 Uncertainties in the Terrestrial Plant Bioaccumulation Test 7-4
7.5 Uncertainties in Terrestrial Food Web Model 7-4
7.6 Uncertainties in Chironomid Survival and Growth Tests 7-6
7.7 Uncertainties in Chironomid Bioaccumulation Test 7-7
7.8 Uncertainties in Benthic Invertebrate Community Analysis 7-7
7.9 Uncertainties in Riparian Food Web Model 7-8
8.0 Summary and Conclusions 8-1
8.1 BERA Summary 8-1
8.1.1 Terrestrial Habitat at the BBGR Ranges 8-1
8.1.2 South Branch of Cane Creek Tributaries Habitat 8-1
8.1.3 Ingram Creek Tributaries Habitat 8-2
8.1.4 Threatened and Endangered Species 8-3
8.1.5 COPECs Identified in the SLERA 8-3
8.1.6 Assessment and Measurement Endpoints 8-4
8.1.7 Data Quality Objectives 8-4
8.1.8 Results of the Terrestrial Plant Bioaccumulation Tests 8-5
8.1.9 Results of the Perennial Rye Grass Toxicity Tests 8-6
8.1.10 Results of Earthworm Toxicity Tests 8-7
8.1.11 Results of Earthworm Bioaccumulation Test 8-8
8.1.12 Results of Terrestrial Food Web Model 8-8
8.1.13 Results of the Chironomid Toxicity Tests 8-10
8.1.14 Results of Chironomid Bioaccumulation Test 8-11

Table of Contents *(Continued)*

Page

8.1.15 Results of Benthic Invertebrate Community Analysis 8-12
8.1.16 Results of Riparian Food Web Model 8-13
8.2 BERA Conclusions 8-14
8.2.1 Terrestrial Ecosystems 8-14
8.2.2 Riparian Ecosystems 8-15
9.0 References 9-1

Attachment 1 - List of Abbreviations and Acronyms
Tables
Figures
Appendix A – Toxicity Test Report
Appendix B – Habitat Assessment Field Data Sheets

List of Tables

Table	*Title*	*Follows Tab*
2-1	Constituents of Potential Ecological Concern in Surface Soil, Range 18 (Parcel 74Q)	
2-2	Constituents of Potential Ecological Concern in Surface Soil, Range 20 (Parcel 76Q)	
2-3	Constituents of Potential Ecological Concern in Surface Soil, Range 23 (Parcel 79Q)	
2-4	Constituents of Potential Ecological Concern in Surface Soil, Range 25 (Parcels 83Q and 118Q-X)	
2-5	Constituents of Potential Ecological Concern in Surface Soil, Range 26 (Parcel 84Q)	
2-6	Constituents of Potential Ecological Concern in Surface Soil, Ranges South of Range 25 (Parcels 224Q and 226Q)	
2-7	Constituents of Potential Ecological Concern in Surface Water, South Branch Tributaries at the Baby Bains Gap Road Ranges	
2-8	Constituents of Potential Ecological Concern in Surface Water, Ingram Creek Tributaries at the Baby Bains Gap Road Ranges	
2-9	Constituents of Potential Ecological Concern in Sediment, South Branch Tributaries at the Baby Bains Gap Road Ranges	
2-10	Constituents of Potential Ecological Concern in Sediment, Ingram Creek Tributaries at the Baby Bains Gap Road Ranges	
2-11	Constituents of Potential Ecological Concern in Groundwater	
3-1	Assessment Endpoints, Risk Hypotheses, and Measurement Endpoints	
5-1	Surface Soil Sample Analytical Results	
5-2	Sediment Sample Analytical Results	
5-3	Plant Tissue Analytical Results	
5-4	Earthworm Tissue Analytical Results	
5-5	Chironomid Tissue Analytical Results	
5-6	COPEC Concentrations in Soil and Terrestrial Plant Tissues	
5-7	Surface Soil COPEC Concentrations and Perennial Rye Grass Toxicity	
5-8	Surface Soil COPEC Concentrations and Earthworm Toxicity	
5-9	COPEC Concentrations in Soil and Earthworm Tissues	
5-10	Sediment COPEC Concentrations and Chironomid Toxicity	
5-11	COPEC Concentrations in Sediment and Chironomid Tissues	

List of Tables *(Continued)*

Table	Title	Follows Tab
5-12	Benthic Invertebrate Habitat Assessment	
5-13	Benthic Invertebrate Identification Summary	
5-14	Rapid Bioassessment Metric Results	
5-15	Benthic Invertebrate Community Bioassessments Results and Sediment COPEC Concentrations	
6-1	Site-Specific Soil Toxicity Values	
6-2	Terrestrial Foodweb Indicator Species Life History Parameters	
6-3	Biological Fate and Transport Properties for the Constituents of Potential Concern in the Terrestrial Foodweb	
6-4	Exposure Point Concentrations for the Constituents of Potential Concern in the Terrestrial Foodweb	
6-5	Maximum Constituent Exposures for White-Footed Mouse	
6-6	Maximum Constituent Exposures for American Robin	
6-7	Maximum Constituent Exposures for Short-Tailed Shrew	
6-8	Maximum Constituent Exposures for American Woodcock	
6-9	Mean Constituent Exposures for White-Footed Mouse	
6-10	Mean Constituent Exposures for American Robin	
6-11	Mean Constituent Exposures for Short-Tailed Shrew	
6-12	Mean Constituent Exposures for American Woodcock	
6-13	Avian Toxicity Reference Values	
6-14	Mammalian Toxicity Reference Values	
6-15	Comparison of Maximum Constituent Exposures with Toxicity Reference Values for White-Footed Mouse	
6-16	Comparison of Maximum Constituent Exposures with Toxicity Reference Values for American Robin	
6-17	Comparison of Maximum Constituents Exposures with Toxicity Reference Values for Short-Tailed Shrew	
6-18	Comparison of Maximum Constituent Exposures with Toxicity Reference Values for American Woodcock	
6-19	Comparison of Mean Constituent Exposures with Toxicity Reference Values for White-Footed Mouse	

List of Tables *(Continued)*

Table	Title	Follows Tab
6-20	Comparison of Mean Constituent Exposures with Toxicity Reference Values for American Robin	
6-21	Comparison of Mean Constituent Exposures with Toxicity Reference Values for Short-Tailed Shrew	
6-22	Comparison of Mean Constituent Exposures with Toxicity Reference Values for American Woodcock	
6-23	Summary of Terrestrial Food Web Hazard Quotients Assuming Maximum Exposure Point Concentrations	
6-24	Summary of Terrestrial Food Web Hazard Quotients Assuming Mean Exposure Point Concentrations	
6-25	Risk-Based Remedial Goals for Terrestrial Food Web Exposures	
6-26	Site-Specific Sediment Toxicity Values	
6-27	Riparian Foodweb Indicator Species Life History Parameters	
6-28	Biological Fate and Transport Properties for the Constituents of Potential Ecological Concern in the Riparian Foodweb	
6-29	Exposure Point Concentrations for the Constituents of Potential Ecological Concern in the Riparian Foodweb	
6-30	Maximum Constituent Exposures for Little Brown Bat	
6-31	Maximum Constituent Exposures for Marsh Wren	
6-32	Mean Constituent Exposures for Little Brown Bat	
6-33	Mean Constituent Exposures for Marsh Wren	
6-34	Avian Toxicity Reference Values for Riparian Food Web	
6-35	Mammalian Toxicity Reference Values for Riparian Food Web	
6-36	Comparison of Maximum Constituent Exposures with Toxicity Reference Values for Little Brown Bat	
6-37	Comparison of Maximum Constituent Exposures with Toxicity Reference Values for Marsh Wren	
6-38	Comparison of Mean Constituent Exposures with Toxicity Reference Values for Little Brown Bat	
6-39	Comparison of Mean Constituent Exposures with Toxicity Reference Values for Marsh Wren	
6-40	Summary of Riparian Food Web Hazard Quotients Assuming Maximum Exposure Point Concentrations	

List of Tables *(Continued)*

Table	*Title*	*Follows Tab*
6-41	Summary of Riparian Food Web Hazard Quotients Assuming Mean Exposure Point Concentrations	
6-42	Risk-Based Remedial Goals for Riparian Food Web Exposures	
8-1	Summary of Potential Ecological Risk-Based Remedial Goals	

List of Figures

Figure	*Title*	*Follows Tab*
2-1	Conceptual Site Model	
5-1	Rye Grass Shoot Length and COPEC Concentrations in Soil	
5-2	Rye Grass Root Length and COPEC Concentrations in Soil	
5-3	Rye Grass Weight and COPEC Concentrations in Soil	
5-4	Chironomid Survival and COPEC Concentrations in Sediment	
5-5	Chironomid Growth and COPEC Concentrations in Sediment	
6-1	Antimony in Soil and Rye Grass Mean Shoot Length	
6-2	Beryllium in Soil and Rye Grass Mean Shoot Length	
6-3	Copper in Soil and Rye Grass Mean Shoot Length	
6-4	Lead in Soil and Rye Grass Mean Shoot Length	
6-5	Zinc in Soil and Rye Grass Mean Shoot Length	
6-6	Antimony in Soil and Rye Grass Mean Root Length	
6-7	Beryllium in Soil and Rye Grass Mean Root Length	
6-8	Copper in Soil and Rye Grass Mean Root Length	
6-9	Lead in Soil and Rye Grass Mean Root Length	
6-10	Zinc in Soil and Rye Grass Mean Root Length	
6-11	Antimony in Soil and Rye Grass Plant Weight	
6-12	Beryllium in Soil and Rye Grass Plant Weight	
6-13	Copper in Soil and Rye Grass Plant Weight	
6-14	Lead in Soil and Rye Grass Plant Weight	
6-15	Zinc in Soil and Rye Grass Plant Weight	
6-16	Soil-to-Plant BAF (Copper)	
6-17	Soil-to-Plant BAF (Lead)	
6-18	Soil-to-Plant BAF (Zinc)	
6-19	Soil-to-Earthworm BAF (Antimony)	
6-20	Soil-to-Earthworm BAF (Copper)	
6-21	Soil-to-Earthworm BAF (Lead)	
6-22	Soil-to-Earthworm BAF (Zinc)	
6-23	Copper in Sediment and Chironomid Survival	
6-24	Lead in Sediment and Chironomid Survival	

List of Figures *(Continued)*

Figure	*Title*	*Follows Tab*
6-25	Copper in Sediment and Chironomid Growth	
6-26	Lead in Sediment and Chironomid Growth	
6-27	Sediment-to-Midge BAF (Copper)	
6-28	Sediment-to-Midge BAF (Lead)	
6-29	Copper Concentrations and RBP II Scores	
6-30	Lead Concentrations and RBP II Scores	
6-31	Gamma-Chlordane Concentrations and RBP II Scores	
6-32	Habitat Assessment Scores and RBP II Scores	

1.0 Introduction

This Baseline Ecological Risk Assessment (BERA) utilizes the results of the Screening-Level Ecological Risk Assessments (SLERA) conducted for the Baby Bains Gap Road Ranges at Fort McClellan (FTMC) (Shaw, 2004a) and builds upon the information presented in the Problem Formulation and Study Design report for these small arms ranges (Shaw, 2006). The methods used in this BERA are consistent with the guidelines set forth in USEPA's *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA, 1997) and *Amended Guidance on Ecological Risk Assessment at Military Bases: Process Considerations, Timing of Activities, and Inclusion of Stakeholders* (USEPA, 2000a).

2.0 Screening-Level Ecological Risk Assessment Summary

A Screening-Level Ecological Risk Assessment (SLERA) was completed for the Baby Bains Gap Road (BBGR) ranges. The SLERA for these small arms ranges indicated that constituents of potential ecological concern were present in one or more environmental media at these sites. The following sections summarize the findings of the SLERA completed at the BBGR ranges.

2.1 Environmental Setting at the Baby Bains Gap Road Ranges

In general, the ecological habitats present at the BBGR Ranges can be categorized into upland and aquatic habitat types. The upland habitats can be described as either "cleared" areas or forested areas. The cleared areas, or old field habitat, are those areas that were formerly maintained as lawns or mowed fields. These areas represent the locations where range activities were most prevalent. Since maintenance activities have ceased in these areas, pioneer species are colonizing these areas. Typically, the species most likely to colonize these 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 abandoned areas are the grasses and herbaceous species. These formerly maintained grassy areas are classified as being in an early old field successional state. Over time, these grass and herbaceous species will be followed by shrubs and small trees. The early old field successional areas at the BBGR Ranges are dominated by various grasses and herbs including dock (*Rumex spp*.), clover (*Trifolium spp*.), vetch (*Astragalus spp*.), milkweed (*Ascelepias spp*.), bed straw (*Galium spp*.), ox-eye daisy (*Chrysanthemum leucanthemum*), and Johnson grass (*Sorghum halepense*). Other old field herbaceous species occurring at the BBGR Ranges are black raspberry (*Rubus occidentalis*), poison ivy (*Toxicodendron radicans*), smooth sumac (*Rubus glabra*), green brier (*Smilax rotundiflora*), Japanese honeysuckle (*Lonicera japonica*), fox grape (*Vitus labrusca*), and multiflora rose (*Rosa multiflora*). Loblolly pine (*Pinus taeda),* longleaf pine *(Pinus palustris),* and shortleaf pine (*Pinus echinata*) saplings have also begun to encroach on the formerly maintained grassy areas of the BBGR Ranges.

The forested areas outside of the cleared areas are best characterized as mixed deciduous/coniferous forest. The canopy species typically found in the forested areas surrounding the BBGR Ranges include yellow poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), white oak (*Quercus alba*), and northern red oak *(Quercus rubra*). The dominant understory species of this area 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 mountain laurel (*Kalmia latifolia*), southern low blueberry *(Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this habitat type.

The aquatic habitats in the vicinity of the BBGR Ranges can be divided into two separate drainages: Ingram Creek tributaries and South Branch of Cane Creek tributaries. The Ingram Creek tributaries drain Ranges 20, 23, 25, 25-East, 26, and Ranges South of Range 25. The South Branch of Cane Creek tributaries drain Range 18. All of these tributaries are small ephemeral streams that normally only exhibit flowing water during periods of heavy precipitation, with the exception of the small tributary that flows from east-to-west across Range 23. This small tributary can be considered perennial as it contains water throughout the majority of the year under normal precipitation conditions. The tributaries in both of these drainages are generally narrow (less than 3 feet wide) and shallow when water is present (less than 6 inches deep). The substrate is mostly sand and gravel with significant amounts of leaf litter within the forested areas.

Terrestrial species that may inhabit the area of the BBGR Ranges include opossum, short-tailed 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 (U.S. Army Corps of Engineers, 1998). Common species expected to occur in the vicinity of the BBGR 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 BBGR Ranges may include northern bobwhite (*Colinus virginianus*), mourning dove (*Zenaida macroura*), and eastern wild turkey (*Meleagris gallopavo*). Woodland hawks (e.g., sharp-shinned hawk) were observed in this area during the ecological investigation (October 2006) and are expected to use this area for a hunting ground. A variety of other raptors (e.g., red-tailed hawk, barred owl, and great horned owl) could also use portions of this area for a hunting ground, particularly the fringe areas where the forested areas abut roads and cleared areas. Although the small size and ephemeral nature of the tributaries at the BBGR Ranges limits the potential for the occurrence of piscivorous birds, they may be present during limited

periods of the year when precipitation is heavy and the tributaries exhibit flowing water. The piscivorous bird species that may be present in the vicinity of the BBGR Ranges include great blue heron (*Ardea herodias*), green-backed heron (*Butorides striatus*), and belted kingfisher (*Ceryle alcyon*).

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 BBGR Ranges are copperhead (*Agkistrodon contortix*), king snake (*Lampropeltis getulus*), black racer (*Coluber constrictor*), fence lizard (*Sceloporour undulatus*), and six-lined racerunner (*Cnemidophorous sexlineatus*).

Descriptions of the habitats at each of the BBGR Ranges are presented in the following sections.

2.1.1 Range 18 Habitat

Range 18 is comprised of two main habitat types: cleared and forested areas. The cleared area comprises the vast majority of Range 18. The entire area of Range 18, including the extensive safety fan is approximately 3,304 acres. The study area of Range 18 is approximately 70 acres in size. The overall elevation of Range 18 ranges from approximately 800 to 875 feet above mean sea level (msl), with the highest elevation at the top of the hill which forms the impact zone in the southern portion of the study area. Ground surface is flat in the firing line area with a slight downward slope to the northeast.

The study area of Range 18 is almost entirely comprised of formerly maintained lawn, mowed fields, and unvegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine (*Pinus taeda*) saplings and shortleaf pine (*Pinus echinata*), have also begun to encroach into these previously maintained areas. Significant portions of Range 18 remain unvegetated, with large areas of bare soil.

The forested areas surrounding Range 18 are best characterized as mixed deciduous/ coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally to the north and east from the impact zone area via several ephemeral creeks and ditches which transect the range immediately behind the 75-meter target line. This water eventually connects to South Branch of Cane Creek northeast of Range 18, flowing further north towards Cane Creek. These ephemeral ditches are generally narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these ditches is mostly sand and gravel with a significant amount of leaf litter present where the ditches run through forested areas. The vegetation within these ditches (when present) is characteristic of the upland habitats surrounding the ditches and is not characteristic of wetland or aquatic habitats, indicating that these ditches only carry water during significant rainfall events. During the majority of the year, these ditches are characteristic of the upland habitats present at the site.

The ephemeral drainage features that flow through Range 18 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). These drainage features have been identified as potential gray bat foraging habitat because they provide habitat for aquatic insects, which are fed upon by the gray bat. However, these drainage features contain water only during periods of significant precipitation and are dry during the majority of the year. Thus, aquatic insects would not be expected to occur in these ditches for the majority of the year. Additionally, gray bats prefer continuous cover while traveling to and from their foraging habitats and while foraging. Due to historical maintenance activities at Range 18, the forest canopy has been eliminated and only grasses and weeds remain over the majority of Range 18. Thus, the currently existing vegetation at Range 18 does not provide the cover favored by gray bats. Studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3D/International [3DI], 1997).

2.1.2 Range 20 Habitat

Range 20 is comprised of two main habitat types: cleared and forested areas. The study area (33.4 acres) extends over the entire range including the M-60 impact zone area (1.7 acres). The range safety fan extends to the southeast, covering an area of approximately 1,505 acres.

A small, ephemeral tributary of Ingram Creek cuts across Range 20 draining surface water runoff west towards Ingram Creek. The overall elevation of Range 20 ranges from approximately 925 to 1,200 feet above msl. The lowest elevation is found near the Ingram Creek tributary and the highest elevation is at the top of the hill in the impact zone. Ground surface is relatively flat in the M-60 firing point and explosive pit area with a 75 foot decent to the tributary in the

infiltration training area and a steep 1,025 foot rise towards the impact zone hillside southeast of the M-60 firing points.

The study area of Range 20 is almost entirely forested, with the exception of the firing point and the impact zone. The former firing point area has been re-graded and is entirely enclosed with chain-link fence, and is currently used by the Corps of Engineers for storage. The impact zone exhibits several areas of bare, un-vegetated ground and the remaining portion is forested with immature (less than 5 years old) mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this area as signs of fire indicate that these vegetative layers have been burned in recent years. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally to the west-southwest from the firing point and impact zone areas via a small, ephemeral stream that bisects the firing point and impact areas. This tributary eventually connects to Ingram Creek west of Range 20. This ephemeral ditch is generally narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in this ditch is mostly sand and gravel with a significant amount of leaf litter.

Ingram Creek and its tributaries in the vicinity of Range 20 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the ephemeral nature of the small tributary at Range 20, aquatic insects would not be expected to occur in this area for significant periods during any given year. Additionally, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.3 Range 23 Habitat

The study area where firing points, lanes, and target mounds are located is defined as approximately 58.6 acres. The range safety fan extends to the southeast and covers an area of approximately 4,566 acres.

Snap Lane forms the western boundary of the Range 23 study area. Three tributaries of Ingram Creek are present at the range, carrying runoff water from the hillside east of the range, through the firing lane area, under Snap Lane, and northwards towards Ingram Creek. Two of these tributaries are ephemeral, while the tributary that flows from the central portion of Range 23

generally contains water throughout the year. The elevation of Range 23 increases from approximately 850 to 900 feet above msl at the firing lane area in the western portion of the range, steeply rising beyond the 300-meter target mounds to an elevation of 1,050 feet at the top of the unnamed hill east of the main study area.

Range 23 exhibits both cleared and forested areas. The 16 firing lanes are characteristic of cleared areas. The firing lanes are separated from each other by narrow strips of forest. The formerly cleared firing lanes are characteristic of old field, early successional habitat. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these areas. There are significant areas within the firing lanes that are un-vegetated, with large areas of bare soil. The narrow strips of forest between the firing lanes can be characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. These forested strips also exhibit dense understory and herbaceous layers. The dominant understory species of this area 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 mountain laurel (*Kalmia latifolia*), southern low blueberry *(Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally from east-to-west across Range 23 via three small creeks/ditches. Two of these creeks (the northern-most and southern-most) are ephemeral in nature, while the third creek contains water throughout the majority of the year. All three of these creeks eventually connect to Ingram Creek west of Range 23. All three of these creeks are narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these creeks is mostly sand and gravel. Vegetation in the two ephemeral tributaries is characteristic of the upland vegetation that occurs in the surrounding area, indicating that these tributaries do not contain water for significant portions of the year.

Ingram Creek and its tributaries in the vicinity of Range 23 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the

ephemeral nature of two of the tributaries at Range 23, aquatic insects would not be expected to occur in these tributaries for significant periods during any given year. Aquatic insects may be present in the central tributary since it is more perennial in nature. However, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.4 Range 25 Habitat

Range 25 is comprised of two main habitat types: cleared and forested areas. The cleared area comprises the vast majority of Range 25. The total area of Range 25, including the extensive safety fan, comprises 1,713 acres. The main study area of Range 25 is approximately 103 acres and is topographically relatively flat. A hillside in the northern portion of the study area forms one of the major impact zones for this range. The study area of Range 25 is comprised almost entirely of formerly maintained lawns, mowed fields, and un-vegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. Significant portions of Range 25 remain un-vegetated, with large areas of bare soil.

The forested area in the northern portion of Range 25 is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Three small, ephemeral tributaries to Ingram Creek and Cane Creek flow from east-to-west across the study area of Range 25. The southern-most tributary and the tributary that crosses through the center of the study area drain to Ingram Creek. The northern-most tributary crosses the study area in the northwest corner of the study area and leads to Cane Creek to the northwest. All three of these tributaries are narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these creeks is mostly sand and gravel except where they traverse wooded areas, in which case they have significant leaf litter. Vegetation in these ephemeral tributaries is characteristic of the upland vegetation that occurs in the surrounding area, indicating that these tributaries do not contain water for significant portions of the year.

The small ephemeral tributaries that cross Range 25 are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC

and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.5 Range 25-East Habitat

It is not certain whether Range 25 East was ever constructed. The study area for Range 25 East encompasses the area where possible firing lines and impact zones are located and is approximately 44 acres. There is no safety fan associated with Range 25 East. The other ranges which overlap this study area include Range 20 and Range 26. If it was constructed in the same layout and orientation as Range 25, Range 25 East would have featured firing lines to the south (in the vicinity of the Range 26 firing line) and a target line berm to the northeast. The hillside that would act as a possible impact zone for this range rises approximately 150 feet in elevation in the northern portion of the study area downrange of the original first firing line. The elevation of Range 25 East ranges from approximately 825 feet (in the southern portion of the study area) to 1,025 feet above msl at the top of the hillside in the possible impact zone. Ground surface rises gradually in the firing line area with a steep rise in the possible impact zone.

The majority of Range 25 East is forested with mixed deciduous/coniferous forest. The southeastern corner of the study area (which overlaps the firing line area of Range 26) is characteristic of old field, early successional habitat. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate the forest habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. The old field, early successional portion of the study area is dominated by various grasses and herbaceous species that have grown uncontrolled since maintenance activities have ceased. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Range 25 East is bisected by Bains Gap Road, which runs through the impact zone area in the northern portion of the range. A small ephemeral tributary of Ingram Creek drains runoff water from the hillside in the northern portion of the study area (where the possible impact zone would be). Two other ephemeral tributaries (discussed in Range 20 and Range 26 sections) drain runoff from the east to the west across the study area of Range 25 East. These tributaries meet near the parking area at Range 26, cross under Bains Gap Road, and continue flowing west towards Ingram Creek. The small ephemeral tributaries that cross Range 25 East are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.6 Range 26 Habitat

The study area for Range 26 (where firing lanes and impact zones are located) is approximately 34.2 acres. The range safety fan extends to the southeast covering an area of approximately 1,267 acres. The overall elevation of the Range 26 study area gradually increases from approximately 875 to 900 feet above msl in the western portion of the study area to approximately 900 to 985 feet above msl at the hillside impact zone in the eastern portion of the study area.

The study area of Range 26 is comprised almost entirely of formerly maintained lawns, mowed fields, and un-vegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. Significant portions of Range 26 remain un-vegetated, with large areas of bare soil.

On the periphery of the study area the habitat is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Two ephemeral tributaries of Ingram Creek collect runoff from the hillsides to the south and northeast of the study area and traverse the study area from east-to-west. Both of these drainage features exhibit vegetation that is characteristic of the surrounding upland vegetation, indicating that these ditches only transmit water during periods of significant precipitation. Additionally, these small ephemeral tributaries that cross Range 26 are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.7 Ranges South of Range 25 Habitat

The Ranges South of Range 25 encompass an area of approximately 6.1 acres and includes portions of several historical ranges, including Parcels 224Q, 226Q, and 227Q. The study area is bounded on the east by Snap Lane and on the north by Bains Gap Road. There are no safety fans associated with the Ranges South of Range 25. The study area is relatively flat with a total elevation change of approximately 25 feet, sloping from east-to-west.

The entire study area is forested with a somewhat immature mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate the forest habitat. The study area exhibits dense understory and herbaceous layers. The dominant understory species of this area 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 mountain laurel (*Kalmia latifolia*), southern low blueberry *(Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

A small ephemeral tributary of Ingram Creek runs through the center of the study area from east-to-west. This drainage feature exhibits vegetation that is characteristic of the surrounding upland vegetation, indicating that this ditch only transmits water during periods of significant precipitation. The small ephemeral tributary that crosses the study area is not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.8 Ingram Creek Tributaries Habitat

The tributaries to Ingram Creek that are present at the BBGR Ranges drain Ranges 20, 23, 25, 25-East, 26, and Ranges South of Range 25. All of these tributaries are small ephemeral streams with the exception of one of the tributaries that flows across Range 23, which is perennial. Surface water runoff from the hills located east and north of the BBGR Ranges is the major source of water for these tributaries. As such, they only convey water during periods of significant precipitation. Flow in these small ephemeral tributaries is highly variable, depending on precipitation in the surrounding watershed. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution from groundwater varies according to the amount of precipitation, with an increase when precipitation raises the potentiometric surface. Downstream (west) of the BBGR Ranges, these small ephemeral tributaries continue to flow in a westerly direction towards Ingram Creek. Ingram Creek (located west of the BBGR Ranges) flows in a northwesterly direction until its confluence with Cane Creek, northwest of the BBGR Ranges.

The headwater areas for these tributaries are relatively undeveloped portions of the Main Post and are almost entirely mixed deciduous/coniferous forest. The physical characteristics of these tributaries at the BBGR Ranges are relatively consistent. All of these tributaries are narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in these tributaries where they pass through forested areas. The vegetation in these tributaries is generally characteristic of the surrounding upland vegetation, indicating that these tributaries only transmit water during periods of significant precipitation.

Although the Ingram Creek tributaries at the BBGR Ranges are ephemeral in nature, they have the potential to support a variety of amphibious species and some small, drought-tolerant fish species. Bullfrog *(Rana catesbeiana*) and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in these small ephemeral tributaries in the vicinity of the BBGR Ranges. Fish species that may be found in the Ingram Creek tributaries in the vicinity of the BBGR Ranges include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*), and various darters (*Etheostoma spp*.). The shallow nature of these ephemeral tributaries limits their ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the Ingram Creek tributaries in the vicinity of the BBGR Ranges because they only hold water during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

The small ephemeral tributaries to Ingram Creek in the vicinity of the BBGR Ranges are either not classified with regard to potential gray bat foraging habitat or are classified as low quality gray bat foraging habitat. The tributaries that are classified as low quality gray bat foraging habitat are classified as such because they provide habitat for aquatic insects, which may be fed upon by the gray bat. However, these drainage features contain water only during periods of significant precipitation and are dry during the majority of the year. Thus, aquatic insects would not be expected to occur in these ditches for the majority of the year. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.1.9 South Branch of Cane Creek Tributaries Habitat

A single tributary to South Branch of Cane Creek occurs within the BBGR Ranges and only at Range 18. This small, ephemeral tributary runs along the western boundary of Range 18 and then transects the range from west-to-east in the northern portion of the range. This small

tributary then turns north along the eastern boundary of Range 18 and joins South Branch of Cane Creek immediately north of the Range 18 study area. Surface water runoff from the hills west and south of Range 18 is the major source of water for this tributary. As such, it only conveys water during periods of significant precipitation. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution from groundwater varies according to the amount of precipitation, with an increase when precipitation raises the potentiometric surface.

The headwater areas for this tributary (immediately south and west of Range 18) are relatively undeveloped portions of the Main Post and are almost entirely mixed deciduous/coniferous forest. The physical characteristics of this tributary are consistent with other small ephemeral drainage features across FTMC. This tributary is narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in this tributary where it passes through forested areas. The vegetation within this tributary is generally characteristic of the surrounding upland vegetation, indicating that this tributary only transmits water during periods of significant precipitation.

Although the South Branch of Cane Creek tributary at the BBGR Ranges is ephemeral in nature, it has the potential to support a variety of amphibious species and some small, drought-tolerant fish species. Bullfrog *(Rana catesbeiana*) and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in this small ephemeral tributary. Fish species that may be found in the South Branch tributary at Range 18 include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*), and various darters (*Etheostoma spp*.). The shallow nature of this ephemeral tributary limits its ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the South Branch tributary in the vicinity of Range 18 because it only has water present during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

The South Branch of Cane Creek tributary within the Range 18 study area is classified as low quality gray bat foraging habitat. South Branch of Cane Creek, located just east and north of the Range 18 study area, is classified as moderate quality gray bat foraging habitat. The tributary is classified as low quality gray bat foraging habitat because it has the potential to provide habitat for aquatic insects, which may be fed upon by the gray bat. However, this drainage feature contains water only during periods of significant precipitation and is dry during the majority of the year. Thus, aquatic insects would not be expected to occur in this ditch for the majority of

the year. Additionally, gray bats prefer continuous cover while traveling to and from their foraging habitats and while foraging. Due to historical maintenance activities at Range 18, the forest canopy has been eliminated and only grasses and weeds remain over the majority of Range 18. Thus, the currently existing vegetation at Range 18 does not provide the cover favored by gray bats. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

2.2 Potential Receptors at the Baby Bains Gap Road Ranges

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 COPECs 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. Exposure via inhalation of fugitive dust is limited to constituents present in surface soils at areas that are devoid of vegetation. The inherent moisture content of the soil and the frequency of soil disturbance also play important roles in the amount of fugitive dust generated at a particular site.

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 previous 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 inorganic compounds that are not prone to volatilization, volatilization from sediments is not an important fate mechanism. 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 Ingram Creek tributaries and South Branch tributaries as feeding areas. 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.

While constituents in soils may leach into groundwater, environmental receptors will not come into direct contact with constituents in groundwater since there is no direct exposure route. The only potential exposure pathways for ecological receptors to groundwater would be via surface water exposure routes. Since no COPECs were identified in surface water or groundwater, ecological exposures to these two environmental media are insignificant at the BBGR ranges.

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. Sediment-borne COPECs may bioaccumulate into aquatic organisms, aquatic plants, or animals which frequent surface waters and then be passed through the food chain to impact organisms at higher trophic levels.

Potential ecological receptors at the BBGR 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 BBGR ranges: herbivores, invertivores, omnivores, carnivores, and piscivores. All of these feeding guilds have the potential to be directly exposed to various combinations of surface soil and sediment via various activities (e.g., feeding, grooming, bathing, etc.). These feeding guilds may also be exposed to site-related chemicals via food web transfers.

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 BBGR 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 BBGR ranges.

Potentially complete exposure pathways are depicted in the SCM for the BBGR ranges as presented in Figure 2-1 and are described in the following sections for the various feeding guilds expected to occur at these ranges.

2.2.1 Herbivorous Feeding Guild

The major route of exposure for herbivores is through ingestion of plants that may have accumulated constituents from the soil, surface water, or sediment. The vegetation at the formerly maintained areas at the BBGR Ranges is mainly grasses and sedges, which are remnants of the maintained grass that was present when the BBGR Ranges were operational. 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.

Typical herbivorous species that could be expected to occur at the BBGR 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*), whitetail deer (*Odocoileus virginianus*), and wild turkey (*Meleagris gallopavo*).

Aquatic herbivores, such as muskrat (*Ondatra zibethica*) and mallard (*Anas platyrhynchos*), could be exposed to site-related constituents in sediment.

2.2.2 Invertivorous Feeding Guild

Invertivores specialize in eating insects and other invertebrates. As such, they may be exposed to site-related chemicals that 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 terrestrial invertivores since much of their food (i.e., earthworms and other invertebrates) lives on or below the soil surface.

Typical terrestrial invertivorous species that could be expected to occur at the BBGR 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 (those species that live in water) could include the wood duck (*Aix sponsa*) and blacknose dace (*Rhinichthys atratulus*).

2.2.3 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 be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities. Omnivores may also be exposed to COPECs in sediment through incidental ingestion of sediment in the ephemeral drainage features at the BBGR Ranges.

Typical omnivorous species expected to occur at the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include red fox (*Vulpes vulpes*), white-footed mouse (*Peromyscus leucopus*), and American robin (*Turdus migratorius*). Aquatic omnivores, such as raccoon (*Procyon lotor*) and creek chub (*Semotilus atromaculatus*) could be exposed to COPECs in sediment in the drainage features at the BBGR Ranges.

2.2.4 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 constituents in their tissues. Carnivores are quite often top predators in a local food web and are often subject to exposure to constituents 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, surface water, sediment, plant, and animal material as food and accumulate COPECs in their tissues. Subsequent ingestion of these prey animals by carnivorous animals would expose them to COPECs. Most inorganic compounds and VOCs are not accumulated in animal tissues to any great extent (Shugart et al., 1990 and U.S. Army Environmental Hygiene Agency, 1994). Therefore, food web exposures to these chemicals are expected to be minimal. PAHs have the potential to accumulate in lower trophic level organisms but not in higher trophic level organisms because they have mechanisms for metabolizing and excreting this class of compounds. 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 carnivores. Carnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities.

Typical carnivorous species expected to occur at the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include red-tailed hawk (*Buteo jamaicensis*), black vulture (*Coragyps atratus*), and bobcat (*Lynx rufus*).

Because the drainage features at the BBGR Ranges are relatively narrow and shallow and ephemeral in nature, they do not have the capability to support large aquatic carnivores on a full-time basis. Carnivorous fish such as largemouth bass (*Micropterus salmoides*) and spotted gar (*Lepisosteus oculatus*) would not be expected to occur in the drainage features at the BBGR Ranges due to the habitat restrictions. Carnivorous mammals such as the mink (*Mustela vison*), may feed along several of the drainage features that have flowing water throughout the majority of the year.

2.2.5 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 small drainage features at the BBGR Ranges during periods of significant precipitation. They may also be

exposed to COPECs in sediment in these drainage features through incidental ingestion of sediment during feeding. Although piscivorous species might visit the drainage features at the BBGR Ranges during certain periods of the year when the creek flow is significant, they would not be expected to live near the BBGR Ranges because these drainage features are not large enough to support larger fish species.

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. VOCs and SVOCs 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 VOCs or SVOCs at the BBGR 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 BBGR Ranges.

Typical piscivorous species that could occur near the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include great blue heron (*Ardea herodias*) and belted kingfisher (*Ceryle alcyon*). Larger piscivorous fish species (e.g., smallmouth bass, spotted gar) and piscivorous mammals (e.g., mink) are not expected to occur in the vicinity of the BBGR Ranges due to the habitat limitations of the small drainage features in this area and their inability to support larger fish and other aquatic species.

2.2.6 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*).

The only Federally listed species that has the potential to occur in the vicinity of the BBGR Ranges is the gray bat (Garland, 1996). The drainage features at the BBGR Ranges have either been designated as providing "low quality" foraging habitat for the gray bat or have not been

classified for their potential to provide gray bat foraging habitat (Garland, 1996). The other Federally listed species occur at Pelham Range or Choccolocco Creek corridor.

The gray bat is almost entirely restricted to cave habitats and, with rare exceptions, roosts in caves year-round. Approximately 95 percent of the entire known population hibernates in only nine caves each winter, with more than half in a single cave. Gray bat summer foraging habitat is found primarily over open water of rivers and reservoirs. They apparently do not forage over sections of rivers or reservoirs that have lost their normal woody vegetation along the banks (USFWS, 1982). Gray bats usually follow wooded corridors from their summer caves to the open water areas used as foraging sites. Forested areas surrounding and between caves, as well as over feeding habitats, are clearly advantageous to gray bat survival as the cover provides increased protection from predators such as screech owls. In addition, surveys have demonstrated that reservoirs and rivers that have been cleared of their adjacent forest canopy are avoided as foraging areas by gray bats (USFWS, 1982).

The gray bat is entirely insectivorous and surveys have shown that gray bats feed almost exclusively on mayflies at certain times of the year (Mount, 1986). Therefore, gray bats could be exposed to site-related constituents that have accumulated in aquatic insects from the drainage features at the BBGR Ranges. Because gray bats are flying mammals and the BBGR Ranges do not provide roosting habitat, no other exposure pathways are potentially complete for the gray bat.

Most foraging occurs within 5 meters of the water's surface, usually near a shoreline or stream bank. Mist net surveys were conducted on and adjacent to FTMC in 1995. Gray bats were captured along both Choccolocco Creek (east of FTMC Main Post) and Cane Creek on Pelham Range (west of FTMC Main Post) during these mist net surveys (Garland, 1996). These preliminary data suggest that these major stream corridors at FTMC may provide at least a minimum foraging habitat for gray bats. Radiotelemetry surveys conducted by 3DI (1997) indicated gray bats do not roost at FTMC and foraging mostly occurs in the vicinity of the Cane Creek golf course. The small, ephemeral streams in the vicinity of the BBGR Ranges do not provide the habitat favored by gray bats and their presence in the vicinity of the BBGR Ranges is unlikely. However, gray bat surveys have not been conducted on the small streams in the vicinity of the BBGR Ranges.

2.3 Constituents of Potential Ecological Concern

The SLERA for the BBGR Ranges initially identified a number of COPECs in soil as well as in sediments in South Branch of Cane Creek and Ingram Creek tributaries. 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.

The ecological screening values (ESVs) that were utilized in the SLERA represent the most conservative values available from various literature sources and have been selected to be protective of the assessment endpoints described in the SLERA. These ESVs were developed specifically for FTMC in conjunction with USEPA Region 4 and are presented in *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000). The ESVs used in the SLERA are based on no-observed-adverse-effect-levels (NOAELs) when available. If a NOAEL-based ESV was not available for a certain COPEC, then the most health-protective value available from the scientific literature was used in the SLERA.

Constituents that were detected in environmental media at the BBGR Ranges were evaluated against the ESVs by calculating a screening-level hazard quotient (HQ_{screen}) for each constituent in each environmental medium. An 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 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 BBGR 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 BBGR Ranges are presented in Tables 2-1 through 2-11.

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.

The comparison of detected constituent concentrations with naturally occurring constituent concentrations was conducted via a three-tier process outlined in a technical memorandum dated March 14, 2005 (Shaw, 2005). The first tier of the background comparison process was a comparison of the maximum detected constituent concentration to the background threshold value (BTV). A study of the natural geochemical composition associated with FTMC (SAIC, 1998) determined the mean concentrations of 24 metals in surface soil, surface water, sediment, and groundwater samples collected from presumably un-impacted areas. Per agreement with EPA Region 4, the 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. Site sample metal concentrations less than or equal to the corresponding BTV represent the natural geochemical composition of media at FTMC, and not contamination associated with site activity. Site sample metal concentrations greater than the corresponding BTV require further background assessment.

If maximum constituent concentrations were greater than the BTV, then the second tier of the background comparison was employed. Tier two of the background comparison consists of statistical comparisons of the site data to background data using the Slippage Test and the Wilcoxon Rank Sum (WRS) Test. If the site data failed either the Slippage Test or the WRS Test, then the site data were subjected to a geochemical evaluation (Tier 3) to determine whether

concentrations of inorganic compounds are naturally occurring or are elevated due to contamination.

Thus, the first step in determining screening-level hazard quotients was a comparison of maximum detected constituent concentrations to appropriate ESVs. Constituents with HQ_{screen} values less than one were considered to pose insignificant ecological risk and were eliminated from further consideration. Constituents with HQ_{screen} values greater than one were eliminated from further consideration if they were macro-nutrients and their detected concentrations were less than 10-times naturally occurring levels. Those constituents that had HQ_{screen} values greater one and were not considered macro-nutrients were then compared to background using the three-tier background screening process. If constituent concentrations were determined to be less than their naturally occurring background concentrations, then a risk management decision could result in eliminating these constituents from further assessment.

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 macronutrient
- The maximum detected constituent concentration exceeded the BTV for inorganics.

If a constituent did not meet all of these conditions, then it was not considered a COPEC at the BBGR 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 was deemed appropriate but did not imply that a particular constituent posed a definite risk to ecological receptors.

In order to focus the BERA efforts on the constituents that are the most prevalent at the BBGR Ranges and have the greatest potential to pose ecological risk, additional lines of evidence were assessed in the SLERA 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 the BERA for the BBGR Ranges. These additional lines of evidence are discussed in the following sections.

2.3.1 COPECs in Surface Soil

Antimony, copper, and lead were frequently detected in surface soil at virtually all of the BBGR Ranges at concentrations that exceeded their respective ESVs and naturally occurring levels. The highest concentrations of these three constituents were found in locations that would be anticipated based on the site usage as firing ranges (i.e., soil berms that act as impact areas). Thus, it could be concluded that these constituents are site-related and could be considered COPECs in surface soil at virtually all of the BBGR Ranges. Beryllium and zinc were also detected in several surface soil samples at elevated concentrations and were considered COPECs in surface soil at the BBGR Ranges.

2.3.2 COPECs in Surface Water

No constituents were detected at elevated concentrations in surface water samples from either South Branch of Cane Creek or Ingram Creek tributaries. Therefore, no COPECs were identified in surface water at the BBGR Ranges.

2.3.3 COPECs in Sediment

No constituents were detected at elevated concentrations in sediment samples from the South Branch of Cane Creek tributaries; therefore, no COPECs were identified in South Branch of Cane Creek tributary sediments. Copper and lead were detected in one sample from the Ingram Creek tributaries at anomalously high concentrations relative to naturally occurring levels. These results indicate that these samples may contain a component of contamination. Copper and lead are also known components of ammunition. For these reasons, copper and lead were identified as COPECs in sediment within the Ingram Creek tributaries.

Gamma-chlordane was detected in two out of 9 sediment samples at elevated concentrations. The calculated HQ_{screen} value for gamma-chlordane was 52.4. Because gamma-chlordane has the potential to bioaccumulate, it was identified as a COPEC in sediment at the BBGR ranges.

2.3.4 COPECs in Groundwater

Several pesticides were detected infrequently in groundwater. It is important to note that none of the pesticides detected in groundwater were detected in surface water samples from the BBGR Ranges. Ecological receptors do not have a direct exposure pathway to groundwater. Ecological receptors can only be exposed to constituents in groundwater if groundwater is expressed at the ground surface as seeps or is discharged to lakes or streams via springs. Exposure of ecological receptors to groundwater could then occur via surface water pathways. Contaminants that may have entered groundwater in the past are likely to have been mostly, if not entirely, transported to surface water bodies by now, and if ongoing groundwater contamination of surface water bodies

were a concern, surface water samples would indicate the presence of groundwater contaminants. Because the pesticides in groundwater were detected infrequently and were not detected in surface water samples, it is expected that potential ecological exposure to these compounds is insignificant. Therefore, these compounds were not considered COPECs in groundwater at the BBGR Ranges.

2.3.5 Summary of COPECs

In order to focus on the constituents that are most prevalent at the BBGR Ranges and 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 were identified at the BBGR Ranges are summarized below:

- ***Surface Soil:*** antimony, beryllium, copper, lead, and zinc
- ***Surface Water:*** none
- ***Sediment:*** copper, lead, and gamma-chlordane
- ***Groundwater:*** none.

3.0 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, the risk hypotheses, and the corresponding measurement endpoints that have been identified for the BBGR ranges,.

3.1 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.

Assessment endpoints for this BERA were selected based on the ecosystems, communities, and species present at the BBGR ranges. Selection of the assessment endpoints was dependent upon the following factors:

- The COPECs, their characteristics, and their concentrations at the BBGR 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 assessment endpoints that were identified for the BBGR ranges are presented in the following sections.

3.1.1 Terrestrial Assessment Endpoints

Given the overall goal of protecting the integrity and quality of the terrestrial forest and old field ecosystems at the BBGR ranges, the terrestrial assessment endpoints focus on critical community niches within the mixed deciduous/coniferous forest and old field ecosystems present at the BBGR ranges. The ecological receptors with the potential for the greatest exposure to COPECs at the BBGR ranges were determined to be invertivorous and omnivorous small mammals and birds. Additionally, the terrestrial plant and terrestrial invertebrate communities have the potential for significant exposure to COPECs. These ecological communities formed 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 BBGR 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 community at the BBGR 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. The assessment endpoint that was identified with respect to the terrestrial invertebrate community is the following:

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

Invertivorous mammals and birds were identified as having significant potential for exposure to COPECs at the BBGR 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. The assessment endpoint that was 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 BBGR ranges.

Omnivorous mammals and birds were identified as having significant potential for exposure to COPECs at the BBGR 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. The assessment endpoint that was 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 BBGR ranges.

The assessment endpoints that have been identified for the BBGR ranges are summarized in Table 3-1.

3.1.2 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 South Branch of Cane Creek and its tributaries, and Ingram Creek and its tributaries at the BBGR ranges. The aquatic assessment endpoints focus on critical community niches within the sediment of the South Branch of Cane Creek and its tributaries, and Ingram Creek and its tributaries. Because no COPECs were identified in surface water, surface water is not a focus of the aquatic assessment at the BBGR Ranges. The ecological receptors with the potential for the greatest exposure to COPECs in the sediment of the South Branch of Cane Creek and its tributaries, and Ingram Creek and its tributaries at the BBGR Ranges are those populations and communities that live in direct contact with the sediment and those feeding guilds that utilize these creek systems 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 BBGR ranges. The assessment endpoint that was identified with respect to the aquatic benthic invertebrate community is the following:

- Survival, growth, and reproduction of aquatic benthic invertebrates in South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR ranges.

Riparian invertivorous mammals and birds were identified as having significant potential for exposure to COPECs at the BBGR 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 (i.e., raptors). The assessment endpoint that was 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 BBGR ranges.

The assessment endpoints that have been identified for the aquatic ecosystems at the BBGR ranges are summarized in Table 3-1.

3.2 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, test hypotheses were developed for the assessment endpoints identified in the previous section.

3.2.1 Terrestrial Risk Hypothesis

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

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

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

The test hypothesis regarding ecological benchmark 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 BBGR ranges. These test hypotheses were the following:

- Are concentrations of COPECs in surface soil at the BBGR ranges greater than ecological benchmarks for the survival or growth of terrestrial invertebrates?

- Is the survival and growth of terrestrial invertebrates exposed to surface soil from the BBGR Ranges significantly lower than that for terrestrial invertebrates exposed to soil from reference sites?

The test hypothesis regarding ecological benchmark 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 invertivorous mammals or birds via consumption of the tissues of prey species and from other media at the BBGR 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 omnivorous small mammals or birds via consumption of tissues of prey species and from other media at the BBGR 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 3-1 presents test/risk 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 $p = 0.05$.

3.2.2 Aquatic Risk Hypotheses

Three test hypotheses were identified as being appropriate to address the assessment endpoint of "survival, growth, and reproduction of aquatic benthic invertebrates in South Branch of Cane Creek and Ingram Creek at the BBGR Ranges." The first test hypothesis relative to benthic invertebrates in South Branch of Cane Creek and Ingram Creek and their tributaries was the following:

- Are the concentrations of COPECs in sediment samples from South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR Ranges greater than ecological benchmark 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 South Branch of Cane Creek and Ingram Creek and their tributaries was the following:

- Is the survival and growth of aquatic benthic invertebrates exposed to sediment from South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR 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 South Branch of Cane Creek and Ingram Creek, and off-site reference sediments in laboratory toxicity tests.

The third test hypothesis relative to benthic invertebrates in South Branch of Cane Creek and Ingram Creek and their tributaries was the following:

- Is the benthic community structure (using Rapid Bioassessment Protocol [RBP] II) significantly different in reaches of the South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR 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 South Branch of Cane Creek and Ingram Creek and their tributaries 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 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 BBGR 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 3-1 presents test/risk 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 $p = 0.05$.

3.3 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. The measurement endpoints that have been selected for the BBGR ranges are presented in the following sections.

3.3.1 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 test 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 BBGR ranges to ecological benchmarks 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 community at the BBGR 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 BBGR Ranges to plants grown in soils from a non-impacted 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 BBGR ranges to ecological benchmarks 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 BBGR Ranges" is the following:

- Statistical comparison of earthworm survival rates and body weights between earthworms exposed to on-site soils from the BBGR Ranges to earthworms exposed to soils from a non-impacted 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 BBGR Ranges" is the following:

- Comparison of calculated total daily doses of COPECs for invertivorous mammal (shorttail shrew) and invertivorous bird (American woodcock) using measured earthworm tissue concentrations of COPECs 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 BBGR Ranges" is the following:

- Comparison of calculated total daily doses of COPECs for omnivorous mammal (white-footed mouse) and omnivorous bird (American robin) using measured earthworm tissue concentrations of COPECs and measured terrestrial vegetation concentrations to TRVs.

In order to estimate the bioavailability of the COPECs in soil at the BBGR 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 BBGR Ranges" and "survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the BBGR Ranges". This measurement endpoint is the following:

- Quantification of COPEC concentrations in tissues of earthworms exposed to soils from the BBGR Ranges and tissues of earthworms exposed to soils from a non-impacted 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 BBGR Ranges, the following measurement endpoint has been identified:

- Quantification of COPEC concentrations in above-ground plant tissues from the BBGR Ranges and above-ground plant tissues from a non-impacted reference location.

These measurement endpoints will provide the necessary data to answer the risk hypotheses for the terrestrial ecosystems at the BBGR 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 (Tannenbaum, 2005). 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. Therefore, ecological and biological significance will be considered within the context of these measurement endpoints.

Table 3-1 presents the measurement endpoints corresponding to each assessment endpoint and risk hypothesis for the terrestrial ecosystems at the BBGR Ranges at FTMC.

3.3.2 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 South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR Ranges to ecological benchmarks 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 South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR Ranges" are the following:

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

- Comparison of the benthic community assemblage from South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR Ranges with the benthic community assemblages from 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 BBGR Ranges" is the following:

- Comparison of calculated total daily doses of COPECs for riparian invertivorous mammal (little brown bat) and invertivorous bird (marsh wren) using modeled and measured tissue concentrations of COPECs in emergent benthic invertebrates 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 South Branch of Cane Creek and Ingram Creek and their tributaries and tissues of chironomids exposed to sediment from a non-impacted reference stream.

These measurement endpoints will provide the necessary data to answer the risk hypotheses for the aquatic ecosystems at the BBGR 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. Therefore, ecological and biological significance will be considered within the context of these measurement endpoints.

Table 3-1 presents the measurement endpoints corresponding to each assessment endpoint and risk hypothesis.

4.0 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).

Based on the findings of the SLERA and Problem Formulation, the objectives of the BERA for the BBGR 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 BBGR 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 BBGR Ranges at which the ecological receptors are at risk.

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

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 BBGR Ranges are available for 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 South Branch of Cane Creek and Ingram Creek and their tributaries 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 risk.

The information required to support the decisions identified above includes the following:

- Surface soil concentrations (mg/kg) of the five surface soil COPECs;

- Earthworm mortality based on earthworm LC_{50} (lethal concentration killing 50 percent of the test population) data (mg/kg);

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

- Bio-uptake and accumulation potential 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 five soil COPECs in the terrestrial invertivorous shorttail shrew and American woodcock, as well as the omnivorous American robin and white-footed mouse (milligrams of COPEC per unit of body mass per day);

- Estimated levels of concern for 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 copper, lead, and gamma-chlordane;

- *Chironomus riparius* mortality based on exposure to various COPEC concentrations in sediment and derivation of sediment LC_{50} values;

- Bio-uptake and accumulation potential based on the ratio of sediment COPEC concentrations in South Branch of Cane Creek and Ingram Creek to *Chironomus sp*. tissue concentrations;

- Projected 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 BBGR 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 BBGR Ranges. The Uncertainty Analysis (Chapter 7) should be consulted for a review of the limitations associated with various assessment techniques and in establishing protective COPEC concentrations.

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 risk and, if necessary, in generating ecological-based cleanup goals. The decision rules proposed for the BBGR Ranges BERA include the following:

- If COPECs in soils from the BBGR 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 risks to terrestrial plants at the BBGR Ranges.

- If terrestrial plants exposed to soils from the BBGR 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 BBGR Ranges cause earthworm toxicity which is statistically greater than earthworm toxicity in soils from a reference site, then there is the potential for unacceptable risks to terrestrial invertebrate receptors at the BBGR Ranges.

- If earthworms exposed to soils from the BBGR 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 risk to terrestrial invertivorous mammals or birds at the BBGR 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 risk to terrestrial omnivorous mammals or birds at the BBGR Ranges.

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

- If COPECs within the sediments of South Branch of Cane Creek or Ingram Creek and their tributaries cause toxicity to the benthic invertebrate *Chironomus sp*., which is statistically greater than toxicity from reference sediments, then there is the potential for risk to emergent benthic invertebrates at the BBGR Ranges.

- If chironomids exposed to sediment from South Branch of Cane Creek or Ingram Creek and their tributaries 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 South Branch of Cane Creek or Ingram Creek and their tributaries at the BBGR Ranges is significantly different than the benthic community assemblage in a non-impacted reference stream, then there is the potential for risk to South Branch of Cane Creek or Ingram Creek benthic ecosystem.

- 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 risk to riparian invertivorous mammals or birds at the BBGR 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 BBGR 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 grading of soil may have removed the very shallow layer of topsoil from certain range areas. Without the layer of topsoil, it is very difficult for certain plant species to establish themselves and grow successfully. Therefore, physical disturbance of a site will also be considered when interpreting the results the established decision rules.

5.0 Site Investigation Results

Site investigation tasks included the following:

- Collection of surface soil samples;
- Collection of terrestrial vegetation samples;
- Collection of sediment samples; and
- Survey of the benthic invertebrate communities in the South Branch of Cane Creek and Ingram Creek and their tributaries.

These site investigation tasks were conducted in October 2006. Toxicity and bioaccumulation tests were subsequently run using the collected surface soil and sediment. The results of the site investigation and toxicity testing are presented in the following sections. Analysis and interpretation of the results are presented in Chapter 6.0, Risk Characterization.

5.1 Results of Abiotic Sampling and Analysis

Surface soil and sediment samples were collected from the Baby Bains Gap Road ranges, and the South Branch of Cane Creek and Ingram Creek and their tributaries in order to conduct toxicity and bioaccumulation studies. These samples were also subjected to chemical analysis to determine the concentrations of COPECs present and aid in risk characterization. The following sections describe the results of the abiotic sampling conducted in conjunction with this BERA.

5.1.1 Surface Soil Sampling and Results

A total of 10 surface soil samples were collected from the Baby Bains Gap Road ranges per the methodologies set forth in the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw, 2006). Nine surface soil samples were collected from suspected areas of contamination within the ranges and one reference surface soil sample was collected from an area that was assumed to be un-impacted from Army activities. These 10 surface soil samples were utilized for earthworm toxicity testing, perennial rye grass toxicity testing, and bioaccumulation analysis as described in later sections of this report. These surface soil samples were also subjected to chemical analysis. The results of the chemical analysis of these 10 surface soil samples are presented in Table 5-1.

Surface soil samples were collected from locations representative of the range of lead concentrations that were shown to elicit adverse effects in terrestrial invertebrates and food web

models in the *Baseline Ecological Risk Assessment for Iron Mountain Road and Bains Gap Road Ranges* (Shaw, 2004b). Sample locations were identified in the field using XRF technology. The results of the 10 surface soil samples collected for this BERA were combined with the results of the previous investigations at the BBGR ranges to form a database of 349 surface soil samples for the BBGR ranges.

The results of the surface soil sampling at the BBGR ranges for this BERA indicated that arsenic, antimony, barium, beryllium, copper, iron, lead, manganese, and zinc were detected in one or more surface soil samples at concentrations that exceeded both their respective ecological screening values (ESV) and background threshold values (BTV). Additionally, 4,4'-DDE, 4,4'-DDT, dieldrin, endrin, fluoranthene, and pyrene were detected at low levels in one or more surface soil samples. These data correspond well with the results of the SLERA for the BBGR ranges in that the SLERA identified antimony, beryllium, copper, lead, and zinc in surface soil as COPECs. All of the other constituents detected in surface soil samples were also detected in surface soil samples assessed in the SLERA and were excluded as COPECs in surface soil due to the consideration of additional lines of evidence, as described in the SLERA (Shaw, 2004a).

5.1.2 Sediment Sampling and Results

A total of fourteen sediment samples were collected from the South Branch of Cane Creek, Ingram Creek, and their tributaries at the Baby Bains Gap Road ranges per the methodologies set forth in the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw, 2006). Thirteen sediment samples were collected from the South Branch of Cane Creek, Ingram Creek, and their tributaries, and one sediment sample was collected from a reference site located on the tributary to Choccolocco Creek that flows in an easterly direction along Bains Gap Road, immediately east of the Bains Gap gate to the Main Post. These fourteen sediment samples were used for chironomid toxicity testing and bioaccumulation analysis described in later sections of this report. These sediment samples were also subjected to chemical analysis. The results of the chemical analysis of these 14 sediment samples are presented in Table 5-2.

The results of the 14 sediment samples collected for this BERA were combined with the results of the 12 sediment samples collected during previous investigations at the BBGR ranges. Arsenic, copper, chromium, lead, nickel, and zinc were detected in one or more sediment samples collected as part of this BERA at concentrations that exceeded both their respective ESVs and BTVs. Additionally, acetone, and gamma-chlordane were detected at low levels in one or more sediment samples. These data correspond well with the results of the SLERA for the BBGR ranges in that the SLERA identified copper, lead, and gamma-chlordane in sediment

as COPECs. All of the other constituents detected in sediment samples were also detected in sediment samples assessed in the SLERA and were excluded as COPECs in sediment due to the consideration of additional lines of evidence, as described in the SLERA (Shaw, 2004a).

5.2 Results of Terrestrial Plant Chemical Analysis

A total of 10 terrestrial plant samples were collected from the Baby Bains Gap Road ranges per the methodologies set forth in the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw, 2006). All of the terrestrial plant samples were co-located with the surface soil samples described above. Nine plant samples were collected from suspected areas of contamination within the ranges and one reference plant sample was collected from an area that was assumed to be un-impacted from Army activities. The terrestrial plants that were sampled were grass species. Grass species were targeted for sampling at the BBGR ranges because the terrestrial plant toxicity tests conducted as part of this BERA utilized perennial rye grass (*Lolium perenne*) as the test species; therefore, results of the toxicity tests and chemical analyses of plant tissues can be correlated because the test species are closely related. The results of the chemical analyses of plant tissues are presented in Table 5-3.

Terrestrial plant samples were collected from locations with soil concentrations representative of the range of lead concentrations that were shown to elicit adverse effects in terrestrial invertebrates and food web models in the *Baseline Ecological Risk Assessment for Iron Mountain Road and Bains Gap Road Ranges* (Shaw, 2004b). Sample locations were identified in the field using XRF technology.

Direct comparison of plant tissue concentrations of COPECs to screening levels could not be made because screening levels in plant tissues are not available. The concentrations of antimony in plant tissue samples ranged from non-detected (less than 0.5 mg/kg) to 0.342 mg/kg. The concentrations of copper in plant tissue samples ranged from 2.38 mg/kg to 9.76 mg/kg. The concentrations of lead in plant tissue samples ranged from 0.256 mg/kg to 43.3 mg/kg. The concentrations of zinc in plant tissue samples ranged from 9.57 mg/kg to 63.4 mg/kg. In general, the concentrations of COPECs in the plant tissue samples were lower than the concentrations of COPECs in the co-located surface soil samples, indicating a low potential for bioaccumulation. A summary of the soil COPEC concentrations in surface soil samples and terrestrial plant tissue samples is presented in Table 5-6. A detailed discussion of the bioaccumulation of COPECs from surface soil to terrestrial plants is presented in Chapter 6 of this report.

5.3 Results of Perennial Rye Grass Toxicity Tests

Surface soil from ten sample locations (one reference soil sample and nine site samples) were used to assess the potential toxicity of soil COPECs to terrestrial plants over a 15-day exposure period. Seed germination and growth tests were conducted in accordance with the ASTM (1998) protocol *Standard Guide for Conducting Terrestrial Plant Toxicity Tests* (E1963-98). The endpoints that were measured were seed germination, shoot length, root length, and total plant weight. The plant species used in the toxicity tests was perennial rye grass (*Lolium perenne*). Perennial rye grass was selected as the test species because a significant database exists for the toxicity of various contaminants on perennial rye grass and also because a significant portion of the ranges at the BBRG ranges are formerly cleared areas that are now vegetated by grasses and sedges. Testing the toxicity of the soil COPECs to a grass species provides an insight to the potential toxicity to existing terrestrial plant species that may currently exist, or could potentially exist in the future, at the BBGR ranges. At the same time, there are noteworthy limitations that are inherent in the use of laboratory-based plant toxicity tests (see Uncertainty Analysis, Chapter 7). Most importantly it should be noted that the perennial rye grass used in the toxicity tests had no prior exposure to the soil COPECs, which is unlike actual exposures that might occur at the BBGR ranges.

According to the ASTM (1998) protocol, all soil samples were first sieved through a 2.0-mm mesh screen to remove large debris and were then thoroughly homogenized prior to distribution into each test chamber. Five replicates were tested for each soil sample. Each test chamber was filled with test or control soil to approximately 1 cm below the edge of the container to avoid overflow. Samples were then hydrated to approximately 100 percent water holding capacity using Culligan water. Ten seeds were then distributed into each test chamber and covered with a light layer of test soil. Finally, each test chamber was lightly watered with Culligan water using a spray bottle. The test was terminated on day-15 by first removing the trays from the environmental chamber and counting the number of seedlings germinated in each test chamber. Seedlings were carefully separated from the soil by immersing them in a water bath and gently wiping away excess soil and seed casings. Shoot and root lengths for each plant were measured separately to the nearest mm from the transition point between root and shoot at the hypocotyl. The shoots and roots were then separated with a razor blade, dried in an oven at approximately 70 °C for 24 hours, and then weighed. A detailed description of the testing protocols and results is presented in Appendix A.

After 15 days, mean germination of rye grass seeds was 94 percent in the laboratory control sample and 90 percent in the reference site sample. Germination of seeds exposed to the site soils ranged from 80 percent in SY0004 (BERA-HR-74Q-GP63+200SE) to 94 percent in

SY0008 (BBGR-BERA-SD04). No statistically significant differences in mean germination were detected at any of the sites compared to the reference location.

Mean shoot length was 90 mm for plants grown in the laboratory control soil sample and 87 mm for those grown in the reference soil sample. Among the site samples, mean shoot length ranged from 64 mm in SY0009 (BERA-HR-74Q-GP66) to 101 mm in SY0001 (BERA-HR-74Q-GP52). Individual t-tests detected a significant difference in shoot length from the reference location for sample SY0009. Mean shoot length was reduced 26 percent in sample SY0009 compared to the reference location. None of the other samples exhibited significant reduction in shoot length. The locations of the soil samples used for plant toxicity testing and the results of the shoot length analyses are presented in Figure 5-1.

Mean root length was 108 mm for plants grown in the laboratory control soil sample and 64 mm for those grown in the reference soil sample. Among the site samples, mean root length ranged from 2.9 mm in SY0009 (BERA-HR-74Q-GP66) to 82 mm in SY0005 (BERA-HR-83Q-SD01). Four samples exhibited significant reduction in root length compared to the mean root length of seeds grown in soil from the reference location. These four samples were SY0009 (BERA-HR-74Q-GP66) with mean root length of 2.9 mm, SY0002 (BERA-CWM-183-SD02) with mean root length of 51mm, SY0004 (BERA-HR-74Q-GP63+200SE) with mean root length of 46 mm, and SY0010 (BERA-HR-118Q-DEP01) with mean root length of 36mm. Although statistically significant, the mean root length for seeds grown in sample SY0002 (BERA-CWM-183-SD02) was only marginally different (20 % reduction) than the root length for seeds grown in the reference soil sample. None of the other samples exhibited significant reduction in root length. The locations of the soil samples used for plant toxicity testing and the results of the root length analyses are presented in Figure 5-2.

Mean weight was 8.77 mg for plants grown in the laboratory control soil sample and 6.75 mg for those grown in the reference soil sample. Among the site samples, mean weight ranged from 3.02 mg in SY0009 (BERA-HR-74Q-GP66) to 7.76 mg in SY0005 (BERA-HR-83Q-SD01). Four samples exhibited significant reduction in total plant mass compared to the total plant mass of plants grown in soil from the reference location. These four samples were SY0009 (BERA-HR-74Q-GP66) with total plant mass of 3.02 mg, SY0004 (BERA-HR-74Q-GP63+200SE) with total plant mass of 4.76 mg, SY0006 (BERA-HR-79Q-SS09) with total plant mass of 5.36 mg, and SY0010 (BERA-HR-118Q-DEP01) with total plant mass of 4.56 mg. Although statistically significant, the total plant mass for plants grown in soil from sample SY0006 (BERA-HR-79Q-SS09) was only marginally different (20 % reduction) than the total plant mass for plants grown in the reference soil sample. None of the other samples exhibited significant reduction in total

plant mass. The locations of the soil samples used for plant toxicity testing and the results of the total plant weight analyses are presented in Figure 5-3. A summary of the COPEC concentrations in surface soil samples and the perennial rye grass toxicity tests is presented in Table 5-7.

5.4 Results of Earthworm Toxicity and Bioaccumulation Tests

Surface soil from ten sample locations (one reference soil sample and nine site samples) within the BBGR ranges was used for toxicity and bioaccumulation testing using the earthworm *Eisenia fetida*. These ten surface soil samples were collected from the same locations as the samples collected for the perennial rye grass toxicity tests. The test procedures utilized in the performance of the 28-day earthworm toxicity and bioaccumulation tests were based upon those procedures described in *Ecological Effects Test Guidelines, Earthworm Subchronic Toxicity Test* (USEPA, 1996) and *Standard Guide for Conducting Laboratory Soil Toxicity or Bioaccumulation Tests with the Lumbricid Earthworm Eisenia fetida* (ASTM, 1999). A detailed description of the testing protocols and results is presented in Appendix A.

Earthworms were exposed to test soils for 28-days to assess survival, growth, and the potential for bioaccumulation. Test chambers consisted of 1.0-L glass jars with perforated lids to allow air exchange. Each soil sample was replicated eight times. All soil samples were sieved through a 6.3-mm stainless steel screen prior to testing to remove large debris and were then thoroughly homogenized prior to distribution into each test chamber. Samples were then hydrated to an appropriate moisture level using DI water. Each replicate test chamber received 290 g of soil. Ten earthworms were added to each test chamber and the worms were not fed during the test period.

Earthworm survival was assessed on both day-14 and at the termination of the test on day-28. A measure of survival at day-14 was accomplished by emptying the contents of four replicate jars (one at a time) into a clean plastic tray and gently sorting the contents to locate the worms. The number of surviving worms was recorded and all of the worms were placed back into the same replicate test chamber to continue the remainder of the 28-day exposure. At the test termination (day-28), each of the eight replicates were emptied (one at a time) into a clean plastic tray and gently sorted to locate the worms. The number of surviving worms in each replicate and their composite wet weight were recorded. Dead worms were removed and discarded. The surviving worms were washed with DI water to remove any soil and placed in a clean 1-L glass jar with moist paper towels to depurate overnight. The next day, worms were removed from the depuration jars, rinsed with DI water, placed in labeled plastic food-grade Ziploc bags, and

immediately placed in a freezer at a temperature of approximately -20 °C. Frozen worms were shipped overnight to the analytical laboratory for chemical analysis.

Day 14 survival ranged from 98 percent to 100 percent. The 28-day survival rates ranged from 93 percent to 100 percent. No statistically significant differences were detected in earthworm survival after 14-day or 28-day exposures to site surface soils. A summary of the COPEC concentrations in surface soil samples and earthworm toxicity test results is presented in Table 5-8.

All samples showed a loss in weight since no food was added during the test. Percent weight loss ranged from 67 percent in the reference sample SY0011 (BBGR-BERA-REF-SS) to 143 percent in site sample SY0009 (BERA-HR-74Q-GP66). Percent weight loss in the laboratory control was 105 percent. A Kruskall-Wallis test did not detect an overall significant difference in weight loss among the worms in any of the samples and individual comparison t-tests did not detect significant decreases in weight at any individual site when compared to the reference soil.

COPEC concentrations in surface soil samples and earthworm tissues are summarized in Table 5-9. Antimony concentrations in earthworm tissues ranged from ND in five earthworm tissue samples to 2.5 mg/kg in sample SY0002 (BBGR-BERA-SS01). Copper in earthworm tissues ranged from 13.6 mg/kg in sample SY0011 (BBGR-BERA-REF-SS) to 89.4 mg/kg in sample SY0010 (BERA-HR-118Q-DEP01). Lead in earthworm tissues ranged from 3.6 mg/kg in sample SY0011 (BBGR-BERA-REF-SS) to 1,140 mg/kg in sample SY0010 (BERA-HR-118Q-DEP01). Zinc in earthworm tissues ranged from 97.9 mg/kg in sample SY0011 (BBGR-BERA-REF-SS) to 128 mg/kg in sample SY0005 (BERA-HR-79Q-GP31). Results of the chemical analysis of earthworm tissues are presented in Table 5-4.

5.5 Results of Midge Survival, Growth, and Bioaccumulation Tests

Sediment samples from twelve locations along the South Branch of Cane Creek, Ingram Creek, and their tributaries at the BBGR ranges were used for toxicity testing using the larval midge *Chironomus tentans*. A reference sample was also collected from the unnamed tributary to Choccolocco Creek that flows in an easterly direction along Bains Gap Road, immediately east of the Bains Gap gate to the Main Post.

The test procedures utilized in the performance of the 10-day midge toxicity tests were based upon those procedures described in *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates* (USEPA, 2000c) and *Standard Guide for Determination of the Bioaccumulation of Sediment-Associated Contaminants*

by Benthic Invertebrates (ASTM, 2000). A detailed description of the testing protocols and results is presented in Appendix A.

Midge larvae were exposed to test sediments for ten days to determine the effects of sediment COPECs on survival and growth. The bioaccumulation potential of the sediment COPECs into midge tissues was tested over a 21-day exposure period. Sediment samples for all screening toxicity tests were replicated five times with 10 organisms per replicate chamber. The test chambers were 1.0-L glass jars containing approximately 2 cm of sediment and approximately 800 mL of overlying Culligan-filtered water. The midge larvae were fed approximately 1 mL of a prepared invertebrate food every 2 to 3 days during the test duration. The number of live and dead animals in each test chamber were enumerated at test termination (day 10) by sieving the sediment through a No. 35 (500-µm) and/or No. 40 (425-µm) sieve. Animals were considered dead if they did not respond to a gentle physical stimulus. Ash-free dry weights were measured by placing animals into an ashed and pre-tared weigh boat and dried at 60 ºC for 24 hours. Weigh boats were placed into a dessicator to cool and then re-weighed. Weigh boats were then placed into a muffle furnace at approximately 600 ºC for 24 hours. Weigh boats were allowed to cool in a dessicator and then re-weighed.

A summary of the midge survival percentages is presented in Table 5-10. Midge survival ranged from 80 percent in site sample SY1009 (BERA-HR-79Q-SD01) and the reference sample SY1015 (BBGR-BERA-REF-SED) to 96 percent in site sample SY1013 (BERA-HR-79Q-SD05). No statistically significant differences in survival were detected for any of the site samples compared to the reference location. The locations of the sediment samples and the results of the midge survival tests are presented in Figure 5-4.

The test procedures utilized in the performance of the 21-day midge bioaccumulation test were based upon those general procedures described above. Test chambers consisted of shallow, 27-L plastic trays (38 cm wide x 56 cm long x 13 cm deep) filled with 1.5 cm of sediment (approximately 1.0 L of sediment per replicate). There were four replicate trays (A – D) per sample. All test chambers were maintained on a flow-through system delivering a volume of water equivalent to approximately two full water changes per day.

At test initiation, four *C. tentans* egg cases were added to replicate test chambers A and B, and three *C. tentans* egg cases were added to replicate test chambers C and D. Each egg case contained approximately 400 – 500 *C. tentans* eggs. The number of egg cases was estimated to yield adequate biomass for chemical analysis at test termination.

The *C. tentans* life cycle is comprised of several distinct stages including egg, larva, pupa, and free-flying adult stages. At the pupa stage of development, the *C. tentans* leave the sediment and enter the water column. In order to maximize both sediment exposure and tissue recovery, bioaccumulation test termination was commenced when the first few larvae reached the pupa stage of development. The goal was to ensure that the majority of the larvae had reached their maximum size, but had not yet transformed into the pupa stage. This point of development was reached on or around day-21 of the exposure period. In order to ensure consistency in the methodology, all sample replicates were terminated on day-21 of the exposure period.

At the test termination, sediment was sorted to extract the larvae. *C. tentans* were gently rinsed with Culligan water to remove debris. All of the recovered organisms were placed in clean aquaria filled with Culligan water and allowed to depurate overnight. The following day, a second cleaning of the organisms was performed to remove any excess debris. All replicates from a single sample were then combined into a single food-grade Ziploc plastic bag and immediately placed in a freezer at a temperature of approximately -20 °C. All of the frozen tissue samples were than shipped overnight to the analytical laboratory.

A summary of the ash-free dry weight measurements of midge are presented in Table 5-10. Mean dry weight after the 10-day exposure period was 1.45 mg in the laboratory control and 1.69 mg in the reference sample. Mean dry weight among the site sediment samples ranged from 1.29 mg in SY1004 (BERA-FTA-163-SD02) to 1.67 mg in SY1014 (BBGR-BERA-SD01). Statistically significant differences in weight were found for site samples SY1002 (BERA-CWM-183-SD02) with mean dry weight of 1.50 mg and SY1005 (BERA-HR-83Q-SD01) with mean dry weight of 1.43 mg. These differences, however, only represent 11 percent and 15 percent reductions in growth from the reference sample and may not be biologically significant. The locations of the sediment samples and the results of the midge growth test are presented in Figure 5-5.

The surviving organisms in the 27-liter plastic trays were utilized solely for generation of tissue for chemical analysis. The number of surviving organisms was estimated for analytical use only. Chironomid tissue samples were analyzed for metals and pesticides content. The concentrations of sediment COPECs (copper, lead, and gamma-chlordane) detected in chironomid tissues are presented in Table 5-11. Copper concentrations in chironomid tissues ranged from 44.2 mg/kg in sample SY1015 (BBGR-BERA-REF-SED) to 361 mg/kg in sample SY1010 (BERA-HR-79Q-SD03). Lead concentrations in chironomid tissues ranged from 4.25 mg/kg in sample SY1015 (BBGR-BERA-REF-SED) to 84.5 mg/kg in sample SY1010 (BERA-HR-79Q-SD03). Gamma-chlordane was not detected in 12 out of the 13 chironomid tissue samples. The only tissue

sample that exhibited a detectable concentration of gamma-chlordane was sample SY1009 (BERA-HR-79Q-SD01) with a detected concentration of 0.0023 mg/kg. Results of the chemical analysis of midge tissues is presented in Table 5-5.

The locations of the sediment samples used in the chironomid toxicity tests and the corresponding chironomid survival and growth rates are presented in Figures 5-4 and 5-5, respectively.

5.6 Results of Benthic Community Analysis

Benthic community assemblages in creeks in the vicinity of the BBGR Ranges and a reference location were assessed using a multi-metric approach in accordance with the USEPA's Rapid Bioassessment Protocols (Plafkin, et al, 1989; Barbour, et al., 1999). The methodologies used in the present assessment are consistent with those employed in the assessment of the Bains Gap Road (BGR) Ranges conducted in 2003. Benthic macroinvertebrate samples, which were co-located with the sediment samples used for chemical analysis and chironomid toxicity testing, were collected from 12 locations on the South Branch of Cane Creek, Ingram Creek and their tributaries that flow through the area of the BBGR ranges. Benthic macroinvertebrate samples were also collected at a reference location. The benthic macroinvertebrate sample locations are shown on Figures 5-4 and 5-5. The specific sampling locations were determined in the field, based on available invertebrate habitat and field observations.

5.6.1 Benthic Invertebrate Habitat Assessment

An important factor in the establishment of a viable benthic community assemblage is the quality of the benthic habitat that is present in the stream of interest. Simply put, the quality of the benthic habitat has a significant impact on the quality of the benthic assemblage present. The benthic invertebrate habitat assessment is based on the *Stream Classification Guidelines for Wisconsin* developed by Ball (1982) and *Methods of Evaluating Stream, Riparian, and Biotic Conditions* developed by Platts, et al. (1983). The benthic invertebrate habitat assessment includes an evaluation of the variety and quality of the substrate, channel morphology, bank structure, and riparian vegetation.

The habitat parameters that are pertinent to the assessment of habitat quality include: epifaunal substrate and available cover; embeddedness; velocity/depth regime; sediment deposition; channel flow; channel alteration; frequency of riffles; bank stability, vegetative protection; and riparian vegetative zone width. The habitat assessment process involves rating these ten habitat parameters as optimal, suboptimal, marginal, or poor based on the criteria included on the Habitat Assessment Field Data Sheets (Appendix B). All parameters are evaluated and rated on

a numerical scale of 0 to 20 (0 being the lowest and 20 being the highest). The ratings are then totaled and compared to a reference condition. Scores increase as the habitat quality increases.

The results of the benthic invertebrate habitat assessment are presented in Appendix B and summarized in Table 5-12. The benthic habitat quality scores ranged from a low of 11.5 (sub-optimal) at sampling location HR-83Q-SD01 to a high of 16.8 (optimal) at sampling location CWM-183-SD02. The reference location had a habitat quality score of 17.1. None of the sampling locations were scored as having marginal or poor quality benthic invertebrate habitat.

5.6.2 Benthic Invertebrate Community Assessment

Benthic macroinvertebrate samples were collected in accordance with the procedures specified in Appendix B of the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw, 2006). Two benthic macroinvertebrate samples were collected at each location, a riffle/run sample and a course particulate organic matter (CPOM) sample. The riffle/run sample was collected with a kick net, and consisted of a composite sample collected from two areas; one from an area of relatively fast current velocity (riffle), and the other from an area of relatively slow current (run). The CPOM sample was hand collected from a depositional area of the creek and consisted of a composite of a variety of leaves, twigs, bark, and other organic matter. All samples were placed into appropriate containers and preserved with buffered 10 percent formalin solution.

Benthic macroinvertebrate samples were processed by Shaw aquatic scientists. Most samples for benthic invertebrate analysis were analyzed in total (i.e., not “split” into sub-samples) because the abundance of invertebrates was not excessively high, ranging from 21 to 345 in the riffle/run samples, and 5 to 255 in the CPOM samples. In accordance with Plafkin et al. (1989), the CPOM data were used only for the calculation of Metric 8, the ratio of the shredder functional feeding group to the total number of organisms in the sample. During the sorting process, all invertebrates found in a sample were separated into major taxa (such as insect Order), were counted, and were placed into a glass vial containing 95 percent ethanol. The vial containing the invertebrates from each sample was labeled with the sample location code. Benthic macroinvertebrates were then identified to Family level using both compound and dissecting microscopes based on the information provided in Merritt and Cummins (1996), Peckarsky, et al. (1990) and Pennak (2001). As the organisms were identified and counted, the information was recorded on bench sheets, and the specimens were placed in labeled vials with ethanol as a preservative and archived for future reference. A summary of the benthic macroinvertebrates collected during this survey are shown in Table 5-13. Also shown on this table are the pollution

tolerance values and the functional feeding groups of each taxon that were used to calculate specific metrics

Using the raw benthic data, a numerical value was calculated for each of the eight Rapid Bioassessment Protocol (RBP) metrics. Calculated values were then compared to values derived from the reference site. Each metric was then assigned a score according to its comparability to the reference location value for that metric. Scores for the eight metrics were then totaled and compared to the total metric score for the reference location. The percent comparison between the total scores for the potentially impacted sites and the reference site provides a final evaluation of biological condition. A site is assigned a biological condition of non-impaired, moderately impaired, or severely impaired in relation to the reference location.

The eight RBP metrics used to evaluate the BBGR sites and the results of the assessment are described below, and are presented in Table 5-14.

Metric 1: Taxa Richness. Taxa richness is calculated by counting the number of taxa present in each sample. Taxa richness reflects the health of the community through a measurement of the variety of taxa present. In general, taxa richness increases with increasing water quality, habitat diversity, and/or habitat suitability. Taxa richness at the BBGR sites ranged from 3 at sample location FTA-163-SD02 to 17 taxa at BBGR-BERA-SD03 and CWM-183-SD02. The reference site BBGR-BERA-SED-REF also had 17 taxa.

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 Bode and Novak (1995) and Hilsenhoff (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 as follows:

$$FBI = \Sigma \frac{x_i t_i}{n}$$

where:

x_i	=	number of individuals within a taxon;
t_i	=	tolerance value of a given taxon; and
n	=	total number of organisms in the sample.

Values for the FBI range from 0 to 10 with higher values corresponding to greater levels of organic pollution as shown in the following table:

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

The family biotic index (FBI) at the BBGR sites ranged from 4.524 at sampling location BBGR-BERA-SD01, to 7.04 at sample location BBGR-BERA-SD02. The FBI score at the reference site was 4.176.

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. The proportion of the two feeding groups is important because predominance of a particular feeding type may indicate an unbalanced community responding to an overabundance of a particular food source. Functional feeding group designations for the taxa identified were obtained from Bode and Novak (1995) and Merritt and Cummins (1984). This metric is calculated by dividing the relative abundance of scrapers by the relative abundance of filter feeding organisms. In general, the ratio increases with increasing water quality.

The ratio of scrapers and filterers was not calculable at two sample locations, FTA-163-SD02 and HR-79Q-SD01 because no organisms of the filtering collector feeding group were present in those samples. The values calculated for the remaining BBGR locations ranged from 0.333 at location HR-83Q-SD01, to 13 at sample location BBGR-BERA-SD01 and HR-79Q-SD05. The ratio at the reference site was 0.508.

Metric 4: Ratio of EPT and Chironomidae Abundances. The ratio of Ephemeroptera, Plecoptera and Trichoptera (EPT) and Chironomidae abundance is 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. Good biotic condition is reflected in communities having a fairly even distribution among all four major groups and with substantial representation in the sensitive groups Ephemeroptera, Plecoptera and Trichoptera. Skewed populations having a disproportionate number of the generally tolerant Chironomidae relative to

the more sensitive insect groups may indicate environmental stress. The EPT/Chironomidae ratio generally increases with increasing water quality.

The EPT/Chironomidae ratio was not calculable at location HR-74Q-SD01, because no Chironomidae were found in the sample, and the ratio was zero at FTA-163-SD02 because there were no EPT taxa in sample. The ratio of EPT and Chironomidae at the eight sample locations that had both Ephemeroptera, Plecoptera, Trichoptera and Chironomidae ranged from 0.029 at HR-79Q-SD03 to 15 at HR-79Q-SD01. The EPT/Chironomidae ratio at the reference location was calculated to be 18.4.

Metric 5: Percent Contribution of Dominant Taxon. The percent contribution of the numerically dominant taxon to the total number of organisms is an indication of community balance at the lowest positive taxonomic level. A community dominated by relatively few species would indicate environmental stress. The percent contribution of the dominant taxon is calculated by dividing the abundance of the taxon that 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. The percent contribution of the dominant taxon at the BBGR sites ranged from 76 at sample location FTA-163-SD02 to 18 at location BBGR-BERA-SD04. The reference location had a 30 percent contribution of the dominant taxon.

Metric 6: EPT Index. The EPT Index is the total number of distinct taxa within the orders Ephemeroptera, Plecoptera, and Trichoptera. This value summarizes taxa richness within the insect orders that are generally considered to be pollution sensitive. The EPT index generally trends higher with increasing water quality. The EPT index of the BBGR sites ranged from 0 at FTA-163-SD02 to 7 at CWM-183-SD02. An EPT index of 7 was also found for the reference location.

Metric 7: Community Loss Index. This index evaluates the benthic populations at specific locations relative to populations present at the reference location. The Community Loss Index measures the loss of benthic species between a reference station and the station of comparison. It is an index of dissimilarity with values increasing as the degree of dissimilarity from the reference station increases. 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:

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

A lower index value corresponds to a smaller difference from the reference location, and therefore indicates higher water quality. The community loss index for the BBGR sites ranged from 5.33 at sample location FTA-163-SD02 to 0.412 at location CWM-183-SD02.

Metric 8: Ratio of Shredder Functional Feeding Group and Total Number of Individuals Collected. The abundance of the shredder functional group relative to the abundance of all other functional groups allows evaluation of potential impairment as indicated by the CPOM-based shredder community. Shredders are sensitive to riparian zone impacts and are particularly good indicators of toxic effects when the toxicants involved are readily absorbed to the CPOM and either affect the microbial communities colonizing the CPOM or the shredders directly. The ratio of the relative abundance of shredders to the abundance of all other functional feeding groups is calculated by dividing the relative abundance of shredders by the total number of organisms in a given sample. The ratio generally increases with increasing water quality. Other factors that may affect the abundance of shredders include climate, season, and vegetation abundance and type within the riparian zone.

No shredder feeding organisms were found at locations BBGR-BERA-SD01, BBGR-BERA-SD04, BBGR-BERA-SD05, HR-79Q-SD03, and HR-83Q-SD01. These locations, therefore, had a shredder abundance to total organism abundance ratio of zero. The highest ratio of shredders at the BBGR sites was 0.395 found at location HR-79Q-SD01, while the reference location had a ratio of 0.632.

The total benthic community score for each sampling location, the relative biological condition, and the concentrations of sediment COPECs are presented in Table 5-15.

6.0 Risk Characterization

This Risk Characterization analyzes and interprets the data collected during the Site Investigation in terms of ecological exposures and effects and estimates the potential for ecological risks. In the case of toxicity tests, the data interpretation attempts to relate effects observed in the toxicity tests to measured exposure concentrations of COPECs in the environmental samples. The result of this interpretation is the determination of toxicity values that relate COPEC concentrations to potential effects. The following toxicity values are developed within the subsequent sections that describe the results of the toxicity tests:

- ***Apparent Effects Threshold (AET)*** – The concentration at and above which effects are always observed. Since all samples with concentrations of a COPEC at or above the AET have observed effects, there is good evidence that the COPEC, or another co-occurring COPEC or stressor, may be causing the observed effects.

- ***No Observable Effect Concentration (NOEC)*** – The highest concentration at and below which effects are never observed. Since all samples with COPEC concentrations at or below the NOEC have no observable effects, there is strong evidence that the COPEC is not toxic at levels at or below the NOEC under those environmental conditions.

- ***Lowest Observable Effect Concentration (LOEC***) – The lowest concentration at which an adverse effect is observed. Often some samples with concentrations between the AET and NOEC are toxic and other samples are not toxic. The LOEC is the lowest concentration of the COPEC in this range that has an observed effect. However, since some samples with COPEC concentrations above the LOEC are not toxic, the evidence that the COPEC is the cause of the observed effect is not as strong. The effects may or may not be related to the COPEC.

A useful parameter to evaluate with regard to these toxicity values is predictability. Predictability is essentially the ability to correctly predict that concentrations above the toxicity value are toxic and that concentrations below the toxicity value are not toxic. The results of samples that are incorrectly predicted are either false positives (i.e., samples that were predicted to be toxic but were found to be non-toxic) or false negatives (i.e., samples that were predicted to be non-toxic but were found to be toxic). By definition, the AET has no false negatives and the LOEC (or NOEC) has no false positives. This terminology will be used in the following sections describing the toxicity tests conducted as part of the Site Investigation.

The following sections describe the results of this baseline ecological risk assessment with respect to the assessment endpoints and decision rules presented in the *Baseline Ecological Risk*

Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges (Shaw, 2006) and summarized in the Chapter 4 of this report.

6.1 Terrestrial Ecosystems

Four assessment endpoints have been identified for the terrestrial ecosystems at the BBGR ranges. These assessment endpoints are the following:

- Survival and growth of the terrestrial plant community at the BBGR Ranges
- Survival and growth of the terrestrial invertebrate community at the BBGR Ranges
- Survival, growth, and reproduction of terrestrial invertivorous small mammals and birds at the BBGR Ranges
- Survival, growth, and reproduction of terrestrial omnivorous small mammals and birds at the BBGR Ranges.

Six decision rules have been defined pertaining to terrestrial ecosystems at the BBGR ranges and have been designed to answer specific questions related to the terrestrial ecosystem assessment endpoints. The results of the Site Investigation and subsequent risk characterization are described herein with respect to each of these decision rules.

6.1.1 Terrestrial Plant Germination and Growth

The decision rule pertaining to terrestrial plant germination and growth sates that "If COPECs in soils from the BBGR 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 risks to terrestrial plants at the BBGR Ranges."

The results of the perennial rye grass toxicity tests indicate that seed germination was not adversely affected at any of the sample locations. None of the seed germination rates were significantly reduced compared to seed germination rates using soil from the reference site.

The results of the perennial rye grass toxicity tests indicate that shoot length was statistically reduced in one sample compared to shoot length of seedlings grown in soil from a reference location. Statistically reduced shoot length was reported in sample SY0009 (BERA-HR-74Q-GP66). Shoot length was reduced 26 percent in this sample compared to shoot length in seedlings grown in soil from the reference site. None of the other samples exhibited statistically reduced shoot growth.

Root length was statistically reduced in four samples compared to root length of seedlings grown in soil from a reference location. Statistically reduced root length was reported in sample SY0002 (BBGR-BERA-SS01), SY0004 (BERA-74Q-GP64+200SE), SY0009 (BERA-HR-74Q-GP66), and SY0010 (BERA-HR-118Q-DEP01).

Total plant mass was statistically reduced in four samples compared to total plant mass of seedlings grown in soil from a reference location. Statistically reduced plant mass was reported in sample SY0004 (BERA-74Q-GP64+200SE), SY0006 (BERA-HR-79Q-SS09), SY0009 (BERA-HR-74Q-GP66), and SY0010 (BERA-HR-118Q-DEP01). The results of the perennial rye grass toxicity tests were summarized in Table 5-7. The locations of the soil samples used for the perennial rye grass toxicity tests and the results of the toxicity tests were also presented in Figures 5-1 through 5-3.

The perennial rye grass seed germination results showed no statistically significant differences in the germination rates of seeds grown in site soils and seeds grown in soil from the reference site after the 15-day exposure period. Therefore, the NOECs based on perennial rye grass germination for the five soil COPECs are defined as the highest concentrations tested. The LOECs and AETs are simply listed as greater than the NOEC concentrations. The NOEC, LOEC, and AET values based on rye grass germination are summarized below and presented in Table 6-1.

Soil COPEC	***Germination NOEC*** *(mg/kg)*	***Germination LOEC*** *(mg/kg)*	***Germination AET*** *(mg/kg)*
Antimony	1,310	> 1,310	> 1,310
Beryllium	1.67	> 1.67	> 1.67
Copper	563	> 563	> 563
Lead	146,000	> 146,000	> 146,000
Zinc	154	> 154	> 154

One soil sample showed a statistically significant difference for mean shoot length. Only sample SY0009 (BERA-HR-74Q-GP66) exhibited statistically significant reduced shoot length compared to the shoot lengths of seedlings grown in soil from the reference location. Measured shoot lengths as a function of the COPEC concentrations in soil are presented in Figure 6-1 for antimony, Figure 6-2 for beryllium, Figure 6-3 for copper, Figure 6-4 for lead, and Figure 6-5 for zinc. The correlation between the soil concentration and mean shoot length was highest for zinc (Pearson's correlation coefficient r^2 was 0.1293), though this is not considered significant. The correlation between mean shoot length and concentrations of the other COPECs was even weaker: $r^2 = 0.01$ for antimony, $r^2 = 0.0772$ for beryllium, $r^2 = 0.0387$ for copper, and $r^2 =$

0.0097 for lead. Since the soil sample with the highest concentrations of COPECs showed no effect on shoot length, the AETs are equal to the highest concentrations tested. The NOECs, LOECs, and AETs based on shoot length, are summarized below and also presented in Table 6-1.

Soil COPEC	***Shoot Length NOEC*** *(mg/kg)*	***Shoot Length LOEC*** *(mg/kg)*	***Shoot Length AET*** *(mg/kg)*
Antimony	4.92	5.29	1,310
Beryllium	0.624	0.685	1.67
Copper	152	184	563
Lead	526	919	146,000
Zinc	37.3	53.5	154

Significant differences in root length were found in four of the soil samples tested compared to root length for seedlings grown in the reference soil. The root length results as a function of the COPEC concentrations in soil are presented in Figure 6-6 for antimony, Figure 6-7 for beryllium, Figure 6-8 for copper, Figure 6-9 for lead, and Figure 6-10 for zinc. Though the correlation of root length with COPEC concentration in soil was highest for copper ($r^2 = 0.2289$), it was still relatively weak. Correlations with the other COPECs were $r^2 = 0.0673$ for antimony, $r^2 = 0.0454$ for beryllium, $r^2 = 0.068$ for lead, and $r^2 = 0.1043$ for zinc. The AETs, based on root length, for antimony, beryllium, copper, lead, and zinc are summarized below. Since the soil with the lowest concentrations of antimony and copper were found to exhibit significantly reduced root lengths, the LOECs for antimony and copper are defined as the lowest concentrations tested (0.724 mg/kg and 16.5 mg/kg, respectively) and the NOEC is less than these concentrations. The NOECs and LOECs, based on root length are summarized below and also presented in Table 6-1.

Soil COPEC	***Root Length NOEC*** *(mg/kg)*	***Root Length LOEC*** *(mg/kg)*	***Root Length AET*** *(mg/kg)*
Antimony	< 0.724	0.724	1,310
Beryllium	0.156	0.444	1.02
Copper	< 16.5	16.5	152
Lead	102	125	146,000
Zinc	22.3	25.4	53.5

Significant differences in total weight were found in seedlings grown in four of the site soil samples tested compared to the total weight of seedlings grown in the reference soil. The total weight results as a function of the COPEC concentrations in soil are presented in Figure 6-11 for antimony, Figure 6-12 for beryllium, Figure 6-13 for copper, Figure 6-14 for lead, and Figure 6-15 for zinc. Though the correlation of total plant weight to COPEC concentration in soil were highest for copper and zinc ($r^2 = 0.23$ for each), these correlations were still relatively weak.

Correlations of total plant weight to COPEC concentrations in soil for the other COPECs were r^2 = 0.0767 for antimony, r^2 = 0.0924 for beryllium, and r^2 = 0.0773 for lead. The AETs, based on total plant weight, for antimony, beryllium, copper, lead, and zinc are 1,310 mg/kg, 1.02 mg/kg, 184 mg/kg, 146,000 mg/kg, and >154 mg/kg, respectively. Since the seedlings grown in soil with the lowest concentrations of antimony and copper were found to have significant reductions in total weight, the LOECs for antimony and copper are defined as the lowest concentrations tested (0.724 mg/kg and 16.5 mg/kg, respectively) and the NOECs are less than these numbers. The NOECs, LOECs, and AETs based on total plant weight are summarized below and presented in Table 6-1.

Soil COPEC	***Total Plant Mass NOEC*** *(mg/kg)*	***Total Plant Mass LOEC*** *(mg/kg)*	***Total Plant Mass AET*** *(mg/kg)*
Antimony	< 0.724	0.724	1,310
Beryllium	0.583	0.597	1.02
Copper	< 16.5	16.5	184
Lead	102	125	146,000
Zinc	22.3	25.4	> 154

Table 6-1 also presents the relative predictability of the AETs and LOECs for the various toxicity endpoints for the soil COPECs. The AETs for copper had fairly good predictability with only 1 false negative for root length and 2 false negatives for total plant weight. The NOEC-LOEC pairs showed poor predictability for all COPECs and all endpoints.

It is also helpful to examine the range of COPEC concentrations between the LOEC and the NOEC. Over this range, the toxicity test results are often mixed, showing both toxic and non-toxic responses. For example, for copper and root length, this uncertainty range was 16.5 mg/kg to 74.1 mg/kg, spanning a factor of about 5. For lead, on the other hand, this range was 125 mg/kg to 1,240 mg/kg, spanning an order of magnitude. The difference in the ranges is reflective of the greater correlation of the reduction in root length with copper concentrations in soil than with concentrations of lead in soil. Within the uncertainty range for copper, however, all six samples that had copper concentrations within this range showed no detectable toxic effect on shoot length. Thus, it seems that the LOEC is an anomalous toxic result in so far as copper is concerned and the AET is the most reliable indicator of toxicity of copper in soils.

Additionally, these toxicity values can be compared to generic screening benchmarks from the literature. Efroymson et al. (1997) developed screening benchmarks designed as conservative estimates of concentrations below which effects on terrestrial plants are not expected. These benchmarks for antimony, beryllium, copper, lead, and zinc are reported as 5 mg/kg, 10 mg/kg, 100 mg/kg, 50 mg/kg, and 50 mg/kg, respectively. The benchmark for copper is, as expected

based on the benchmark's conservative nature, below the AET but well above the LOEC, also supporting the use of the AET as the most reliable indicator of copper toxicity.

6.1.2 Terrestrial Plant Bioaccumulation

In order to assess the potential for bioaccumulation of soil COPECs into terrestrial plants, samples of above-ground vegetation were collected concurrently with the soil samples used in the toxicity tests and analyzed for TAL metals. The concentrations of COPECs in above-ground vegetation as a function of the soil COPEC concentrations are presented in Figure 6-16 for copper, Figure 6-17 for lead, and Figure 6-18 for zinc. These figures show graphically the potential bioavailability and bioaccumulation of the COPECs in the soil to terrestrial plants. Antimony and beryllium were not detected in sufficient numbers of vegetation samples to examine for accumulation. The correlation between concentrations of COPECs in vegetation and soil was strongest for lead ($r^2 = 0.3393$). The correlations for copper ($r^2 = 0.0284$) and zinc ($r^2 = 0.0368$) were very weak. Though lead concentrations in soil and vegetation exhibited a fairly good correlation ($r^2 = 0.3393$), the highest soil concentration of lead did not result in an increase in vegetation concentration above those from soils with much lower concentrations of lead. This apparent lack of bioavailability of lead (at least within the above-ground portion of the vegetation) may explain the relative absence of toxic effects on shoot length in the sample with the highest lead soil concentrations.

In order to define the site-specific relationship between soil concentrations and plant tissue concentrations of COPECs, site-specific bioaccumulation factors were estimated by comparing the plant tissue concentrations to soil concentrations of COPECs at each sampling location in the following manner.

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

where:

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

Site-specific BAFs were estimated for each soil sample location and a mean soil-to-plant BAF was then calculated for each soil COPEC. The soil-to-plant BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-6:

Surface Soil COPEC	Soil-to-Plant BAF
Copper	0.13 *
Lead	0.03
Zinc	1.1 *

* Due to poor correlations these site-specific $BAF_{soil\text{-}to\text{-}earthworm}$ were not used in the food web models.

It is important to note that, as stated previously, there are very poor correlations between plant tissue concentrations and soil concentrations for antimony, beryllium, copper, and zinc. In fact, beryllium was not detected in any of the plant tissue samples and antimony was only detected in three out of ten plant tissue samples.

These soil-to-plant BAFs were used in the terrestrial food web model to estimate the transfer of COPECs from surface soil to terrestrial plants. Based on the sparsity of plant tissue data for antimony and beryllium and the poor correlation between plant tissue concentrations and soil concentrations for copper and zinc, default soil-to-plant BAFs (USEPA, 1999) were assumed for these COPECs and used in the food web model. The site-specific soil-to-plant BAF for lead was used in the food web model.

6.1.3 Earthworm Survival and Growth

The decision rule pertaining to earthworm survival and growth states that "If COPECs in soils from the BBGR Ranges cause earthworm toxicity which is statistically greater than earthworm toxicity in soils from a reference site, then there is the potential for unacceptable risks to terrestrial invertebrate receptors at the BBGR Ranges."

The results of the earthworm toxicity testing indicate that earthworm survival and growth were not adversely affected at any of the sample locations. None of the survival or growth rates were significantly reduced after either 14 days or 28 days of exposure compared to survival and growth rates using soil from the reference site (Table 5-8). Therefore, the NOECs for both earthworm survival and growth are the highest concentrations tested, and the LOECs and AETs are defined as greater than the NOEC concentrations. These values are summarized below and also presented in Table 6-1.

Soil COPEC	Earthworm NOEC (mg/kg)	Earthworm LOEC (mg/kg)	Earthworm AET (mg/kg)
Antimony	1310	> 1310	> 1310
Beryllium	1.67	> 1.67	> 1.67
Copper	563	> 563	> 563
Lead	146,000	> 146,000	> 146,000
Zinc	154	> 154	> 154

6.1.4 Earthworm Bioaccumulation

The decision rule pertaining to accumulation of COPECs from surface soil by earthworms states that "If earthworms exposed to soils from the BBGR 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."

The earthworm tissue COPEC concentrations as a function of the soil COPEC concentrations are presented in Figure 6-19 for antimony, Figure 6-20 for copper, Figure 6-21 for lead, and Figure 6-22 for zinc. Beryllium was not detected in any of the earthworm tissue samples; therefore, a graph of concentrations is not presented. The correlations between tissue concentrations and soil concentrations were strongest for copper ($r^2 = 0.94$) and lead ($r^2 = 0.90$). Antimony was only detected in five of the earthworm tissue samples; therefore, the correlations were derived for these five samples only (Figure 6-19). The correlation was weak for antimony ($r^2 = 0.172$), as well as zinc ($r^2 = 0.0511$). These figures help relate the potential bioavailability of the COPECs in soil to the earthworm toxicity test results.

In order to define the site-specific relationship between soil concentrations and earthworm tissue concentrations of COPECs, site-specific bioaccumulation factors were estimated by comparing the earthworm tissue concentrations to soil concentrations of COPECs at each sampling location in the following manner.

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

where:

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

Site-specific BAFs were estimated for each soil sample location and a mean soil-to-earthworm BAF was then calculated for each soil COPEC. The soil-to-earthworm BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-9:

Surface Soil COPEC	Soil-to-Earthworm BAF
Copper	0.5
Lead	0.53
Zinc	3.17 *

* Due to poor correlation this site-specific $BAF_{soil\text{-}to\text{-}earthworm}$ was not used in the food web models

It is important to note that, as stated previously, there are very poor correlations between earthworm tissue concentrations and soil concentrations for antimony, beryllium, and zinc. In fact, beryllium was not detected in any of the earthworm tissue samples and antimony was only detected in half of the earthworm tissue samples.

These soil-to-earthworm BAFs were used in the terrestrial food web model to estimate the transfer of COPECs from surface soil to terrestrial invertebrates. Based on the sparsity of earthworm tissue data for antimony and beryllium and the poor correlation between earthworm tissue concentrations and soil concentrations for zinc, default soil-to-earthworm BAFs (USEPA, 1999) were assumed for these COPECs and used in the food web model. Site-specific soil-to-earthworm BAFs for copper and lead were used in the food web model.

6.1.5 Terrestrial Food Web Exposures

Two decision rules pertaining to terrestrial food web exposures to surface soil-related COPECs were summarized in Chapter 5. The first decision rule pertaining to terrestrial food web exposures is "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 BBGR Ranges." The second decision rule pertaining to terrestrial food web exposures is "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 omnivorous mammals or birds at the BBGR Ranges."

In order to assess the potential hazards to various terrestrial-based feeding guilds from soil-related constituents, a food web was constructed. Based on the fact that the COPECs in surface soil at the BBGR ranges (antimony, beryllium, copper, lead, and zinc) do not bioconcentrate or biomagnify appreciably through the food chain and do not accumulate appreciably in plant tissues (Kabata-Pendias and Pendias, 1992), the terrestrial ecological receptors with the potential for the greatest exposure to COPECs were determined to be invertivorous and omnivorous small mammals and birds. Herbivores were considered to have a lower exposure potential to COPECs

because the COPECs do not accumulate appreciably in plant tissues, the herbivores' main food source. Carnivores were determined to have lower exposure potential to COPECs because the COPECs do not biomagnify in the food chain and would not be expected to occur at elevated concentrations in prey animal tissues. Additionally, carnivores in general have larger home ranges which would tend to minimize their exposures to COPECs at the BBGR ranges. Likewise, piscivores were determined to have lower exposure potential to COPECs because COPECs were not identified in surface water at the BBGR ranges, the COPECs in sediment do not bioconcentrate or biomagnify in fish tissue to any appreciable extent, and fish are not readily found in the South Branch of Cane Creek or Ingram Creek tributaries at the BBGR ranges. Therefore, the terrestrial food web model focused on the protection of the terrestrial omnivorous and invertivorous feeding guilds present at the BBGR ranges.

Daily doses of COPECs for terrestrial invertivorous and omnivorous small mammals and birds were calculated using standard exposure algorithms. These algorithms incorporated species-specific natural history parameters (i.e., feeding rates, water ingestion rates, dietary composition, etc.) and also utilize site-specific area use factors (AUF). Additionally, site-specific soil-to-earthworm BAFs and soil-to-plant BAFs were used, where the data supported their derivation, to calculate COPEC concentrations in the invertebrate and plant portions of the food of the terrestrial invertivorous and omnivorous small mammals and birds. Literature-derived bioaccumulation factors were used to estimate COPEC concentrations in the terrestrial invertebrates and vegetation portions of the receptor species' diets where site-specific BAFs could not be calculated due to site-specific data constraints.

In order to calculate COPEC exposures, indicator species that represent the feeding guilds of interest were identified. For this risk assessment, the small terrestrial invertivorous mammal was represented by the shorttail shrew (*Blarina brevicauda*) and the terrestrial invertivorous bird was represented by the American woodcock (*Philohela minor*). The small terrestrial omnivorous mammal was represented by the white-footed mouse (*Peromyscus leucopus*) and the terrestrial omnivorous bird was represented by the American robin (*Turdus migratorius*). Natural history parameters for these indicator species (Table 6-2) were used in combination with site-specific exposure parameters to estimate exposures to terrestrial invertivorous and omnivorous small mammals and birds at the BBGR ranges.

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

$$TDD_{wildlife} = \left[\left(IR_{food} \times f_{worm} \times \left\{C_{soil} \times BAF_{worm}\right\}\right) + \left(IR_{food} \times f_{veg} \times \left\{C_{soil} \times BAF_{plant}\right\}\right) + \left(IR_{food} \times f_{soil} \times C_{soil}\right)\right] \times AUF$$

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_{soil}	=	concentration of COPEC in surface soil (mg/kg)
BAF_{worm}	=	soil-to-earthworm bioaccumulation factor (unitless)
f_{veg}	=	fraction of daily diet comprised of vegetation (percent)
BAF_{plant}	=	soil-to-plant bioaccumulation factor (unitless)
f_{soil}	=	fraction of daily diet comprised of soil (percent)
AUF	=	area use factor (fraction of site used by receptor species (percent).

Because portions of the receptor species' diets consist of vegetative material, COPEC concentrations in terrestrial plant matter needed to be estimated in order to calculate a total COPEC dose. The COPEC concentrations in terrestrial plant matter were estimated using the empirically-derived plant BAF value for lead and literature-based values for antimony, beryllium, copper, and zinc as described previously. These soil-to-plant BAFs were applied to the soil concentrations of COPECs to estimate concentrations of COPECs in terrestrial vegetative food material in the following manner:

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

where:

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

The COPEC concentrations in terrestrial invertebrate tissues were estimated using the empirically-derived soil-to-worm BAF values for copper and lead, and literature-based values for antimony, beryllium, and zinc as described previously. These soil-to-worm BAFs were 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);
$BAF_{soil\text{-}to\text{-}worm}$ = soil-to-worm bioaccumulation factor (unitless).

The soil ingestion rates for the receptor species are most often represented as a percentage of a receptor species' diet. 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-dry weight)
$Diet_{soil}$ = portion of diet that is soil (percent).

Most wildlife food ingestion rates are presented in units of mg/kg/day wet weight. In order to convert these ingestion rates to dry weight, the moisture content of the wildlife species' diets must be known or approximated. The moisture contents of the invertebrate and vegetative material in the receptor species' diets were referenced from the EPA's *Wildlife Exposure Factors Handbook* (USEPA, 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%	

		Percent Moisture	Weighted-Average Moisture Content
dAmerican 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 was simplified for modeling purposes but incorporates the major food types for the different feeding guilds. It was assumed that food intake for invertivores is comprised almost entirely of terrestrial invertebrates (i.e., earthworms). It was also assumed that omnivores consume both plant and animal material, a portion of which consists of terrestrial invertebrates.

The AUFs for each of the indicator species take into account the home range and habitat requirements for each species and the size of the contaminated areas and viable habitat at the BBGR ranges. The biological fate and transport properties that affect the exposure routes of the receptor species are summarized in Table 6-3 and the exposure point concentrations for the receptor species are summarized in Table 6-4.

The terrestrial food web model was executed with maximum and mean exposure point concentrations in order to estimate a range of potential terrestrial wildlife exposures. The food web model was executed with site-specific area use factors, site-specific soil-to-earthworm BAFs (where the data are sufficiently robust), and site-specific soil-to-plant BAFs (where the data are sufficiently robust). Default, literature-derived BAFs were utilized for COPECs whose supporting data do not support site-specific BAF development. Maximum and mean exposures for the receptor species in the terrestrial food web are summarized in Table 6-5 through Table 6-12. The avian and mammalian toxicity reference values used in the terrestrial food web model are presented in Tables 6-13 and 6-14, respectively.

Estimated maximum and mean exposures for the receptor species were compared to the toxicity reference values to estimate maximum and mean hazard quotients for each receptor species.

These calculated hazard quotients are presented in Tables 6-15 through 6-22 and are summarized below.

	Maximum Exposure		*Mean Exposure*	
	NOAEL HQ	**LOAEL HQ**	**NOAEL HQ**	**LOAEL HQ**
White-footed Mouse:				
Antimony	17.4	1.74	1.69	0.169
Beryllium	0.0746	0.00746	0.0118	0.00118
Copper	64.3	50.1	1.13	0.882
Lead	56.6	5.66	2.36	0.236
Zinc	0.947	0.0308	0.0408	0.00133
American Robin :				
Antimony	31	6.2	3.01	0.601
Beryllium	ND	ND	ND	ND
Copper	26.1	19.8	0.459	0.349
Lead	252	85.9	10.5	3.58
Zinc	0.17	0.017	0.00732	0.000732
Short-tail Shrew :				
Antimony	16.8	1.68	1.63	0.163
Beryllium	0.202	0.0202	0.0321	0.0321
Copper	60.5	47.1	1.06	0.829
Lead	137	13.7	5.7	0.57
Zinc	2.69	0.0876	0.116	0.00378
American Woodcock:				
Antimony	32.7	6.53	3.17	0.634
Beryllium	ND	ND	ND	ND
Copper	26.5	20.2	0.466	0.355
Lead	511	174	21.3	7.25
Zinc	0.387	0.0387	0.0167	0.00167

The results of the food web model indicate that there are potential risks to terrestrial receptors at the BBGR ranges from exposures to COPECs at these ranges through food web interactions. Namely, antimony, copper, and lead in surface soil have the potential to pose risks to all of the terrestrial receptors assessed via the food web model. The terrestrial food web model indicates that beryllium does not pose a risk to any terrestrial wildlife receptors at the BBGR ranges, and zinc is unlikely to pose any significant risk to terrestrial wildlife receptors at the BBGR ranges. The hazard quotients for terrestrial receptors based on maximum exposures are summarized in Table 6-23 and the hazard quotients for terrestrial receptors based on mean exposures are summarized in Table 6-24.

The terrestrial food web model was also executed in order to estimate soil concentrations of COPECs that would result in hazard quotients of one or lower. These estimated soil

concentrations (assuming all of the conservatism inherent in the food web model) based on an HQ equal to 1.0 are presented in Table 6-25 and are summarized below.

	Soil Conc. Based on NOAEL TRV and HQ = 1.0 (mg/kg)	**Soil Conc. Based on LOAEL TRV and HQ = 1.0** (mg/kg)
White-Footed Mouse :		
- antimony	4.98	49.8
- copper	234	300
- lead	413	4,131
- zinc	1,327	40,800
American Robin :		
- antimony	2.8	14
- copper	578	757
- lead	92.5	272
- zinc	7,390	73,900
Short-tailed Shrew :		
- antimony	5.15	51.5
- copper	249	319
- lead	171	1,705
- zinc	465	14,300
American Woodcock :		
- antimony	2.65	13.25
- copper	567	747
- lead	45.7	134
- zinc	3,230	32,400

Soil concentrations based on an HQ equal to 1.0 were not calculated for beryllium because the terrestrial food web model did not indicate any risk to terrestrial wildlife species due to food web exposures to beryllium.

6.2 Aquatic Ecosystems

Two assessment endpoints have been identified for the aquatic ecosystems at the BBGR ranges. These assessment endpoints are the following:

- Survival, growth, and reproduction of aquatic benthic invertebrates in South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR Ranges.

- Survival, growth, and reproduction of riparian invertivorous small mammals and birds at the BBGR Ranges.

Four decision rules have been defined pertaining to aquatic ecosystems at the BBGR ranges and have been designed to answer specific questions related to the aquatic ecosystem assessment endpoints. The results of the Site Investigation and subsequent risk characterization are described herein with respect to each of these decision rules.

6.2.1 Chironomid Survival and Growth

The decision rule pertaining to chironomid survival and growth states that "If COPECs within the sediments of South Branch of Cane Creek or Ingram Creek and their tributaries cause acute toxicity to the benthic invertebrate *Chironomus sp*., which is statistically greater than toxicity from reference sediments, then there is the potential for risk to emergent benthic invertebrates at the BBGR Ranges."

The results of the 10-day chironomid survival test show that survival was not adversely affected at any of the sample locations. None of the chironomid survival rates were significantly reduced compared to survival rates using soil from the reference location (Table 5-10).

Chironomid growth was found to be statistically reduced in two sediment samples compared to chironomid growth in sediment from a reference location. Mean dry weight after 10 days was 1.45 mg/organism in the laboratory control sample and 1.69 mg/organism in the reference sample. Mean dry weight among the site sediment samples ranged from 1.29 mg/organism in SY1004 (BERA-FTA-163-SD02) to 1.67 mg/organism in SY1014 (BBGR-BERA-SD01). Statistically significant differences in weight were found for site samples SY1002 (BERA-CWM-183-SD02) with mean dry weight of 1.50 mg/organism and SY1005 (BERA-HR-83Q-SD01) with mean dry weight of 1.43 mg/organism (Table 5-10). These differences, however, only represent 11 percent and 15 percent reductions in growth from the reference sample and may not be biologically significant.

The 10-day chironomid survival results showed no statistically significant differences between survival in site sediment samples and reference sediment samples. Therefore, the NOECs, based on chironomid survival, for the three sediment COPECs are defined as the highest concentrations tested, and the LOECs and AETs are simply listed as greater than the NOEC concentrations. The sediment values based on chironomid survival are presented in Table 6-26 and summarized below.

Sediment COPEC	***Chironomid Survival & Growth NOEC*** *(mg/kg)*	***Chironomid Survival & Growth LOEC*** *(mg/kg)*	***Chironomid Survival & Growth AET*** *(mg/kg)*
Copper	22.5	> 22.5	> 22.5
Lead	112	> 112	> 112
gamma-Chlordane	0.0091	> 0.0091	> 0.0091

Similar to the results for chironomid survival, results of the toxicity tests for chironomid growth showed no statistically and biologically significant differences between site samples and reference samples after 10 days of exposure. Therefore, the NOECs, AETs, and LOECs for

chironomid growth are the same as those for chironomid survival. These values are presented in Table 6-26 and summarized above. Chironomid survival as a function of COPEC concentrations in sediment are presented in Figure 6-23 for copper and Figure 6-24 for lead. The correlations between copper and lead in sediment and chironomid survival were weak (r^2 = 0.2984 for copper and r^2 = 0.0857 for lead). Chironomid growth as a function of COPEC concentrations in sediment are presented in Figure 6-25 for copper and Figure 6-26 for lead. The correlations between copper and lead in sediment and chironomid growth were weak (r^2 = 0.1537 for copper and r^2 = 0.1348 for lead).

Because none of the samples elicited significant impacts to chironomid survival or growth, the predictability of the established NOEC, LOEC, and AET could not be estimated.

6.2.2 Chironomid Bioaccumulation

The decision rule pertaining to accumulation of sediment-associated COPECs by benthic invertebrates is "If chironomids exposed to sediment from South Branch of Cane Creek or Ingram Creek and their tributaries 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."

Data collected as part of the Site Investigation indicate that copper and lead may accumulate in chironomid tissues. Since gamma-chlordane was only detected in one chironomid tissue sample very close to the detection limit, it was determined that accumulation of gamma-chlordane in chironomid tissues at the BBGR Ranges is negligible.

A summary of the COPEC concentrations in sediment samples and the corresponding chironomid tissue concentrations was presented in Table 5-11. In order to define the relationship between sediment concentrations and chironomid tissue concentrations of COPECs, the data were plotted for copper and lead and the line that fit the data best was drawn. The graphs depicting the sediment-to-chironomid BAFs for copper and lead are presented in Figure 6-27 for copper and Figure 6-28 for lead. These figures help relate the potential bioavailability of the COPECs in the sediment to the toxicity test results. Gamma-chlordane was detected in only three sediment samples and in only one chironomid tissue sample and the sample with a detected tissue concentration of gamma-chlordane had no gamma-chlordane detected in the sediment. Therefore, gamma-chlordane bioaccumulation is not presented as a figure. The correlation of copper and lead in sediment and corresponding concentrations in chironomid tissues was strongest for lead (r^2 = 0.7841). The correlation coefficient for copper in sediment and chironomid tissue was r^2 = 0.4399.

In order to define the site-specific relationship between sediment concentrations and chironomid tissue concentrations of COPECs, site-specific bioaccumulation factors were estimated by comparing the chironomid tissue concentrations to sediment concentrations of COPECs at each sampling location in the following manner.

$$BAF_{sed-to-chironomid} = \frac{C_{chironomid}}{C_{sediment}}$$

where:

$BAF_{sed\text{-}to\text{-}chironomid}$	=	sediment-to-chironomid bioaccumulation factor (unitless);
$C_{chironomid}$	=	COPEC concentration in chironomid tissue (mg/kg); and
$C_{sediment}$	=	COPEC concentration in sediment (mg/kg).

Site-specific BAFs were estimated for each sediment sample location and a mean sediment-to-chironomid BAF was then calculated for each sediment COPEC. The sediment-to-chironomid BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-11:

Sediment COPEC	Sediment-to-Chironomid BAF
Copper	11.81*
Lead	0.67*

* Due to poor correlation this site-specific $BAF_{sed\text{-}to\text{-}chironomid}$ was not used in the food web models.

It is important to note that, as stated previously, there are very poor correlations between chironomid tissue concentrations and sediment concentrations for copper and lead, and gamma-chlordane was only detected in one chironomid tissue sample. Based on the poor correlation between chironomid tissue concentrations and sediment concentrations for copper and lead, and the sparsity of data for gamma-chlordane, default sediment-to-invertebrate BAFs (USEPA, 1999) were assumed for these COPECs and used in the riparian food web model to estimate the transfer of COPECs from sediment to riparian invertivores.

6.2.3 Benthic Invertebrate Community Assemblages

The decision rule pertaining to benthic invertebrate community assemblages, as stated in the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw 2006), is as follows: "If the benthic community assemblage in South Branch of Cane Creek or Ingram Creek and their tributaries at the BBGR Ranges is significantly different than the benthic community assemblage in a non-impacted reference stream, then there

is the potential for risk to South Branch of Cane Creek or Ingram Creek benthic ecosystem." As presented in the previous chapter, benthic invertebrate samples were collected from 12 locations in the South Branch of Cane Creek, Ingram Creek, and their tributaries within the BBGR ranges and one reference station, for a total of 13 benthic invertebrate sampling locations. A riffle/run sample and a course particulate organic matter (CPOM) sample were collected from each sampling location.

Using the raw benthic data, a numerical value was calculated for the eight metrics (measures of benthic macroinvertebrate community quality) described previously. Calculated values were then compared to values derived from the reference site. Each metric was then assigned a score of 6, 3 or 0 according to the comparability (percent similarity) of calculated and reference values (Plafkin et al., 1989). Scores for the eight metrics were then totaled and compared to the total metric score for the reference location. The percent comparison between the total scores provides the evaluation of biological condition, and places each site into one of three categories as being either non-impaired, moderately impaired or severely impaired in relation to the reference location. The inclusion of an integrated multi-metric approach to benthic habitat integrity incorporates many components of benthic community structure, and provides a more reliable assessment than obtainable with a single metric analysis.

A summary of the benthic macroinvertebrates which were collected during this survey was shown in Table 5-13. The values for the individual metrics, percent comparison to the reference location, and the habitat quality scores are presented in Table 5-14. The metrics and their significance are described below. During sample collection, observations were made and recorded regarding the specific physical habitat parameters at each sample location in accordance with the visual-based habitat assessment guidelines provided in Barbour, et al. (1999). The scores assigned to each sample location are presented in Table 5-12. These physical habitat parameters and the assigned scores were used to aid in the interpretation of the RBP scores calculated for each of the sample locations associated with the BBGR Ranges.

6.2.3.1 Taxa Richness

Taxa richness was calculated by counting the number of taxa present in each sample. Taxa richness is an unambiguous and easily understandable indication of the biological health of a stream community, and reflects the health of the community through a measurement of the variety of taxa present. In general, taxa richness increases with increasing water quality, habitat diversity, and/or habitat suitability. Taxa richness at the BBGR sites ranged from 3 at sample location FTA-163-SD02 to 17 taxa at BBGR-BERA-SD03 and CWM-183-SD02. The reference site BBGR-BERA-SED-REF also had 17 taxa.

Although, as stated above, two locations had the same number of taxa as the reference location, the majority of the BBGR sample locations had fewer taxa, thereby indicating that the habitat quality at these locations may be lower than the reference location.

6.2.3.2 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 Bode and Novak (1995) and Hilsenhoff (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.

Values for the FBI range from 0 to 10 with higher values corresponding to greater levels of organic pollution in seven categories ranging from excellent water quality to very poor water quality as shown in tabular form in Section 5.6. The family biotic index (FBI) at the BBGR Ranges sites ranged from 4.524, indicating good water quality, at sampling location BBGR-BERA-SD01, to 7.04, indicating fairly poor water quality, at sample location BBGR-BERA-SD02. The FBI score at the reference site was 4.176, which was in the range indicating very good water quality.

6.2.3.3 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. The proportion of the two feeding groups is important because predominance of a particular feeding type may indicate an unbalanced community responding to an overabundance of a particular food source. Functional feeding group designations for the taxa identified were obtained from Bode and Novak (1995) and Merritt and Cummins (1984).

This metric is calculated by dividing the relative abundance of scrapers by the relative abundance of filter feeding organisms. In general, the ratio increases with increasing water quality. The ratio of scrapers and filterers was not calculable at two sample locations, FTA-163-SD02 and HR-79Q-SD01 because no organisms of the filtering collector feeding group were present in those samples. The values calculated for the remaining BBGR locations ranged from 0.333 at location HR-83Q-SD01, to 13 at BBGR-BERA-SD01 and HR-79Q-SD05. The ratio at the reference site was 0.508.

6.2.3.4 Ratio of EPT and Chironomidae Abundances

The ratio of Ephemeroptera, Plecoptera and Trichoptera (EPT) and Chironomidae abundance is calculated by dividing the abundance of EPT taxa by the abundance of Chironomidae. The ratio of EPT to Chironomidae indicates the relative representation of the pollution sensitive EPT taxa and more pollution tolerant Chironomidae. High quality benthic habitat condition is indicated by communities having a fairly even distribution among several major taxa and with substantial representation among the sensitive taxa. Skewed invertebrate communities having an abundance of the generally tolerant Chironomidae and few or none of the more sensitive insect groups may indicate environmentally stressed conditions and poor habitat quality. Chironomids also have been found to increase in relative abundance with increasing heavy metals concentration (Ferrington, 1987, Hayford and Ferrington, 2005), and are therefore an appropriate indicator species for the BBRG ranges evaluation.

The EPT/Chironomidae ratio was not calculable at location HR-74Q-SD01, because no Chironomidae were found in the sample, and the ratio was zero at FTA-163-SD02 because there were no EPT taxa in sample. The ratio of EPT and Chironomidae at the eight sample locations that had Ephemeroptera, Plecoptera, Trichoptera and Chironomidae ranged from 0.029 at HR-79Q-SD03 to 15 at HR-79Q-SD01. The EPT/Chironomidae ratio at the reference location was calculated to be 18.4.

6.2.3.5 Percent Contribution of Dominant Taxon

Somewhat related to the previous metric is the percent contribution of the numerically dominant taxon to the total number of organisms. This metric is an indication of community balance at the lowest positive taxonomic level. A community dominated by relatively few species likely indicates stressed conditions and low quality habitat. Conversely, a low percent contribution of the dominant family indicates a balanced community and generally indicates a higher quality habitat.

The percent contribution of the dominant taxon at the BBGR sites ranged from 76 at sample location FTA-163-SD02 to 18 at location BBGR-BERA-SD04. The reference location had a 30 percent contribution of the dominant taxon. The numerically dominant taxon at FTA-163-SD02 was oligochaete worms. (Only three taxa were found in the sample from this location, the lowest of the BBGR sample locations.). Taxa that dominated at other locations included crayfish (HR-83Q-SD01), Pleurocerid snails (HR-79Q-SD05), Chironomini midges (HR-79Q-SD03) and netspinner caddisflies (CWM-183-SD02).

6.2.3.6 EPT Index

The EPT Index is the total number of distinct taxa within the orders Ephemeroptera, Plecoptera, and Trichoptera. This value summarizes taxa richness within three insect orders that are generally considered to be pollution sensitive. In general, the EPT index increases with increasing water quality.

The EPT index of the BBGR sites ranged from 0 at sample location FTA-163-SD02 (no EPT taxa present), to 7 at location CWM-183-SD02. An EPT index of 7 was also found for the reference location.

6.2.3.7 Community Loss Index

This index evaluates the benthic populations at on-site locations relative to populations present at the reference location. The Community Loss Index measures the loss of benthic species between a reference station and the location of comparison. It is an index of dissimilarity with values increasing as the degree of dissimilarity from the reference station increases. The community loss index is calculated by subtracting the number of taxa common to both locations from the number of taxa present at the reference location, divided by the number of taxa present at the potential impact location.

A lower index value corresponds to a smaller difference from the reference location, and therefore indicates higher water quality. The community loss index for the BBGR sites ranged from 5.33 at sample location FTA-163-SD02 to 0.412 at location CWM-183-SD02

6.2.3.8 Ratio of Shredder Functional Feeding Group and Total Number of Individuals Collected

The abundance of the shredder functional group relative to the abundance of all other functional groups allows evaluation of potential impairment as indicated by the CPOM-based shredder community. Shredders are sensitive to riparian zone impacts and are good indicators of toxic effects, particularly when the toxicants are readily absorbed to CPOM and affect the microbial communities colonizing the CPOM and/or the shredders directly (Plafkin, et al, 1989). The ratio of the relative abundance of shredders to the abundance of all other functional feeding groups is calculated by dividing the relative abundance of shredders by the total number of organisms in a given sample.

No shredder feeding organisms were found at locations BBGR-BERA-SD01, BBGR-BERA-SD04, BBGR-BERA-SD05, HR-79Q-SD03, and HR-83Q-SD01. These locations therefore had shredder abundance to total organism abundance ratios of zero. The highest ratio of shredders at

the BBGR sites was 0.395 found at location HR-79Q-SD01, while the reference location had a ratio of 0.632.

6.2.3.9 Total Metric Scores and Bioassessment

The percent comparison to reference for all eight metrics is presented in Table 5-14. The percentages are used to assess the benthic invertebrate habitat quality at each sampling location in relation to the reference location, as shown in the following table, (adapted from Plafkin, et al., 1989).

Percent Comparability to Reference Score[a]	Relative Benthic Invertebrate Community Assessment	Attributes
Greater Than 79 Percent	Non-impaired	Comparable to the best benthic invertebrate community to be expected for stream size and habitat quality
29 – 72 Percent	Moderately Impaired	Fewer species due to loss of intolerant taxa.
Less Than 21 Percent	Severely Impaired	Few taxa present. May have high numbers of individuals of tolerant taxa.

[a] Percentage values that are intermediate to the table ranges are interpreted using best professional judgment.

The benthic invertebrate communities at ten of the twelve BBGR Ranges sampling locations were evaluated as being moderately impaired relative to the reference location, with comparability percentages ranging from 33% at locations HR-79Q-SD03 and HR-83Q-SD01, to 67% at BBGR-BERA-SD03 (Table 5-14). One location, FTA-163-SD02, was evaluated as severely impaired, with a comparison to reference percentage of 20%, and one location, CWM-183-SD02, was determined to be non-impaired in relation to the reference location, with an 80% comparison to reference (Table 5-14).

Three analytes, copper, lead and gamma-chlordane were identified as COPECs in sediment at the BBGR Ranges. Sediment analytical results for all sampling locations were reviewed for concentrations of COPECs that may have affected the benthic community assemblages in creeks in the vicinity of the BBGR Ranges. Sediment COPEC concentrations and benthic invertebrate metric scores were presented in Table 5-15. Values that are shaded and with an asterisk on Table 5-15 exceeded the ecological screening values and/or the background threshold values. Six of the twelve BBGR locations had COPEC concentrations that exceeded at least one ecological screening or background threshold value, and these locations were determined to be moderately impaired in relation to the reference location. However, four other locations deemed to be moderately impaired did not have COPEC concentrations that exceeded screening or background

values. In addition, location FTA-163-SD02, the only sample location determined to be severely impaired in comparison to the reference location, had COPEC concentrations that were among the lowest of the twelve BBGR sampling locations. Figures 6-29 through 6-31 present the RBP scores as a function of sediment concentrations of copper, lead, and gamma-chlordane, respectively. No significant correlation between biological condition and sediment COPEC concentration are indicated for any of the three sediment COPECs (r^2 = 0.0349, 0.0766 and 0.0036 for copper, lead, and gamma-chlordane, respectively). Therefore, the data do not indicate that site-related COPECs are significantly impacting the benthic communities at sites associated with the BBGR Ranges.

A possible explanation for the differences in benthic community quality is the natural physical habitat characteristics at the various sampling locations. During the collection of benthic invertebrate samples, visual observations were made and recorded for ten habitat parameters using field data sheets based on those provided in Barbour, et al. (1999). The parameters were assigned numerical scores within four categories indicating optimal, sub-optimal, marginal and poor conditions. The habitat scores for the twelve BBGR locations and the reference location are shown in Table 5-12. While none of the sample locations were found to have poor or marginal conditions, the average habitat score for nine of the sampling locations was in the range indicating sub-optimal benthic habitat. Three BBGR locations and the reference location had scores in the range indicating optimal habitat. Figure 6-32 presents the RBP scores as a function of the habitat quality scores. This comparison shows that the quality of the benthic community as represented by the RBP scores is fairly well correlated with the physical quality of the habitat (r^2 = 0.4008). The results of the benthic community analysis indicate that the quality of the benthic invertebrate communities at locations associated with the BBGR ranges is more dependant upon physical habitat parameters than the presence of site-related sediment COPECs.

6.2.4 Riparian Food Web Exposures

The decision rule pertaining to riparian food web exposures to sediment-related COPECs was summarized in Chapter 5 and is the following: “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 BBGR Ranges.”

In order to assess the potential hazards to various riparian-based feeding guilds from sediment-related constituents, a riparian food web was constructed. Riparian invertivorous mammals and birds were identified as having significant potential for exposure to COPECs at the BBGR 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 (i.e., 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 BBGR ranges.

Daily doses of COPECs for riparian invertivorous mammals and birds were calculated using standard exposure algorithms. These algorithms incorporated species-specific natural history parameters (i.e., feeding rates, water ingestion rates, dietary composition, etc.) and also utilize site-specific area use factors (AUF). Although site-specific sediment-to-chironomid BAFs were derived for copper and lead as described previously, the correlation coefficients for these relationships were very low; thus, the reliability of these BAFs were very low. In order to maintain a conservative assessment, default sediment-to-invertebrate BAFs referenced from the literature were used to estimate COPEC concentrations in aquatic invertebrates.

In order to calculate COPEC exposures, indicator species that represent the feeding guilds of interest were identified. For this risk assessment, the riparian invertivorous mammal was represented by the little brown bat (*Myotis lucifugus*) and the riparian invertivorous bird was represented by the marsh wren (*Cistothorus palustris*). Natural history parameters for these indicator species (Table 6-27) were used in combination with site-specific exposure parameters to estimate exposures to riparian invertivorous mammals and birds at the BBGR ranges.

The algorithm that was used to estimate exposures to COPECs by riparian invertivorous mammals and birds is the following:

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

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);
BAF_{invert}	=	sediment-to-benthic invertebrate bioaccumulation factor (unitless);
AUF	=	area use factor (fraction of site used by receptor species) (percent).

It was assumed that the receptor species' diets consist entirely of emergent benthic invertebrates; therefore, it was necessary to estimate or measure COPEC concentrations in benthic invertebrate tissues. For this assessment, the COPEC concentrations in benthic invertebrate tissue were calculated using literature-derived sediment-to-invertebrate BAFs described previously. The total daily doses of COPECs received by the riparian invertivorous mammals and birds did 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. Since no COPECs were identified in surface water at the BBGR Ranges, exposure to surface water was not considered a complete exposure pathway.

Most wildlife food ingestion rates are presented in units of mg/kg/day wet weight. In order to convert these ingestion rates to dry weight, the moisture content of the wildlife species' diets must be known or approximated. The moisture contents of the aquatic invertebrates that make up the entirety of the riparian receptor species' diets were assumed to be 83 percent and were referenced from the EPA's *Wildlife Exposure Factors Handbook* (USEPA, 1993).

The AUFs for each of the indicator species take into account the home range and habitat requirements for each species and the size of the contaminated areas and viable habitat at the BBGR ranges. The biological fate and transport properties used to estimate the exposure point concentrations of COPECs and the calculated exposure point concentrations of COPECs are presented in Tables 6-28 and 6-29, respectively.

The riparian food web model was executed with site-specific sediment concentrations, site-specific area use factors, literature-derived sediment-to-invertebrate BAFs, and literature-derived values for the other input variables. The model was executed with both maximum and mean detected concentrations as the exposure point concentrations for sediment in order to calculate a range of potential risks to riparian receptors. The estimated maximum and mean doses received by the riparian receptor species are presented in Tables 6-30 through 6-33. The avian and mammalian toxicity reference values used to assess the toxicity of the COPECs are presented in Tables 6-34 and 6-35, respectively.

Estimated maximum and mean exposures for the receptor species were compared to the toxicity reference values to estimate maximum and mean hazard quotients for each receptor species

(Tables 6-36 through 6-39). The results of the model indicate that there is a potential risk to riparian invertivorous mammals and riparian invertivorous birds from exposure to lead in sediment. The calculated hazard quotients for the sediment COPECs (copper, lead, and gamma-chlordane) are summarized below.

Sediment COPEC	*Little Brown Bat*		*Marsh Wren*	
	NOAEL HQ	***LOAEL HQ***	***NOAEL HQ***	***LOAEL HQ***
Copper	1.87E-01	1.45E-01	1.51E-01	1.15E-01
Lead	1.42E+00	1.42E-01	9.35E+00	3.19E+00
Gamma-Chlordane	1.10E-03	5.48E-04	7.45E-03	1.49E-03

Lead was the only COPEC in sediment that indicated the potential for adverse affects from exposure to even the maximum detected concentrations. Tables 6-36 through 6-39 present the estimated hazard quotients for the riparian receptor species assuming maximum and mean exposures. Summaries of the estimated maximum and mean hazard quotients are presented in Tables 6-40 and 6-41, respectively.

The riparian food web model was also executed to determine the sediment concentrations of COPECs that would result in hazard quotients of one or lower. The estimated sediment concentrations based on HQs equal to 1.0 are presented in Table 6-42 and are summarized below.

Riparian Receptor Species	**Sediment Conc. Based on NOAEL TRV and HQ = 1.0** (mg/kg)	**Sediment Conc. Based on LOAEL TRV and HQ = 1.0** (mg/kg)
Little Brown Bat :		
Lead	375	3,750
Marsh Wren :		
Lead	57	168

Sediment concentrations based on an HQ = 1.0 were not calculated for copper or gamma-chlordane because the riparian food web model did not indicate any risk to riparian wildlife species due to food web exposures to copper or gamma-chlordane.

7.0 Uncertainty Analysis

There are numerous uncertainties that are inherent in the ecological risk assessment process. Although this BERA was designed to reduce the level of uncertainty that was present in the SLERA and employed documented ecological risk assessment methodologies and procedures, the uncertainties that remain were nevertheless unavoidable. Risk managers should consider the collective uncertainty treatment presented below when contemplating the need for remedial options. The following sections describe the uncertainties present in the various lines of evidence utilized in this baseline ecological risk assessment.

7.1 Uncertainties in Earthworm Survival and Growth Tests

There are several sources of uncertainty that are evident in the earthworm survival and growth tests. Because all of the on-site sample results were compared to the results from a reference location, it was critical to identify a representative reference site. Every effort was made to identify a reference location that was representative of the soil type most prevalent at the BBGR Ranges and from areas thought to be unaffected by Army activities. However, subtle differences in the soil chemistry or physical make-up may be present between the on-site soils and reference soil that could affect the toxicity of the soils or the ability of earthworms to accumulate chemicals from the soil. Also, although statistically significant differences in test response may be noted between organisms exposed to site soil and reference site soil, it is unknown how much of a difference is considered ecologically relevant (Tannenbaum, 2001).

It is also important to note that the earthworm species used in the toxicity tests (*Eisenia fetida*) may not be endemic to the BBGR ranges. Endemic terrestrial invertebrates may be more or less sensitive to the soil COPECs found at the BBGR ranges than the species that was tested. Since no characterization/speciation of the endemic terrestrial invertebrate community at the BBGR ranges has been completed, the relative sensitivity of the test species to endemic species is unknown, and consequently, the results of the earthworm toxicity tests should be interpreted with caution.

Earthworms used in the toxicity testing did not have prior exposure to the soil COPECs (unlike native organisms). Thus they represent a departure from actual site receptors that may have lived amid site contaminants for a significant period of time, and may have acclimated to the existing site conditions. Additionally, the toxicological responses of commercially available test organisms may not accurately mimic the responses of site organisms.

One of the strengths of earthworm toxicity tests is that the tests directly relate earthworm toxicity to on-site soil. However, the main weakness of the earthworm toxicity test is that it does not identify the toxicant(s) that causes the adverse effect. In the specific case of the earthworm survival and growth tests run for the BBGR Ranges, no statistically significant affects were observed in any of the tests. Therefore; no earthworm toxicants were identified.

7.2 Uncertainties in Earthworm Bioaccumulation Test

Many of the uncertainties related to the earthworm survival and growth tests are also inherent in the earthworm bioaccumulation test. One source of uncertainty in the earthworm bioaccumulation test is the duration of the test (28 days). Although the 28-day exposure period is the standard test duration, earthworm tissues may not reach equilibrium with the surrounding soils in 28 days. This uncertainty could serve to underestimate earthworm tissue concentrations and subsequent bioaccumulation potential.

Another source of uncertainty is in the assumption that a single species of earthworms (*Eisenia fetida*) adequately represents all terrestrial invertebrates present at the BBGR Ranges. This assumption is an obvious over-simplification of a complex and diverse ecosystem. The terrestrial invertebrates that naturally occur at the BBGR Ranges may accumulate soil COPECs to a greater or lesser extent than the earthworms tested in this BERA.

As was the case with the earthworm toxicity tests, earthworms used in the bioaccumulation testing did not have prior exposure to the soil COPECs (unlike native organisms). Thus they represent a departure from actual site receptors that may have lived amid site contaminants for a significant period of time, and may have acclimated to the existing site conditions. The acclimatization process that may occur in endemic terrestrial invertebrates may result in higher or lower bioaccumulation of soil COPECs.

The presence of other chemicals in the soil and the physical characteristics of the soil may serve to enhance or inhibit the bioaccumulation of soil COPECs by earthworms. This uncertainty could serve to either over- or under-estimate the bioaccumulation potential of certain soil COPECs.

7.3 Uncertainties in the Perennial Rye Grass Germination and Growth Tests

There are several uncertainties inherent in the performance and interpretation of the perennial rye grass toxicity tests. The use of a single plant species (perennial rye grass, *Lolium perenne*) as a surrogate for all terrestrial plants potentially growing at the BBGR Ranges is a gross simplification of the diverse natural plant ecosystems present at FTMC. There are hundreds of

terrestrial plant species growing at the BBGR Ranges and these naturally occurring terrestrial plants may be more or less tolerant to the soil COPECs than the plant species used in the toxicity tests (perennial rye grass).

Because all of the on-site sample results were compared to the results from a reference location, it was critical to identify a representative reference site. Every effort was made to identify a reference location that was representative of the soil type most prevalent at the BBGR Ranges and from areas thought to be unaffected by Army activities. However, subtle differences in the soil chemistry or physical make-up may be present between the on-site soils and reference soil that could affect the toxicity of the soils to perennial rye grass.

Also, although statistically significant differences in test response may be noted between plants grown in site soil and reference site soil, it is unknown how much of a difference is considered ecologically relevant (Tannenbaum, 2001).

It is also important to note that the plant species used in the toxicity tests (*Lolium perenne*) may not be endemic to the BBGR ranges. Endemic terrestrial plants may be more or less sensitive to the soil COPECs found at the BBGR ranges.

Perennial ryegrass seeds used in the toxicity testing did not have prior exposure to the soil COPECs (unlike native plants). Thus they represent a departure from actual site plants that may have lived amid site contaminants for decades, and may have acclimated to the existing site conditions. Additionally, the toxicological responses of commercially available perennial ryegrass seeds may not accurately mimic the responses of site plants.

One of the strengths of plant toxicity tests is that the tests directly relate plant toxicity to on-site soil. However, the main weakness of the plant toxicity test is that it does not identify the toxicant(s) that causes the adverse effect. Although the plant toxicity tests identified copper, lead, and zinc as the most likely toxicants that induced the observed adverse effects in perennial rye grass growth, there may be other chemicals or physical characteristics of the soil itself that may also contribute to the observed adverse effects (reduced growth). This is especially true because the correlation coefficients for COPEC concentrations in soil and reduced growth were weak for all of the COPECs measured. The weak correlation coefficients for all of the soil COPECs indicates that other factors (non-COPEC related) may be inducing the observed adverse effects.

7.4 Uncertainties in the Terrestrial Plant Bioaccumulation Test

Many of the uncertainties related to the terrestrial plant germination and growth tests are also inherent in the plant bioaccumulation test. A significant level of uncertainty in the plant bioaccumulation test is the fact that terrestrial plant tissues used for the accumulation measurements were collected in the field co-located with the soil samples used in the toxicity tests. Because these plants were grown in the wild without laboratory controls, there is the possibility of significant natural variability in their COPEC accumulation potential. However, collected plant tissues grown on-site gives a more accurate measure of the potential for COPEC accumulation given the environmental conditions that exist at the BBGR Ranges compared to the controlled conditions that exist in a laboratory.

Another source of uncertainty is the fact that perennial grasses were collected for tissue measurement of COPECs. The fact that these grasses grow for several months and then die-off during the winter months only allows for accumulation of COPECs during the plant's growing season. A longer-lived plant might allow for more accumulation of COPECs from the soil simply to the exposure duration.

The presence of other chemicals in the soil and the physical characteristics of the soil may serve to enhance or inhibit the bioaccumulation of soil COPECs by terrestrial plants. This uncertainty could serve to either over- or under-estimate the bioaccumulation potential of certain soil COPECs.

Additionally, only the above-ground portion of the plants was sampled for chemical analysis. The above-ground portion of the plants was sampled because most terrestrial species are likely to consume only the above-ground portion of the plants. Some terrestrial species, however, are known to eat the roots and other below-ground portions of plants. The COPEC accumulation in these below-ground portions of the plant is unknown.

7.5 Uncertainties in Terrestrial Food Web Model

There are a number of uncertainties in food web models in general and specifically the terrestrial food web model used in this risk assessment. Uncertainty is introduced into the terrestrial food web model by using a single surrogate species to represent an entire feeding guild. Different species and individuals within a given species exhibit different feeding habits and susceptibilities to stressors. Likewise, the use of a single value to represent each exposure parameter (e.g., food ingestion rate, body weight, foraging area, etc.) is an over-simplification of a natural system with a great deal of natural variability in order to establish a useable model. Most of the receptor-specific input parameters are the mean of several values presented in the literature. Additionally,

the terrestrial food web model utilized simplified feeding preferences in order to simplify the modeling process. Simplified transfer rates were used to estimate the transfer of contaminants from abiotic media (e.g., soil, surface water, and sediment) to food items for the receptors of interest. These transfer rates introduce an unknown level of uncertainty into the terrestrial food web model.

A significant level of uncertainty is inherent in the food web models due to the fact that they assess chemicals individually and not as a diverse mixture of chemicals. Most receptors are continuously integrating their exposures to multiple chemicals over time and space. The current state-of-the-science of risk assessment does not provide proven methodologies for integrating exposures temporally or spatially.

The mammalian and avian toxicity reference values used to assess toxicity were based on literature values derived in laboratory toxicity tests using different species than the receptor species and also using a single toxicant, as opposed to a number of co-occurring toxicants as occurs at FTMC. Laboratory animals respond differently to toxicants than do wild animals and environmental conditions at the site may act to attenuate or exacerbate the toxicity of certain stressors to wild animals. The toxicity reference values used in the terrestrial food web were derived from no-observed-adverse-effects-levels (NOAEL); therefore, they represent doses at which no adverse effects occur in the test animals. NOAEL-based TRVs are the most protective values that could be used and result in very conservative risk estimates. Uncertainty factors are also used in the derivation of the NOAEL-based TRVs, introducing more uncertainty to these values.

The terrestrial food web model used in this risk assessment did not take into account the potential contribution of dermal absorption or inhalation to the total dose of contaminant received by a receptor. The relative contribution of contaminant dose from inhalation and dermal absorption is expected to be insignificant compared to the dose received through ingestion, particularly with respect to the COPECs (inorganics) assessed in this risk assessment. However, there is a certain level of uncertainty introduced into the assessment by not quantifying these potential exposures.

A significant level of uncertainty associated with the food web models concerns the use of hazard quotients (HQ) to express potential ecological hazards. There are a number of limitations to the use of HQs that limit their ability to determine the presence of potential ecological hazards. These limitations include: the HQ does not express risk, but expresses instead “hazard”; the HQ is not population-based; the HQ is not linearly scaled; naturally-occurring background concentrations of inorganic compounds may produce “unacceptable” HQs; HQs are often

calculated that are unrealistically high and toxicologically impossible; and HQs cannot adequately address temporal scales (Tannenbaum, et al., 2003).

7.6 Uncertainties in Chironomid Survival and Growth Tests

A number of uncertainties are inherent in the chrionomid survival and growth tests. Because all of the on-site sample results were compared to the results from a reference stream, it was critical to identify a representative reference site. For the chironomid tests, a reference sample was collected from a tributary to Choccolocco Creek that flows in an easterly direction along Bains Gap Road, immediately east of the Bains Gap gate to Main Post (the same reference location used for the benthic community analyses). Every effort was made to identify a reference location that was similar to the on-site sampling locations with respect to stream gradient, substrate, and surrounding habitat. However, subtle differences in the surface water or sediment chemistry or physical make-up of the substrate may be present between the on-site sample locations and reference location that could affect the toxicity of the sediment to chironomids.

One of the strengths of chrionomid toxicity tests is that the tests directly relate chironomid toxicity to on-site sediment. However, the main weakness of the chironomid toxicity test is that it does not identify the toxicant(s) that causes the adverse effect. Although the chironomid toxicity tests identified lead and copper as the most likely toxicants that induced the observed adverse effects in chironomids, there may be other chemicals or physical characteristics of the sediment itself that may also contribute to the observed adverse effects (reduced survival and/or growth). As indicated by the benthic invertebrate habitat assessment, the adverse affects noted in the benthic invertebrate community structure are more closely associated with physical habitat constraints than COPEC concentrations in the sediment.

Also, although statistically significant differences in test response may be noted between organisms exposed to site sediment and reference site sediment, in it unknown how much of a difference is considered ecologically relevant (Tannenbaum, 2001).

It is also important to note that the benthic invertebrate species used in the toxicity tests (*Chironomus tentans*) may not be endemic to the streams at the BBGR ranges. Endemic benthic invertebrates may be more or less sensitive to the sediment COPECs found at the BBGR ranges.

Chironomids used in the toxicity testing did not have prior exposure to the sediment COPECs (unlike native organisms). Thus they represent a departure from actual site receptors that may have lived amid site contaminants for long periods of time, and may have acclimated to the

existing site conditions. Additionally, the toxicological responses of commercially available test organisms may not accurately mimic the responses of site organisms.

7.7 Uncertainties in Chironomid Bioaccumulation Test

There are several uncertainties that are evident in the chironomid bioaccumulation test. Based on the small size of individual chironomids and the volume requirements of the analytical laboratory, a large number of chironomids were required to conduct the required chemical analyses on the chironomid tissues. These volume requirements necessitated the use of non-standard sample volumes and test chambers. These variances from the standard testing protocols may have introduced some uncertainty into the results of the chironomid bioaccumulation test.

As was the case with the chironomid toxicity tests, chironomids used in the bioaccumulation testing did not have prior exposure to the sediment COPECs (unlike native organisms). Thus they represent a departure from actual site receptors that may have lived amid site contaminants for a significant period of time, and may have acclimated to the existing site conditions. The acclimatization process that may occur in endemic benthic invertebrates may result in higher or lower bioaccumulation of sediment COPECs.

Another source of uncertainty in the chironomid bioaccumulation test is the duration of the test (21 days). Although the 21-day exposure period is approximately the maximum time the chironomid egg cases can remain in the sediment prior to adult emergence, the chironomid tissues may not have reached equilibrium with the surrounding sediments in 21 days. This uncertainty could serve to underestimate chironomid tissue concentrations and subsequently the bioaccumulation potential of the sediment COPECs.

7.8 Uncertainties in Benthic Invertebrate Community Analysis

There are a number of uncertainties inherent in assessing benthic invertebrate community structure using the RBP techniques. The RBP assessment provides a “snapshot” of the benthic community structure at the time of sampling. However, it does not provide an assessment of the benthos over time. As such, the assessment is subject to the environmental conditions at the time of sampling or just prior to sampling. For instance, the small tributaries of the South Branch of Cane Creek and Ingram Creek in the vicinity of the BBGR Ranges exhibited very low flow conditions at the time of sampling (October 2006). In fact, several of the proposed sediment and benthic invertebrate sampling locations could not be sampled due to the absence of water. These low flow conditions could affect the benthic assemblages present in the sediment as different benthos could be present during higher flow regimes.

Another source of uncertainty in the RBP assessment is the fact that several of the metrics used to quantitatively assess the community structure are based on the assessment of organic pollutants. Since most of the COPECs in sediment at the BBGR Ranges are metals, some of the metrics used in the RBP technique may not be applicable.

Another source of uncertainty is the fact that the RBP technique identifies areas of impact regardless of the stressor that causes the impact. The RBP technique cannot differentiate between chemical, biological, physical, or other stressors.

Another area of uncertainty with regards to the benthic community analysis is the habitat quality in the areas surrounding the sampling locations. While the assessment of the key physical characteristics of the water body and surrounding land is critical to the overall evaluation of impairment, these characteristics can only be evaluated qualitatively, and as such, contain an unknown level of uncertainty. This fact is especially true when comparing on-site stream characteristics to a reference stream. Although a thorough effort was made to identify and sample a reference location from a stream with similar physical characteristics as the South Branch of Cane Creek and Ingram Creek at the BBGR Ranges, subtle differences in the physical makeup of the stream or surrounding habitat may cause differences in the benthic invertebrate communities.

7.9 Uncertainties in Riparian Food Web Model

There are a number of uncertainties in food web models in general and specifically the riparian food web model used in this risk assessment. Uncertainty is introduced into the riparian food web model by using a single surrogate species to represent an entire feeding guild. Different species and individuals within a given species exhibit different feeding habits and susceptibilities to stressors. Likewise, the use of a single value to represent each exposure parameter (e.g., food ingestion rate, body weight, foraging area, etc.) is an over-simplification of a natural system with a great deal of natural variability in order to establish a useable model. Most of the receptor-specific input parameters are the mean of several values presented in the literature. Additionally, the riparian food web model utilized simplified feeding preferences in order to simplify the modeling process. In the case of the riparian food web model, the receptors were assumed to eat only emergent benthic invertebrates. Simplified transfer rates were used to estimate the transfer of contaminants from abiotic media (e.g., surface water and sediment) to food items for the receptors of interest. These transfer rates introduce an unknown level of uncertainty into the riparian food web model. Although site-specific sediment-to-invertebrate BAFs were estimated using site-specific sediment and chironomid data, the correlation coefficients were too low to make reliable bioaccumulation estimates. Therefore, these site-specific data were not used in the

riparian food web model. The use of literature-based sediment-to-invertebrate BAFs introduced significant uncertainty to this important COPEC fate process.

A significant level of uncertainty is inherent in the food web models due to the fact that they assess chemicals individually and not as a diverse mixture of chemicals. Most receptors are continuously integrating their exposures to multiple chemicals over time and space. The current state-of-the-science of risk assessment does not provide proven methodologies for integrating ecological exposures temporally or spatially.

The mammalian and avian toxicity reference values used to assess toxicity were based on literature values derived in laboratory toxicity tests using different species than the receptor species and also using a single toxicant, as opposed to a number of co-occurring toxicants as occurs at FTMC. Laboratory animals respond differently to toxicants than do wild animals and environmental conditions at the site may act to attenuate or exacerbate the toxicity of certain stressors to wild animals. The toxicity reference values used in the riparian food web were derived from no-observed-adverse-effects-levels (NOAELs); therefore, they represent doses at which no adverse effects occur in the test animals. NOAEL-based TRVs are the most protective values that could be used and result in very conservative risk estimates. Uncertainty factors are also used in the derivation of the NOAEL-based TRVs, introducing more uncertainty to these values.

A significant level of uncertainty associated with the food web models concerns the use of hazard quotients (HQ) to express potential ecological hazards. There are a number of limitations to the use of HQs that limit their ability to determine the presence of potential ecological hazards. These limitations include: the HQ does not express risk, but expresses instead “hazard”; the HQ is not population-based; the HQ is not linearly scaled; naturally-occurring background concentrations of inorganic compounds may produce “unacceptable” HQs; HQs are often calculated that are unrealistically high and toxicologically impossible; and HQs cannot adequately address temporal scales (Tannenbaum, et al., 2003).

8.0 Summary and Conclusions

A Screening-Level Ecological Risk Assessment (SLERA) was completed for the BBGR ranges. The SLERA for this group of small arms ranges indicated COPECs were present in surface soil and sediment and that additional assessment was warranted to further identify and refine the potential ecological risks at these sites. In order to accomplish these tasks, a BERA was conducted for the BBGR ranges.

8.1 BERA Summary

The BERA for the BBGR Ranges is based on the information presented in the *Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges* (Shaw, 2006). The following sections describe the BERA that was conducted for the BBGR ranges at FTMC.

8.1.1 Terrestrial Habitat at the BBGR Ranges

The SLERA for the BBGR Ranges indicated that the habitats and COPECs present at the individual ranges that make up the BBGR Range complex were similar. The terrestrial habitat at the BBGR ranges fall into two general categories: "cleared" areas and forested areas. The cleared areas are those areas that were formerly maintained as lawns or mowed fields. Since maintenance activities have ceased in these areas, pioneer species are now colonizing these ranges. Typically, the species most likely to colonize these 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 abandoned areas are the grasses and herbaceous species. These formerly maintained grassy areas are classified as being in an early old field successional state. Over time, the grass and herbaceous species will be followed by shrubs and small trees. Many of these formerly maintained areas of the BBGR ranges are now being colonized by loblolly pine and longleaf pine saplings. The forested areas outside of the cleared areas are best characterized as mixed deciduous/coniferous forest. These rich and relatively unaltered forested regions represent the large safety fans across the Main Post.

8.1.2 South Branch of Cane Creek Tributaries Habitat

A single tributary to South Branch of Cane Creek occurs within the BBGR Ranges and only at Range 18. This small, ephemeral tributary runs along the western boundary of Range 18 and then transects the range from west-to-east in the northern portion of the range. This small tributary then turns north along the eastern boundary of Range 18 and joins South Branch of Cane Creek immediately north of the Range 18 study area. Surface water runoff from the hills

west and south of Range 18 is the major source of water for this tributary. As such, it only conveys water during periods of significant precipitation. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface.

This tributary is narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in this tributary where it passes through forested areas. The vegetation within this tributary is generally characteristic of the surrounding upland vegetation, indicating that this tributary only transmits water during periods of significant precipitation.

The shallow nature of this ephemeral tributary limits its ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the South Branch tributary in the vicinity of Range 18 because it only has water present during periods of significant precipitation, and when water is present, it is too shallow to support larger fish. Although the South Branch of Cane Creek tributary at the BBGR Ranges is ephemeral in nature, it has the potential to support a variety of amphibious species and some small, drought-tolerant fish species.

The South Branch of Cane Creek tributary within the Range 18 study area is classified as low quality gray bat foraging habitat. South Branch of Cane Creek, located just east and north of the Range 18 study area, is classified as moderate quality gray bat foraging habitat.

8.1.3 Ingram Creek Tributaries Habitat

The tributaries to Ingram Creek that are present at the BBGR Ranges drain Ranges 20, 23, 25, 25-East, 26, and Ranges South of Range 25. All of these tributaries are small ephemeral streams with the exception of one of the tributaries that flows across Range 23, which is perennial. Surface water runoff from the hills located east and north of the BBGR Ranges is the major source of water for these tributaries. As such, they only convey water during periods of significant precipitation. Flow in these small ephemeral tributaries is highly variable, depending on precipitation in the surrounding watershed. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. Downstream (west) of the BBGR Ranges, these small ephemeral tributaries continue to flow in a westerly direction towards Ingram Creek. Ingram Creek (located west of the BBGR Ranges) flows in a northwesterly direction until its confluence with Cane Creek, northwest of the BBGR Ranges.

The physical characteristics of these tributaries at the BBGR Ranges are relatively consistent. All of these tributaries are narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in these tributaries where they pass through forested areas. The vegetation in these tributaries is generally characteristic of the surrounding upland vegetation, indicating that these tributaries only transmit water during periods of significant precipitation.

Although the Ingram Creek tributaries at the BBGR Ranges are ephemeral in nature, they have the potential to support a variety of amphibious species and some small, drought-tolerant fish species. The shallow nature of these ephemeral tributaries limits their ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the Ingram Creek tributaries in the vicinity of the BBGR Ranges because they only hold water during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

The small ephemeral tributaries to Ingram Creek in the vicinity of the BBGR Ranges are either not classified with regard to potential gray bat foraging habitat or are classified as low quality gray bat foraging habitat.

8.1.4 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*); and
- Tennessee yellow-eyed grass (*Xyris tennesseensis*).

The only federally listed species that has the potential to occur in the vicinity of the BBGR ranges is the gray bat (Garland, 1996). The drainage features at the BBGR Ranges have either been designated as providing "low quality" foraging habitat for the gray bat or have not been classified for their potential to provide gray bat foraging habitat (Garland, 1996). The other federally listed species occur at Pelham Range or Choccolocco Creek.

8.1.5 COPECs Identified in the SLERA

The COPECs that were identified in the SLERA and formed the basis for the BERA at the BBGR ranges are the following:

- ***Surface Soil*:** antimony, beryllium, copper, lead, and zinc; and
- ***Sediment*:** copper, lead, and gamma-chlordane.

8.1.6 Assessment and Measurement Endpoints

Given the overall goal of protecting the integrity and quality of the terrestrial old field ecosystems and mixed deciduous/coniferous forests at the BBGR ranges, the terrestrial assessment endpoints focused on critical community niches within these two ecosystems. The overall goal of the aquatic assessment endpoints was the protection of the integrity and quality of the semi-aquatic ecosystems in the South Branch of Cane Creek and Ingram Creek and their tributaries at the BBGR ranges. The aquatic assessment endpoints focused on critical community niches within these semi-aquatic and aquatic ecosystems. The assessment endpoints, risk hypotheses, and measurement endpoints that were identified for the BBGR ranges were summarized in Table 3-1.

8.1.7 Data Quality Objectives

Based on the findings of the SLERA and Problem Formulations conducted for the BBGR ranges, the objectives of this BERA were identified as 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 BBGR 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 BBGR Ranges at which the ecological receptors are at risk.

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

The following decisions required site-specific data in order to address the issues identified in the objectives presented above.

- Determine if the COPECs at the BBGR Ranges are available for 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 South Branch of Cane Creek and Ingram Creek and their tributaries 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 risk.

These data were used to help determine whether COPECs in surface soil or sediment at the BBGR ranges are, or will, present significant risk to ecological receptors.

8.1.8 Results of the Terrestrial Plant Bioaccumulation Tests

In order to assess the potential for bioaccumulation of soil COPECs into terrestrial plants, samples of above-ground vegetation were collected concurrently with the soil samples used in the toxicity tests and analyzed for TAL metals. Site-specific bioaccumulation factors were estimated by comparing the plant tissue concentrations to soil concentrations of COPECs at each sampling location. Site-specific BAFs were estimated for each soil sample location and a mean soil-to-plant BAF was then calculated for each soil COPEC. The soil-to-plant BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-6:

Surface Soil COPEC	Soil-to-Plant BAF
Copper	0.13 *
Lead	0.03
Zinc	1.1 *

* Due to poor correlations these site-specific $BAF_{soil\text{-}to\text{-}earthworm}$ were not used in the food web models.

It is important to note that, as stated previously, there are very poor correlations between plant tissue concentrations and soil concentrations for antimony, beryllium, copper, and zinc. Based on the sparsity of plant tissue data for antimony and beryllium and the poor correlation between plant tissue concentrations and soil concentrations for copper and zinc, default soil-to-plant BAFs (USEPA, 1999) were assumed for these COPECs and used in the terrestrial food web model. The site-specific soil-to-plant BAF for lead was used in the terrestrial food web model.

8.1.9 Results of the Perennial Rye Grass Toxicity Tests

The results of the perennial rye grass toxicity tests indicate that seed germination was not adversely affected at any of the sample locations. None of the seed germination rates were significantly reduced compared to seed germination rates using soil from the reference site. The NOEC, LOEC, and AET values based on rye grass germination are summarized below and presented in Table 6-1.

Soil COPEC	***Germination NOEC*** *(mg/kg)*	***Germination LOEC*** *(mg/kg)*	***Germination AET*** *(mg/kg)*
Antimony	1,310	> 1,310	> 1,310
Beryllium	1.67	> 1.67	> 1.67
Copper	563	> 563	> 563
Lead	146,000	> 146,000	> 146,000
Zinc	154	> 154	> 154

The results of the perennial rye grass toxicity tests indicate that shoot length was statistically reduced in one sample compared to shoot length of seedlings grown in soil from a reference location. Statistically reduced shoot length was reported in sample SY0009 (BERA-HR-74Q-GP66). The NOECs, LOECs, and AETs based on shoot length, are summarized below and also presented in Table 6-1.

Soil COPEC	***Shoot Length NOEC*** *(mg/kg)*	***Shoot Length LOEC*** *(mg/kg)*	***Shoot Length AET*** *(mg/kg)*
Antimony	4.92	5.29	1,310
Beryllium	0.624	0.685	1.67
Copper	152	184	563
Lead	526	919	146,000
Zinc	37.3	53.5	154

Root length was statistically reduced in four samples compared to root length of seedlings grown in soil from a reference location. Statistically reduced root length was reported in sample SY0002 (BBGR-BERA-SS01), SY0004 (BERA-74Q-GP64+200SE), SY0009 (BERA-HR-74Q-

GP66), and SY0010 (BERA-HR-118Q-DEP01). The NOECs, LOECs, and AETs based on root length are summarized below and also presented in Table 6-1.

Soil COPEC	***Root Length NOEC*** *(mg/kg)*	***Root Length LOEC*** *(mg/kg)*	***Root Length AET*** *(mg/kg)*
Antimony	< 0.724	0.724	1,310
Beryllium	0.156	0.444	1.02
Copper	< 16.5	16.5	152
Lead	102	125	146,000
Zinc	22.3	25.4	53.5

Total plant mass was statistically reduced in four samples compared to total plant mass of seedlings grown in soil from a reference location. Statistically reduced plant mass was reported in sample SY0004 (BER-74Q-GP64+200SE), SY0006 (BERA-HR-79Q-SS09), SY0009 (BERA-HR-74Q-GP66), and SY0010 (BERA-HR-118Q-DEP01). The NOECs, LOECs, and AETs based on total plant weight are summarized below and presented in Table 6-1.

Soil COPEC	***Total Plant Mass NOEC*** *(mg/kg)*	***Total Plant Mass LOEC*** *(mg/kg)*	***Total Plant Mass AET*** *(mg/kg)*
Antimony	< 0.724	0.724	1,310
Beryllium	0.583	0.597	1.02
Copper	< 16.5	16.5	184
Lead	102	125	146,000
Zinc	22.3	25.4	> 154

8.1.10 Results of Earthworm Toxicity Tests

The results of the earthworm toxicity testing indicate that earthworm survival and growth were not adversely affected at any of the sample locations. None of the survival or growth rates were significantly reduced after either 14 days or 28 days of exposure compared to survival and growth rates using soil from the reference site. Therefore, the NOECs for both earthworm survival and growth are the highest concentrations tested, and the LOECs and AETs are defined as greater than the NOEC concentrations. These values are summarized below and also presented in Table 6-1.

Soil COPEC	***Earthworm NOEC*** *(mg/kg)*	***Earthworm LOEC*** *(mg/kg)*	***Earthworm AET*** *(mg/kg)*
Antimony	1310	> 1310	> 1310
Beryllium	1.67	> 1.67	> 1.67
Copper	563	> 563	> 563
Lead	146,000	> 146,000	> 146,000
Zinc	154	> 154	> 154

8.1.11 Results of Earthworm Bioaccumulation Test

In order to define the site-specific relationship between soil concentrations and earthworm tissue concentrations of COPECs, site-specific bioaccumulation factors were estimated by comparing the earthworm tissue concentrations to soil concentrations of COPECs at each sampling location. Site-specific BAFs were estimated for each soil sample location and a mean soil-to-earthworm BAF was then calculated for each soil COPEC. The soil-to-earthworm BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-9:

Surface Soil COPEC	Soil-to-Earthworm BAF
Copper	0.5
Lead	0.53
Zinc	3.17 *

* Due to poor correlation this site-specific $BAF_{soil\text{-}to\text{-}earthworm}$ was not used in the food web models.

It is important to note that, as stated previously, there are very poor correlations between earthworm tissue concentrations and soil concentrations for antimony, beryllium, and zinc. Based on the sparsity of earthworm tissue data for antimony and beryllium and the poor correlation between earthworm tissue concentrations and soil concentrations for zinc, default soil-to-earthworm BAFs (USEPA, 1999) were assumed for these COPECs and used in the terrestrial food web model. Site-specific soil-to-earthworm BAFs for copper and lead were used in the terrestrial food web model.

8.1.12 Results of Terrestrial Food Web Model

A terrestrial food web model was designed to assess the potential risks to various terrestrial-based feeding guilds from soil-related COPECs. The terrestrial food web model was executed with maximum and mean exposure point concentrations in order to estimate a range of potential terrestrial wildlife exposures. The food web model was executed with site-specific area use factors, site-specific soil-to-earthworm BAFs and site-specific soil-to plant BAFs (where they are supported by the site-specific data). Literature-derived bioaccumulation factors were used to estimate COPEC concentrations in the terrestrial invertebrates and vegetation portions of the receptor species' diets where site-specific BAFs could not be calculated due to site-specific data constraints.

Estimated maximum and mean exposures for the receptor species were compared to the toxicity reference values to estimate maximum and mean hazard quotients for each receptor species. The results of the food web model indicate that there are potential risks to terrestrial receptors at the BBGR ranges from exposures to COPECs in surface soil at these ranges through food web

interactions. Namely, antimony, copper, and lead in surface soil result in hazard quotients greater than one for all of the terrestrial receptors assessed via the terrestrial food web model. The terrestrial food web model indicates that beryllium in surface soil does not result in hazard quotients greater than one for any terrestrial wildlife receptors at the BBGR ranges, and zinc is unlikely to pose any significant risk to terrestrial wildlife receptors at the BBGR ranges.

	Maximum Exposure		***Mean Exposure***	
	NOAEL HQ	**LOAEL HQ**	**NOAEL HQ**	**LOAEL HQ**
White-footed Mouse:				
Antimony	17.4	1.74	1.69	0.169
Beryllium	0.0746	0.00746	0.0118	0.00118
Copper	64.3	50.1	1.13	0.882
Lead	56.6	5.66	2.36	0.236
Zinc	0.947	0.0308	0.0408	0.00133
American Robin :				
Antimony	31	6.2	3.01	0.601
Beryllium	ND	ND	ND	ND
Copper	26.1	19.8	0.459	0.349
Lead	252	85.9	10.5	3.58
Zinc	0.17	0.017	0.00732	0.000732
Short-tail Shrew :				
Antimony	16.8	1.68	1.63	0.163
Beryllium	0.202	0.0202	0.0321	0.0321
Copper	60.5	47.1	1.06	0.829
Lead	137	13.7	5.7	0.57
Zinc	2.69	0.0876	0.116	0.00378
American Woodcock:				
Antimony	32.7	6.53	3.17	0.634
Beryllium	ND	ND	ND	ND
Copper	26.5	20.2	0.466	0.355
Lead	511	174	21.3	7.25
Zinc	0.387	0.0387	0.0167	0.00167

The terrestrial food web model was also executed in order to calculate soil concentrations of COPECs that would result in hazard quotients of one or lower. The estimated soil concentrations based on HQs equal to 1.0 are presented in Table 6-25 and are summarized below.

	Soil Conc. Based on NOAEL TRV and HQ = 1.0 (mg/kg)	**Soil Conc. Based on LOAEL TRV and HQ = 1.0** (mg/kg)
White-Footed Mouse :		
- antimony	4.98	49.8
- copper	234	300
- lead	413	4,131
- zinc	1,327	40,800

	Soil Conc. Based on NOAEL TRV and HQ = 1.0 (mg/kg)	Soil Conc. Based on LOAEL TRV and HQ = 1.0 (mg/kg)
American Robin :		
- antimony	2.8	14
- copper	578	757
- lead	92.5	272
- zinc	7,390	73,900
Short-tailed Shrew :		
- antimony	5.15	51.5
- copper	249	319
- lead	171	1,705
- zinc	465	14,300
American Woodcock :		
- antimony	2.65	13.25
- copper	567	747
- lead	45.7	134
- zinc	3,230	32,400

Soil concentrations based on an HQ = 1.0 were not calculated for beryllium because the terrestrial food web model did not indicate any risk to terrestrial wildlife species due to food web exposures to beryllium.

8.1.13 Results of the Chironomid Toxicity Tests

The results of the 10-day chironomid survival test show that survival was not adversely affected at any of the sample locations. None of the chironomid survival rates were significantly reduced compared to survival rates using soil from the reference location (Table 5-10).

Chironomid growth was found to be statistically reduced in two sediment samples compared to chironomid growth in sediment from a reference location. Statistically significant differences in weight were found for site samples SY1002 (BERA-CWM-183-SD02) with mean dry weight of 1.50 mg/organism and SY1005 (BERA-HR-83Q-SD01) with mean dry weight of 1.43 mg/organism (Table 5-10). These differences, however, only represent 11 percent and 15 percent reductions in growth from the reference sample and may not be biologically significant.

The NOECs, based on chironomid survival, for the three sediment COPECs are defined as the highest concentrations tested, and the LOECs and AETs are defined as greater than the NOEC concentrations. The sediment values based on chironomid survival are presented in Table 6-26 and summarized below.

Sediment COPEC	***Chironomid Survival & Growth NOEC*** *(mg/kg)*	***Chironomid Survival & Growth LOEC*** *(mg/kg)*	***Chironomid Survival & Growth AET*** *(mg/kg)*
Copper	22.5	> 22.5	> 22.5
Lead	112	> 112	> 112
gamma-Chlordane	0.0091	> 0.0091	> 0.0091

The results of the toxicity tests for chironomid growth showed no statistically and biologically significant differences between site samples and reference samples after 10 days of exposure. Therefore, the NOECs, AETs, and LOECs for chironomid growth are the same as those for chironomid survival.

8.1.14 Results of Chironomid Bioaccumulation Test

Data collected as part of the Site Investigation indicate that copper and lead may accumulate in chironomid tissues. Since gamma-chlordane was only detected in one chironomid tissue sample very close to the detection limit, it was determined that accumulation of gamma-chlordane in chironomid tissues at the BBGR Ranges is negligible.

In order to define the site-specific relationship between sediment concentrations and chironomid tissue concentrations of COPECs, site-specific bioaccumulation factors were estimated by comparing the chironomid tissue concentrations to sediment concentrations of COPECs at each sampling location. Site-specific BAFs were estimated for each sediment sample location and a mean sediment-to-chironomid BAF was then calculated for each sediment COPEC. The sediment-to-chironomid BAFs calculated using the site-specific data are summarized below and were also presented in Table 5-11:

Sediment COPEC	**Sediment-to-Chironomid BAF**
Copper	11.81*
Lead	0.67*

* Due to poor correlation this site-specific $BAF_{sed\text{-}to\text{-}chironomid}$ was not used in the food web models

It is important to note that, as stated previously, there are very poor correlations between chironomid tissue concentrations and sediment concentrations for copper and lead, and gamma-chlordane was only detected in one chironomid tissue sample. Based on the poor correlation between chironomid tissue concentrations and sediment concentrations for copper and lead, and the sparsity of data for gamma-chlordane, default sediment-to-invertebrate BAFs (USEPA, 1999)

were assumed for these COPECs and used in the riparian food web model to estimate the transfer of COPECs from sediment to riparian invertivores.

8.1.15 Results of Benthic Invertebrate Community Analysis

Benthic invertebrate samples were collected from 12 locations in South Branch of Cane Creek, Ingram Creek, and their tributaries within the BBGR ranges and one reference station. A riffle/run sample and a course particulate organic matter sample were collected from each sampling location. Using the raw benthic data, a numerical value was calculated for the eight metrics (measures of benthic macroinvertebrate community quality) described previously. Calculated values were then compared to values derived from the reference site. Each metric was then assigned a score of 6, 3 or 0 according to the comparability (percent similarity) of calculated and reference values (Plafkin et al., 1989). Scores for the eight metrics were then totaled and compared to the total metric score for the reference location. The percent comparison between the total scores provides a final evaluation of biological condition.

The benthic invertebrate communities at ten of the twelve BBGR Ranges sampling locations were evaluated as being moderately impaired relative to the reference location, with comparability percentages ranging from 33% at locations HR-79Q-SD03 and HR-83Q-SD01, to 67% at BBGR-BERA-SD03 (Table 5-14). One location, FTA-163-SD02, was evaluated as severely impaired, with a comparison to reference percentage of 20%, and one location, CWM-183-SD02, was determined to be non-impaired in relation to the reference location, with an 80% comparison to reference (Table 5-14).

Three analytes, copper, lead and gamma-chlordane were identified as COPECs in sediment at the BBGR Ranges. Sediment analytical results for all sampling locations were reviewed for concentrations of COPECs that may have affected the benthic community assemblages in creeks in the vicinity of the BBGR Ranges. Sediment COPEC concentrations and benthic invertebrate metric scores were presented in Table 5-15. No significant correlation between biological condition and sediment COPEC concentration are indicated for any of the three sediment COPECs (r^2 = 0.0349, 0.0766 and 0.0036 for copper, lead, and gamma-chlordane, respectively). Therefore, the data do not indicate that site-related COPECs are significantly impacting the benthic communities at sites associated with the BBGR Ranges.

A possible explanation for the differences in benthic community quality is the natural physical habitat characteristics at the various sampling locations. While none of the sample locations were found to have poor or marginal habitat conditions, the average habitat score for nine of the sampling locations was in the range indicating sub-optimal benthic habitat. Three BBGR

locations and the reference location had scores in the range indicating optimal habitat. The results of the benthic community analysis indicate that the quality of the benthic invertebrate communities at locations associated with the BBGR ranges is more dependant upon physical habitat parameters than the presence of site-related sediment COPECs.

8.1.16 Results of Riparian Food Web Model

In order to assess the potential risks to various riparian-based feeding guilds from sediment-related constituents, a riparian food web was constructed. The riparian food web model was executed with site-specific sediment concentrations, site-specific area use factors, and literature-derived values for the other input variables. Although site-specific sediment-to-chironomid BAFs were derived for copper and lead as described previously, the correlation coefficients for these relationships were very low; thus, the reliability of these BAFs were very low. In order to maintain a conservative assessment, default sediment-to-invertebrate BAFs referenced from the literature were used to estimate COPEC concentrations in aquatic invertebrates**.** The model was executed with both maximum and mean detected concentrations as the exposure point concentrations for sediment in order to calculate a range of potential risks to riparian receptors.

Estimated maximum and mean exposures for the receptor species were compared to the toxicity reference values to estimate maximum and mean hazard quotients for each receptor species. The results of the model indicate that there is a potential risk to riparian invertivorous mammals and riparian invertivorous birds from exposure to lead in sediment. The calculated hazard quotients for the sediment COPECs (copper, lead, and gamma-chlordane) are summarized below.

Sediment COPEC	*Little Brown Bat*		*Marsh Wren*	
	NOAEL HQ	*LOAEL HQ*	*NOAEL HQ*	*LOAEL HQ*
Copper	1.87E-01	1.45E-01	1.51E-01	1.15E-01
Lead	1.42E+00	1.42E-01	9.35E+00	3.19E+00
Gamma-Chlordane	1.10E-03	5.48E-04	7.45E-03	1.49E-03

The riparian food web model was also executed to determine the sediment concentrations of COPECs that would result in hazard quotients of one or lower. The estimated sediment concentrations based on HQs equal to 1.0 are presented in Table 6-42 and are summarized below.

Riparian Receptor Species	Sediment Conc. Based on NOAEL TRV and HQ = 1.0 (mg/kg)	Sediment Conc. Based on LOAEL TRV and HQ = 1.0 (mg/kg)
Little Brown Bat :		
Pb	1,200	12,000
Marsh Wren :		
Pb	91	268

Sediment concentrations based on an HQ = 1.0 were not calculated for copper or gamma-chlordane because the riparian food web model did not indicate any risk to riparian wildlife species due to food web exposures to copper or gamma-chlordane.

8.2 BERA Conclusions

The results of this BERA indicate that one or more constituents in surface soil and sediment have the potential to pose adverse effects to sensitive ecological receptors in the terrestrial and riparian ecosystems at the BBGR ranges.

In order to identify site-specific concentrations of the COPECs that are protective of the sensitive receptors potentially present at the BBGR ranges, the various lines of evidence collected during the BERA were analyzed. From the various lines of evidence were derived risk-based remedial goals (RBRG). These RBRGs are concentrations of COPECs in the various environmental media that are protective of the assessment endpoints described in Chapter 3 of this report. Because several lines of evidence were assessed for some of the assessment endpoints and COPECs, several RBRGs were also derived for these assessment endpoints and COPECs. The selection of the most reasonable or appropriate RBRG for a given endpoint is a risk management decision and is not the province of this BERA.

8.2.1 Terrestrial Ecosystems

Earthworm toxicity tests (survival and growth endpoints) indicated that COPECs in soil at the BBGR ranges are not likely to any cause adverse effects in terrestrial invertebrates. Similarly, germination of terrestrial plants was not adversely affected by COPECs in soil based on the results of the plant germination tests. However, COPECs in soil may cause adverse effects in some sensitive terrestrial plants (i.e., reduced shoot and root length, and reduced overall plant weight). The results of the terrestrial plant and terrestrial invertebrate bioaccumulation tests indicated that none of the COPECs in soil accumulate to any significant extent in terrestrial invertebrates or terrestrial plants. The terrestrial food web model indicated that sensitive

terrestrial receptors could experience adverse effects due to exposure to COPECs in soil at the BBGR ranges through food web interactions.

The calculated RBRGs for soil COPECs at the BBGR Ranges ranged from <0.724 to 1,310 mg/kg for antimony, 0.156 to >1.67 mg/kg for beryllium, <16.5 to 757 mg/kg for copper, 45.7 to >146,000 mg/kg for lead, and 22.3 to 73,900 mg/kg for zinc. The potential ecological RBRGs for surface soil at the BBGR ranges are presented in Table 8-1. It is important to note that these soil RBRGs represent only those remedial goals resulting from the specific assessments conducted as part of this BERA. Any number of other remedial goal options is available based on human health risk, ARARs, and other rationale. It is a risk management decision as to which remedial option, if any, is most appropriate for a given site.

8.2.2 Riparian Ecosystems

Chironomid toxicity tests (survival and growth endpoints) indicated that COPECs in sediment at the BBGR Ranges are unlikely to effect benthic macroinvertebrate survival. Results of chironomid toxicity tests indicated that although sediment COPECs may adversely affect benthic macroinvertebrate growth, these potential adverse effects are not expected to be significant at the community level. The results of the chironomid bioaccumulation tests indicated that none of the COPECs in sediment at the BBGR Ranges are likely to accumulate to any significant extent in benthic macroinvertebrates. The riparian food web model indicated that sensitive riparian receptors could experience adverse effects due to exposures to lead in sediment through food web interactions.

The calculated RBRGs for sediment COPECs at the BBGR Ranges ranged from 22.5 to >22.5 mg/kg for copper, 91 to 12,000 mg/kg for lead, and 0.0091 to >0.0091 mg/kg for gamma-chlordane. The potential ecological RBRGs for sediment at the BBGR ranges are presented in Table 8-1. It is important to note that these sediment RBRGs represent only those remedial goals resulting from the specific assessments conducted as part of this BERA. Any number of other remedial goal options are available based on human health risk, ARARs, and other rationale. It is a risk management decision as to which remedial option, if any, is most appropriate for a given site.

9.0 References

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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
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
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
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™
'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
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
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&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	installation-wide work plan
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