

**Final  
Data Evaluation Report and  
Screening Level Ecological Risk Assessment  
for Bains Gap Road Ranges  
Fort McClellan, Calhoun County, Alabama**

**Volume I of III**

**Text and Appendices A - E**

**August 2002**

**Final  
Data Evaluation Report and  
Screening Level Ecological Risk Assessment  
for Bains Gap Road Ranges**

**Range 24 Upper, Parcel 80Q  
Range 21, Parcel 77Q  
Range 22, Parcel 78Q  
Former Mortar Range (Firing Line), Parcel 109Q  
Range 27, Parcel 85Q**

**Fort McClellan  
Calhoun County, Alabama**

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## **1.0 Introduction**

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Fort McClellan (FTMC) is undergoing closure by the Base Realignment and Closure (BRAC) Commission under Public Laws 100-526 and 101-510. The 1990 Base Closure Act, Public Law 101-510, established the process by which U.S. Department of Defense installations will be closed or realigned. The BRAC Environmental Restoration Program requires investigation and cleanup of federal properties prior to transfer to the public domain. In addition, the Community Environmental Response Facilitation Act (CERFA) (Public Law 102-426) requires federal agencies to identify real property on military installations scheduled for closure that can be transferred to the public for redevelopment or reuse. The BRAC environmental restoration program at FTMC follows the Comprehensive Environmental Response, Compensation, and Liability Act process.

Ultimately, the U.S. Army will prepare an engineering evaluation/cost analysis (EE/CA) report for Range 24 Upper, Range 21, Range 22, Former Mortar Range (Firing Line), and Range 27 that represents Army compliance with both BRAC and CERFA requirements for investigation and cleanup of the range parcels. This data evaluation report and screening level ecological risk assessment (SLERA) represent the first stage of the EE/CA report. This document is being submitted before the entire EE/CA report is completed to summarize the nature and extent of contamination found at these ranges and to present a SLERA for regulatory review and comment. The data evaluation and SLERA will be incorporated into the draft EE/CA once final approval is obtained from Alabama Department of Environmental Management and U.S. Environmental Protection Agency (EPA) Region IV.

IT Corporation (IT) has been contracted by the U.S. Army Corps of Engineers (USACE), Mobile District, to perform the EE/CA for Range 24 Upper (Defensive Techniques), Parcel 80Q; Range 21 (Field Fire Range), Parcel 77Q; Range 22 (Zero Range), Parcel 77Q; Former Mortar Range (Firing Line), Parcel 109Q; and Range 27 (Special Operations Range), Parcel 85Q. These four ranges are located along the Bains Gap Road (BGR) at FTMC in Anniston, Alabama. This work is being performed under Contract No. DACA21-96-D-0018, Task Order CK11.

The purpose of this document is to summarize the site setting and history and its physical characteristics. The analytical data collected to date from environmental samples of the various site media are summarized, and the SLERA for the BGR ranges is presented. The Army's BGR range activities were terminated in October 1998 as part of the overall Base closure. Chapter 2 presents a site history of each range. Surface and subsurface soil sampling and visual surveying, sediment and surface water sampling, monitoring well installation and groundwater sampling,

and surveying for lead in the range safety fans using an x-ray fluorescence (XRF) instrument were performed at these sites by IT in 2000 and 2001. The results of these activities and an evaluation of the nature and extent of contamination at the BGR ranges are included in Chapter 3. The SLERA is found in Chapter 4. Supporting information and documentation of the samples collected are included in the following appendices:

- Appendix A - Sample Collection Logs and Analysis Request/Chain-of-Custody Records
- Appendix B - Boring Logs and Well Construction Logs
- Appendix C - Well Development Logs
- Appendix D - Survey Data
- Appendix E - Variance Reports.
- Appendix F - Summary of Validated Analytical Data
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## ***2.0 Site Description and Characteristics***

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The four range sites that comprise the BGR ranges are nearly adjacent to each other and lie directly south of BGR in the central-eastern section of the FTMC Main Post. An unnamed hill (1,230 feet above mean sea level [amsl]) separates these ranges from Range 20 further to the south. Near this hill, the area becomes heavily vegetated with dense hardwood forest along its north and east slopes. Jones Hill (1,300 feet amsl) rises to the north and Marcheta Hill (1,300 feet amsl) rises to the east of the ranges, enclosing Range 24 Upper, Range 21, Range 22, and Range 27 on the floor of a shallow valley. The headwaters of Cane Creek are located in this valley. Eight small tributary streams flow from the surrounding hillsides, meet in this area, then flow west towards the FTMC Main Post. Figure 2-1 shows the location of the ranges on the Main Post, and Figure 2-2 shows the details of the BGR ranges. This section summarizes available data on the physical, demographic, and other characteristics of the sites and surrounding areas.

The BGR ranges are among the best understood and documented of all FTMC sites. Their uses were singular and known with certainty, i.e., all were used as firing ranges for various kinds of firearms. The locations of the firing lines as well as impact areas, berms, and safety fans are known with certainty, which logically defines the limits of potentially contaminated areas. The nature of the potential contamination is also known and is generally limited to metals (specifically lead), organic explosives, and other components of gunpowder associated with firearm ammunition. Fortuitously, lead is likely to be the most widespread of the potential constituents. Furthermore, the impact of bullets or shot with soil leaves particles of lead that are plainly visible on the surface, providing a simple, low-cost method of confirming the extent of potential contamination. Thus, lead serves as an excellent indicator chemical for locating potentially contaminated areas. In other words, the absence of visible lead particles at the surface may indicate that an area is not contaminated with other firing-range-related metals as well. The presence of other potential firing-range-related chemicals, such as gun cleaning solvents, will be characterized by collecting and evaluating additional analytical data.

The Archive Search Report (ASR) (USACE, 1999) indicates that these ranges are located within the impact area of the World War I Artillery Impact Area and should be considered an area where unexploded ordnance may exist. The ASR and the Environmental Baseline Survey (EBS) (Environmental Science and Engineering, Inc. [ESE], 1998) summarize much of what is known about these ranges.

## **2.1 Description and Background of the Sites**

### **2.1.1 Range 24 Upper**

Range 24 Upper, Defensive Techniques Range, Parcel 80Q, was constructed between 1983 and 1989 (ESE, 1998). The range was inactivated in 1990. Weapons fired at this range consisted of M-16 rifles with tracers (white phosphorus) and flares. Range activities occurred only in 1989 and 1990, when defensive techniques were practiced here (ESE, 1998). The site area is approximately 11 acres and has no defined safety fan. The 1998 aerial photograph (Figure 2-3) shows bare areas south of the EBS parcel boundary where defensive maneuvers may possibly have been conducted during the operation of Range 24 Upper. The bare areas were apparently small impact areas (soil mounds) and excavations such as foxholes or defensive positions that were used in defensive training activities. These areas have since been disturbed and graded, and there is not any evidence of the foxholes or defensive positions remaining. A map of this area is shown in detail on Figure 2-4. Apparently, much of the training activities occurred in this area just south of the EBS parcel boundary of Range 24 Upper. It is likely that live fire was directed from the parcel to the south.

In addition to the small arms training, Plate 5 of the ASR shows an 81-millimeter (mm) mortar range fan listed as old Range 28, covering the area of Range 24 Upper, Parcel 80Q (USACE, 1999). This mortar range appears to have the firing point just south of BGR (Parcel 109Q at Range 22). The impact area appears to be due south toward Reeves Hill and beyond. The southern end of the mortar range fan covers part of the Artillery and Mortar Impact Areas, Parcels 138Q-X, 141Q-X, and 142Q-X. According to the ASR, this mortar range first appears on a 1950 range map as an 81-mm mortar range. By 1958, the range is designated as a 61-mm mortar range, with a new orientation showing the firing line west of the original firing line. The mortar range was abandoned in 1967 (USACE, 1999).

Range 24 Upper is a densely wooded area immediately south of BGR. One unimproved road connects this area to BGR east of the parcel, and a second unimproved road on the southwest side of the parcel connects Range 24 Upper to the area east of Range 21. This site has few remaining distinguishing features. The aforementioned soil mounds and depressions are found in the south and east sections of the parcel between the unimproved roads. Bullet fragments have been found in these areas. Site drainage is to the west (towards Range 21) along two ditches that join west of the parcel.

### **2.1.2 Range 21**

Range 21, the Field Fire Range, Parcel 77Q, was in use from 1951 through installation closure in 1999. Weapons fired at this range consisted of M-16 rifles (5.56 mm) with tracer. Unspecified small arms were used at this range prior to the advent of the M-16 (ESE, 1998). The total site, including the extensive range fan, comprises 2,249 acres. The main area of the site subject to this evaluation is limited to approximately 15 acres. The 1998 aerial photograph (Figure 2-5) and a map of this area (Figure 2-6) show that Range 21 includes a single firing line and target lines at 75 meters, 175 meters, and 300 meters downrange of the firing line. These target lines contain a series of formerly electrified concrete target coffins that were used to store and present pop-up targets during training exercises.

Range 21 is a flat, open area with a grass covering and wooded areas east and south of the site. The firing line and the target lines of the range are all located within the flat, open area, with the wooded area beginning approximately 200 to 500 feet south (downrange) of the 300-meter target line. This site lacks a definitive soil berm downrange that would usually form the main impact zone; instead, the range main impact zones are found between the target lines and downrange of the 300-meter line. Bullet channels in these areas are evident on Figure 2-5. The area between Cane Creek and the 175-meter target line is eroded, and bullet fragments have been observed on the surface.

Range 21 is separated from Range 22 on the west by an unimproved road and a perpendicular soil berm. Cane Creek flows across the center of the range west towards Range 22, through the dividing soil berm via a large concrete culvert. No structures other than the culvert and two wood bridges are currently present at the site. Site access is via a semicircular gravel road that connects the firing line area to BGR.

### **2.1.3 Range 22**

Range 22, Zero Range, was in operation from 1961 through the installation closure in 1999. This range was in continuous use as a rifle range. Weapons fired at this range consisted of M-16 rifles (5.56 mm) with tracer. The Former Mortar Range (Firing Line), Parcel 109Q, is a 1.5-acre parcel located within the Range 22 study area. Weapons fired at the mortar range possibly include 81-mm and 60-mm mortar (ESE, 1998). Probable impact zones for this range are located south of Range 24 Upper and have been investigated and reported in a separate activity. Parcel 109Q refers to the mortar range firing line only, which was investigated as part of Range 22. The total site area of Range 22, including the extensive range fan, comprises 1,810 acres. The study area of the site subject to this evaluation is limited to approximately 12.5 acres (which includes Parcel 109Q). The 1998 aerial photograph (Figure 2-7) and site map (Figure 2-8) show

the details of Range 22, which include a single firing line and a single target line at 25 meters from the firing line.

Range 22 is a flat, open area with a rocky soil berm that forms the main impact zone for the majority of the site. On the eastern portion of the range, the berm height is much reduced and the impact zone extends into the wooded area beyond. Cane Creek flows west along the toe of the berm slope, and soil erosion along the berm face into the creek is evident. Downrange (south) of the berm at Range 22, the maximum site elevation rises to approximately 1,230 feet amsl at the top of an unnamed hill located southwest of Range 22, while the minimum site elevation of 900 feet amsl is found near BGR.

West of this range, a soil berm that is perpendicular to BGR separates Range 22 from Range 27. Cane Creek flows west along the entire length of the base of the berm in the impact zone and flows behind the dividing berm and into Range 27. No structures are currently present at the site. Access to the site is via a semicircular gravel road that connects the firing line area to BGR.

#### **2.1.4 Range 27**

Historical records are conflicting on the site history of Range 27, the Special Operations Range, Parcel 85Q. The ASR states that the range “was built after World War II. It appears on the 1958 Range Map as Close Combat 1 & 2” (USACE, 1999). The EBS states that the range “has been in use from 1976 through the present. Weapons fired at this range consisted of M-16 rifles (5.56 mm) between 1983 and 1989; and 9-mm pistol, 12-gauge shotgun, and .45-caliber pistol and machine gun from 1989 to present” (ESE, 1998). Base range control Regulation 350-2 also indicates that a rappelling tower and obstacle course were located here and that .38-caliber pistol ordnance was used here. This range has been historically subdivided into four main areas: Range 27A - Shooting House, Range 27B - Live Fire and Maneuver Close Quarters Battle Range, Range 27C - Stress Pistol and Shotgun Range, and Range 27D - Pistol and Submachine Gun Qualification Range. The total site, including the extensive range fan, comprises 954 acres. The main area of the range subject to this investigation is estimated to be approximately 16 acres. The 1998 aerial photograph (Figure 2-9) and site map (Figure 2-10) show the details of Range 27.

Range 27A is referred to as the “shooting house” or “tire house.” This structure is constructed of stacks of tires that have been staked upright using 4 inch by 4 inch wood posts and filled with sand to form the walls of rooms. It has a gravel floor and no roof. The Army has used the shooting house for training exercises with live ammunition. Wooden doors and interior divider

walls in the house have sustained heavy damage from training, and bullets are present in the tires and wood. The shooting house is still present at FTMC.

Range 27B consists of a flat, open area between two perpendicular berms. To the south, Cane Creek flows west. A small tributary begins in this area north of Cane Creek and flows west to meet Cane Creek west of Range 27.

Range 27C is a large, flat, open area separated from Range 27B on the east and Range 27D on the west by perpendicular soil berms. As indicated in the FTMC range control Regulation 350-2, this area may have once contained a rappelling tower and obstacle course. No structures currently remain.

Range 27D is a narrow range on the far west portion of Range 27. In the impact zone of this portion of the range, high concentrations of bullet fragments have been observed along the base of the unnamed hill to the south of the range and in the sediments of Cane Creek.

Range 27 is separated from Range 22 on the east by a perpendicular soil berm. The main branch of Cane Creek flows along the southern edge of the range. An unnamed hill (1,230 feet amsl) separates this range from Range 20 further to the south. West of Range 27, two tributaries from the south merge with Cane Creek and flow west under BGR through a culvert and on towards the Main Post. Access to the site is provided by a semicircular gravel road that connects the firing-line areas of 27B and 27C to BGR. Gravel drives connect 27A (east) and 27D (west) to the main semicircular road.

## **2.2 Physical Characteristics**

### **2.2.1 Physiography and Topography**

With the exception of Range 24 Upper, the BGR ranges are grass-covered and surrounded by trees to the east, south, and west. BGR forms the northern boundary of these ranges. Range 24 Upper is about 150 feet south of BGR and is located within a wooded area. Elevations range from about 975 to 1,050 feet amsl at Range 24 Upper, 900 to 960 feet amsl at Range 21, 900 to 975 feet amsl at Range 22, and about 875 to 1,000 feet amsl at Range 27 (Figure 2-2). Ground surface across the ranges is generally flat, with a gradual slope to the north-northwest towards BGR.

## **2.2.2 Hydrology**

### **2.2.2.1 Regional Hydrology**

Precipitation in the form of rainfall averages about 53 inches annually in Anniston, Alabama, with infiltration rates annually exceeding evapotranspiration rates (U.S. Department of Commerce, 1998). Portions of three drainage basins (Cane Creek, Choccolocco Creek, and Tallasseeatchee Creek) are found within the Main Post of Fort McClellan. All three of these drainage basins eventually empty into the Coosa River, which is located approximately 16 miles west of Main Post. Figure 2-11 is a map showing the surface water hydrology and drainage basins of the Main Post and Choccolocco Corridor.

The Choccolocco Mountains located in the eastern portion of the Main Post and the Skeleton Mountains located in the southern portion of the Main Post form a major surface-water divide. East of the Choccolocco Mountains, FTMC consists of a narrow strip called Choccolocco Corridor, which extends approximately 3.5 to 4 miles from the mountains across the floodplain of Choccolocco Creek to the base of Rattlesnake Mountain. Choccolocco Creek and its tributaries drain all of Choccolocco Corridor along with the eastern and southern-most portions of the Main Post (Figure 2-11). The Choccolocco Creek Drainage Basin covers approximately 7.5 square miles of the Main Post.

On the western side of Choccolocco Mountains are the Cane Creek and Tallasseeatchee Creek Drainage Basins (Figure 2-11). The headwaters of the Cane Creek Drainage Basin originate in the Choccolocco Mountains and flow west through the main cantonment. Cane Creek has four named tributaries on Post (Cave, Remount, South Branch, and Ingram Creeks). Cave Creek is a tributary to Cane Creek; however, Cave Creek occurs as a separate drainage basin within the confines of the Main Post. Cave Creek eventually joins Cane Creek off Post east of the unincorporated development of Sherman Heights near McMinn Airfield. Cane Creek and its tributaries receive surface runoff from the central portion of the Post. Cane Creek exits the reservation at Baltzell Gate. Cave Creek and its unnamed tributaries drain the north-central portion of the Post. Cave Creek exits the Post near the unincorporated development of Sherman Heights. The on-Post drainage area of the Cane Creek Basin covers approximately 19.6 square miles.

The Tallasseeatchee Creek Drainage Basin drains the northern-most portions of the Main Post (Figure 2-11). Most of the surface runoff from this portion of Main Post collects in unnamed tributaries to Reilly Lake. Reilly Lake then empties into Dothard Creek, a tributary to the

Tallaseehatchee, and exits the Main Post north of Reilly Airfield. The Tallaseehatchee Drainage Basin covers approximately 2.5 square miles of Main Post.

Most surface water bodies are fed at least in part by freshwater springs. Freshwater springs occur abundantly on installation lands, often appearing along the trace of thrust faults. Karst features, including developed caves and sinkholes, have been identified in the FTMC area (Science Applications International Corporation [SAIC], 2000).

Freshwater marshes are located along Cane Creek; most are limited to the cumulatively larger downstream watershed of Pelham Range. Only one major area, the 25-acre marsh near Reilly Lake, occurs on the Main Post (SAIC, 2000).

#### **2.2.2.2 Local Hydrology**

Cane Creek and its headwater tributaries are located within BGR ranges and the creek flows from the east to the west towards the Main Post (see Figure 2-2). This creek has several tributaries. Cane Creek is formed by the tributaries south of Range 24 Upper and is joined further west in Range 21 by a tributary that flows southeast under BGR from the north. Other tributaries flow west through the impact zone of Range 21 and join with Cane Creek in Range 22 and in Range 27. Another tributary to Cane Creek flows south under BGR and joins near the perpendicular soil berm that separates Range 21 and Range 27. On Range 27, another drainage feature is present in the center of the range that flows west and joins Cane Creek near its crossing with BGR west of the Range 27 study area.

#### **2.2.3 Geology**

##### **2.2.3.1 Regional Geology**

Calhoun County includes parts of two physiographic provinces, the Piedmont Upland Province and the Valley and Ridge Province. The Piedmont Upland Province occupies the extreme eastern and southeastern portions of the county and is characterized by metamorphosed sedimentary rocks. The generally accepted range in age of these metamorphics is Cambrian to Devonian.

The majority of Calhoun County, including the Main Post of FTMC, lies within the Appalachian fold-and-thrust structural belt (Valley and Ridge Province) where southeastward-dipping thrust faults with associated minor folding are the predominant structural features. The fold-and-thrust belt consists of Paleozoic sedimentary rocks that have been asymmetrically folded and thrust-faulted, with major structures and faults striking in a northeast-southwest direction.

Northwestward transport of the Paleozoic rock sequence along the thrust faults has resulted in the imbricate stacking of large slabs of rock, referred to as thrust sheets. Within an individual thrust sheet, smaller faults may splay off the larger thrust fault, resulting in imbricate stacking of rock units within an individual thrust sheet (Osborne and Szabo, 1984). Geologic contacts in this region generally strike parallel to the faults, and repetition of lithologic units is common in vertical sequences. Geologic formations within the Valley and Ridge Province portion of Calhoun County have been mapped by Warman and Causey (1962), Osborne and Szabo (1984), and Moser and DeJarnette (1992) and vary in age from Lower Cambrian to Pennsylvanian.

The basal unit of the sedimentary sequence in Calhoun County is the Cambrian Chilhowee Group. The Chilhowee Group consists of the Cochran, Nichols, Wilson Ridge, and Weisner Formations (Osborne and Szabo, 1984) but in Calhoun County is either undifferentiated or divided into the Cochran and Nichols Formations and an upper, undifferentiated Wilson Ridge and Weisner Formation. The Cochran is composed of poorly sorted arkosic sandstone and conglomerate with interbeds of greenish gray siltstone and mudstone. Massive to laminated greenish gray and black mudstone makes up the Nichols Formation, with thin interbeds of siltstone and very fine-grained sandstone (Osborne et al., 1988). These two formations are mapped only in the eastern part of the county.

The Wilson Ridge and Weisner Formations are undifferentiated in Calhoun County and consist of both coarse-grained and fine-grained clastics. The coarse-grained facies appears to dominate the unit and consists primarily of coarse-grained, vitreous quartzite and friable, fine- to coarse-grained, orthoquartzitic sandstone, both of which locally contain conglomerate. The fine-grained facies consists of sandy and micaceous shale and silty, micaceous mudstone, which are locally interbedded with the coarse clastic rocks. The abundance of orthoquartzitic sandstone and quartzite suggests that most of the Chilhowee Group bedrock in the vicinity of FTMC belongs to the Weisner Formation (Osborne and Szabo, 1984).

The Cambrian Shady Dolomite overlies the Weisner Formation northeast, east, and southwest of the Main Post and consists of interlayered bluish gray or pale yellowish gray sandy dolomitic limestone and siliceous dolomite with coarsely crystalline, porous chert (Osborne et al., 1989). A variegated shale and clayey silt have been included within the lower part of the Shady Dolomite (Cloud, 1966). Material similar to this lower shale unit was noted in core holes drilled by the Alabama Geologic Survey on FTMC (Osborne and Szabo, 1984). The character of the Shady Dolomite in the FTMC vicinity and the true assignment of the shale at this stratigraphic interval are still uncertain (Osborne, 1999).

The Rome Formation overlies the Shady Dolomite and locally occurs to the northwest and southeast of the Main Post as mapped by Warman and Causey (1962) and Osborne and Szabo (1984), and immediately to the west of Reilly Airfield (Osborne and Szabo, 1984). The Rome Formation consists of variegated, thinly interbedded grayish-red-purple mudstone, shale, siltstone, and greenish red and light gray sandstone, with locally occurring limestone and dolomite. The Conasauga Formation overlies the Rome Formation and occurs along anticlinal axes in the northeastern portion of Pelham Range (Warman and Causey, 1962; Osborne and Szabo, 1984) and the northern portion of the Main Post (Osborne et al., 1997). The Conasauga Formation is composed of dark gray, finely to coarsely crystalline, medium- to thick-bedded dolomite with minor shale and chert (Osborne et al., 1989).

Overlying the Conasauga Formation is the Knox Group, which is composed of the Copper Ridge and Chepultepec dolomites of Cambro-Ordovician age. The Knox Group is undifferentiated in Calhoun County and consists of light medium gray, fine to medium crystalline, variably bedded to laminated, siliceous dolomite and dolomitic limestone that weather to a chert residuum (Osborne and Szabo, 1984). The Knox Group underlies a large portion of the Pelham Range area.

The Ordovician Newala and Little Oak Limestones overlie the Knox Group. The Newala Limestone consists of light to dark gray, micritic, thick-bedded limestone with minor dolomite. The Little Oak Limestone consists of dark gray, medium- to thick-bedded, fossiliferous, argillaceous to silty limestone with chert nodules. These limestone units are mapped as undifferentiated at FTMC and in other parts of Calhoun County. The Athens Shale overlies the Ordovician limestone units. The Athens Shale consists of dark gray to black shale and graptolitic shale with localized interbedded dark gray limestone (Osborne et al., 1989). These units occur within an eroded “window” in the uppermost structural thrust sheet at FTMC and underlie much of the developed area of the Main Post.

Other Ordovician-aged bedrock units mapped in Calhoun County include the Greensport Formation, Colvin Mountain Sandstone, and Sequatchie Formation. These units consist of various siltstones, sandstones, shales, dolomites and limestones and are mapped as one, undifferentiated unit in some areas of Calhoun County. The only Silurian-age sedimentary formation mapped in Calhoun County is the Red Mountain Formation. This unit consists of interbedded red sandstone, siltstone, and shale with greenish gray to red silty and sandy limestone.

The Devonian Frog Mountain Sandstone consists of sandstone and quartzitic sandstone with shale interbeds, dolomudstone, and glauconitic limestone (Osborne et al., 1988). This unit locally occurs in the western portion of Pelham Range.

The Mississippian Fort Payne Chert and the Maury Formation overlie the Frog Mountain Sandstone and are composed of dark to light gray limestone with abundant chert nodules and greenish gray to grayish red phosphatic shale, with increasing amounts of calcareous chert toward the upper portion of the formation (Osborne and Szabo, 1984). These units occur in the northwestern portion of Pelham Range. Overlying the Fort Payne Chert is the Floyd Shale, also of Mississippian age, which consists of thin-bedded, fissile brown to black shale with thin intercalated limestone layers and interbedded sandstone. Osborne and Szabo (1984) reassigned the Floyd Shale, which was mapped by Warman and Causey (1962) on the Main Post of FTMC, to the Ordovician Athens Shale based on fossil data.

The Pennsylvanian Parkwood Formation overlies the Floyd Shale and consists of a medium to dark gray, silty, clay shale and mudstone with interbedded light to medium gray, very fine to fine grained, argillaceous, micaceous sandstone. Locally the Parkwood Formation also contains beds of medium to dark gray, argillaceous, bioclastic to cherty limestone and beds of clayey coal up to a few inches thick (Raymond et al., 1988). The Parkwood Formation in Calhoun County is generally found within a structurally complex area known as the Coosa Deformed Belt. In the deformed belt, the Parkwood Formation and Floyd Shale are mapped as undifferentiated because their lithologic similarity and significant deformation make it impractical to map the contact (Thomas and Drahovzal, 1974; Osborne et al., 1988). The undifferentiated Pennsylvanian Parkwood Formation and Mississippian Floyd Shale are found throughout the western quarter of Pelham Range.

The Jacksonville Thrust Fault is the most significant structural geologic feature in the vicinity of the Main Post of FTMC, both for its role in determining the stratigraphic relationships in the area and for its contribution to regional water supplies. The trace of the fault extends northeastward for approximately 39 miles between Bynum, Alabama, and Piedmont, Alabama. The fault is interpreted as a major splay of the Pell City Fault (Osborne and Szabo, 1984). The Ordovician sequence that makes up the Eden thrust sheet is exposed at FTMC through an eroded “window,” or fenster, in the overlying thrust sheet. Rocks within the window display complex folding, with the folds overturned and tight to isoclinal. The carbonates and shales locally exhibit well-developed cleavage (Osborne and Szabo, 1984). The FTMC window is framed on the northwest by the Rome Formation; north by the Conasauga Formation; northeast, east, and southwest by the Shady Dolomite; and southeast and southwest by the Chilhowee Group (Osborne et al.,

1997). Two small klippen of the Shady Dolomite, bounded by the Jacksonville Fault, have been recognized adjacent to the Pell City Fault at the FTMC window (Osborne et al., 1997).

The Pell City Fault serves as a fault contact between the bedrock within the FTMC window and the Rome and Conasauga Formations. The trace of the Pell City Fault is also exposed approximately nine miles west of the FTMC window on Pelham Range, where it traverses northeast to southwest across the western quarter of Pelham Range. The trace of the Pell City Fault marks the boundary between the Pell City Thrust Sheet and the Coosa Deformed Belt.

The eastern three-quarters of Pelham Range is located within the Pell City Thrust Sheet, while the remaining western quarter of Pelham is located within the Coosa Deformed Belt. The Pell City Thrust Sheet is a large-scale thrust sheet containing Cambrian and Ordovician rocks and is relatively less structurally complex than the Coosa Deformed Belt (Thomas and Neathery, 1982). The Pell City Thrust Sheet is exposed between the traces of the Jacksonville and Pell City Faults along the western boundary of the FTMC window and along the trace of the Pell City Fault on Pelham Range (Thomas and Neathery, 1982; Osborne et al., 1988). The Coosa Deformed Belt is a narrow northeast-to-southwest-trending linear zone of complex structure (approximately 5 to 20 miles wide and approximately 90 miles long) consisting mainly of thin imbricate thrust slices. The structure within these imbricate thrust slices is often internally complicated by small-scale folding and additional thrust faults (Thomas and Drahovzal, 1974).

### **2.2.3.2 Site Geology**

The soils mapped within the area of investigation for the BGR ranges are the Anniston and Allen gravelly loam, the Anniston and Allen stony loam, the Jefferson gravelly fine sandy loam, the Jefferson stony fine sandy loam, and the Stony Rough Land sandstone. The Stony Rough Land sandstone underlies the majority of the area of investigation, with only the northern portion of the area of investigation underlain by the Jefferson gravelly and stony fine sandy loams and the extreme southwestern portion of the area of investigation underlain by the Anniston and Allen gravelly and stony fine sandy loams (U.S. Department of Agriculture [USDA], 1961) (Figure 2-12).

Stony Rough Land sandstone is a land type that is found in rough, steep areas with many outcrops of sandstone or quartzite bedrock, loose rock fragments, and scattered patches of sandy soil material. The soil materials found in these areas are generally thin. The runoff in these area is high, the infiltration is slow, and the capacity for available moisture is low (USDA, 1961).

The Jefferson stony fine sandy loam and Jefferson gravelly fine sandy loam are friable soils derived from old local alluvium. These soils are generally found on the foot slopes at the base of steep hills and on ridges underlain by sandstone and shale. The surface soil is generally dark grayish brown. The subsoil generally consists of a yellowish brown stony, gravelly, fine sandy loam. The Jefferson stony fine sandy loam can have numerous pieces of sandstone and quartzite up to eight inches in diameter (USDA, 1961).

The Anniston and Allen stony loam and Anniston and Allen gravelly loam have developed in old alluvium on the foot slopes and fans along the base of steep hills. The subsoil of these units is generally dark brown to grayish brown in color. The subsurface soil consists of a dark red to reddish brown stony, gravelly, fine sand, clay loam. These soils generally have medium infiltration and a high capacity for available moisture. The Anniston and Allen stony loam can have numerous stones between three to eight inches in diameter (USDA, 1961).

The BGR ranges are located long the eastern boundary of the Fort McClellan window (Figure 2-13). The mapping units exposed within this area of investigation are the Cambrian Chilhowee Group, the Cambrian Shady Dolomite, the Cambrian Rome Formation, and the Mississippian/Ordovician Floyd and Athens Shale, undifferentiated. The Chilhowee Group underlies the majority of the area of investigation. The Shady Dolomite and Rome Formation are found along the eastern boundary of the area of investigation. The Shady Dolomite, Little Oak and Newala Limestones, and Floyd and Athens Shales are found along the western boundary of the area of investigation (Osborne et al., 1997 and 1988).

The monitoring wells and soil boring locations were focused near the impact areas for the BGR ranges. Hence, a majority of the geologic data collected for this investigation describe an area located along Cane Creek in the north portion of the area of investigation. The geologic data collected during drilling activities at Parcels 77Q, 78Q, 80Q, and 85Q revealed a light brown to reddish orange cobbly, gravelly, sandy loam from ground surface to between five to twenty feet below ground surface (bgs). The cobbles and gravel generally consisted of quartz sandstone and quartzite. Below the cobbly, gravelly, sandy loam a sandy, silty, clay residuum with a variety of colors ranging from dark reddish brown to light tan to yellowish orange to olive to purple was encountered to the total depth of the borings. However, quartz sandstone was encountered at one monitoring well location (HR-80Q-MW01) at a depth of approximately seven feet bgs, and mudstone was encountered at monitoring well HR-85-MW02 at a depth of approximately 82 feet bgs. The soils and residuum encountered appear consistent with soils mapped as the Jefferson gravelly fine sandy loam, Jefferson stony fine sandy loam, and the Stony Rough Land sandstone.

Bedrock encountered in the borings is consistent with bedrock mapped within the Chilhowee Group.

## **2.2.4 Hydrogeology**

### **2.2.4.1 Regional Hydrogeology**

The hydrogeology of Calhoun County has been investigated by the Geological Society of America (Moser and DeJarnette, 1992), the U.S. Geological Survey in cooperation with the Geological Society of America (Warman and Causey, 1962), and the Alabama Department of Environmental Management (Planert and Pritchette, 1989). Groundwater in the vicinity of FTMC occurs in residuum derived from bedrock decomposition, within fractured bedrock, along fault zones, and from the development of karst frameworks. Groundwater flow may be estimated to be toward major surface water features. However, because of the impacts of differential weathering, variable fracturing, and the potential for conduit flow development, caution must be exercised in the use of surface topography as an indicator for groundwater flow direction in the area. Areas with well-developed residuum horizons may subtly reflect the surface topography, but the groundwater flow direction also may exhibit the influence of pre-existing structural fabrics or the presence of perched water horizons on unweathered ledges or impermeable clay lenses.

Precipitation and subsequent infiltration provide recharge to the groundwater flow system in the region. The main recharge areas for the aquifers in Calhoun County are located in the valleys. The ridges generally consist of sandstones, quartzite, and slate, which are resistant to weathering, relatively unaffected by faulting, and, therefore, relatively impermeable. The ridges have steep slopes and thin to no soil cover, which enhance runoff to the edges of the valleys (Planert and Pritchette, 1989).

The thrust fault zones typical of the county form large storage reservoirs for groundwater. Points of discharge occur as springs, effluent streams, and lakes. Coldwater Spring is the largest spring in the state of Alabama, with a discharge of approximately 32 million gallons per day. This spring is the main source of water for the Anniston Water Department, from which FTMC obtains its water. The spring is located approximately 5 miles southwest of Anniston and discharges from the brecciated zone of the Jacksonville Fault (Warman and Causey, 1962).

Shallow groundwater on FTMC occurs principally in the residuum developed from Cambrian sedimentary and carbonate bedrock units of the Weisner Formation and Shady Dolomite, and locally in lower Ordovician carbonates and shales. The shallow groundwater in this area

generally flows to the northwest and discharges to streams in the area. The residuum may yield adequate groundwater for domestic and livestock needs but may go dry during prolonged dry weather. Groundwater within the residuum serves as a recharge reservoir for the underlying bedrock aquifers. Bedrock permeability is locally enhanced by fracture zones associated with thrust faults and by the development of solution (karst) features.

Two major aquifers were identified by Planert and Pritchette (1989), the Knox-Shady and Tuscumbia-Fort Payne Aquifers. The continuity of the aquifers has been disrupted by the complex geologic structure of the region, such that each major aquifer occurs repeatedly in different areas. The Knox-Shady Aquifer group occurs over most of Calhoun County and is the main source of groundwater in the county. It consists of the Cambrian and Ordovician-aged quartzite and carbonates. The Conasauga Formation is the most utilized unit of the Knox-Shady aquifer, with twice as many wells drilled as any other unit (Moser and DeJarnette, 1992).

The Tuscumbia-Fort Payne Aquifer occurs in the extreme northwestern portion of the county. This aquifer consists of Mississippian-age carbonates and shales. Because of its limited outcrops in the recharge area and the rugged terrain of the outcrop area, the Tuscumbia-Fort Payne Aquifer is not considered a major groundwater supply in Calhoun County (Moser and DeJarnette, 1992). However, it is an important source of groundwater in counties to the west (Planert and Pritchette, 1989). Neither the Knox-Shady Aquifer nor the Tuscumbia-Fort Payne Aquifer is in the vicinity of the BGR ranges.

#### **2.2.4.2 Local Hydrogeology**

Reeves-Truitt Spring is located south of Bains Gap Road between Range 22 and Range 27 (Moser and DeJarnette, 1992). Reeves-Truitt Spring reportedly had a measured discharge of approximately 58 gallons per minute in October 1957 (Moser and DeJarnette, 1992). For the purpose of the BGR range investigations, Reeves-Truitt Spring is identified as SAR-85-SEEP01.

Static groundwater levels were measured in the permanent residuum monitoring wells at the ranges south of Bains Gap Road, Parcels 77Q, 78Q, 80Q, and 85Q, on January 7, 2002. Depth to groundwater measurements were taken from the top of casing, following procedures outlined in Section 4.18 of the installation-wide sampling and analysis plan (SAP) (IT, 2000a). A piezometric surface map constructed with static groundwater elevation data for the residuum water-bearing unit is provided on Figure 2-14. Groundwater elevation data revealed an east-to-west groundwater flow direction across the study area (Figure 2-14). The horizontal hydraulic gradient across this area is calculated to be approximately 0.07 feet per foot. Based on the piezometric surface and the depth groundwater was encountered during well installation

activities, it appears that groundwater at the BGR ranges is under confined to semiconfined conditions.

### ***2.2.5 Current and Future Land Use***

The current land use at all four ranges is open space. A comprehensive reuse plan was completed for FTMC in November 1997 (USACE, 1997). All four of the BGR ranges are planned for transfer to the U.S. Fish and Wildlife Service (USFWS) as part of the FTMC nature preserve. The USFWS has expressed an interest in using land at Range 21 to establish a welcome center with hiking trails nearby.

## **3.0 Investigation Summary**

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### **3.1 Field Investigation**

Investigations have been conducted in different stages at the BGR ranges. Sampling procedures followed methodology outlined in the SAP (IT, 2000a). Three sampling and analysis plans (IT, 2000b; IT, 2001a; IT, 2001b) were prepared and used to direct the field investigation activities. In July/August 2000, surface soil, subsurface soil, and surface water/sediment sampling was conducted at Range 21, Range 22, and Range 27 (IT, 2000b). A supplemental sampling event was conducted at Range 21, Range 22, and Range 27 in August/November 2001, when additional surface soil and surface water/sediment samples were collected (IT, 2001a). During this supplemental phase, 10 residuum and 1 bedrock monitoring wells were installed and sampled and an XRF survey for lead was performed on Range 21, Range 22, and Range 27 safety fan surface soils.

Range 24 Upper was investigated separately from June to September 2001, with surface soil, subsurface soil, and surface water/sediment samples collected (IT, 2001b). Six residuum monitoring wells were installed and five were sampled at Range 24 Upper in August 2001. Additional investigation sampling at Range 24 Upper was conducted in January and February 2002 which included surface soil, subsurface soil, and surface water/sediment sampling.

Table 3-1 summarizes the number and type of the samples collected at the BGR ranges. All of the available data from these activities at the four BGR range parcels have been combined into this investigation summary chapter. Sample documentation, boring logs, well construction diagrams, well development logs, survey data, variance reports, validated data, and quality are presented in Appendices A through G. Tables 3-2 through 3-7 summarize the sample results. Figures 3-1 through 3-5 show the surface soil, subsurface soil, surface water/sediment, and XRF survey sampling locations and selected data.

#### **3.1.1 Soil Samples and Visual Survey**

A total of 168 surface soil, 2 depositional soil, and 71 subsurface soil samples were collected and analyzed. Soil samples were collected following the methodology outlined in the SAP (IT, 2000a). In addition to sampling, a visual survey and mapping of surface soils was performed at Range 21, Range 22, and Range 27 to ascertain the extent of bullet fragments (i.e., the source of the metals contamination). The results of the visual survey are shown on Figure 3-6. The main objective for collecting the soil data and performing the visual survey was to delineate contamination from weapons firing and gather data useful in determining the potential

excavation area for each range for the EE/CA. The visual survey of the surficial bullet fragments was subjective in nature as far as distinguishing “light,” “moderate,” and “dense” lead density. The objective of this qualitative survey was to provide preliminary surface areas for calculating soil volumes for potential excavation within the context of scoping and planning the BGR EE/CA. The final planned excavation areas, which will be presented in the BGR EE/CA document, will be based on a combination of factors, including the soil analytical data and the established remedial goals for the planned remedial action. Figure 3-1 is the surficial soil lead isopleth map which summarizes the analytical data for lead.

During sample collection, lead fragments (e.g., bullets fragments, shot pellets) were removed from the soil/sediment samples to improve their representativeness. Samples were analyzed for a variety of chemicals to document the concentrations of constituents of potential concern at the ranges’ firing lines and impact zones (Table 3-1). The analysis suites included target compound list (TCL) volatile organic compounds (VOC), TCL semivolatile organic compounds (SVOC), target analyte list (TAL) metals, chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, polychlorinated biphenyl compounds (PCB), nitroaromatic explosive compounds, cyanide, and perchlorate. Since lead was expected to be the constituent most indicative of soil contamination due to weapons firing, several soil samples were analyzed only for lead.

### **3.1.2 Surface Water/Sediment Samples**

In Cane Creek and its adjoining tributaries on the BGR ranges, 33 surface water and 38 sediment samples were collected to measure the impact of firing activities on these media. At the time of collection, the following sample locations did not have sufficient surface water to collect a sample and only sediment was collected at these locations: SAR-77-SW/SED17, SAR-77-SW/SED18, SAR-77-SW/SED21, and SAR-77-SW/SED24 at Range 21 and location SAR-78-SW/SED11 at Range 22. Surface water and sediment samples were collected following methodology outlined in the SAP (IT, 2000a).

In the field, water quality measurements were performed that included specific conductivity, dissolved oxygen, oxidation reduction potential, temperature, turbidity, and pH. Table 3-8 summarizes these field-measured parameters for the surface water samples. Surface water and sediment samples were analyzed for TCL VOCs, TCL SVOCs, TAL metals, chlorinated pesticides, organophosphorus pesticides, chlorinated herbicides, PCBs, nitroaromatic explosive compounds, cyanide, and perchlorate. Sediment samples were also analyzed for total organic carbon (TOC) and grain size. Since lead was expected to be the constituent most indicative of soil contamination due to weapons firing, several surface water/sediment samples were analyzed only for lead.

### **3.1.3 Well Installation and Groundwater Samples**

Seventeen monitoring wells (one bedrock, 16 residuum) were installed at the BGR ranges. Two of the residuum monitoring wells (HR-77Q-MW02 and HR-85Q-MW02) were originally proposed to be bedrock wells (IT, 2001a); however, bedrock was not encountered during drilling within 100 feet of the ground surface. Since the likelihood of groundwater contamination from activities at the BGR ranges being present at depths greater than 100 feet bgs is very small, drilling was not continued and the wells were installed as residuum wells. One residuum monitoring well (HR-80Q-MW01) was not sampled due to insufficient water. The monitoring well locations are shown on Figure 2-14. Table 3-9 summarizes the well construction details as well as the static water levels recorded to produce the piezometric surface map (Figure 2-14). Groundwater samples from Parcels 77Q, 78Q, and 85Q were analyzed for TCL VOCs, TCL SVOCs, TAL metals, explosives, and perchlorate. Groundwater samples from Parcel 80Q were analyzed for TAL metals and explosives.

### **3.1.4 Safety Fan XRF Survey for Lead**

A calibrated, field-portable XRF instrument was taken into the Range 21, Range 22, and Range 27 safety fans to analyze surface soil collected at 40 preplanned locations. The sample locations were found in the safety fans using a global positioning system to navigate the sampling team to the coordinates shown in the work plan tables. The analysis was performed in the field using the instrument's cadmium-109 source. The lead concentration and standard deviation data from the instrument display were copied onto the XRF laboratory results form by the analyst, and the sampled soil was enclosed in an aluminum pan and sealed in a labeled Ziploc™ bag for on-site archive. Surface soil samples for the XRF survey were collected and analyzed following methodology specified in the site-specific sampling and analysis plan (IT, 2001a). Table 3-10 summarizes the XRF-measured lead concentrations.

Ten percent of the samples (4 samples) were selected for confirmation analysis at the fixed-base laboratory using EPA Method 6010B for lead. Samples that contained both the highest and the lowest concentrations of lead were selected for confirmation analysis. Table 3-11 summarizes the confirmation analysis results. The confirmation data agreed with the results of the on-site measurement.

## **3.2 Nature and Extent of Contamination**

Lead from weapons training activities constitutes the largest single source of environmental contamination at the BGR ranges; therefore, a special separate discussion on lead contamination is presented in this section for each of the sampled site media. To evaluate whether lead and the other detected constituents present an unacceptable risk to human health and the environment,

the analytical results were compared to the human health site-specific screening levels (SSSL) and ecological screening values (ESV) for FTMC. Specifically, residential SSSLs or recreational site user SSSLs (for surface water/sediment) were used. The SSSLs and ESVs were developed by IT for human health and ecological risk evaluations as part of the ongoing site investigations performed under the BRAC Environmental Restoration Program at FTMC (IT, 2000c).

Metals concentrations exceeding the SSSLs and ESVs were subsequently compared to metals background screening values to determine if the metal concentrations are within natural background concentrations (SAIC, 1998). Summary statistics for background metal samples collected at FTMC are included in Appendix H.

The following sections and Tables 3-2 through 3-7 summarize the results of the comparison of detected constituents to the SSSLs, ESVs, and background screening values. Complete analytical results are presented in Appendix F.

### **3.2.1 Surface Soil and Depositional Soil**

A total of 168 surface soil and two depositional soil samples were collected and analyzed at the BGR ranges. The analytical data from these samples are compared to SSSLs, ESVs, and background screening values in Table 3-2.

**TAL Metals (Except Lead).** Twenty-two TAL metals (in addition to lead) were detected in surface soil and depositional soil samples collected at BGR ranges. Eleven metals exceeded the SSSLs; however, all of those results were below background or within the upper background range (UBR), except for the following:

- Antimony (4.5 to 335 milligrams per kilogram [mg/kg]) in 14 samples
- Arsenic (71 mg/kg) at SAR-85-SS02
- Copper (2,620 to 85,900 mg/kg) in 6 samples
- Iron (71,400 to 75,200 mg/kg) in 2 samples
- Zinc (2,900 to 9,670 mg/kg) in 2 samples.

Fourteen metals exceeded the ESVs; however, all of those results were below background or within the UBR, except for the following:

- Antimony (4.5 to 335 mg/kg) in 10 samples
- Arsenic (71 mg/kg) at SAR-85-SS02
- Beryllium (1.3 to 3.63) in 5 samples
- Copper (47 to 85,900 mg/kg) in 18 samples

- Iron (71,400 to 75,200 mg/kg) in 2 samples
- Selenium (2.85 mg/kg) at HR-85Q-SS02
- Silver (2.11 to 2.74 mg/kg) in 3 samples
- Zinc (444 to 9,670 mg/kg) in 5 samples.

**Lead.** A lead isopleth map for surface soils using the ESV (50 mg/kg), residential SSSL (400 mg/kg), industrial site user SSSL (880 mg/kg), and recreational site user SSSL (7,600 mg/kg) is presented on Figure 3-1. Areas of lead contamination were detected at concentrations ranging from 3.9 mg/kg (at HR-80Q-GP04) to 114,000 mg/kg (at SAR-78-SS34) at the BGR ranges, Parcels 80Q, 77Q, 78Q, and 85Q.

As shown on Figure 3-1, lead concentrations exceeding 7,600 mg/kg (recreational site user SSSL) were detected at the small arms impact area at Range 24 Upper, near the 175-meter target line at Range 21, in the bullet channels at Range 22, and in an area densely covered with bullet fragments at Range 27 (Figure 3-6). Concentrations exceeding the residential SSSL and industrial site worker SSSL extend the length of the ranges (east to west), to the 300-meter firing line at Range 21, and cover the impact zones of Range 27 and Range 22. Ground surface at these locations was observed to be moderately to densely covered with bullet fragments (Figure 3-6). Lead concentrations above the ESV (50 mg/kg) encompass an area approximately 4,500 feet (east to west) by 1,400 feet (north to south) (Figure 3-1).

Elevated concentrations of lead are generally elongate in an east-west direction, adjacent to (and south of) the locations of the former firing lines. Areas with the highest concentrations of lead typically occur along the hillside south of the firing lines. Figure 3-1 shows that the extent of lead in surface soil within the site investigation area has been defined to the ESV. Sample locations below the ESV represent areas of lead concentrations within or below the background range determined by SAIC (1998).

**VOCs.** Twelve VOCs were detected in BGR range surface soil samples. All results were less than their SSSLs and ESVs.

**SVOCs.** Nineteen SVOCs were detected in BGR range surface soil samples. Thirteen of the 19 SVOCs were polyaromatic hydrocarbon (PAH) compounds. Benzo(a)pyrene results in two samples (SAR-77-SS02 and SAR-78-SS02) exceeded its SSSL. One of the two values (SAR-77-SS02) also exceeded the ESV; however, both results were less than the PAH background value for benzo(a)pyrene (IT, 2000c). Three other SVOCs exceeded ESVs: fluoranthene (in samples SAR-77-SS02, SAR-77-SS07, SAR-78-SS02), pyrene (in samples SAR-77-SS02 and SAR-78-

SS02), and butyl benzyl phthalate (SAR-85-SS38). All six of these results are estimated values below the method reporting limit.

**Chlorinated Pesticides.** Three chlorinated pesticide compounds (aldrin, endrin, alpha-betahexachlorocyclohexane [BHC]) were detected in sample HR-80Q-GP21; 4,4'-dichlorodiphenyldichloroethene (DDE) was detected in sample HR-77Q-SS01; and 4-4'-dichlorodiphenyltrichloroethane (DDT) was detected in sample HR-78Q-SS01 at concentrations that exceeded their ESVs. None of the five pesticides also exceeded its SSSL. Six other chlorinated pesticides were also detected at estimated concentrations (flagged with a "J" data qualifier) below their SSSLs and ESVs.

**Chlorinated Herbicides.** The chlorinated herbicide propionic acid (MCP) was detected in three samples (HR-80Q-DEP01, HR-80Q-GP05, HR-80Q-GP21) at concentrations that exceeded its ESV; however, no concentrations also exceeded its SSSL. Three other chlorinated herbicides (2,4-DB, dichloroprop, and dinoseb) were also detected at estimated concentrations (flagged with a "J" data qualifier) below their SSSLs and ESVs.

**Nitroaromatic Explosive Compounds.** One nitroaromatic explosive compound, nitrobenzene, was detected at one location (HR-80Q-GP21) at a concentration that was below its SSSL and ESV. One other nitroexplosive, 2,4-dinitrotoluene (DNT), was detected by Method 8270C in a total of 10 samples at Range 21, Range 22, and Range 27 at estimated concentrations below the method reporting limit. None of these estimated concentrations exceeded any of the screening values.

No surface soil samples contained detectable concentrations of organophosphorus pesticide compounds, PCBs, cyanide, or perchlorate.

### **3.2.2 Subsurface Soil**

A total of 71 subsurface soil samples were collected at BGR ranges. The analytical data from these samples are compared to the SSSLs and background screening values in Table 3-3.

**TAL Metals (Except Lead).** Eight TAL metals (other than lead) were detected in subsurface soil samples collected at BGR ranges. All eight metals exceeded the SSSLs; however, those results were below background or within the UBR, except for the following:

- Aluminum (33,800 mg/kg) at HR-80Q-GP24
- Antimony (5.67 to 8.16 mg/kg) in 3 samples
- Arsenic (56.3 mg/kg) at SAR-78-SS07

- Copper (132 mg/kg) at SAR-78-SS07
- Iron (77,100 to 88,000 mg/kg) in 5 samples.

**Lead.** A lead isopleth map for subsurface soils using the human health SSSL (400 mg/kg), the industrial SSSL (880 mg/kg), and the recreational site user SSSL (7,600 mg/kg) is presented on Figure 3-2. The lead concentrations in the subsurface soil samples at Parcels 77Q, 78Q, 80Q, and 85Q ranged from less than 0.37 mg/kg to 43,100 mg/kg.

As shown on Figure 3-2, the highest concentrations of lead were generally found in impact areas located along the hillside south of the firing lines. The area that exceeded the residential human health SSSL for lead extended from the eastern portion of Range 21 between the 75 and 300-meter target lines, narrowing to the west only to encompass the hillsides located south of Range 22 and 27. Within this area there are three zones that have lead concentrations that exceed the industrial SSSL. One of these zones is located in the central portion of Range 21, near the 175-meter target line; the second and third zones are located along hillsides in the central portions of Range 22 and 27. The highest concentrations of lead in the subsurface soil are found along the hillside in the central portion of Range 22 (Figure 3-2). The lead results at two sample locations (SAR-78-SS25 and SAR78-SS36) within this area exceed the recreational site user SSSL (7,600 mg/kg) with concentrations of 31,100 and 43,100 mg/kg, respectively.

An isolated area of high lead concentrations in subsurface soil was also noted south of Parcel 80Q. Within this area two subsurface soil sample locations had lead concentrations that exceeded the residential human health SSSL, and one sample location (HR-80Q-GP18) had a lead concentration that exceeded the industrial SSSL (Figure 3-2).

**VOCs.** Four VOCs were detected in BGR range subsurface soil samples. All results were less than their SSSLs.

**SVOCs.** Fifteen SVOCs were detected in BGR range subsurface soil samples. Twelve of the 15 SVOCs were PAH compounds. One result, the benzo(a)pyrene at SAR-77-SS02 (0.12 mg/kg, estimated), exceeded its SSSL (0.09 mg/kg).

**Chlorinated Pesticides.** One chlorinated pesticide compound, 4-4'-DDT, was detected in sample HR-80Q-GP21 but it did not exceed its SSSL.

**Chlorinated Herbicides.** One chlorinated herbicide compound, MCPP, was detected in sample HR-80Q-GP21 but it did not exceed its SSSL.

No subsurface soil samples contained detectable concentrations of organophosphorus pesticide compounds, nitroaromatic explosive compounds, or PCBs.

### **3.2.3 Sediment**

A total of 38 sediment samples were collected from Cane Creek and its tributaries at the BGR ranges. The analytical data from these samples are compared to the SSSLs, ESVs, and background screening values in Table 3-4.

**TAL Metals (Except Lead).** Twenty TAL metals (other than lead) were detected in sediment samples collected at the BGR ranges. None of these metals concentrations exceeded their SSSLs.

Two of the metals exceeded their ESVs and exceeded the background screening values:

- Antimony (19.1 mg/kg) at SAR-78-SW/SD10
- Copper (59.8 to 738 mg/kg) in 7 samples.

Four metals that do not have ESVs established were detected at concentrations that exceeded background: barium, iron, manganese, and thallium. One result, barium at HR-80Q-SW/SD04 (438 mg/kg), exceeded the range of background values (UBR is 272 mg/kg).

**Lead.** A map depicting lead concentrations in sediment at the BGR ranges is presented on Figure 3-3. In addition to the BGR range sediment data, sediment data collected during the site investigation at the mortar impact areas south of BGR, Parcel 138Q-X, were included in this figure to provide additional perspective. Lead concentrations were compared to the ESV (30.2 mg/kg), recreational site user SSSL (400 mg/kg), and industrial SSSL (880 mg/kg). On Figure 3-3, stream reaches containing sediment samples with lead concentrations exceeding the ESV are depicted in yellow, reaches with sediment samples exceeding the recreational user SSSL are depicted in orange, and reaches with sediment samples exceeding the industrial SSSL are depicted in red.

The lead concentrations in the sediment samples at the BGR ranges, Parcels 80Q, 77Q, 78Q, and 85Q ranged from 5.7 to 3,280 mg/kg. In general, low concentrations of lead in sediment are found in the upper reaches of the tributaries to Cane Creek. Concentrations of lead in sediment samples increase in Cane Creek and the lower reaches of unnamed streams B, C, D, E, F, and G, where most of the range activity has historically taken place. Concentrations of lead in sediment generally increase to levels exceeding the ESV in the stream segments that cross or are located adjacent to the BGR ranges. Lead concentrations in sediment exceeding the recreational site

user SSSL are generally found adjacent to the stream reaches that flow through the BGR range impact zone. The stream segments with the highest concentrations of lead in sediment (i. e., exceeding the industrial site user SSSL) are generally located adjacent to bullet impact areas. These segments include the portion of Cane Creek near the hillside south of the firing line for Range 22, the portion of unnamed stream A near a target area at Range 27, and along unnamed stream E near the 300-meter target line for Range 21.

**VOCs.** Seven VOCs were detected in BGR range sediment samples. No concentrations exceeded the SSSLs. One VOC, trichlorofluoromethane, was detected four sediment samples at estimated concentrations (0.0032 to 0.0047 mg/kg) that exceeded its ESV (0.003 mg/kg).

**SVOCs.** Two SVOCs, bis(2-ethylhexyl)phthalate and di-n-butyl phthalate, were detected in two sediment samples from the BGR ranges. No concentrations exceeded the SSSLs or ESVs.

**Chlorinated Pesticides.** Three chlorinated pesticide compounds, gamma-chlordane, alpha-BHC, and 4,4'-dichlorodiphenyldichloroethane (DDD), were detected in four BGR range sediment samples. No concentrations exceeded the SSSLs or ESVs.

**Chlorinated Herbicides.** Two chlorinated herbicide compounds, MCPP and 2,4-DB, were detected in two BGR range sediment samples, but they did not exceed the SSSLs.

**Nitroaromatic Explosive Compounds.** One nitroaromatic explosive compound, 2,4-DNT was detected at HR-80Q-SW/SD03. Its estimated concentration (0.14 mg/kg) exceeded the ESV (0.075 mg/kg) but was below the SSSL.

**TOC.** All sediment samples were analyzed for TOC. TOC concentrations ranged from 23.5 mg/kg to 42,700 mg/kg.

**Grain Size.** All sediment samples were analyzed for grain size. Appendix F contains the results of the grain size analysis.

No sediment samples contained detectable concentrations of organophosphorus pesticide compounds, PCBs, cyanide, or perchlorate.

### 3.2.4 Surface Water

A total of 33 surface water samples were collected from Cane Creek and its tributaries at the BGR ranges. The analytical data from these samples are compared to the SSSLs, ESVs, and background screening values in Table 3-5. Table 3-8 summarizes the results of the field-measured water quality parameters.

**TAL Metals (Except Lead).** Eighteen TAL metals (other than lead) were detected in surface water samples collected at the BGR ranges. Metals concentrations at HR-80Q-SW/SD03 (arsenic) and SAR-85-SW/SD09 (iron) exceeded their SSSLs but were below the UBR. Two thallium results at HR-80Q-SW/SD06 and SAR-78-SW/SD10 exceeded both the SSSL and the range of background values (0.0042 milligrams per liter [mg/L]).

Three TAL metals (other than lead) exceeded their ESVs and the background screening values:

- Copper (.0728 to 0.129 mg/L) in 4 samples
- Sodium (722 mg/L) at HR-85Q-SW/SD02
- Thallium (0.00711 to 0.0075 mg/L) in 2 samples.

**Lead.** A map depicting lead concentrations in surface water at the BGR ranges, Parcels 77Q, 78Q, 80Q, and 85Q, is presented on Figure 3-4. In addition to the BGR range surface water data, surface water and seep sample data collected during the site investigation at the mortar impact areas south of BGR, Parcel 138Q-X, were included in this figure to provide additional perspective. Lead concentrations were compared to its ESV (0.00132 mg/L), recreational site user SSSL (0.01500 mg/L), and upper background value (0.047 mg/L) as presented in the *Final Background Metals Survey Report* for FTMC (SAIC, 1998). On Figure 3-4, stream reaches containing surface water samples with lead concentrations exceeding the ESV are depicted in yellow, reaches with surface water samples exceeding the recreational site user SSSL are depicted in orange, and reaches with surface water samples exceeding the upper background value are depicted in red.

Based on the distribution of lead in surface water, it appears that, as the surface water flows through the BGR ranges, the concentration of lead in the water generally increases.

In general, Figure 3-4 shows that low concentrations of lead in surface water, exceeding the ESV (depicted in yellow), are found in the upper reaches of the tributaries to Cane Creek.

Concentrations of lead in surface water samples from the upper reach of Cane Creek, the lower reaches of unnamed streams F and G, and near the confluence of unnamed streams B, C, and D

within Cane Creek exceed the recreational site user SSSL (depicted in orange). However, the highest concentrations of lead in surface water (depicted in red) were found in Cane Creek from the central portion Range 22 to western end of the area of investigation and in unnamed stream A located south of the firing line for Range 27. Most of the lead concentrations in the surface water samples collected from this portion of Cane Creek and unnamed stream A exceeded the recreational site user SSSL and the upper background value. Figure 3-4 and Table 3-5 show that the extent of lead in surface water exceeding the upper background value and the SSSL has not been defined west of the area of investigation (i.e., west of Range 27).

**VOCs.** Two VOCs (methylene chloride and acetone) were detected in eleven BGR range surface water samples. All of these results were flagged with the "B" data qualifier, indicating that the same compounds were also detected in the associated blanks. No concentrations exceeded the SSSLs or ESVs.

**Chlorinated Pesticides.** One chlorinated pesticide compound, 4,4'-DDE, was detected at HR-78Q-SW/SD01. Its concentration did not exceed the SSSL or ESV.

**Nitroaromatic Explosive Compounds.** One nitroaromatic explosive compound, 2-nitrotoluene, was detected at HR-80Q-SW/SD02. Its concentration did not exceed the SSSL or ESV.

No surface water samples contained detectable concentrations of SVOCs, chlorinated herbicides, organophosphorus pesticide compounds, PCBs, cyanide, or perchlorate.

### **3.2.5 Groundwater**

A total of 17 groundwater samples were collected at the BGR ranges. The analytical data from these samples are compared to the SSSLs and background screening values in Table 3-6. Table 3-8 summarizes the results of the field-measured water quality parameters. Table 3-9 is a summary of the monitoring well construction details.

**Metals.** Seventeen metals were detected in groundwater samples collected at the BGR ranges. Lead was detected in five locations at concentrations below the SSSL and background. The concentrations of eight metals (aluminum, arsenic, barium, chromium, iron, manganese, selenium, and thallium) exceeded SSSLs; however, all the concentrations were either below their respective background concentration or within the UBR.

Arsenic was detected at two locations (HR-78Q-MW01 and HR-78Q-MW02). The arsenic result from HR-78Q-MW01 was flagged with a “J” data qualifier, signifying that the compound was positively identified, but the concentration is estimated. The arsenic result from HR-78Q-MW02 was flagged with a “B” data qualifier, signifying that the metal was also detected in an associated laboratory or field blank sample.

Chromium was detected at one location (HR-77Q-MW02). The result was flagged with a “B” data qualifier, signifying that the compound was also detected in an associated laboratory or field blank sample.

Selenium was detected at one location (HR-80Q-MW02). The result was flagged with a “B” data qualifier, signifying that the compound was also detected in an associated laboratory or field blank sample.

Thallium was detected at one location (HR-77Q-MW02). The result was flagged with a “B” data qualifier, signifying that the compound was also detected in an associated laboratory or field blank sample.

**VOCs.** Six VOCs, including acetone, bromodichloromethane, chloroform, dibromochloromethane, and methylene chloride, were detected in groundwater samples collected at the BGR ranges. All of the detected concentrations were below their respective SSSLs. VOCs were detected at only three sample locations (HR-7Q-MW02, HR-77Q-MW04, and HR-85Q-MW01). All of the detected compounds were either flagged with a “B” data qualifier, signifying that the compound was also detected in an associated laboratory or field blank sample, or were flagged with a “J” data qualifier, signifying that the compound was positively identified, but the concentration is estimated.

**Explosives.** Four explosives (2,4-DNT; 2-amino-2,6-DNT; 2-nitrotoluene; and 4-amino-2,6-DNT) were detected at six locations (HR-77Q-MW01, HR-77Q-MW02, HR-77Q-MW03, HR-77Q-MW04, HR-80Q-MW06, and HR-85Q-MW01). Four of the detected concentrations exceeded the SSSLs. 2,4-DNT (0.0002 mg/L) exceeded its SSSL at one location (HR-85Q-MW01). 2-amino-2,6-DNT (0.00032 mg/L) exceeded the SSSL at one location (HR-77Q-MW01). 4-amino-2,6-DNT (0.00021 to 0.002 mg/L) exceeded its SSSL at four locations (HR-77Q-MW02, HR-77Q-MW04, HR-80Q-MW06, and HR-85Q-MW01). All the detected concentrations exceeding the SSSLs (except 4-amino-2,6-DNT at HR-80Q-MW06) were flagged with a “J” data qualifier, signifying that the compounds were positively identified, but the concentrations are estimated.

### **3.2.6 Seep Water**

One seep sample, SAR-85-SEEP01, was collected at the BGR ranges. The analytical data from this sample are compared to the SSSLs and background screening values in Table 3-7.

**Metals.** Seven metals were detected in the seep sample, including aluminum, barium, calcium, iron, magnesium, manganese, and sodium. Two of these, aluminum and barium, exceeded their ESVs, but both values were less than the UBR.

The seep sample did not have any detectable concentrations of VOCs, SVOCs, chlorinated pesticides, organophosphorus pesticides, or chlorinated herbicides.

### **3.2.7 Safety Fan Surface Soil (XRF Survey)**

An XRF survey of 40 surface soil locations was conducted for lead analysis within the safety fans for Range 21, Range 22, and Range 27 (Table 3-10, Figure 3-5). Ten percent of the samples (4 samples) were selected for confirmation analysis for lead at the fixed-base laboratory using EPA Method 6010B (Table 3-11). The confirmation data verified the results of the XRF measurement. The results indicate that lead concentration levels within the range fans are generally within the range of background (40 to 83 mg/kg) (SAIC, 1998).

## **3.3 Summary of Findings**

Chemical analysis of samples collected at the BGR ranges indicates that various metals, especially lead, were detected in various environmental media sampled. VOCs, SVOCs, chlorinated pesticides, chlorinated herbicides, and nitroaromatic explosive compounds were also detected. Analytical results were compared to the SSSLs and ESVs for FTMC that were developed by IT for human health and ecological risk evaluations as part of the ongoing site investigation process performed under the BRAC Environmental Restoration Program at FTMC. Additionally, metals and PAH compound concentrations that exceeded SSSLs and ESVs were compared to medium-specific background screening values (SAIC, 1998).

Although the BGR ranges are projected to be used for passive recreation, the environmental sampling data were screened against residential human health SSSLs to evaluate the site for unrestricted land reuse. Surface water and sediment data were compared to recreational site user SSSLs. With the exceptions of benzo(a)pyrene in two surface soil samples and one subsurface soil sample and nitroaromatic explosive compounds in five monitoring wells, no organic compounds were detected at concentrations that exceeded their SSSLs. Several metals exceeded their SSSL screening values and the range of background values. These metals include lead, antimony, arsenic, copper, iron, and zinc in surface soils and lead, aluminum, antimony, arsenic

copper, and iron in subsurface soils. Lead also exceeded its SSSL and range of background values in sediment. In surface water, lead and thallium exceeded their SSSLs and the range of background values.

Ecological risk was evaluated by screening the analytical results against ESVs. With the following exceptions, no other organic compounds were detected that exceeded ESVs:

- The SVOC benzo(a)pyrene in one surface soil sample
- Five chlorinated pesticides in 3 surface soil samples
- The herbicide MCPP in three surface soil samples and one sediment sample
- The VOC trichlorofluoromethane in 4 sediment samples
- The nitroaromatic explosive 2,4-DNT in one sediment sample.

The following metals exceeded both ESVs and the range of background screening values:

- Surface soils: Lead, antimony, arsenic, beryllium, copper, iron, selenium, silver, zinc
- Sediment: Lead, antimony, copper
- Surface water: Lead, copper, sodium, thallium.

Barium and thallium were detected above background concentrations in sediment samples at BGR. Neither of these two metals has an established ESV in the sediment matrix. Cobalt was detected in three surface water samples at concentrations that exceeded its ESV; however, no background value exists for cobalt in the surface water matrix.

Based on the results of these investigations, past operations at the BGR ranges appear to have impacted the environment. The concentrations of metals and organic compounds detected in the various site media may pose an unacceptable risk to human health and the environment. IT recommends additional assessment and a possible constituent source removal action at the BGR ranges.

## 4.0 Screening-Level Ecological Risk Assessment

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### 4.1 Introduction

The ranges on BGR include Range 21 (Parcel 77Q), Range 22 (Parcel 78Q), Range 24 Upper (Parcel 80Q), and Range 27 (Parcel 85Q). In order to determine the potential for ecological risks posed by site-related chemicals at the ranges on BGR at FTMC, a SLERA was conducted. This SLERA consists of a description of the habitat(s) in and around the BGR ranges, a discussion of the constituents detected in samples collected from environmental media at the BRG ranges, a discussion of the conceptual site model, an estimation of the screening-level risk, the identification of the constituents of potential ecological concern (COPEC), an uncertainty analysis, a discussion of the different lines of evidence, and a summary of the results and conclusions.

### 4.2 Environmental Setting

The terrestrial habitat occurring at the BGR ranges falls into two general categories: “cleared” areas and forested areas. The “cleared” areas 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, 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, the grass and herbaceous species will be followed by shrubs and small trees. The early old field successional areas at the BGR ranges are dominated by various grasses and herbs, including *Rumex spp.* (dock), *Trifolium spp.* (clover), *Astragalus spp.* (vetch), *Asclepias spp.* (milkweed), *Galium spp.* (bed straw), *Chrysanthemum leucanthemum* (ox-eye daisy), and *Sorghum halepense* (Johnson grass). Other old field herbaceous species occurring at the BGR ranges are *Rubus occidentalis* (black raspberry), *Toxicodendron radicans* (poison ivy), *Rubus glabra* (smooth sumac), *Smilax rotundiflora* (green brier), *Lonicera japonica* (Japanese honeysuckle), *Vitis labrusca* (fox grape), and *Rosa multiflora* (multiflora rose). Loblolly pine (*Pinus taeda*) saplings have also begun to encroach on the formerly maintained grassy areas of the BGR 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 BGR 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 vines (*Vitis rotundifolia*) are also present in this area.

Terrestrial species that may inhabit the area of the BGR 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 wood rat, and hispid cotton rat), and eastern cottontail. Approximately 200 avian species reside at FTMC at least part of the year (USACE, 1998). Common species expected to occur in the vicinity of the BGR ranges include northern cardinal (*Cardinalis cardinalis*), northern mockingbird (*Mimus polyglottus*), warbler (*Dendroica spp.*), indigo bunting (*Passerina cyanea*), red-eyed vireo (*Vireo olivaceus*), American crow (*Corvus brachyrhynchos*), bluejay (*Cyanocitta cristata*), several species of woodpeckers (e.g., *Melanerpes spp.*, *Picoices spp.*), and Carolina chickadee (*Parus carolinensis*). Game birds present in the vicinity of the BGR 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 (September 2000) 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. Due to the presence of Cane Creek, piscivorous bird species may also be present in the vicinity of the BGR ranges. These piscivorous birds may 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 BGR ranges are copperhead (*Agkistrodon contortix*), king snake (*Lampropeltis getulus*), black racer (*Coluber*

*constrictor*), fence lizard (*Sceloporous undulatus*), and six-lined racerunner (*Cnemidophorous sexlineatus*).

Descriptions of the habitats at each of the ranges on Bains Gap Road are presented in the following sections.

#### **4.2.1 Range 21 Habitat**

Range 21 consists of two main habitat types: “cleared” and forested areas. The cleared area comprises the vast majority of Range 21. The entire area of Range 21, including the extensive safety fan, is approximately 2,249 acres. The study area of Range 21 is approximately 15 acres in size and is topographically relatively flat. It is bounded on the north by Bains Gap Road, on the east and south by mixed deciduous/coniferous forest, and on the west by Range 22. A soil berm separates Range 21 from Range 22 to the west. The study area of Range 21 consists almost entirely 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. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. Significant portions of Range 21 remain unvegetated, with large areas of bare soil.

The forested areas to the east and south of Range 21 are best characterized as mixed deciduous/coniferous forest. Scrub 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.

Cane Creek flows east-to-west across Range 21 towards Range 22 and through a large concrete culvert beneath the soil berm that separates Range 21 from Range 22 to the west. Several small tributaries also flow across the southern portion of Range 21 and along the eastern and western boundaries of Range 21. Cane Creek is relatively narrow (4 to 6 feet) and shallow (0.5 to 1.0 feet) along its length within Range 21, with steep embankments approximately four feet high. The substrate of Cane Creek is mostly boulders and cobbles, with a few small depositional areas with sand substrate. The water level in Cane Creek is highly variable, depending on the amount of precipitation received by the local watershed. Cane Creek is a perennial creek and, as such, maintains water flow even during periods of drought. The vegetative canopy of the Cane Creek corridor within Range 21 is characterized by low-level shrubs and tree saplings (less than 8 feet high) that form a low, dense canopy over the creek. This vegetation extends less than six feet from the creek bed itself. Thus this vegetative canopy is narrow, low, and dense. Because Cane

Creek bisects Range 21 between the firing line and several of the target areas, vegetation along Cane Creek was previously maintained at a low level so that the target areas would not be obstructed.

The portion of Cane Creek that flows through Range 21 has been identified as low-quality foraging habitat for the federally listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). This section of Cane Creek has been identified as a gray bat foraging area because it provides habitat for aquatic insects, which are fed upon by the gray bat. However, the gray bat requires continuous cover while traveling to and from its foraging habitats and while foraging. Due to historical maintenance activities along Cane Creek, the forest canopy has been eliminated and has only recently been replaced by low-lying shrubs and saplings. Thus, the currently existing vegetative cover along the Cane Creek corridor may not provide the cover favored by gray bats. In the future, growth of a high canopy of trees along the Cane Creek corridor may improve the gray bat foraging habitat potential.

#### **4.2.2 Range 22 Habitat**

Range 22 consists of two main habitat types: “cleared” and forested areas. The cleared area comprises the vast majority of Range 22. The total area of Range 22, including the extensive safety fan, is 1,810 acres. The main study area of Range 22 is approximately 12.5 acres and is topographically relatively flat. A rocky soil berm forms the southern boundary of the main study area. Range 22 is bounded on the north by Bains Gap Road, on the east by Range 21, on the south by mixed deciduous/coniferous forest, and on the west by Range 27. A soil berm separates Range 22 from Range 27 to the west. The study area of Range 22 consists almost entirely of formerly maintained lawns, 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. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. Significant portions of Range 22 remain unvegetated, with large areas of bare soil.

The forested area south of Range 22 is best characterized as mixed deciduous/coniferous forest. Scrub 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.

Cane Creek flows east-to-west across Range 22 towards Range 27. A small tributary that originates southeast of Range 22 flows into Cane Creek at Range 22. Another small tributary

that originates north of Range 22 flows into Cane Creek between Ranges 22 and 27. Cane Creek is relatively narrow (4 to 6 feet) and shallow (0.5 to 1.0 feet) along its length within Range 22. The northern bank of Cane Creek is relatively steep and approximately four feet high. The southern bank of Cane Creek is very steep and rises significantly to an elevation of approximately 330 feet above the firing line elevation. This southern bank effectively forms the impact zone for the majority of Range 22. The substrate of Cane Creek is mostly boulders and cobbles, with a few small depositional areas with sand substrate. The water level in Cane Creek is highly variable, depending on the amount of precipitation received by the local watershed. Cane Creek is a perennial creek and, as such, maintains water flow even during periods of drought.

The vegetation adjacent to Cane Creek within Range 22 is sporadic, and where it is present is best characterized by low-level shrubs and tree saplings (less than 8 feet high). This vegetation extends less than six feet from the creek bed to the north. On the southern side of the creek, the bank rises steeply and is largely devoid of vegetation. Because Cane Creek flows along the southern boundary of the impact zone, vegetation along Cane Creek was previously maintained at a low level so that the target areas would not be obstructed.

The portion of Cane Creek that flows through Range 22 has been identified as low-quality foraging habitat for the federally listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). This section of Cane Creek has been identified as a gray bat foraging area because it provides habitat for aquatic insects, which are fed upon by the gray bat. However, the gray bat requires continuous cover while traveling to and from its foraging habitats and while foraging. Due to historical maintenance activities along Cane Creek, the forest canopy has been eliminated and has only recently been replaced by low-lying shrubs and saplings. Thus, the currently existing vegetative cover along the northern border of the Cane Creek corridor may not provide the cover favored by gray bats. In the future, growth of a high canopy of trees along the Cane Creek corridor may improve the gray bat foraging habitat potential.

#### **4.2.3 Range 24 Upper Habitat**

Range 24 Upper consists of two main habitat types: “cleared” and forested areas. The forested area comprises the northern half of Range 24 Upper. The southern half is a “cleared” area. The total area of Range 24 Upper is approximately 11 acres, and there is no defined safety fan. The main study area of Range 24 Upper is on a south-facing slope immediately south of Bains Gap Road. It is bounded on the north by Bains Gap Road, on the east and south by mixed deciduous /coniferous forest, and on the west by Range 21. The northern half of the site slopes from an elevation of approximately 1,050 feet amsl to 975 feet amsl. This south-facing hillside is best

characterized as mixed deciduous/coniferous forest. Scrub 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. At the base of this slope is a cleared area that is best characterized as old field, early successional habitat. Various grasses and herbaceous species dominate this habitat type. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these areas. Significant portions of the southern half of Range 24 Upper remain unvegetated, with large areas of bare soil.

Two small, ephemeral tributaries of Cane Creek occur at the base of the slope and in the cleared area south of the hill at Range 24 Upper. These tributaries only have water after significant rainfall events and are dry most of the year. Their substrates are cobbles and boulders, and there are very few areas with sandy deposits. These tributaries run east-to-west across Range 24 Upper and join west of the range to form a portion of the headwaters of Cane Creek. These tributaries have not been identified as being capable of supporting gray bat habitat (Garland, 1996).

#### **4.2.4 Range 27 Habitat**

Range 27 consists of two main habitat types: “cleared” and forested areas. The cleared area comprises the vast majority of Range 27. The total area of Range 27, including the extensive safety fan, is 954 acres. The main study area of Range 27 is approximately 16 acres and is topographically relatively flat. A rocky soil berm forms the southern boundary of the main study area. Range 27 is bounded on the north by Bains Gap Road, on the east by Range 22, and on the south and west by mixed deciduous/coniferous forest. A soil berm separates Range 27 from Range 22 to the east. The study area of Range 27 consists almost entirely of formerly maintained lawns, 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. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. Significant portions of Range 27 remain unvegetated, with large areas of bare soil.

The forested area south of Range 27 is best characterized as mixed deciduous/coniferous forest. Scrub 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.

Cane Creek flows east-to-west across the southern boundary of Range 27. A small tributary that originates southeast of Range 27 flows into Cane Creek at Range 27. Cane Creek is relatively narrow (4 to 6 feet) and shallow (0.5 to 1.0 feet) along its length within Range 27. The northern bank of Cane Creek is gently sloping to a height of approximately three feet. The southern bank of Cane Creek is very steep and rises significantly to an elevation of approximately 330 feet above the firing line elevation. This southern bank effectively forms the impact zone for the majority of Range 27. The substrate of Cane Creek is mostly boulders and cobbles, with a few small depositional areas with sand substrate. The water level in Cane Creek is highly variable, depending on the amount of precipitation received by the local watershed. Cane Creek is a perennial creek and, as such, maintains water flow even during periods of drought.

The vegetation adjacent to Cane Creek within Range 27 is sporadic, ranging from areas completely devoid of vegetation to areas with relatively mature forest canopy. The western portion of the Cane Creek corridor at Range 27 is mostly devoid of vegetation, while the eastern portion exhibits mature forest vegetation. On the southern side of the creek, the bank rises steeply and the vegetation is mostly brush and weed species for a distance of approximately 50 feet until it transitions to a mixed deciduous/coniferous forest.

The portion of Cane Creek that flows through Range 27 has been identified as low-quality foraging habitat for the federally listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). This section of Cane Creek has been identified as a gray bat foraging area because it provides habitat for aquatic insects, which are fed upon by the gray bat. However, the gray bat requires continuous cover while traveling to and from its foraging habitats and while foraging. The forest canopy along the Cane Creek corridor at Range 27 is sporadic and not continuous. Thus, the currently existing vegetative cover along the Cane Creek corridor may not provide the cover favored by gray bats. In the future, growth of a high canopy of trees along the Cane Creek corridor may improve the gray bat foraging habitat potential.

#### **4.2.5 Cane Creek Habitat**

Cane Creek in the vicinity of the BGR ranges is a perennial stream that flows east-to-west across the ranges at Bains Gap Road. The physical characteristics of Cane Creek at the BGR ranges are relatively consistent; however, they differ both upstream and downstream of the BGR ranges. The BGR ranges lie within an east-west trending valley that is formed by Jones Hill, Mount Tylo, several unnamed hills north of the ranges, and Marcheta Hill and several unnamed hills south of the ranges. Upstream (one-half to three-quarters of a mile east) of the BGR ranges, the headwaters of Cane Creek are formed by several small tributaries that are created by runoff from the hills north, south, and east of the ranges. These headwater streams are small ephemeral

streams with boulder and cobble substrate that carry runoff during storm events but are dry during significant portions of the year. The headwater areas are relatively undeveloped portions of the Main Post and are almost entirely mixed deciduous/coniferous forest.

Downstream (west) of the BGR ranges, Cane Creek continues to flow in a westerly direction across the developed portion of the Main Post (including the Cane Creek golf course) and off site along the west-northwest boundary of the Main Post.

In general, the portion of Cane Creek that flows through the BGR ranges is a low-gradient perennial stream with widths ranging from 4 to 10 feet and depths ranging from 0.5 to 2.0 feet. The banks of Cane Creek are steep (4 to 8 feet) and exhibit erosional features characteristic of occasional high-velocity flow (i.e., during significant storm events). The substrate of Cane Creek is mostly cobbles and boulders. There is very little evidence of organic matter present as substrate in Cane Creek in the vicinity of the BGR ranges. In fact, large sections of the creek bed in this area are made up of exposed bedrock.

The vegetation surrounding Cane Creek at the BGR ranges is variable. Because Cane Creek bisects these ranges, routine maintenance activities historically controlled or eliminated the vegetation along the creek banks. Since maintenance activities have ceased, vegetative species have begun to recolonize the creek banks. Therefore, weeds, low-lying shrubs, and tree saplings dominate the creek banks. The area directly north of Cane Creek is best characterized as old field early successional habitat, and the area directly south of Cane Creek is mixed deciduous /coniferous forest, except for Range 21. Cane Creek flows through the center of Range 21 the habitat on both the north and south sides of Cane Creek at Range 21 is characterized as old field, early successional.

The headwaters of Cane Creek are formed by runoff from the hills north, east, and south of the BGR ranges. There also appears to be localized contribution to the creek flow (mainly in the lower reaches) 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. Flow in Cane Creek is highly variable, depending on precipitation in the surrounding watershed.

Although relatively shallow (less than two feet deep) over its entire length at the BGR ranges, Cane Creek has the potential to support a variety of amphibious species and some small fish species. Bullfrog (*Rana catesbeiana*) and leopard frog (*Rana sphenoccephala*) are examples of amphibians that may be found in Cane Creek in the vicinity of the BGR ranges. Fish species that

may be found in Cane Creek in the vicinity of the BGR 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 Cane Creek 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 Cane Creek due to its shallow nature.

Cane Creek in the area of the BGR ranges has been identified as providing low quality foraging habitat for the federally listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Two major requirements for gray bat foraging habitat are contiguous forest cover and habitat for aquatic insects (one of the gray bat's preferred dietary items). Although aquatic insects may be present in Cane Creek at the BGR ranges, the forest canopy is sporadic and may not provide the cover required by gray bats. In the future, growth of a high canopy of trees along the Cane Creek corridor may improve the gray bat foraging habitat potential.

#### **4.2.6 Wetland/Seep Habitat**

The wetland/seep habitat present in the vicinity of the BGR Ranges is limited to the area south of Range 21. This area is known as the Marcheta Hill Orchid Seep special interest natural area (SINA). SINAs at FTMC are those biological communities that harbor federal, candidate, or state-listed species and those habitats containing single or groups of unique or unusual species. SINAs have been identified at Fort McClellan at both the regional and community levels. Additional information regarding SINAs and their management practices at Fort McClellan is presented in the *Endangered Species Management Plan for Fort McClellan, Alabama* (Garland, 1996). The only SINA that could potentially be impacted by activities (both past and future) at the BGR ranges is the Marcheta Hill Orchid Seep SINA, as it is located directly adjacent to Range 21. The Bains Gap Seep SINA is located approximately 2,000 feet northeast of the closest BGR range (Range 24 Upper) and is also "upstream" of all of the BGR ranges. Surface water and sediment contaminants from the BGR ranges cannot impact the Bains Gap Seep SINA; and, based on the distance separating the BGR ranges and the Bains Gap Seep SINA, it is highly unlikely that soil contaminants at the BGR ranges could impact the Bains Gap Seep SINA. Figure 4-1 presents the relative locations of the Marcheta Hill Orchid Seep SINA, Bains Gap Seep SINA, and the BGR ranges.

The spring seepage to the west of Marcheta Hill constitutes one of the more important SINAs on the Main Post at FTMC. The boundary of the wetland seep encompasses approximately 7.2 acres; however, the integrity of the adjacent watershed is critical to the maintenance of this seep. The area is located directly south of Range 21. This wetland is the largest forested seepage on

the installation and contains two federal candidate 2 species: white fringeless orchid (*Plantanthera integrilabia*) and Diana butterfly (*Speyeria diana*). The population of white fringeless orchid is particularly significant, with over 250 individuals recorded. Additional plants on the Alabama Natural Heritage Program tracking list include rose pink (*Sabatia capitata*) and soapwort gentian (*Gentiana saponaria*).

The ecological significance of this wetland has been recognized for several years. “Do Not Disturb Endangered Species Area” signs have been posted along the wetland’s boundary. The continuation of the existing fire regime is considered the most critical management requirement. According to verbal accounts, this area experienced a wildfire at least once every two years while the facility was active. Many of these wildfires were due to the training activities that took place at Range 21. In order to ensure this fire frequency in the future, the management plan for this area prescribes that a burn will be instituted if the area has not experienced a fire by March 1 of the second year. This permissive burn policy concerning wildfires will benefit this wetland area.

#### **4.3 Constituents Detected on Site**

The sampling and analysis programs conducted at the BGR ranges were designed based on a number of factors, including:

- Site history
- Results of the EBS
- Results of previous sampling and analysis programs.

The sampling and analysis programs at the BGR ranges are described in Section 3.1 of this report. Constituents detected in surface soil at the BGR ranges and in sediment and surface water in Cane Creek adjacent to the BGR ranges are summarized in Sections 3.2 and 3.3 of this report.

In general, inorganic constituents were commonly detected in soils, but organic compounds (i.e., SVOCs and chlorinated pesticides) were less frequently detected and at relatively low concentrations. Antimony, copper, lead, and zinc were the most commonly detected inorganic compounds and were detected at elevated concentrations (with respect to ecological screening values) in surface soil at all of the BGR ranges. At Range 21, antimony, copper, and zinc were detected at elevated concentrations in three out of nine surface soil samples. In addition to the elevated concentrations of antimony, copper, and zinc in surface soil at Range 21, arsenic and selenium were also detected at elevated concentrations in surface soil. Arsenic was detected in

two samples out of nine at elevated concentrations. Selenium was detected in one sample out of nine at a concentration that slightly exceeded its ESV.

Three PAH compounds (benzo[a]pyrene, fluoranthene, and pyrene) were detected in surface soil at Range 21. These compounds were infrequently detected (one or two detections out of nine samples), and the estimated concentrations were relatively low. The detected PAHs were found in only two samples from Range 21 located near the firing line. The chlorinated pesticide 4,4'-DDE was also detected in a single surface soil sample from Range 21, albeit at an estimated concentration that only slightly exceeded its ESV.

As with the other ranges at BGR, surface soil at Range 22 exhibited elevated concentrations of antimony, copper, lead, and zinc with respect to ESVs. Antimony was detected at elevated concentrations in two surface soil samples, and zinc was detected at an elevated concentration in one sample. Beryllium was also detected in a single surface soil sample at a slightly elevated concentration. Fluoranthene and pyrene were also detected in a single surface soil sample at slightly elevated concentrations. The only other compound that was detected in surface soil from Range 22 at an elevated concentration was 4,4'-DDT, which was detected in a single sample at a slightly elevated concentration. The elevated concentrations of beryllium, fluoranthene, pyrene, and 4,4'-DDT were all from three samples collected adjacent to the firing line at Range 22.

Range 24 Upper exhibited elevated concentrations of a number of inorganic compounds in surface soil. As was the case with the other ranges at BGR, antimony, copper, lead, and zinc were commonly detected and exhibited elevated concentrations with respect to ESVs. Additionally, barium, chromium, manganese, and mercury were detected in one sample out of 38 samples collected, at concentrations that exceeded either their respective ESVs or background threshold values. Aluminum was detected in five samples at elevated concentrations, selenium was detected in four samples at elevated concentrations, beryllium and iron were detected in four samples at elevated concentrations, and silver was detected in two samples at elevated concentrations.

The chlorinated herbicide MCPP was detected at elevated concentrations in three of five surface soil samples at Range 24 Upper. The chlorinated pesticides aldrin, alpha-BHC, and endrin were detected in a single sample at concentrations that slightly exceeded their respective ESVs.

Surface soil at range 27 exhibited elevated concentrations of a number of inorganic compounds. As was the case with the other ranges at BGR, antimony, copper, lead, and zinc were commonly detected in surface soil and exhibited elevated concentrations with respect to ESVs.

Additionally, arsenic, beryllium, manganese, selenium, and silver were detected in one or two surface soil samples at concentrations that exceeded either their respective ESVs or background threshold values. Arsenic, beryllium, manganese, selenium, and silver were detected at elevated concentrations in one or two samples from the hillside that forms the impact zone on the southern boundary of the study area of Range 27. Butylbenzyl phthalate was also detected in a single surface soil sample from the hillside along the southern boundary of Range 27 at a concentration that slightly exceeded its ESV.

Surface water from Cane Creek and its tributaries in the vicinity of the BGR ranges exhibited elevated concentrations of copper and lead in a number of samples. In addition, three of 24 surface water samples exhibited elevated concentrations of cobalt with respect to its ESV. Two of 24 surface water samples exhibited slightly elevated concentrations of thallium.

Sediment from Cane Creek and its tributaries exhibited elevated concentrations of copper and lead in a number of samples. Barium, iron, manganese, and thallium do not have ESVs, but they were detected in several samples at elevated concentrations relative to their background threshold values. Barium was detected in 4 of 21 samples at elevated concentrations, iron was detected in 2 of 21 samples at elevated concentrations, manganese was detected in 2 of 21 samples at elevated concentrations, and thallium was detected in 1 of 21 samples at elevated concentrations, relative to background. Antimony was detected in one sediment sample at a concentration that slightly exceeded its ESV. 2,4-DNT was detected in one sample at an estimated concentration that slightly exceeded its ESV. Trichlorofluoromethane was detected in four sediment samples at estimated concentrations that exceeded its ESV.

Several inorganic constituents were detected in groundwater at concentrations that exceeded the ESVs for surface water. Aluminum, barium, copper, iron, and manganese were detected at concentrations in groundwater that exceeded their surface water ESVs.

In addition to the samples collected and analyzed in a laboratory, soil samples for XRF analysis were collected from 40 gridded locations within the range safety fans of the BGR ranges. The locations of the XRF samples are presented in Figure 3-5. These 40 samples were analyzed in situ using an energy-dispersive portable XRF instrument. Although the XRF instrument will measure a number of metals present in a sample, lead was selected as an indicator of contamination from range-related activities. The results of the XRF analyses are presented in Table 3-10.

Statistical comparisons (t-test, Mann-Whitney u-test) of lead concentrations in “background” soil (SAIC, 1998) to XRF data collected in the safety fans at the BGR ranges indicated statistically significant differences between lead concentrations in these two sample populations. However, all of the XRF lead concentrations from the range safety fans were within the range of “background” lead concentrations, with the exception of two sample locations (BGR-XRF05 and BGR-XRF04). BGR-XRF05 is located in the impact zone of Range 24 Upper, and BGR-XRF04 is immediately adjacent to the impact zone for Range 21. These two sample locations are more representative of the range impact areas than the range safety fans. If the XRF results from these two sample locations are removed from the data set, then the mean lead concentration in soil in the safety fans is equal to the ESV and all of the lead concentrations from the safety fans are within the range of background lead at FTMC (Table 4-1). Based on these data, historical operational records for these ranges indicating no Army activity in the range safety fans, the physical configuration of the range safety fans in relation to the firing areas, and the lack of visual evidence of bullets or bullet fragments, it is suggested that the lead concentrations in soil in the range safety fans are within the range of naturally occurring background concentrations of lead at FTMC and surrounding areas.

#### ***4.4 Site Conceptual Model***

The ecological site conceptual model (SCM), presented in Figure 4-2, is a schematic diagram of possible exposure pathways and the means by which constituents are transported from the primary constituent source(s) to ecological receptors. The exposure scenarios include the sources, environmental transport, partitioning of the constituents amongst various environmental media, potential chemical/biological transformation processes, and identification of potential routes of exposure for the ecological receptors. In this section, the SCM is described in relation to constituent fate and transport properties, the ecotoxicity of the various constituents, potential ecological receptors at the BGR ranges, and the complete exposure pathways expected to exist at the BGR ranges.

##### ***4.4.1 Constituent Fate and Transport***

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

#### **4.4.1.1 Fate and Transport in Soil**

Constituents in surface soil at the BGR ranges have the potential to be transported from their source area to other areas within their respective ranges and to off-site locations by a number of mechanisms, including volatilization, dust entrainment, surface runoff, and infiltration to subsurface soil/groundwater.

Several VOCs were identified in the upper soil horizons at the BGR ranges. These volatile constituents have a high potential to volatilize to the atmosphere and be transported from their source area via air movement. The concentrations of VOCs detected in surface soil at the BGR ranges are low; therefore, this transport mechanism is expected to be insignificant with respect to other transport mechanisms active at these sites. Most of the metals and SVOCs in the surface soil at the BGR ranges are not expected to volatilize to any great extent, with the exception of mercury, which would be expected to volatilize relatively rapidly. Most of the metals and SVOCs in the surface soil at the BGR ranges are closely associated with particulate matter and would be transported from their source areas by fugitive dust generation and entrainment by the wind. Subsequent dispersion by atmospheric mixing could transport particulate-associated constituents to other parts of the BGR ranges and to off-site locations. The generation of fugitive dust and subsequent transport by the wind is potentially a significant transport mechanism at the BGR ranges, based on the presence of unvegetated areas and areas of sparse vegetation within certain areas of these ranges (e.g., impact areas and soil berms).

The transport of surface soil-associated constituents by surface runoff is another potentially significant transport mechanism. Surface soil constituents may be solubilized by rainwater and subsequently transported to drainage ditches, low-lying areas, and Cane Creek via surface runoff. The solubility of inorganics in rainwater is largely dependent upon the pH of the rainwater. Because the rainwater in this region is most likely slightly acidic, the inorganic constituents in surface soil are likely to solubilize to some degree in the rainwater and be subject to transport via runoff. Most of the SVOCs are strongly associated with soil particles and would not solubilize to a large extent. Constituents that may be more strongly bound to particulate matter in surface soil (e.g., SVOCs and some of the inorganics) may be entrained in surface water runoff and transported to drainage ditches, low-lying areas, and Cane Creek via surface runoff. Many of the metals and SVOCs are strongly sorbed to soil particles and could be transported from their source areas via this mechanism.

Constituents in surface soil may be transported vertically to subsurface soils and groundwater via solubilization in rainwater and infiltration. Subsequent groundwater transport to surface water in Cane Creek could result in exposure of aquatic receptors to soil constituents. Migration in this

manner is dependent upon constituent solubility and frequency of rainfall. Although the soil types (sand, stone, and gravel) in the vicinity of the BGR ranges are expected to promote relatively rapid infiltration of rainwater, the less soluble constituents (i.e., SVOCs) found at the BGR ranges are not likely to migrate to any great extent vertically due to their relatively low solubilities. Inorganics in soil at the BGR ranges may migrate vertically due to the acidic nature of the rainwater in this area and the increased solubility of metals that it produces. Based on surface water data from Cane Creek, vertical migration of soil-related constituents with subsequent discharge to surface water is expected to be insignificant compared to other transport mechanisms (e.g., surface runoff).

The transfer of constituents in surface soil to terrestrial plants through root uptake and to terrestrial animals through ingestion and other pathways are potentially significant transfer mechanisms. Many metals are readily absorbed from soil by plants, but they are not biomagnified to a great extent through the food web. There are several exceptions to this, namely, mercury and selenium, which may bioconcentrate and/or biomagnify (Agency for Toxic Substances and Disease Registry [ATSDR], 1999 and 1996a). Many of the SVOCs have the potential to bioaccumulate in lower trophic level organisms (e.g., terrestrial invertebrates), but most higher trophic level animals have the ability to metabolize these compounds rapidly, precluding the potential for bioconcentration (Eisler, 1987a).

VOCs in the surface soil at the BGR ranges are expected to volatilize and/or photolyze rapidly (half-lives of 3 hours to 5 days) when exposed to sunlight (Burrows et al., 1989). The other surface soil constituents (metals, SVOCs, and chlorinated pesticides/herbicides) are expected to remain in the soil relatively unchanged by physical and/or chemical processes for much longer periods of time.

#### **4.4.1.2 Fate and Transport in Surface Water**

In general, constituents present in the surface water associated with the BGR ranges (Cane Creek and tributaries) are the result of erosion and runoff from the ranges. Constituents in surface water at the BGR ranges may be transported from their sources to other areas at the ranges or to off-site locations by the following mechanisms: 1) volatilization; 2) transfer to groundwater; 3) transfer to sediment; and 4) flow downstream. VOCs in surface water would be expected to rapidly volatilize from the water-air interface and be dispersed in the atmosphere. Therefore, transport of VOCs in surface water is not expected to occur for any significant distance.

Water in Cane Creek originates mainly from overland flow from the surrounding watershed. There also appear to be localized and sporadic contributions to creek flow from groundwater

where the potentiometric surface exceeds the creek bed surface. The flow contribution in Cane Creek from groundwater varies according to the amount of precipitation received by the surrounding watershed, with an increase in groundwater contribution when precipitation raises the potentiometric surface. Thus, constituents in groundwater could migrate to surface water in Cane Creek and its tributaries. This transport mechanism appears to be relatively insignificant compared to other transport mechanisms at these ranges (e.g., surface runoff).

Constituent transfer to sediments represents another significant transfer mechanism, especially where constituents are in the form of suspended solids or are hydrophobic substances (e.g., PAHs) that can become adsorbed to organic matter in the sediments. The metals detected in surface water have the potential to associate with suspended particulate matter. Sequestration of constituents in sediment is expected to be minimal at the BGR ranges due to the lack of organic matter in the sediments of Cane Creek. As presented previously, Cane Creek substrates are boulders and cobbles, with very little evidence of fine-grained organic sediment.

Constituents in surface water can be transported to other ranges along Bains Gap Road or off site via Cane Creek. Transfer of constituents in surface water to aquatic organisms is also a potentially significant transfer pathway. Some of the inorganic constituents detected in surface water may bioaccumulate in lower trophic level organisms. Most of the inorganics detected in surface water are not highly bioconcentratable; therefore, transfer through the food web is expected to be minimal for these compounds.

#### ***4.4.1.3 Fate and Transport in Sediment***

Constituent transfer between sediment and surface water potentially represents a significant transfer mechanism, especially when constituents are in the form of suspended solids. Sediment/surface water transfer is reversible; sediments often act as temporary repositories for constituents and gradually release constituents to surface waters. This is especially true in surface water systems that are acidic, as is the case with Cane Creek in the vicinity of the BGR ranges. Sorbed or settled constituents can be transported with the sediment to downstream locations. Much of the substrate of Cane Creek and its tributaries in the vicinity of the BGR ranges is best characterized as boulders or cobbles. Very few areas of high organic content sediment or muck are present. The very low organic content of boulders and cobbles creates a substrate with very low binding capacity; therefore, constituents released to Cane Creek and its tributaries via surface runoff or other transport mechanisms would most likely remain suspended in the surface water and be transported downstream and would not be sequestered in the stream substrate directly adjacent to the BGR ranges.

Although transfer of sediment-associated constituents to bottom-dwelling biota also represents a potentially significant transfer mechanism, it is not expected to be a major mechanism at the BGR ranges. Lower trophic level organisms may accumulate metals and PAHs; however, higher trophic level organisms have the ability to metabolize PAHs and therefore reduce their accumulative properties. Most of the inorganics detected in sediment are not bioaccumulative.

#### **4.4.1.4 Fate and Transport in Groundwater**

The soils mapped within the area of investigation for the BGR ranges are the Anniston and Allen gravelly loam, the Anniston and Allen stony loam, the Jefferson gravelly fine sandy loam, the Jefferson stony fine sandy loam, and the Stony Rough Land sandstone. The Stony Rough Land sandstone underlies the majority of the area of investigation, with only the northern portion of the area underlain by the Jefferson gravelly and stony fine sandy loams and the extreme southwestern portion of the area underlain by the Anniston and Allen gravelly and stony fine sandy loams (USDA, 1961) (Figure 4-3).

Stony Rough Land sandstone is a land type that is found in rough, steep areas with many outcrops of sandstone or quartzite bedrock, loose rock fragments, and scattered patches of sandy soil material. The soil materials that are found in these areas are generally thin. The runoff in these areas is high, the infiltration is slow, and the capacity for available moisture is low (USDA, 1961).

The Jefferson stony fine sandy loam and Jefferson gravelly fine sandy loam are friable soils derived from old local alluvium. These soils are generally found on the foot slopes at the base of steep hills and on ridges underlain by sandstone and shale. The surface soil is generally dark grayish brown. The subsoil generally consists of a yellowish brown stony, gravelly, fine sandy loam. The Jefferson stony fine sandy loam can have numerous pieces of sandstone and quartzite up to eight inches in diameter throughout (USDA, 1961).

The Anniston and Allen stony loam and Anniston Allen gravelly loam have developed in old alluvium on the foot slopes and fans at the base of steep hills. The subsoil of these units is generally dark brown to grayish brown in color. The subsurface soil consists of a dark red to reddish brown stony, gravelly, fine sand, clay, and loam. These soils generally have medium infiltration and a high capacity for available moisture. The Anniston and Allen stony loam can have numerous stones from three to eight inches in diameter throughout (USDA, 1961).

The BGR ranges are located along the eastern boundary of the Fort McClellan geological window (Figure 4-4). The mapping units exposed within this area of investigation are the

Cambrian Chilhowee Group, undifferentiated, the Cambrian Shady Dolomite, and the Mississippian/Ordovician Floyd and Athens Shale, undifferentiated. The Chilhowee Group underlies the majority of the area. The Shady Dolomite is found to the west of the Chilhowee Group. The Little Oak and Newala Limestones and Floyd and Athens Shales are found along the western boundary of the area of investigation within the geological window. The boundary of the window is defined by the Jacksonville fault, which is the major structural feature in this area (Osborne et al., 1997 and 1988).

The geological data collected during drilling activities at Parcels 77Q, 78Q, 80Q, and 85Q revealed a light brown to reddish orange sandy loam with cobbles and gravel from ground surface to between five to twenty feet bgs. The cobbles and gravel generally consisted of quartz sandstone and quartzite. Below this, a clay with sand and silt ranging from dark reddish brown to light tan to yellowish orange to olive to purple was encountered to the total depth of the borings. However, quartz sandstone was encountered at boring HR-80Q-MW01 at a depth of approximately seven feet bgs, and mudstone was encountered at HR-85-MW02 at a depth of approximately 82 feet bgs. The soils and residuum encountered appear consistent with soils mapped as the Jefferson gravelly fine sandy loam, Jefferson stony fine sandy loam, and the Stony Rough Land sandstone. Bedrock encountered in the borings is consistent with bedrock mapped within the Chilhowee Group, undifferentiated.

Precipitation in the form of rainfall averages about 53 inches annually in Anniston, Alabama, with infiltration rates annually exceeding evapotranspiration rates (U.S. Department of Commerce, 1998). The major surface water features at the Main Post of FTMC include Remount Creek, Cane Creek, and Cave Creek. These waterways flow in a general northwest to westerly direction towards the Coosa River on the western boundary of Calhoun County.

Cane Creek and its headwater tributaries are located within BGR ranges. Cane Creek is formed by a number of tributaries flowing to the west from Choccolocco Mountain, which is located to the east of the study area.

Surface elevations range from about 975 to 1,050 feet amsl at Range 24 Upper, 900 to 960 feet amsl at Range 21, 900 to 975 feet amsl at Range 22, and about 875 to 1,000 feet amsl at range 27. Ground surface across the ranges is generally flat with a gradual slope to the north-northwest, towards BGR. Within the study area the ground surface generally slopes to the west following the gradient of Cane Creek and its tributaries. Surface water runoff in the area of the BGR ranges follows topography, flowing into Cane Creek and its tributaries, which then flow to the west.

Reeves-Truitt Spring is located south of Bains Gap Road between Range 22 and Range 27 (Moser and DeJarnette, 1992). Reeves-Truitt Spring reportedly had a measured discharge of approximately 58 gallons per minute in October 1957 (Moser and DeJarnette, 1992). For the purpose of the BGR range investigations, Reeves-Truitt Spring is identified as SAR-85-SEEP01.

During soil boring and well installation activities, groundwater was encountered at depths ranging from 4 to 87 feet bgs. Static groundwater levels were collected at the site on January 7, 2002 (Table 4-2). A piezometric surface map constructed with static groundwater elevation data for the residuum water-bearing unit is provided on Figure 4-5. Groundwater elevation data revealed an east-to-west groundwater flow direction across the study area (Figure 4-5). The horizontal hydraulic gradient across this area is calculated to be approximately 0.07 feet per foot. Based on the piezometric surface and the depth groundwater was encountered during well installation activities, it appears that groundwater at the BGR ranges is under confined to semiconfined conditions.

Comparing the stream bed elevations within the study area to the elevation of groundwater encountered during drilling and the static water levels from January 2000, it appears that groundwater in residuum at the BGR ranges does not contribute to Cane Creek and its tributaries in the upper reaches of these creeks (i.e., within Range 24 Upper and the southeast portion of Range 21). This is based on the observed potentiometric surface in nearby wells being below the base of the creek bed, suggesting that the creeks are losing water to the surrounding residuum. However, further downstream in the study area it appears that the groundwater residuum does contribute to Cane Creek and its tributaries, as the potentiometric surface intersects the creek and, therefore, groundwater in residuum is contributing to the surface water flow within Cane Creek.

#### **4.4.2 Ecotoxicity**

The ecotoxicological properties of the constituents detected in the various environmental media at the BGR ranges are discussed in the following sections.

##### **4.4.2.1 Aluminum**

Aluminum is the most abundant element in the Earth's crust. Minimal evidence exists concerning the essentiality of aluminum. Aluminum is generally considered to have low mammalian toxicity (Hayes, 1994).

**Plants.** Aluminum appears to be essential for the growth of some plant species (Kabata-Pendias and Pendias, 1992). Higher concentrations of aluminum are usually detected in older rather than

younger leaves (Bollard, 1983). Differences in the toxicity of aluminum to plants is closely linked to different uptake and transport of calcium (Foy, 1974). Interactions of aluminum with potassium, silicon, and organic acids have also been reported (Foy, 1974). According to Foy (1974), aluminum toxicity in plants usually does not occur in soils with pH values above 5.5. Toxicity is, however, common at soil pH values below 5.0 (Foy, 1974). The addition of nitrogenous fertilizers to soil increases the toxicity of aluminum to plants by displacing exchangeable aluminum into soil solution and lowering soil pH (Foy, 1974).

Concentrations of aluminum in leaf tissue that are excessive or toxic to various plant species range from 5 to 10 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 50 mg/kg (dry weight) has been proposed by Efraymson et al. (1997) as a benchmark screening value for aluminum phytotoxicity. Signs of aluminum toxicity in plants include overall stunting of growth, the presence of dark green leaves, purpling of stems, death of leaf tips, and coralloid and damaged root systems (Kabata-Pendias and Pendias, 1992)

**Mammals.** Aluminum is not an essential element for animal growth and development. Limited data exist on the concentrations and effects of aluminum on wildlife. Most absorbed aluminum is eliminated through the kidney (National Library of Medicine [NLM], 1996).

Data are scarce on the effects of aluminum on wild mammals. Laboratory studies have shown inhalation of aluminum dust to induce infections and diseases of the lung (NLM, 1996). Laboratory-derived toxicity data from studies conducted with mice fed aluminum ( $\text{AlCl}_3$ ) in their drinking water were used to derive a no observed adverse effect level (NOAEL) value of 1.93 milligrams per kilogram per day (mg/kg/day) (Ondreicka et al., 1966). Reproduction was the endpoint for these studies.

**Birds.** Dietary ingestion of aluminum at concentrations of approximately 1,400 mg/kg produced declines in inorganic phosphorus levels in blood and resulted in the development of severe rickets in chickens (NLM, 1996). No adverse effects were observed in black ducks (*Anas rubripes*) fed diets containing 1,000 mg/kg aluminum as aluminum sulfate over a period of 12 days (Sparling, 1990). Diets with low calcium and phosphorus concentrations adversely affected the response of the ducks to aluminum (Sparling, 1990). An acute  $\text{LD}_{50}$  (lethal dose that will result in 50 percent mortality in a test population) of 111 mg/kg has been reported for exposure of birds to aluminum (Schafer et al., 1983).

**Aquatic Life.** Bioconcentration of aluminum has been reported for several freshwater species. A bioconcentration factor for daphnids exposed to aluminum is 574 (Cowgill and Burns, 1975).

Crayfish have been reported to have a bioconcentration factor for aluminum of 1,305 (Malley et al., 1987). The National Recommended Water Quality Criteria (EPA, 1999a) for aluminum are 750 and 87 micrograms per liter ( $\mu\text{g/L}$ ) for acute and chronic exposures, respectively. The lowest chronic values for aluminum toxicity reported in the literature for fish and daphnids are 3,290 and 1,900  $\mu\text{g/L}$ , respectively (Suter and Tsao, 1996). The test  $\text{EC}_{20}$  (the concentration that will result in a specified effect on 20 percent of the test population) for fish can be used as a benchmark indicative of production within a population. The  $\text{EC}_{20}$  value for aluminum is 4,700  $\mu\text{g/L}$  (Suter and Tsao, 1996). The  $\text{EC}_{20}$  value for aluminum with respect to daphnids is 540  $\mu\text{g/L}$  (Suter and Tsao, 1996).

#### **4.4.2.2 Antimony**

Antimony binds to soil and particulates (especially those containing iron, manganese, or aluminum) and is oxidized by bacteria in soil. Exposure routes for aquatic organisms include ingestion and gill uptake. Antimony bioconcentrates in aquatic organisms to a small degree. Exposure routes for mammals include ingestion and inhalation. It does not biomagnify in terrestrial food chains (Ainsworth, 1988). Antimony is not significantly metabolized and is excreted in the urine and feces. Antimony causes reproductive, pulmonary, and hepatic effects in mammals (EPA, 1999b).

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

**Mammals.** Female mice exposed to 5.0 mg/L antimony as antimony potassium tartrate in their drinking water showed a reduction in their lifespan. This dose was equivalent to a lowest observed adverse effects level (LOAEL) of 1.25 mg/kg/day, which can be converted to a NOAEL of 0.125 mg/kg/day (Integrated Risk Information Service [IRIS], 2002).

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

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

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

#### **4.4.2.3 Arsenic**

Arsenic occurs naturally as sulfides and as complex sulfides of iron, nickel, and cobalt (Woolson, 1975). In one form or another, arsenic is present in rocks, soils, water, and living organisms at concentrations of parts per billion (ppb) to parts per million (ppm) (National Academy of Sciences [NAS], 1977a). Arsenic is ubiquitous in living tissue and is constantly being oxidized, reduced, or otherwise metabolized. In soils, insoluble or slightly soluble arsenic compounds are constantly being resolubilized, and the arsenic is being presented for plant uptake or reduction by organisms and chemical processes (NAS, 1977a). Among elements, arsenic ranks 20th in abundance in the Earth's crust (1.5 to 2 mg/kg), 14th in sea water, and 12th in the human body (Woolson, 1975). It occurs in various forms, including inorganic and organic compounds, and in trivalent and pentavalent states (Pershagen and Vahter, 1979).

**Plants.** The NAS (1977a) reports background arsenic concentrations in terrestrial plants as ranging from 0.01 to 5 mg/kg (dry weight). Generally, the roots of a plant contain higher concentrations of arsenic than leaves. The toxicity of arsenic to plants may differ due to different soil conditions. Various chemical forms of arsenic have different phytotoxicities. In general, arsenates are less toxic to plants than arsenites. Concentrations of arsenic in leaf tissue that are excessive or toxic to various plant species range from 5 to 20 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 10 mg/kg has been proposed by Efrogmson et al. (1997) as a benchmark screening value for phytotoxicity in soils. General symptoms of arsenic toxicity in plants include the presence of red-brown necrotic spots on old leaves, yellowing or browning roots, depressed tillering, wilting of new leaves, and root discoloration (Kabata-Pendias and Pendias, 1992).

**Mammals.** Arsenic is more toxic to wildlife in its trivalent form as compared to either pentavalent arsenic or organic arsenicals. Arsenic poisoning has been reported in acute episodes, but chronic arsenosis is rarely encountered (NAS, 1977a). The probability of chronic arsenic poisoning from continuous ingestion of small doses is rare, because detoxification and excretion are rapid (Woolson, 1975). General signs of arsenic toxicosis include intense abdominal pain, staggering gait, extreme weakness, trembling, fast and feeble pulse, collapse, and death (Eisler, 1988a). Adverse effects in mammals were noted in single oral doses of 2.5 to 33 mg/kg body weight and at chronic oral doses of 1 to 10 mg/kg body weight. As little as 1.7 mg/kg has been shown to produce fetal death and malformations during critical stages of hamster embryogenesis.

Various species of rodents exposed to cacodylic acid through various routes exhibited lethal dose for 50 percent population tested (LD<sub>50</sub>) ranging from 470 to 830 mg/kg body weight (Hood, 1985).

Laboratory data for arsenic toxicity (as arsenite) through drinking water in laboratory mice were used to derive a NOAEL value of 0.126 mg/kg/day (Schroeder and Mitchener, 1971).

Reproduction was the endpoint for these laboratory tests. Arsenic concentrations of greater than 10 mg/kg (wet weight) in tissue are usually indicative of arsenic poisoning (Goede, 1985).

Detoxification and excretion of arsenic are relatively rapid processes, making the probability of chronic arsenic poisoning from the continuous ingestion of small amounts of arsenic a rare event (Eisler, 1988a).

**Birds.** As with mammals, arsenic poisoning in birds has been reported in acute episodes, but chronic arsenosis is rarely encountered. Signs of inorganic trivalent arsenite poisoning in birds include muscular incoordination, debility, slowness, jerkiness, falling hyperactivity, immobility, and seizures (Eisler, 1988a). Studies reported by Hudson et al. (1984) using mallard ducks (*Anas platyrhynchos*) fed sodium arsenite determined an LC<sub>50</sub> of 323 mg/kg body weight. Copper acetoarsenite fed to the northern bobwhite (*Colinus virginianus*) at 480 mg/kg proved fatal to 50 percent of the test organisms in 11 days (NAS, 1977a). The grey partridge (*Perdix perdix*) succumbed to 300 mg/kg body weight of lead arsenate in 52 hours (NAS, 1977a).

**Aquatic Life.** Toxic and other effects of arsenicals to aquatic life are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977a; National Research Council of Canada [NRCC], 1978; EPA, 1980a; Howard et al., 1984; Michnowicz and Weak, 1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The LC<sub>50</sub> values, for example, are markedly affected by water temperature, pH, oxidation-reduction potential, organic content, phosphate concentration, suspended solids, and the presence of other substances and toxicants, as well as arsenic speciation and duration of exposure. In general, inorganic arsenicals are more toxic to aquatic biota than organoarsenicals, and trivalent species are more toxic than pentavalent species. Early life stages are most sensitive, and large interspecies differences have been recorded, even among those closely related taxonomically. Juvenile bluegills (*Lepomis macrochirus*) exhibited reduced survival after sixteen weeks when exposed to a single treatment of trivalent arsenic at 0.69 mg/L (EPA, 1980a, 1985a). An adult bluegill population was reduced 42 percent after several monthly applications of 4 mg/L trivalent arsenic (NAS, 1977a).

As with fish, toxic and other effects of arsenicals to aquatic invertebrates are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977a; NRCC,

1978; EPA, 1980a; Howard et al., 1984; Michnowicz and Weaks, 1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The cladoceran *Daphnia magna* population exposed to 4.3 mg/L trivalent arsenic exhibited a 50 percent immobilization after 96 hours, and the amphipod *Gammarus pseudolimnaeus* experienced 50 percent immobilization following 96 hours of 0.96 mg/L trivalent arsenic exposure (Lima et al., 1984).

#### **4.4.2.4 Barium**

Barium, a silvery-white metal, is used in various alloys, in paints, soap, paper, and rubber, and in the manufacture of ceramics and glass. Two forms of barium, barium sulfate and barium carbonate, are often found in nature as underground ore deposits. Barium is relatively abundant in nature and is found in plants and animal tissue. Plants can accumulate barium from the soil.

Most of the barium that enters an animal's body is removed within a few days, and almost all of it is gone within 1 to 2 weeks. Most of the barium that stays in the body goes into the bones and teeth. Rats exposed to barium in their diet at lower doses, but for longer time periods, showed increased blood pressure and changes in the function and chemistry of the heart (ATSDR, 1992a).

**Plants.** Background concentrations of barium in various food and feed plants are reported to range from 1 to 198 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Concentrations are often highest in the leaves of cereals and legumes and lowest in grains and fruits (Kabata-Pendias and Pendias, 1992). The availability of barium to plants is greatly influenced by the pH of the soil, with barium more available under acidic soil conditions (Kabata-Pendias and Pendias, 1992). The concentration of barium in leaf tissue that has been reported as excessive or toxic to various plant species is 500 mg/kg (Kabata-Pendias and Pendias, 1992). A soil concentration of 500 mg/kg has been proposed by Efrogmson et al. (1997) as a benchmark screening value for barium phytotoxicity.

**Mammals.** Barium administered to rats via their drinking water at doses of 1, 10, and 100 mg/L had no effect on food or water consumption or growth. Because the highest dose tested (100 mg/L) did not elicit any adverse effects, it was considered the NOAEL (5.1 mg/kg/day) (IRIS, 2002). Laboratory rat toxicity data for barium chloride in drinking water were used to calculate a NOAEL value of 5.1 mg/kg/day. Growth and hypertension were the test endpoints.

**Aquatic Life.** The chronic value for daphnids is from a 21-day test on *Daphnia magna* by Biesinger and Christensen (1972) which resulted in 16 percent reproductive impairment. The Tier II secondary acute water quality value and secondary chronic water quality value for

barium, as calculated by the method described in the EPA's *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 110 and 4.0 µg/L, respectively.

#### **4.4.2.5 Beryllium**

In environmental media, beryllium usually exists as beryllium oxide. Beryllium has limited solubility and mobility in sediment and soil.

**Plants.** Beryllium uptake by plants occurs when beryllium is present in the soluble form. The highest levels of beryllium are found in the roots, with lower levels in the stems and foliage (EPA, 1985c).

Soluble forms of beryllium are easily taken up by plants, probably in a manner similar to calcium and magnesium, but it is not readily translocated from roots to shoots (Peterson and Girling, 1981). Beryllium has been reported to inhibit seed germination, enzyme activation, and uptake of calcium and magnesium by roots. Common symptoms of beryllium toxicity to plants are brown, retarded roots and stunted foliage (Romney and Childress, 1965). The phytotoxicity benchmark value for beryllium (10 mg/kg) is based on unspecified toxic effects on plants grown in surface soil amended with 10 mg/kg beryllium (Kabata-Pendias and Pendias, 1992).

**Mammals.** The major exposure route for mammals is inhalation. Based on animal studies, beryllium is poorly absorbed from the gastrointestinal tract and is not absorbed through intact skin to any significant degree. The most important route of exposure for beryllium is inhalation, although absorption by this route does not appear to be extensive. Mammals exposed via inhalation exhibit pulmonary effects that may last long after exposure ceases. Once beryllium is absorbed, it is circulated in the blood as an orthophosphate colloid and is then distributed primarily to the bone, liver, and kidneys in both humans and animals. Beryllium and its compounds are not biotransformed, but soluble beryllium compounds are partially converted to more insoluble forms in the lungs (Reeves and Vorwald, 1967).

Following inhalation of soluble beryllium compounds in both humans and animals, the lung appears to be the main target organ for toxicity. Acute exposure may cause chemical pneumonitis; chronic exposure to insoluble forms may lead to chronic beryllium disease (berylliosis), a fibrotic lung disease (ATSDR, 1993a). Laboratory data based on beryllium sulfate exposures to rats through their drinking water were used to estimate a NOAEL value of 0.66 mg/kg/day (Schroeder and Mitchener, 1975). Longevity and weight loss were the endpoints in this study.

A variety of beryllium compounds have been demonstrated to cause pulmonary tumors following inhalation in animals. However, it is thought that oral administration does not lead to carcinogenesis, due to poor absorption of the constituent from the gastrointestinal tract. The NOAEL for a rat lifetime chronic exposure to beryllium in drinking water was 0.54 mg/kg/day (Health Effects Assessment Summary Tables [HEAST], 1997).

**Aquatic Life.** Exposure routes for aquatic organisms include ingestion and gill uptake. Beryllium does not bioconcentrate in aquatic organisms. Beryllium uptake from water is low, resulting in low bioconcentration rates. Biomagnification of beryllium in aquatic food chains does not occur (Fishbein, 1981). Beryllium can be toxic to warm-water fish, especially in soft water.

The Tier II secondary acute water quality value and secondary chronic water quality value for beryllium, as calculated by the method described in the EPA's *Proposed Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 35 and 0.66 µg/L, respectively.

The effects concentration for 20 percent of a population (EC<sub>20</sub>) for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC<sub>20</sub> value for beryllium is 148 µg/L. A similar value can be determined for daphnids, which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC<sub>20</sub> for daphnids is 3.8 µg/L (Suter and Tsao, 1996).

#### **4.4.2.6 Chromium**

Chromium in the trivalent form is required for normal glucose metabolism and as an insulin cofactor, and thus chromium is an essential trace element.

**Plants.** Chromium does not play an essential role in plant metabolism. The concentration of chromium in terrestrial plants is controlled primarily by soluble chromium in the soil (Kabata-Pendias and Pendias, 1992). Chromium concentrations in plants are usually higher in roots than in leaves or shoots. Concentrations of chromium in leaf tissue that are excessive or toxic to various plant species range from 5 to 30 mg/kg soil (Kabata-Pendias and Pendias, 1992). General symptoms of chromium toxicity in plants include chlorosis of new leaves, necrotic spots and purpling tissues, and injured root growth (Kabata-Pendias and Pendias, 1992).

**Mammals.** Chromium is a required element in animal nutrition. In general, hexavalent chromium compounds are more toxic than the trivalent chromium compounds. Adverse effects on blood and serum chemistry and morphological changes in liver have been reported in rabbits and rats exposed to chromium concentrations of 1.7 mg/kg/day for six weeks. Rats exposed to hexavalent chromium concentrations of 134 mg/L in drinking water over a two to three month period were found to develop lesions in kidney and liver tissues (Eisler, 1986).

Laboratory data based on rats exposed to chromium (as Cr<sub>2</sub>O<sub>3</sub>) in their diets were used to derive a NOAEL value for trivalent chromium of 2,737 mg/kg/day (Ivankovic and Preussmann, 1975). Reproduction and longevity were the endpoints in this study. Laboratory data based on rats exposed to chromium (as K<sub>2</sub>Cr<sub>2</sub>O<sub>4</sub>) in their drinking water were used to derive a NOAEL value for hexavalent chromium of 3.28 mg/kg/day (MacKenzie et al., 1958). Body weight and food consumption were the endpoints in this study. Mammalian laboratory studies have shown chromium to be mutagenic, carcinogenic and teratogenic (Eisler, 1986).

**Birds.** Data on the effects of chromium to avian species is limited. Haseltine et al. (1985) did not observe changes in survival, reproduction or blood chemistry following exposure of adult black ducks (*Anas rubripes*) to diets containing between 10 and 50 mg/kg chromium III (as CrK[SO<sub>4</sub>]<sub>2</sub>). Based on these data, a NOAEL value of 1 mg/kg/day has been derived.

**Aquatic Life.** In freshwater systems, hexavalent chromium appears to be more toxic than the trivalent form. Water quality standards for chromium are set for both chromium III and chromium VI. Federal Water Quality Criteria for the protection of aquatic life set the criteria for acute exposure at 1.7 mg/L for chromium III and 0.016 mg/L for chromium VI. Chronic exposure values are 0.21 mg/L for chromium III and 0.011 mg/L for chromium VI (EPA, 1999a), based on a water hardness of 100 mg/L.

#### **4.4.2.7 Cobalt**

Cobalt is a natural element that is widely distributed in the Earth's crust at 0.001 to 0.002 percent (Merck Index, 1983). Small amounts of cobalt are found in rocks, soil, and surface and groundwater. Natural cobalt can stay airborne for a few days but will stay for years in the soil. In most soils, the transfer of cobalt from soils to plants is not significant, although higher transfer rates have been observed in some higher plants and in acidic soils (Boikat et al., 1985; Francis et al., 1980). Some cobalt may seep from acid soil into groundwater. It is present in trace quantities in most foods and is readily absorbed by the gut in humans (International Commission on Radiological Protection, 1979).

**Plants.** Although cobalt is essential to some blue-green algae, fungi, and microorganisms, it apparently is not essential for the growth of higher plants (Kabata-Pendias and Pendias, 1992). Several abiotic factors govern the availability of cobalt to plants. Soil factors include organic matter and clay content, pH, leachability, and concentrations of manganese and iron oxides. Uptake of cobalt can occur via the roots or leaves of a plant (Kabata-Pendias and Pendias, 1992).

Concentrations of cobalt in leaf tissue that are excessive or toxic to various plant species range from 15 to 50 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 20 mg/kg (dry weight) has been proposed by Efroymsen et al. (1997) as a benchmark screening value for cobalt phytotoxicity. General symptoms of cobalt toxicity in plants include interveinal chlorosis in new leaves, followed by induced iron chlorosis, white leaf margins, and damaged root tips (Kabata-Pendias and Pendias, 1992).

**Mammals.** Cobalt is a component of vitamin B<sub>12</sub> and, therefore, is an essential micronutrient for animal growth. No information has been located at this time on chronic toxic effects of cobalt to terrestrial wildlife; however, some acute studies have been completed. Additionally, there is little biomagnification of cobalt in animals of higher trophic levels (Jenkins, 1980).

Young rats are unable to survive repeated 30 mg doses of cobalt metal powder in their diet for a month (total dosage about 900 mg), whereas they can tolerate 1,250 mg of the metal in a single dose (Venugopal and Luckey, 1978). Cobalt was embryotoxic to rat fetuses when it was administered during the entire gestation (dose of 0.05 mg/kg). A dose of 0.005 mg/kg was nontoxic to the females; however, the progeny of treated females had a reduced survival rate (Shepard, 1986). At doses under 2 milligrams per kilogram of body weight per day (mg/kg-bw/day), no adverse effects to sheep were noted. However, at 6 mg/kg-bw/day, sheep exhibited loss of appetite, loss of weight, and debilitation (National Research Council, 1977).

**Birds.** No information has been located at this time on chronic toxic effects of cobalt to birds; however, some acute studies have been completed. Additionally, there is little biomagnification of cobalt in animals of higher trophic levels (Jenkins, 1980).

Chickens were administered 50 mg/kg of diet/day with acute effects of loss of appetite, loss of weight, and debilitation. At doses under 2 mg/kg-bw/day, no adverse effects to chickens were noted (National Research Council, 1977).

**Aquatic Life.** In most surface water bodies, cobalt is primarily associated with the sediment. However, some mobilization may occur in acidic water and in the presence of chloride ions or

chelating agents. Bioaccumulation factors for freshwater fish range from 40 to 1,000 (Smith and Carson, 1981). Research by Evans et al. (1988) indicates that cobalt does not significantly bioaccumulate in benthic bottom feeders.

#### **4.4.2.8 Copper**

Copper is ubiquitously distributed in nature in its free state and in sulfides, arsenides, chlorides, and carbonates. Several copper-containing proteins have been identified in biological systems as oxygen-binding hemocyanin, cytochrome oxidase, tyrosinase, and laccase. Copper has also been identified with the development of metalloproteins employed in the sequestering and cellular detoxification of metals.

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

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

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

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

**Mammals.** Copper is an essential trace element to plants and animals (Callahan et al., 1979) but becomes toxic at concentrations only slightly higher than essential levels (EPA, 1985b). Copper is an essential element for hemoglobin synthesis and oxidative enzymes in animals. Copper is absorbed by mammals following ingestion, inhalation, and dermal exposure. Once absorbed, copper is distributed to the liver. Copper is not metabolized (Marceau et al., 1970). No evidence of bioaccumulation was obtained in a study of pollutant concentrations in the muscles and livers of ten species of herbivorous, omnivorous, and carnivorous animals in Donana National Park in Spain (Hernandez et al., 1985). Copper concentrations in small mammals collected from various uncontaminated sites ranged from 8.3 to 13.4 mg/kg (whole-body concentrations) (Talmage and Walton, 1991). Highest concentrations of copper tend to be in hair, followed in decreasing concentration by liver, kidney, and whole body (Hunter and Johnson, 1982). Among the small mammals collected, Hunter and Johnson (1982) found shrews (*Sorex araneus*) to contain the highest concentrations of copper. Mice were found to contain the lowest copper concentrations. Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg/day of copper as copper sulfate (Lecyk, 1980). Increased mortality rates in mink offspring have been observed at levels above 3.21 mg/kg/day (Aulerich et al., 1982).

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

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

**Aquatic Life.** Invertebrates inhabiting “polluted” freshwaters worldwide have been known to have tissue residues of copper ranging from 5 to 200 ppm (Moore and Ramamoorthy, 1984). Field studies have shown that there is virtually no accumulation of this metal through the food chain (Fuller and Averett, 1975). Studies by Kosalwat and Knight (1987) indicated that copper present in the substrate or sediment was significantly less toxic to chironomid species than overlying water column levels. The substrate copper concentration at which chironomid larval growth was reduced 50 percent (EC<sub>50</sub>) was 1,602 mg/kg. These researchers found that deformities in larval mouth parts were observed at elevated concentrations, and adult emergence was inhibited when the sediment concentration exceeded 1,800 mg/kg. Carins et al. (1984)

reported copper toxicity in sediment for several chironomus midges and cladocerans with LC<sub>50</sub>s ranging from 681 to 2,296 mg/kg.

#### **4.4.2.9 Iron**

Iron is an essential trace element, required as a constituent of oxygen-carrying and oxidative-reductive macro-molecules such as hemoglobin, myoglobin, and cytochrome P-450. As such, most iron-related health concerns are induced by insufficient iron intake, rather than excess iron intake (Hayes, 1994).

**Plants.** Wallihan (1966) reported unspecified reductions in plant growth in a solution culture with the addition of 10 ppm iron. Wallace et al. (1977) evaluated the effects of iron (as FeSO<sub>4</sub>) on leaf, stem, and root weights of bush bean seedlings grown for 15 days in nutrient solution. Iron at 28 ppm reduced all three measures 67, 52, and 67 percent, respectively, while 11.2 ppm iron had no effect. After 55 days, cabbage seedling plant weight was reduced 45 percent by 50 ppm iron added as FeSO<sub>4</sub> to nutrient solution, while 10 ppm had no effect on growth (Hara et al., 1976).

Iron is the key metal required for energy transformations needed for cellular function. It occurs in heme and nonheme proteins and is concentrated in chloroplasts. Organic iron complexes are involved in phototsynthetic electron transfer. Plant symptoms of toxicity are not specific and differ among plant species and growth stages (Foy et al., 1978).

**Mammals.** Iron is an essential nutrient for most wildlife species and is necessary to maintain homeostasis; therefore, it is only toxic at very high concentrations. Bioaccumulation factors have been calculated for several small mammal species. Small herbivorous mammals were estimated to have an iron bioaccumulation factor of 0.0127, and small omnivorous mammals were estimated to have an iron bioaccumulation factor of 0.01209. These bioaccumulation factors indicate that iron is not accumulated in small mammal tissues (Sample et al., 1998a). Additionally, the bioaccumulation factor for earthworms has been estimated to be 0.038, indicating that iron is not accumulated in earthworm tissues (Sample et al., 1998b).

**Aquatic Life.** The national recommended water quality criteria for iron (1,000 µg/L) is based on field study at a site receiving acid mine drainage (EPA, 1999a). The lowest chronic value for daphnids (158 µg/L) is a threshold for reproductive effects from a 21-day test using iron chloride with *Daphnia magna* (Dave, 1984). It is considerably lower than the 4,380 µg/L concentration causing 16 percent reproductive decrement in another test using iron chloride with *Daphnia magna* (Biesinger and Christensen, 1972). The lowest chronic value for fish (1,300 µg/L) is a

concentration that caused 100 percent mortality in an embryo-larval test with rainbow trout exposed to dissolved iron salts (Amelung, 1981).

The Ontario Ministry of the Environment has prepared provincial sediment quality guidelines using the screening-level concentration approach. This approach estimates the highest concentration of a particular contaminant in sediment that can be tolerated by approximately 95 percent of benthic fauna (Neff et al., 1988). These values are based on Ontario sediments and benthic species from a wide range of geographical areas within the province (Persaud et al., 1993). The lowest effect level (Low) is the level at which actual ecotoxic effects become apparent. The severe effect level (Severe) represents contaminant levels that could potentially eliminate most of the benthic organisms (Persaud et al., 1993). The “Low” and “Severe” levels for iron in sediment are 2 percent (20,000 ppm) and 4 percent (40,000 ppm), respectively.

#### **4.4.2.10 Lead**

Global production of lead from both smelter and mining operations has been high throughout the last 100 years. Lead is commonly used in storage batteries as well as in ammunition, solder, and casting materials. In addition, tetraethyl lead was a principal additive to gasolines as an anti-knock agent and was commonly used as an additive in paints. In short, lead is one of the most ubiquitous pollutants in the civilized world.

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

**Plants.** Although lead is not an essential nutrient for plant growth, it is detected in plant tissues due to the prevalence of lead in the environment. The bioavailability to plants of lead in soil is limited. Bioavailability may be enhanced by a reduction in soil pH, a reduction in the content of organic matter and inorganic colloids in soil, a reduction in iron oxide and phosphorous content, and increased amounts of lead in soil (NRCC, 1973). Plants can absorb lead from soil and air. Aerial deposition of lead can also contribute significantly to the concentration of lead in above-ground plant parts. Lead is believed to be the metal of least bioavailability and the most highly accumulated metal in root tissue (Kabata-Pendias and Pendias, 1992).

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

substantial amounts must accumulate before it can affect the growth of higher plants (Boggess, 1977).

**Mammals.** As with plants, lead is not considered an essential nutrient for mammalian life. Lead is transported in the blood first to the soft tissues, then to the bone, where ninety percent of the total body burden is normally found. Ingestion is the major route of exposure for wildlife. Lead tends to accumulate in bone, hair, and teeth. Biomagnification of lead is negligible (Eisler, 1988b). Lead uptake may increase when dietary iron and calcium are deficient. Absorption of lead in food ranges from two percent to sixteen percent in adult organisms, whereas the young may absorb up to 45 to 50 percent of ingested lead (Shore and Rattner, 2000). Reduced survival was reported at acute oral doses as low as 5 mg/kg body weight in rats, at a chronic dose of 0.3 mg/kg body weight in dogs, and at a dietary level of 1.7 mg/kg body weight in horses (Eisler, 1988b). Laboratory data from studies of rats fed lead acetate in their diets were used to estimate a NOAEL value of 8.0 mg/kg/day (Azar et al., 1973). Reproduction was the endpoint for this study. Symptoms of lead poisoning in mammals are diverse and depend on the form of lead ingested, the concentration, and the species and its age. These symptoms may include reproductive impairment, decreased body weight, vomiting, uncoordinated body movements, visual impairment, reduced life span, renal disorders, and abnormal social behavior (Eisler, 1988b).

In laboratory studies, breeding mice exposed to low doses of lead in drinking water (25 ppm) resulted in loss of the strain in two generations with many abnormalities (Schroeder and Mitchener, 1971). Exposure of rats in this same experiment resulted in many early deaths and runts. Blood  $\delta$ -aminolevulinic acid dehydratase activity associated with exposure to lead was reduced in white-footed mice living near a metal smelter (Beyer et al., 1985). Amounts of whole-body lead content and feeding habits of roadside rodents have been correlated with highest body burdens in insectivores such as shrews, intermediate in herbivores, and lowest in granivores (Boggess, 1977; Getz et al., 1977).

**Birds.** Most of the information on the effects of lead to terrestrial vertebrates is concerned with the poisoning of waterfowl by lead shot. Apparent symptoms include loss of appetite and mobility, avoidance of other birds, lethargy, weakness, emaciation, tremors, dropped wings, green feces, impaired locomotion, loss of balance and depth perception, nervous system damage, inhibition of heme synthesis, damage to kidneys and liver, and death (Eisler, 1988b; Mudge, 1983). Anemia, kidney disease, testicular and liver lesions, and neurological disorders have been associated with high brain lead concentrations in mourning doves (*Zenaida macroura*) (Kendall, 1992). Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively more

tolerant to moderate lead exposure, including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2,000 ppm (Hoffman et al., 1985; Eisler, 1988b).

Toxicity of lead to birds is dependent upon the form of lead, the route of exposure and exposure duration, and the species and age of the bird. Laboratory toxicity data for American kestrels fed metallic lead in their diet were used to estimate a NOAEL value of 3.85 mg/kg/day (Pattee, 1984). Reproduction was the endpoint for this study.

**Aquatic Life.** All life stages are sensitive to the toxic effects of lead; however, embryos are more sensitive to lead than are later juvenile stages (Davies et al., 1976). Lead uptake depends on exposure time, aqueous concentration, pH, temperature, salinity, diet, and other factors. For example, gill, liver, kidney, and erythrocytes accumulate lead from aqueous sources in proportion to exposure time and concentration (Holcombe et al., 1976). Direct erythrocyte injury is considered the first and most important sign of lead poisoning in catfish (Dawson, 1935). Respiratory distress occurs in fish living in rivers receiving lead mining wastes in England (Carpenter, 1924, 1925, 1926). Fish are thought to be asphyxiated as a result of a mucous coating over the gills (NAS, 1972).

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

The majority of benthic invertebrates do not bioconcentrate lead from water or abiotic sediment particles. There is some evidence of bioaccumulation through the food web of organic forms of lead, such as tetraethyl lead. Anderson et al., (1980) reported lead LC<sub>50</sub>s of 258 ppm for the chironomid and that growth of this amphipod was not reduced above this level in freshwater sediments. In addition, Suter and Tsao (1996) reported effect levels in the water flea (*Daphnia magna*) to be in the 12.26 ppb range, while Khangrot and Ray (1989) reported a *D. magna* LC<sub>50</sub> of 4.89 ppm.

#### **4.4.2.11 Manganese**

Manganese, a silver-colored metal with chemical properties similar to iron, is a naturally occurring substance found in many minerals. Manganese is usually combined with oxygen, sulfur, and/or chlorine. Manganese is present in all living organisms and is an essential element for adequate nutritional needs in mammals and many other organisms. Manganese is poorly absorbed from the intestinal tract; about 3 to 5 percent of the oral dose of manganese is absorbed. Absorption efficiency is also related to dietary intake of iron and calcium. Sufficient body stores of iron decrease absorption of manganese (ATSDR, 1992b).

**Plants.** Manganese is an essential element for plant growth. Uptake of manganese may occur via root or leaves (Kabata-Pendias and Pendias, 1992). The concentration of manganese in plants is dependent upon plant and soil characteristics. Plants grown on flooded or acid soils tend to contain higher concentrations of manganese than plants grown in other, uncontaminated soils. In addition, concentrations of manganese in plants are positively correlated with soil organic matter (Kabata-Pendias and Pendias, 1992). Concentrations of manganese in leaf tissue that are excessive or toxic to various plant species range from 400 to 1,000 mg/kg dry weight (Kabata-Pendias and Pendias, 1992). A soil concentration of 500 mg/kg (dry weight) has been proposed by Efroymson et al. (1997) as a benchmark screening value for manganese phytotoxicity. General symptoms of manganese toxicity in plants include the presence of chlorosis and necrotic lesions on old leaves, blackish-brown or red necrotic spots, dried leaf tips, and stunted root and plant growth (Kabata-Pendias and Pendias, 1992).

**Mammals.** Manganese is an essential nutrient that is homeostatically regulated in vertebrates (Vanderploeg et al., 1975). Liver and kidney tissues generally contain the highest concentrations of manganese in the body. Manganese in the body is primarily excreted in the feces (Gregus and Klaassen, 1986).

Divalent manganese is more toxic than the trivalent form. Exposure to manganese dust via inhalation is usually of greater toxicological concern than ingestion (Klaassen et al., 1991). Laboratory data for rats fed manganese oxide in their diet were used to estimate a NOAEL value of 88 mg/kg/day (Laskey et al., 1982). Reproduction was the endpoint for this study. Laboratory studies with rats have found no hematologic, behavioral, or histologic effects in animals exposed to manganese dioxide at concentrations of 47 milligrams per cubic meter for five hours per day, five days a week, for 100 days (Klaassen et al., 1991).

**Aquatic Life.** As discussed previously, manganese is a required nutrient for plant and animal life. Manganese concentrations in most vertebrates are homeostatically controlled (Vanderploeg

et al., 1975). Bioconcentration factors for freshwater macrophytes have been reported to range from 190 to approximately 25,000 (Vanderploeg et al., 1975). With regard to freshwater fish, concentrations of manganese in fish muscle are generally less than 0.5 mg/kg and range from 3 to 10 mg/kg in whole fish (Vanderploeg et al., 1975). Bioconcentration factors from water to whole fish range from 40 to 2,300. A bioconcentration factor of 10,000 was also suggested for crustaceans (Vanderploeg et al., 1975).

No federal water quality criteria exist for the protection of freshwater biota from elevated manganese concentrations. Suter and Tsao (1996) have estimated acute and chronic advisory levels for manganese to be 1,470 and 80.3 µg/L, respectively. The EC<sub>20</sub> for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC<sub>20</sub> value for manganese is 1,270 µg/L. A similar value can be determined for daphnids which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC<sub>20</sub> for daphnids is less than 1,100 µg/L (Suter and Tsao, 1996).

#### **4.4.2.12 Mercury**

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

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

**Plants.** Mercury is not required for plant growth. Background concentrations of mercury in plants usually range from 0.0026 to 0.086 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Pine needles have been reported to be good biomonitors of mercury-contaminated environments (Kabata-Pendias and Pendias, 1992). In general, the concentration of mercury in plants will be elevated when mercury concentrations in soils are high. Mercury concentrations in

plants, however, generally do not exceed those in associated soils (Lisk, 1972). Methyl mercury is more available to plants than either phenyl- or sulfide-mercury. In addition to mercury uptake from the soil, plants can also absorb mercury vapor (Browne and Fang, 1978).

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

**Mammals.** Mercury is not an essential element for animal life. Background mercury concentrations in wildlife tend to be less than 1.0 mg/kg (wet weight) (Eisler, 1987b). Biomonitoring studies have shown that mercury concentrations in mammals are highest in hair, followed by kidney and liver tissues (Bull et al., 1977; Klaassen, 1991; Wren, 1986). Mercury is bioaccumulated and biomagnified in terrestrial food chains (Eisler, 1987b; Talmage and Walton, 1993). Talmage (1989) has shown the insectivorous shorttail shrew (*Blarina brevicauda*) to be a better monitor of environmental mercury contamination than the granivorous white-footed mouse (*Peromyscus leucopus*). Mink (*Mustela vison*) and river otter (*Lutra canadensis*) have been shown to be good monitors of mercury contamination within river environments due to their consumption of contaminated fish (Kucera, 1983).

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

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

**Birds.** Concentrations of mercury that are acutely toxic to birds following oral exposure range from 2.2 to 31 mg/kg body weight (Eisler, 1987b). Mercury concentrations in the livers of methylmercury-poisoned birds ranged from 17 to 70 mg/kg (dry weight) (Solonen and Lodenius, 1984). Methylmercury is more toxic to avian species than inorganic mercury (Hill, 1981). In

addition to the form of mercury to which the bird is exposed, the species, gender, age, and health of the individual may also influence the toxic response (Fimreite, 1979). Physical signs of mercury poisoning in birds include muscular incoordination, falling, slowness, fluffed feathers, calmness, withdrawal, hyporeactivity, and eyelid drooping (Eisler, 1987b).

***Aquatic Life.*** Concentrations of mercury in freshwater fish collected from 12 monitoring stations in the United States from 1978 to 1981 ranged from 0.1 to 1.1 mg/kg (wet weight), with an average of 0.11 mg/kg (Lowe et al., 1985). Elevated concentrations of mercury in fish have often been associated with low pH, low calcium concentrations in the water, and low water hardness (Eisler, 1987b). Methylating bacteria in sediments actively convert inorganic mercury into methylmercury. This results in an increase in the bioavailability of mercury. Fish absorb methylmercury more easily than inorganic mercury from the water column (Huckabee et al., 1979). Because exposure of fish to methylmercury can occur via ingestion of contaminated prey, methylmercury concentrations are usually highest in organisms near the top of the food chain, such as carnivorous fish (Huckabee et al., 1979).

Exposure of aquatic organisms to elevated mercury concentrations can result in reduced growth and reproduction (Eisler, 1987b). The National Recommended Water Quality Criteria for acute and chronic exposure to mercury in freshwater systems are 1.4 and 0.77  $\mu\text{g/L}$ , respectively (EPA, 1999a). The test  $\text{EC}_{20}$  for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The  $\text{EC}_{20}$  value for methylmercury is less than 0.03  $\mu\text{g/L}$  (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The  $\text{EC}_{20}$  benchmark for daphnids has been determined to be 0.87  $\mu\text{g/L}$  (Suter and Tsao, 1996).

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

#### **4.4.2.13 Selenium**

Selenium is distributed widely in nature and is found in most rocks and soils at concentrations between 0.1 and 2.0 mg/kg (Fishbein, 1981). The primary factor determining the fate of selenium in the environment is its oxidation state. Selenium is stable in four valence states (-2, 0, +4, and +6) and forms chemical compounds similar to those of sulfur. The selenides (-2) are insoluble in water, as is elemental selenium. The inorganic alkali selenites (+4) and the selenates (+6) are soluble in water and are, therefore, more bioavailable.

Conditions such as pH, oxidation-reduction potential, and the presence of metal oxides affect the partitioning of the various compounds of selenium in the environment. In general, elemental selenium is stable in soils and is found at low levels in water because of its ability to co-precipitate with sediments. The soluble selenates are readily taken up by plants and converted to organic compounds such as selenomethionine, selenocysteine, dimethyl selenide, and dimethyl diselenide. Selenium is bioaccumulated by aquatic organisms and may also biomagnify in aquatic organisms.

**Plants.** The role of selenium in plant growth is not fully understood. It is generally not considered essential in plant nutrition (Kabata-Pendias and Pendias, 1992). The concentration of selenium in plants has been shown to be positively correlated with the concentration of selenium in soil. Soil parameters such as pH, oxidation-reduction potential, and moisture content determine the amount of selenium available for plant uptake. Concentrations of selenium in leaf tissues that have been shown to be toxic to various plant species range from 5 to 30 mg/kg (Kabata-Pendias and Pendias, 1992). General symptoms of selenium toxicity in plants include the signs of interveinal chlorosis or black spots in plants containing approximately 4 mg/kg selenium, complete bleaching or yellowing of younger leaves at higher concentrations, and the presence of pinkish spots on roots (Kabata-Pendias and Pendias, 1992).

**Mammals.** Selenium is an essential trace element for animal life. Concentrations that are essential to animals are in the range of 0.05 to 0.1 mg/kg in the diet (Arthur et al., 1992). According to Ganther (1974), selenium concentrations in healthy, unexposed laboratory animals and livestock range between 0.1 and 1 mg/kg. Selenium offers a protective effect against some carcinogens such as benzo(a)pyrene and benzo(a)anthracene (Hammond and Beliles, 1980). Selenium also functions as an antidote to the toxic effects of mercury, thallium, copper, arsenic and cadmium (Frost and Lish, 1975).

Acute poisoning has been reported in livestock that consumed plant material containing 400 to 800 mg/kg selenium (Eisler, 1985). Signs of acute poisoning in livestock include abnormal

movements, lowered head, drooped ears, diarrhea, elevated temperature, rapid pulse, labored breathing, bloating with abdominal pain, increased urination, and dilated pupils (Eisler, 1985). Chronic poisoning may occur in animals exposed to dietary selenium concentrations between 1 and 44 mg/kg (Eisler, 1985). Laboratory data from studies using rats fed potassium selenate in their drinking water were used to derive a NOAEL value of 0.2 mg/kg/day (Rosenfeld and Beath, 1954). Reproduction was the endpoint in this study.

**Birds.** Toxicity from selenium has also been documented in birds. The major toxic effect of selenium on avian species is on reproductive success. Both sodium selenite and selenomethionine have been reported to be embryotoxic and teratogenic (Heinz et al., 1987). Reproductive impairment is likely to occur as concentrations of selenium approach 5 mg/kg. Mortality in mallard ducklings does not occur until selenium concentrations in the diet reach 40 mg/kg. Extrapolated NOAELs for chronic exposure of various avian wildlife species to selenomethionine, based on an estimated NOAEL for mallards of 0.4 mg/kg/day, are 0.3 mg/kg/day for the great blue heron and 0.385 mg/kg/day for the red-tailed hawk. The NOAEL for selenomethionine consumed in drinking water has been estimated to be 6.8 mg/L for wild birds (Sample et al., 1996).

**Aquatic Life.** Selenium is an essential micronutrient for fish. Dietary requirements of selenium for fish range from 0.07 to 0.25 mg/kg, depending on the fish species (Gatlin and Wilson, 1984). The bioconcentration of selenium from water is highly dependent on the species of selenium present. Laboratory studies have shown bioconcentration factors for selenomethionine to be greater than those for selenite and selenate. Bioconcentration factors for aquatic biota exposed to 1 µg/L selenomethionine were approximately 16,000 for algae, 200,000 for daphnids, and 5,000 for bluegills (Besser et al., 1993).

The EPA's National Recommended Water Quality Criteria for Priority Toxic Pollutants for selenium in freshwater is 5 µg/L for chronic exposure (EPA, 1999a). The toxicity of selenium to freshwater fish appears to be correlated more closely with dietary than waterborne exposure (Coyle et al., 1993). Sulfate concentrations in water may also influence the toxicity of selenium to aquatic invertebrates (Maier et al., 1993).

The test EC<sub>20</sub> for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC<sub>20</sub> for selenium is 40 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration

causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC<sub>20</sub> benchmark for daphnids is 25 µg/L selenium (Suter and Tsao, 1996).

#### **4.4.2.14 Silver**

Silver is a rare but naturally occurring element. It is often found deposited as a mineral ore in association with other elements. Silver occurs primarily as sulfides, in association with iron (pyrite), lead (galena), and tellurides, and with gold. Silver is found in surface water in various forms: 1) as the monovalent ion (e.g., sulfide, bicarbonate, or sulfate salts); 2) as part of more complex ions with chlorides and sulfates; and 3) adsorbed onto particulate matter (ATSDR, 1990).

**Plants.** Silver is not considered essential for plant growth. Silver concentrations in plants generally range between 0.03 and 0.5 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). The availability of silver in soil to plants is dependent on soil pH, the organic matter content of the soil, and the concentration of manganese oxides in the soil (Kabata-Pendias and Pendias, 1992). Concentrations of silver in leaf tissue that are excessive or toxic to various plant species range from 5 to 10 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 2 mg/kg has been proposed by Efroymson et al. (1997) as a benchmark screening value for silver phytotoxicity.

**Mammals.** Silver is not an essential element for animal life. The highest concentrations of silver in soft tissues of wildlife occur in the liver and spleen (NLM, 1996). The biological half-life of silver in animals is only a few days (NLM, 1996).

The toxicity of silver is dependent on the form of silver and the route of exposure. Ingestion, inhalation, and dermal exposure to silver can induce a toxic response in mammals. Internal antagonistic interactions have been noted between silver and selenium (NLM, 1996). Repeated exposure of animals to silver may produce anemia, enlargement of the heart, growth retardation, and degenerative changes in the liver (NLM, 1996). A value of 89 mg/kg/day has been estimated as the LOAEL (chronic) for rats (IRIS, 2002).

**Aquatic Life.** Accumulation of silver has been reported in algae, daphnia, freshwater mussels, and fathead minnows (NLM, 1996). Biomagnification of silver, however, has not been observed in freshwater systems. The deposition of silver into sediment is dependent on the concentrations of magnesium dioxides, ferric compounds, and clay minerals (NLM, 1996).

The national recommended water quality criteria for the protection of freshwater aquatic life for acute exposure to silver is 3.4 µg/L. The lowest chronic values of silver reported in the literature for fish and daphnia are 0.12 and 2.6 µg/L, respectively (Suter and Tsao, 1996). The test EC<sub>20</sub> for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC<sub>20</sub> value for silver is 0.2 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids that represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC<sub>20</sub> benchmark for daphnids has been determined to be less than 0.56 µg/L (Suter and Tsao, 1996).

#### **4.4.2.15 Thallium**

Thallium is widely distributed in trace amounts in the Earth's crust and is one of the more toxic metals. In the environment, thallium exists in either the monovalent (thallous) or trivalent (thallic) form. Thallium is chemically reactive with air and moisture, undergoing oxidation. Thallium is relatively insoluble in water. Thallium adsorbs to soil and sediment and is not transformed or biodegraded (Callahan et al., 1979).

**Plants.** Thallium is not essential for plant growth. When soluble forms are available, thallium is readily taken up by plants and translocated to aerial parts, probably because of its similarity to potassium. Toxic effects on plants include impairment of chlorophyll synthesis and seed germination, reduced transpiration due to interference in stomatal processes, growth reduction, stunting of roots, and leaf chlorosis (Adriano, 1986). The phytotoxicity benchmark value of 1.0 mg/kg is based on unspecified toxic effects on plants grown in surface soil amended with 1.0 mg/kg thallium (Kabata-Pendias and Pendias, 1992).

**Mammals.** Birds and mammals are exposed to thallium via ingestion of soil, water, and plant material. In mammals, thallium is absorbed primarily from ingestion and is distributed to several organs and tissues, with the highest levels reported in the kidneys (Manzo et al., 1982). Thallium exposure in mammals causes cardiac, neurologic, reproductive, and dermatological effects. Various effects and toxic responses have been reported, including paralysis and pathological changes in the liver, kidneys, and stomach mucosa in rabbits exposed to thallium (Tikhonova, 1967). Testicular toxicity in rats has also been reported (Formigli et al., 1986).

Laboratory toxicity data for rats exposed to thallium sulfate in their drinking water were used to estimate a NOAEL value of 0.0074 mg/kg/day (Formigli et al., 1986). Reproduction was the endpoint for this study.

**Aquatic Life.** In aquatic organisms, thallium is absorbed primarily from ingestion and thereafter bioconcentrates in the organism. Toxic effects have been observed in numerous aquatic organisms, including daphnia, fat-head minnow, bluegill sunfish, and others (EPA, 1980b). The Tier II secondary acute water quality value and secondary chronic water quality value for thallium, as calculated by the method described in the EPA's *Proposed Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 110 and 12 µg/L, respectively.

The test EC<sub>20</sub> for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC<sub>20</sub> value for thallium is 81 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids that represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC<sub>20</sub> benchmark for daphnids has been determined to be less than 64 µg/L (Suter and Tsao, 1996).

#### **4.4.2.16 Zinc**

Zinc is a naturally occurring element that may be found in both organic and inorganic forms and, as such, is commonly found in the environment. In general, zinc is concentrated in the sediments of water bodies. The NAS (1977b) has reported that zinc will probably be detected in 75 percent of all water bodies examined for the compound at various locations. The fate of zinc in soils appears to have a pH basis. Studies have shown that a pH of less than 7 often favors zinc desorption (EPA, 1984).

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

Certain species of plants, particularly those from the families Caryophyllaceae, Cyperaceae, and Plumbaginaceae, and some tree species are extremely tolerant to elevated zinc concentrations (Kabata-Pendias and Pendias, 1992). Concentrations of zinc in these plants may reach 1 percent (dry weight) in the plant. Concentrations in leaf tissue that are excessive or toxic to various plant

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

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

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

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

**Aquatic Life.** Zinc residues in freshwater and marine fish are generally much lower than those found in algae and invertebrates. Thus there is little evidence for bioaccumulation (Moore and Ramamoorthy, 1984). Rainbow trout (*Oncorhynchus mykiss*) have the ability to detect and avoid

areas of water containing 5.6 ppb zinc (Sprague, 1968). Cairns and Scheier (1968) reported 96-hour LC<sub>50</sub>s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC<sub>50</sub>s ranging from 2.86 to 3.78 ppm in soft water. These results demonstrate that water hardness affects the toxicity of zinc to fish. Chronic toxicity tests have been conducted with five species of freshwater fish. Chronic values ranged from 47 µg/L for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinalis*) (EPA, 1980c).

Acute toxicity to freshwater invertebrates is relatively low and, as with other metals, increasing water hardness decreases the toxicity of zinc (Moore and Ramamoorthy, 1984). As reported by Baudouin and Scoppa (1974), the 48-hour LC<sub>50</sub> for the cladoceran *Daphnia hyalina* was 0.055 mg/L, and 5.5 mg/L for the copepod *Cyclops abyssorum*. Four chronic toxicity tests are reported for *Daphnia magna*, with chronic values ranging from 47 to 136 µg/L (EPA, 1980c). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1980c).

#### **4.4.2.17 Polycyclic Aromatic Hydrocarbons**

PAHs are a diverse group of organic chemicals consisting of substituted and unsubstituted polycyclic and heterocyclic aromatic rings in which interlinked rings have at least two carbon atoms in common (Zander, 1983). They are formed as a result of incomplete combustion of organic materials such as wood, coal, and oil and exist in the environment in quantity, both from anthropogenic and natural sources. Activities associated with large releases of PAHs include coke production; petroleum refining; the manufacture of carbon black, coal tar pitch and asphalt; heating and power generation; and emissions from internal combustion engines. It is estimated that approximately 270,000 metric tons of PAHs reach the environment yearly (Eisler, 1987a).

**Plants.** Some PAHs are synthesized by plants at very low concentrations (Sims and Overcash, 1983). Background concentrations of specific PAH compounds usually range from 22 to 88 micrograms per kilogram (µg/kg) in tree leaves, 48 to 66 µg/kg in cereal crop plants, 0.05 to 50 µg/kg in leafy vegetables, 0.01 to 6 µg/kg in underground vegetables, and 0.02 to 0.04 µg/kg in fruits (Sims and Overcash, 1983). In general, PAH concentrations are usually greater in aboveground plant parts than in belowground parts and are greater on plant surfaces than within internal tissues (Eisler, 1987a).

Lower-molecular-weight PAHs are taken up from soil by plants more readily than higher-molecular-weight PAHs (Eisler, 1987a). Soil-to-plant concentration ratios for total PAHs have been reported to range from 0.001 to 0.183 (Talmage and Walton, 1990). Atmospheric

deposition is believed to be the usual source of PAHs in plants, not uptake from soil (Sims and Overcash, 1983).

Limited data exist on the phytotoxicity of PAHs to plants. Benzo(b)fluoranthene concentrations of 6,254  $\mu\text{g}/\text{kg}$  in soil were reported to reduce stem growth in wheat but did not affect rye plants. Benzo(a)pyrene and benzo(b)fluoranthene soil concentrations of up to 18,000  $\mu\text{g}/\text{kg}$  do not appear to be severely toxic to higher plants. There is some evidence that low concentrations of some PAHs may actually stimulate plant growth (Sims and Overcash, 1983).

**Mammals.** Most of the PAHs taken into the body are not accumulated but are oxidized and the metabolites excreted (NLM, 1996). In fact, most PAH compounds are detoxified and excreted from the body (Klaassen et al., 1991). PAHs are metabolized in vertebrates by a group of enzymes in the liver known as mixed-function oxidases. A few laboratory studies on rodents have revealed that acute oral toxicities of PAHs are greatest for benzo(a)pyrene, followed in decreasing order of toxicity by phenanthrene, naphthalene, and fluoranthene (Sims and Overcash, 1983). Laboratory toxicity data for mice fed benzo(a)pyrene through oral intubation were used to estimate a NOAEL value of 1.0 mg/kg/day (Mackenzie and Angevine, 1981). Reproduction was the endpoint for this study.

Sims and Overcash (1983) have reported  $\text{LC}_{50}$  values for rodents (*Rattus* spp. and *Mus* spp.) as 50 mg/kg/day benzo(a)pyrene, 700 mg/kg/day phenanthrene, and 2,000 mg/kg/day fluoranthene. Sublethal effects manifested as decreased pup weight in mice have been reported at 10 mg/kg/day benzo(a)pyrene (MacKenzie and Angevine, 1981). Subchronic and chronic effects of exposure to PAHs in rats include liver and kidney damage, unspecified changes in peripheral blood pattern, body weight loss, genetic aberrations, and increased serum aminotransferase activity (Knobloch et al., 1969).

**Birds.** Hoffman and Gay (1981) measured embryotoxicity of various PAHs applied externally to the surface of mallard duck eggs. Approximately 0.002  $\mu\text{g}/\text{egg}$  of 7,12-dimethylbenz(a)anthracene (DMBA) caused 26 percent mortality in 18 days and, among the survivors, produced significant reduction in embryonic growth and a significant increase in the percent of abnormalities, e.g., incomplete skeletal ossification, defects in eye, brain, liver, feathers, and bill. At 0.1  $\mu\text{g}$  DMBA/egg, only 10 percent survived to day 18.

**Aquatic Life.** In general, PAHs as a group are not appreciably acutely toxic (Eisler, 1987a; Neff, 1985). The toxicity of PAH compounds to fish is related to the solubility of the compound in water. The toxicity of PAHs to aquatic organisms is very species-specific and related to the

organisms' ability to metabolize and excrete the compound (Eisler, 1987a). For aquatic organisms, only PAHs in the molecular weight range from naphthalene to pyrene are considered acutely toxic. Toxicity in this group increases with increasing molecular weight. There is some evidence to suggest that PAHs are responsible for reproductive and teratogenic effects in eggs of the sand sole (*Psettichthys melanostictus*) exposed to 0.1 µg benzo(a)pyrene/L for 5 days. The eggs showed reduced and delayed hatch and, when compared to controls, produced larvae with high accumulations (2.1 mg/kg fresh weight) and gross abnormalities, such as tissue overgrowths, in 50 percent of the test larvae (Hose et al., 1982).

Inhibited reproduction of daphnids and the delayed emergence of larval midges by fluorene was reported by Finger et al. (1985). When sediment PAH levels are elevated, benthic organisms obtain a majority of their PAHs from sediments through their ability to mobilize PAHs from the sediment/pore water matrix. The elevated levels in the tissues of these organisms could provide a significant source of PAHs to predatory fish. However, fish do have the ability to efficiently metabolize and degrade PAHs.

#### **4.4.2.18 4,4'-DDT and Metabolites**

DDT is a chlorinated pesticide that has been banned in the United States since 1972.

Dichlorodiphenyldichloroethene (DDE) and DDD are metabolites of DDT. DDT and its metabolites will adsorb very strongly to soil and are subject to evaporation and photodegradation at the soil surface. DDT and related compounds are very persistent in soils (NLM, 1996).

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

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

The most significant exposure route for vertebrates to DDT and its metabolites is oral exposure. Dermal exposure is believed to be very limited, and inhaled DDT and associated particulates are believed to be deposited in the upper respiratory tract and eventually swallowed. The toxicity of DDT and its metabolites to mammalian and avian wildlife is dependent on the fat content within

the animal. In both mammals and birds, the storage of DDT and DDE in fat is protective, because it decreases the amount of chemical in circulation that may reach the brain, which is the site of toxic action (NLM, 1996).

Laboratory toxicity data for rats exposed to DDT in their diet were used to estimate a NOAEL value of 0.8 mg/kg/day (Fitzhugh, 1948). Reproduction was the endpoint for these studies. Signs of acute DDT poisoning in animals include paresthesia of the tongue, lips and face; apprehension; dizziness; tremor; disturbed equilibrium; and convulsions (Klaassen et al., 1991).

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

**Aquatic Life.** DDT in freshwater environments partitions primarily into sediment. Biodegradation of DDT in sediment may be significant. Bioconcentration factors between 51,000 and 100,000 have been reported for fathead minnows exposed to DDT (NLM, 1996). The federal water quality criteria for the protection of aquatic life for acute and chronic exposures to DDT and its metabolites in freshwater systems are 1.1 and 0.001 µg/L, respectively (EPA, 1995).

#### **4.4.2.19 Phthalates**

Phthalates are a class of predominantly man-made compounds which do not naturally occur in nature. They are manufactured and commonly used to produce flexible plastics, wetting agents, insecticidal sprays, paints, and glues (Hazardous Substances Data Bank, 1996a,b; ATSDR, 1993b). Because of their many uses, phthalates are widespread in the environment and have been identified at low levels in the air, water, and soil. In air, phthalates may be adsorbed to particulate matter and can be transferred to water by wet or dry deposition. In water and soil phthalates are subject to microbial degradation. Both aerobic and anaerobic degradation have been reported. Inman et al. (1984) demonstrated that di-n-butyl phthalate in soil was completely degraded within 100 days. Di-n-butyl phthalate and di-n-octyl phthalate have strong ultraviolet absorption bands at 274 nanometers extending beyond 290 nanometers and are therefore strong candidates for photolysis. However, the estimated photolysis half-life in natural waters is 144

days for both compounds (Callahan et al., 1979). There is some evidence that phthalate esters might be biosynthesized and occur naturally in some plants and organisms (Callahan et al., 1979).

**Mammals.** No studies were located on the effects of phthalate exposure to wildlife. Effects of phthalate esters in laboratory animals were seen at only very high doses (one to two percent di-n-butyl phthalate in the diet in oral studies). The male reproductive system appears to be the most sensitive target organ for acute-duration oral exposure to di-n-butyl phthalate in animals. A LOAEL of 1,000 mg/kg-day was established for decreased testis weight in rats (Oishi and Hiraga, 1980). The mechanism of testicular damage by di-n-butyl phthalate may involve interference with zinc metabolism (Foster et al., 1980). After oral administration, butyl benzyl phthalate was rapidly excreted. Rats and mice exposed to high concentrations of butyl benzyl phthalate lost weight, had testicular atrophy, hemorrhages, and hepatomegaly. LD<sub>50</sub> values for these experiments were 2.3 g/kg for rats and 4.2 to 6.2 g/kg for mice (DIALOG, 1996).

**Birds.** No data were found regarding the toxicity of phthalates to avian species.

**Aquatic Life.** Studies by Sasaki (1978) indicate that both di-n-butyl phthalate and di-n-octyl phthalate are non or low bioaccumulative in fishes. Studies by Streufert et al. (1981) showed the acute 48-hour LC<sub>50</sub>s of di-2-ethylhexyl phthalate and di-n-butyl phthalate to the midge larvae *Chironomus plumosus* to be 18 mg/L and 0.76 mg/L, respectively. Chronic life cycle toxicity tests showed no effect up to 0.36 mg/L di-2-ethylhexyl phthalate on midge emergence, egg production, or egg hatchability.

#### **4.4.2.20 Aldrin**

Aldrin and dieldrin are active against insects by contact or ingestion (Hayes, 1994). Thus, their primary use in the past was for control of corn pests by application to soil and in the citrus industry (EPA, 1980d). Other past uses included general crop protection from insects, timber preservation, and termite-proofing of plastic and rubber coverings of electrical and telecommunication cables, and plywood and building boards (Worthington and Walker, 1987). Aldrin is readily converted to dieldrin, which is ubiquitous in the environment. Dieldrin persists because it is more resistant to biotransformation and abiotic degradation than aldrin and, as a result, it is found at low levels in all media, even at distances from the site of concentration.

**Mammals.** Oral LD<sub>50</sub> values for single doses of aldrin in rats ranged from 39 to 64 mg/kg (Gaines, 1960). Decreased survival in animals consuming aldrin over longer periods was seen at lower doses. Rats exposed to aldrin for six weeks exhibited an increase in mortality at doses of

aldrin of 8 mg/kg/day (National Cancer Institute, 1978). When exposed for two years or more, rats exhibited decreased survival at doses of 2.5 to 5 mg/kg/day aldrin (Deichmann et al., 1970).

A number of adaptive changes characteristically produced by halogenated hydrocarbon pesticides were observed in livers of dogs, mice, and rats exposed to aldrin. These changes include an increase in liver weight and/or size, liver cell enlargement, cytoplasmic eosinophilia with migration of basophilic granules, an increase in the smooth endoplasmic reticulum, an increase in microsomal protein, an increase in cytochrome P-450 content, and an increase in microsomal enzyme activity (Wright et al., 1972).

Several metabolic studies indicate that aldrin is absorbed rapidly from the gastrointestinal tract. Following dosing with radio-labeled aldrin, high levels of radioactivity were detected in the liver, blood, and stomach and/or duodenum of dosed rats within one to five hours. Twenty-four hours following a single oral administration to rats of 10 mg/kg, 50 percent of the dose was found in fat (Hayes, 1994), indicating a strong tendency for aldrin/dieldrin to be sequestered in the fatty tissues of animals.

Laboratory toxicity data for rats exposed to aldrin in their diet were used to estimate a NOAEL value of 0.2 mg/kg/day (Treon and Cleveland, 1955a). Reproduction was the endpoint for this study.

**Aquatic Life.** Aldrin has a potential for high bioaccumulation, as indicated by a log  $K_{ow}$  value that ranges from 4.32 to 6.2 (Briggs, 1981). Measured bioconcentration factors for aldrin were 2,700 for fish and 61,657 for snails (Metcalf et al., 1973). The National Recommended Water Quality Criteria for acute exposures to aldrin in fresh water is 3.0  $\mu\text{g/L}$  (EPA, 1999a). Lethal concentrations ( $LC_{50}$ ) for crustaceans range from 8 mg/L for *Asellus brevicaudus* to 9,800 mg/L for *Gammarus lacustris*. Lethal concentrations ( $LC_{50}$ ) for fish range from 4.0  $\mu\text{g/L}$  for carp to 2,735  $\mu\text{g/L}$  for mosquito fish (Verschueren, 1983).

#### **4.4.2.21 Endrin**

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

**Mammals.** Studies have demonstrated that the nervous system is the primary target for endrin toxicity (ATSDR, 1996b). Exposure of animals to endrin causes central nervous system effects, particularly convulsions (Deichmann et al., 1970). Nonspecific degeneration of the liver, kidney, and brain was observed in animals exposed to lethal doses of endrin (Treon et al., 1955b). Endrin can cause abnormal bone formation, hyperactivity, and death in fetuses of dams exposed during gestation (Chernoff et al., 1979). Endrin appears to be well absorbed orally, and distribution is primarily to fat and skin (ATSDR, 1996b).

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

Laboratory toxicity data for mice exposed to endrin in their diet were used to estimate a NOAEL value of 0.092 mg/kg/day (Good and Ware, 1969). Reproduction was the endpoint for this study.

**Aquatic Life.** When released to water, endrin strongly adsorbs to sediment and bioconcentrates significantly in aquatic organisms (ATSDR, 1996b). Endrin appears to be biomagnified only slightly through various levels of the food chain (Metcalf et al., 1973). Bioconcentration factors for endrin in aquatic organisms range from 80 for fathead minnows (*Pimephales promelas*) to 49,000 for snails (*Physa sp.*) (ATSDR, 1996b). The estimated half-life of endrin in water is more than 4 years (ATSDR, 1996b). The National Recommended Water Quality Criteria for endrin in freshwater are 0.086 and 0.036 µg/L for acute and chronic exposures, respectively (EPA, 1999a). The Ontario Ministry of the Environment (1992) has identified a lowest effect level in sediment of 3 µg/kg and a severe effect level in sediment of 1,300 µg/kg (Persaud, 1993).

Acute toxicity (LC<sub>50</sub>) for various freshwater fish species ranged from 0.5 µg/L for *Oncorhynchus kisutch* to 314 µg/L for mosquito fish (Verschuere, 1983). Acute toxicity (LC<sub>50</sub>) for various aquatic insects ranged from 0.03 µg/L for *Acroneuria pacifica* to 2.4 µg/L for *Pteronarcys californica*.

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

#### **4.4.2.22 2,4-Dinitrotoluene**

Technical grade dinitrotoluene (DNT) is typically composed of 78 percent 2,4-DNT, 19 percent 2,6-DNT, and small amounts of 3,4-DNT, 2,3-DNT, and 2,5-DNT (Dunlap, 1978). 2,4-DNT is primarily used as a chemical intermediate in the manufacture of polyurethanes but also serves as a component of military and commercial explosives (Etnier, 1987).

**Mammals.** DNT is absorbed through the gastrointestinal tract, respiratory tract, and skin in most species (EPA, 1986). Lee et al. (1975) reported that female CD rats absorbed 80 to 90 percent of the orally administered 2,4-DNT within 24 hours. Absorption was 90 to 100 percent in male and female CD rats following subchronic or chronic feeding of 2,4-DNT (Ellis et al., 1979). 2,4-DNT metabolites that are excreted into the bile are subsequently reabsorbed from the intestine (Medinsky and Dent, 1983).

Oral LD<sub>50</sub> values for 2,4-DNT range from 268 to 650 mg/kg for rats and from 1,250 to 1,954 mg/kg for mice (Etnier, 1987). Acute toxic effects in animals include central nervous system depression resulting in ataxia, respiratory depression, and death after a few hours (Ellis et al., 1979). Subchronic exposure of rats to dietary levels of 35, 100, or 350 mg/kg/day 2,4-DNT for 13 weeks resulted in decreased weight gain at all dose levels. Increased relative liver, kidney, and brain weights, decreased spermatogenesis, reticulocytosis, and splenic hemosiderosis occurred at the two higher dose levels. Also observed was neuromuscular dysfunction associated with demyelination of the cerebellum and brain stem at 350 mg/kg/day (Lee et al., 1975).

**Aquatic Life.** The EPA has estimated a bioconcentration factor of 3.8 for aquatic organisms that contain about 7.6 percent lipid (NLM, 1996). Concentrations of 2,4-DNT as low as 330 and 230 µg/L are toxic to freshwater aquatic life following acute and chronic exposure, respectively (EPA, 1986). The criterion to protect freshwater aquatic life as derived using procedures other than the EPA guidelines is 620 µg/L as the 24-hour average, and the concentration should not exceed 1,400 µg/L at any time (Dacre, 1980).

#### **4.4.2.23 MCPP**

Mecoprop is commonly called MCPP. MCPP is a selective, hormone-type phenoxy herbicide. It is applied post-emergence and is used on ornamentals and sports turf, for forest site preparation, and on drainage ditch banks for selective control of surface creeping broadleaf weeds such as clovers, chickweed, lambsquarters, ivy, plantain, and other weeds. MCPP is absorbed by plant leaves and translocated to the roots. It affects enzyme activity and plant growth. It acts relatively slowly, requiring three to four weeks for weed control. The EPA has classified MCPP as toxicity class III-slightly toxic.

The duration of MCPP's residual activity in soil is about two months. Adsorption of MCPP increases with an increase in organic matter in soil. Un-aged MCPP and its salt forms are very mobile in a variety of soils. However, in general, phenoxy herbicides such as MCPP are not sufficiently persistent to reach groundwater.

**Mammals.** MCPP has a low acute toxicity to test animals. The LD<sub>50</sub> is 930 to 1,210 mg/kg for rats and 650 mg/kg for mice. The LD<sub>50</sub> for rats exposed dermally is greater than 4,000 mg/kg. MCPP is a teratogen in rats at moderate to high doses. Oral doses of 125 mg/kg/day of MCPP in pregnant rats from days 6 to 15 of gestation caused increased intra-uterine deaths, decreased body lengths, and an increased incidence of delayed or absent bone formation in offspring (Extension Toxicology Network, 2002).

**Aquatic Life.** MCPP is virtually nontoxic to fish. Available data indicate a low potential for MCPP to bioaccumulate in fish. The 96-hour LC<sub>50</sub> is 124 ppm for rainbow trout and greater than 100 ppm for bluegill sunfish (Extension Toxicology Network, 2002).

#### **4.4.3 Potential Receptors**

Potential ecological receptors at the BGR 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 BGR 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 at the BGR ranges and surface water and sediment in Cane Creek and its tributaries in the vicinity of the BGR ranges via various activities (e.g., feeding, drinking, grooming, bathing). These feeding guilds may also be exposed to site-related chemicals via food web transfers.

Dermal absorption of PAHs and chlorinated herbicides and pesticides from soil is a potential pathway for all feeding guilds at the BGR ranges; however, birds and mammals are less susceptible to dermal exposures because their feathers or fur prevents skin from coming into direct contact with the soil (EPA, 1993). Dermal absorption of inorganic compounds from direct contact with soil is expected to be minimal due to the low dermal permeability of these compounds. Although VOCs have been detected in surface soil at the BGR ranges, they have only been detected sporadically and at relatively low concentrations. Therefore, inhalation of volatiles is not a significant exposure pathway at the BGR ranges. Inhalation of constituents sorbed to soil particles and inhaled as dust is a potential pathway for all of the feeding guilds at the BGR ranges.

Terrestrial species may also be exposed to constituents in surface water in Cane Creek through ingestion of and dermal contact with surface water in Cane Creek. The perennial nature of Cane Creek in the vicinity of the BGR ranges provides a drinking water source throughout the year for many wildlife species.

Aquatic and semi-aquatic (i.e., amphibian) species have a greater potential for exposure to COPECs in surface water or sediment, as they spend a majority of their lifetime in close proximity to water bodies. Aquatic and semi-aquatic species could potentially be exposed to COPECs in surface water and sediment via direct contact, ingestion of surface water and sediment, and ingestion of aquatic vegetation and/or aquatic invertebrates that may have accumulated site-related constituents. Inorganic compounds, one SVOC, and one VOC were the only constituents detected in sediment samples at elevated concentrations relative to ESVs. Although these compounds may accumulate in lower trophic level organisms at the BGR ranges, biomagnification through the aquatic food web is not expected to be significant.

#### **4.4.3.1 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 BGR ranges is mainly grasses and sedges, which are remnants of the maintained grass that was present when the BGR 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 BGR 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 surface water and sediment in Cane Creek at the BGR ranges.

#### **4.4.3.2 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 invertivores, since much of their food (e.g., earthworms and other invertebrates) lives on or below the soil surface.

Typical invertivorous species that could be expected to occur at the BGR 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 could include the wood duck (*Aix sponsa*) and blacknose dace (*Rhinichthys atratulus*).

#### **4.4.3.3 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 surface water through ingestion of water in Cane Creek at the BGR ranges.

Typical omnivorous species expected to occur at the BGR ranges and 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 surface water and sediment in Cane Creek at the BGR ranges.

#### **4.4.3.4 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., 1991; 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 BGR ranges and commonly used as sentinel species in ecological risk assessment include red-tailed hawk (*Buteo jamaicensis*), black vulture (*Coragyps atratus*), and bobcat (*Lynx rufus*).

Because Cane Creek and its tributaries at the BGR ranges are relatively narrow and shallow, 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 Cane Creek at the BGR ranges due to the habitat restrictions. Carnivorous mammals such as the mink (*Mustela vison*) may feed along Cane Creek during certain periods of the year when significant water is present in the creek but most likely would not live adjacent to Cane Creek at the BGR ranges because of the creek's inability to support large fish or other aquatic species.

#### **4.4.3.5 Piscivorous Feeding Guild**

Piscivores are specialists that feed mostly on fish. Therefore, they may be exposed to site-related chemicals that have accumulated in small fish that may inhabit Cane Creek at the BGR ranges. They may also be exposed to surface water and sediment in the creek through ingestion of drinking water and during feeding. Cane Creek is a perennial creek at the BGR ranges and, as such, has flowing water throughout the year. Therefore, it is expected that Cane Creek could be used for drinking purposes by a number of different species. Although piscivorous species could be expected to visit the areas around Cane Creek at the BGR ranges during certain periods of the year when the creek flow is significant, they would not be expected to live near the BGR ranges due to the fact that Cane Creek is 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. SVOCs and VOCs 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 COPECs at the BGR ranges through the food web.

Typical piscivorous species expected to occur near the BGR ranges and commonly used as sentinel species in ecological risk assessment include great blue heron (*Ardea herodias*) and belted kingfisher (*Megaceryle alcyon*). Larger piscivorous fish species (e.g., smallmouth bass, spotted gar) and piscivorous mammals (e.g., mink) are not expected to occur in Cane Creek due to the habitat limitations of Cane Creek in this area and its inability to support larger fish and other aquatic species.

#### **4.4.3.6 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 caerulea*)
- 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 BGR ranges is the gray bat (Garland, 1996). Cane Creek at the BGR ranges has been designated as providing "low quality" foraging habitat for the gray bat (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 of gray bats 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 Cane Creek. Because gray bats are flying mammals and the BGR 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. However, gray bat surveys have not been conducted on Cane Creek in the vicinity of the BGR ranges.

Although not officially listed by USFWS as threatened or endangered, two species that are candidates for federal listing are known to occur at the Marcheta Hill Orchid Seep located directly south of Range 21: the white fringeless orchid (*Plantanthera integrilabia*) and the Diana butterfly (*Speyeria diana*). The white fringeless orchid occurs in bogs and seepages along wooded stream banks and ravines from the coastal plain of Mississippi through Alabama, Georgia, Tennessee, Kentucky, the Carolinas, and Virginia. The plant was recorded in two SINAs on Main Post: Marcheta Hill Orchid Seep and Cave Creek Seep (Garland, 1996).

The other candidate species that is known to occur at the Marcheta Hill Orchid Seep is the Diana butterfly (*Speyeria diana*). Habitat affinity for this butterfly includes wet, rich forested valleys and mountainsides and relatively undisturbed forests, especially near streams (Garland, 1996).

#### **4.4.4 Complete Exposure Pathways**

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

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

If any of these four components is absent, then a pathway is generally considered incomplete. Potentially complete exposure pathways are depicted in the SCM as Figure 4-2.

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.

Ecological receptors could be exposed to constituents in surface water via direct contact or through consumption of water. Aquatic organisms inhabiting contaminated waters would be in constant contact with the COPECs.

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

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. As described in previous sections of this report, groundwater discharge to surface water in Cane Creek is a potentially viable transport mechanism for dissolved constituents during periods of heavy precipitation; however, exposure to these constituents by

ecological receptors is only possible via surface water exposure routes. Potential exposure to groundwater-related constituents is expected to be insignificant compared to other exposure pathways (e.g., exposure to constituents in surface water as a result of surface runoff), since groundwater discharge to Cane Creek is expected to be localized and sporadic.

Groundwater discharge to the ground surface via seeps is also a potential exposure pathway for a number of different organisms in the vicinity of the BGR ranges. Specifically, the area south of Range 21, known as the Marcheta Hill Orchid Seep SINA, is dominated by groundwater seeps. This SINA, described in Section 4.2.6 of this report, is also described in detail in the *Endangered Species Management Plan for Fort McClellan, Alabama* (Garland, 1996). These groundwater seep areas could be used by various animals as a source of drinking water, and they also provide unique habitat for a number of plant and animal species. Semi-aquatic organisms (e.g., amphibians) could use these seeps as breeding grounds, as they are inundated during portions of the year when precipitation is heavy and they support vegetation that is characteristic of saturated soils throughout the year.

Secondary exposure pathways involve constituents that are transferred through different trophic levels of the food chain and may be bioaccumulated and/or bioconcentrated. These 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.

In general, the constituents detected in surface soil at the BGR ranges may bioaccumulate in lower trophic level organisms (e.g., terrestrial invertebrates may bioaccumulate inorganic compounds and PAHs detected in soil); however, they will not bioconcentrate through the food chain. Inorganic compounds generally do not bioconcentrate to any great extent, and PAHs are readily metabolized by higher trophic level organisms. However, several chlorinated herbicides and pesticides detected in surface soil have a propensity to bioconcentrate (4,4'-DDE, 4,4'-DDT, MCP, aldrin, alpha-BHC, and endrin). These chlorinated herbicides and pesticides have a propensity to bioconcentrate through the food chain and therefore may be available to higher trophic level organisms through food chain interactions.

The constituents detected in sediment may bioaccumulate in lower trophic level organisms (e.g., benthic invertebrates may bioaccumulate inorganic compounds detected in sediment); however, they will not bioconcentrate through the food chain. Inorganic compounds and volatile organics

generally do not bioconcentrate to any great extent. The constituents detected in groundwater are not expected to bioaccumulate or bioconcentrate significantly.

Summaries of the potentially complete exposure pathways for the terrestrial and aquatic ecosystems at the BGR ranges are presented in Tables 4-3 and 4-4, respectively.

#### **4.5 Screening-Level Risk Estimation**

A screening-level estimation of potential risk can be accomplished by comparing the exposure point concentration of each detected constituent in each environmental medium to a corresponding screening-level ecological toxicity value. In order to conduct the SLERA, the following steps must be followed:

- Determine appropriate screening assessment endpoints
- Determine the ecological toxicity values that are protective of the selected assessment endpoints
- Determine the exposure point concentrations of constituents detected at the site
- Calculate screening-level hazard quotients.

These steps are summarized below.

##### **4.5.1 Ecological Screening Assessment Endpoints**

Most ecological risk assessments focus on population measures as endpoints, since population responses are more well-defined and predictable than are community or ecosystem responses. For screening-level assessments such as this SLERA, an assessment endpoint is any adverse effect on ecological receptors, where receptors are plant and animal populations and communities, habitats, and sensitive environments.

Adverse effects on populations can be inferred from measures related to impaired reproduction, growth, and survival. Adverse effects on communities can be inferred from changes in community structure or function. Adverse effects on habitats can be inferred from changes in composition and characteristics that reduce the ability of the habitat to support plant and animal populations and communities.

Due to the nature of the SLERA process, most of the screening assessment endpoints are generic in nature (e.g., protection of sediment benthic communities from adverse changes in structure or function).

The assessment endpoints for this SLERA identified for each environmental medium are summarized below:

- **Soil**
  - Protection of the terrestrial invertebrate community from adverse changes in structure and function
  - Protection of the terrestrial plant community from adverse changes in structure and function.
- **Surface Water**
  - Protection of the aquatic community from adverse changes in structure and function.
- **Sediment**
  - Protection of the benthic community from adverse changes in structure and function.

#### **4.5.2 Ecological Screening Values**

The ecological screening values (ESV) used in this assessment represent the most conservative values available from various literature sources and have been selected to be protective of the assessment endpoints described above. These ESVs have been developed specifically for FTMC in conjunction with EPA Region IV and are presented in the *Final Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000a). The ESVs used in this assessment are based on 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 this assessment.

For each environmental medium sampled at the BGR ranges (soil, surface water, and sediment), a hierarchy has been developed that presents an orderly method for selection of ESVs. The hierarchy for selecting ESVs for soil is as follows:

- EPA Region IV constituent-specific ESVs
- EPA Region IV ESVs for general class of constituents
- EPA Region V ecological data quality levels (EDQL)
- EPA Region III Biological Technical Advisory Group (BTAG) values
- ESVs from Talmage et al., 1999.

The hierarchy for selecting ESVs for surface water is as follows:

- EPA Region IV constituent-specific ESVs

- NOAA Screening Quick Reference Tables (SQRT), chronic freshwater ambient water quality criteria
- EPA Region V EDQLs
- Office of Solid Waste and Emergency Response (OSWER) Ecotox Threshold values
- EPA Region III BTAG values
- Lowest chronic value from Suter and Tsao, 1996
- ESVs from Talmage et al., 1999.

The hierarchy for selecting ESVs for sediment is as follows:

- EPA Region IV constituent-specific ESVs
- NOAA SQRTs, chronic freshwater ambient water quality criteria
- EPA Region V EDQLs
- OSWER ecotox threshold values
- EPA Region III BTAG values
- Lowest effect levels from Ontario Ministry of the Environment (1992) presented in Jones et al., (1997)
- ESVs from Talmage et al., 1999
- Sediment quality adverse effect threshold (AET) values from the Puget Sound Estuary Program.

#### **4.5.3 Determination of Exposure Point Concentrations**

Exposure point concentrations represent the chemical concentrations in environmental media that a receptor may contact. Since the exposure point concentration is a value that represents the most likely concentration to which receptors could be exposed, a value that reflects the central tendency of the data set is most appropriate to use. However, at the screening-level stage, the data sets are generally not robust enough for statistical analysis and the level of conservatism in the exposure estimates is high to account for uncertainties. Therefore, in the screening-level stage, the maximum detected constituent concentration in each environmental medium is used as the exposure point concentration. The use of the maximum detected constituent concentration as the exposure point concentration ensures that the exposures will not be underestimated and that constituents will not thereby be inadvertently eliminated from further assessment.

The statistical summaries (including the exposure point concentrations) for soil at the BGR ranges, surface water and sediment in Cane Creek, and groundwater in the vicinity of the BGR ranges are presented in Tables 4-5 through 4-11.

#### **4.5.4 Screening-Level Hazard Quotients**

In order to estimate whether constituents detected in environmental media at the site have the potential to pose adverse ecological risks, screening-level hazard quotients were developed. The screening-level hazard quotients were developed via a three-step process as follows:

- Comparison to ESVs
- Identification of essential macronutrients
- Comparison to naturally occurring background concentrations.

Constituents that were detected in environmental media at the BGR 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 indicated that the MDCC was equal to the chemical's conservative ESV and was interpreted in this assessment as a constituent that does not pose the potential for adverse ecological risk. An  $HQ_{screen}$  value less than one indicated that the MDCC was less than the conservative ESV, and that the chemical is not likely to pose adverse ecological hazards to most receptors. 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 BGR ranges, a mean hazard quotient was also calculated by comparing the arithmetic mean constituent concentrations in each environmental medium to the corresponding ESVs. The calculated

screening-level hazard quotients for surface soil, surface water, sediment, and groundwater at the BGR ranges are presented in Tables 4-5 through 4-11.

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

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

In order to determine whether metals detected in site samples were the result of site-related activities or were indicative of naturally occurring conditions, the maximum metal concentrations measured in site samples were compared to their corresponding BTV. 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 represent constituents that may be the result of site-related activities and require further assessment.

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 macronutrients. Those constituents that had  $HQ_{screen}$  values greater one and were not considered macronutrients were then compared to their corresponding BTVs. If constituent concentrations were determined to be less than their

naturally occurring background concentration, then a risk management decision could result in eliminating these constituents from further assessment.

#### **4.6 Identification of Constituents of Potential Ecological Concern**

Constituents were identified as COPECs if the following conditions were met:

- The MDCC exceeded the ESV, or
- The MDCC was 10 times the BTV if the constituent was identified as a macronutrient, or
- The MDCC exceeded the BTV for inorganics.

If a constituent in a given environmental medium did not meet these conditions, then it was not considered a COPEC at the BGR ranges and was not considered for further assessment. If a constituent met these conditions, then it was considered a COPEC. Identification of a constituent as a COPEC indicates that further assessment of that particular constituent in a given environmental medium may be appropriate. It does not imply that a particular constituent poses risk to ecological receptors.

The COPECs that have initially been identified for surface soil, surface water, sediment, and groundwater at the BGR ranges are presented in Tables 4-5 through 4-11. Figures 4-6 through 4-10 show where the COPECs were quantified in samples of the various site matrices.

In order to focus future ecological risk assessment efforts on the constituents that are the most prevalent at the BGR ranges and have the greatest potential to pose ecological risk, additional lines of evidence were assessed. These additional lines of evidence were scrutinized to aid in the decision process of whether or not to include a constituent as a COPEC in future ecological assessments at the BGR ranges. These additional lines of evidence are discussed in the following sections.

##### **4.6.1 COPECs in Surface Soil**

Antimony, copper, lead, and zinc were frequently detected in surface soil at all of the BGR ranges at concentrations that exceeded their respective ESVs. The highest concentrations of these four constituents were found in locations that are associated with small arms use at these ranges (i.e., soil berms that are the impact areas). Thus, it could be concluded that these constituents are site-related and could be considered COPECs in surface soil at all of the BGR ranges.

#### **4.6.1.1 Surface Soil at Range 21**

In addition to the four inorganic constituents discussed above, arsenic, selenium, 4,4'-DDE, benzo(a)pyrene, fluoranthene, and pyrene were also detected in surface soil samples from Range 21 at concentrations that exceeded their respective ESVs (Table 4-5). Arsenic was detected in two samples out of nine at elevated concentrations ( $HQ_{\text{screen}}$  values ranged from 0.92 to 3.65). Selenium was detected in one sample out of nine at a concentration that slightly exceeded its ecological screening value ( $HQ_{\text{screen}} = 1.11$ ). Based on the fact that these inorganic constituents were infrequently detected, their maximum detected concentrations only slightly exceeded their ESVs, and they were not detected in a pattern that would associate their elevated concentrations with Range 21 activities, it could be concluded that these inorganic constituents may not be COPECs in surface soil at Range 21.

Three PAH compounds (benzo[a]pyrene, fluoranthene, and pyrene) were also detected in surface soil at Range 21. These compounds were infrequently detected (one or two detections out of nine samples) and the estimated concentrations were relatively low ( $HQ_{\text{screen}}$  values ranged from 2.1 to 3.5). The detected PAHs were found in only two samples from Range 21 located near the firing line. The chlorinated pesticide 4,4'-DDE was detected in one surface soil sample out of two surface soil samples analyzed for 4,4'-DDE at Range 21, albeit at an estimated concentration that only slightly exceeded its ESV ( $HQ_{\text{screen}} = 1.32$ ). It could be concluded that, based on the infrequency of detection and the highly conservative nature of the ESVs, these PAH compounds and 4,4'-DDE may not be COPECs in surface soil at Range 21.

#### **4.6.1.2 Surface Soil at Range 22**

As with the other ranges at BGR, surface soil at Range 22 exhibited elevated concentrations of antimony, copper, lead, and zinc with respect to ESVs (Table 4-6). Antimony was detected at elevated concentrations in two surface soil samples, and zinc was detected at an elevated concentration in one sample. Beryllium was also detected in a single surface soil sample out of seven samples at a slightly elevated concentration ( $HQ_{\text{screen}} = 1.18$ ). Fluoranthene and pyrene were detected in a single surface soil sample out of eight total samples at slightly elevated concentrations ( $HQ_{\text{screen}}$  values for fluoranthene and pyrene were 1.1 and 1.5, respectively). The only other compound that was detected in surface soil from Range 22 at an elevated concentration was 4,4'-DDT, which was detected in a single sample at a slightly elevated concentration ( $HQ_{\text{screen}} = 1.28$ ). The elevated concentrations of beryllium, fluoranthene, pyrene, and 4,4'-DDT were all from three samples collected adjacent to the firing line at Range 22. Based on the relative infrequency of detection and the fact that the maximum detected concentrations of these constituents only slightly exceeded their respective ESVs, it could be concluded that these constituents may not be COPECs in surface soil at Range 22.

#### **4.6.1.3 Surface Soil at Range 24-Upper**

Range 24-Upper exhibited elevated concentrations of a number of inorganic compounds in surface soil (Table 4-7). As was the case with the other ranges at BGR, antimony, copper, lead, and zinc were commonly detected and exhibited elevated concentrations with respect to ESVs. Additionally, barium, chromium, manganese, and mercury were detected in one surface soil sample of 38 samples collected, at concentrations that exceeded either their respective ESVs or background threshold values. Aluminum was detected in five of 38 samples at elevated concentrations; selenium, beryllium, and iron were detected in four of 38 samples at elevated concentrations; and silver was detected in 2 of 38 samples at elevated concentrations. Based on the fact that these inorganic constituents were detected relatively infrequently at concentrations that exceeded their respective ESVs and were not detected in a pattern that would associate their elevated concentrations with activities at Range 24-Upper, and in light of the highly conservative nature of the ESVs, it could be concluded that these inorganic constituents may not be COPECs in surface soil at Range 24-Upper.

The chlorinated pesticides aldrin, alpha-BHC, and endrin were detected in a single sample out of five total samples at concentrations that slightly exceeded their respective ESVs ( $HQ_{screen}$  values ranged from 2.48 to 6.0). The detected concentrations of these pesticides do not suggest significant potential for toxicity to ecological receptors at Range 24-Upper. Based on the relative infrequency of detection, the low potential for toxicity from detected concentrations, and the fact that the maximum detected concentrations only slightly exceeded the respective ESVs, it could be concluded that these chlorinated pesticides may not be COPECs in surface soil at Range 24-Upper.

The chlorinated herbicide MCPP was detected at elevated concentrations in three out of five surface soil samples at Range 24-Upper. Studies of the fate and transport of MCPP have shown that the duration of MCPP's residual activity in soil is about two months. Because this range has not been active for a number of years, it is expected that the MCPP detected in soil no longer exhibits any residual activity. Additionally, MCPP has been shown to be practically nontoxic to birds ( $LC_{50} > 5,620$  ppm for mallards and  $LC_{50} > 5,000$  ppm for bobwhite quail), nontoxic to fish ( $LC_{50} = 124$  ppm for rainbow trout and  $LC_{50} > 100$  ppm for bluegill sunfish), and nontoxic to bees (EXTOXNET, 2002). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2002). Because the detected concentrations of MCPP present a low potential for toxicity, MCPP does not have the propensity to bioaccumulate, and it has aged sufficiently to render it inactive, it could be concluded that MCPP may not be a COPEC in soil at Range 24-Upper.

#### **4.6.1.4 Surface Soil at Range 27**

Surface soil at Range 27 exhibited elevated concentrations of a number of inorganic compounds (Table 4-8). As was the case with the other ranges at BGR, antimony, copper, lead, and zinc were commonly detected in surface soil and exhibited elevated concentrations with respect to ESVs. Additionally, arsenic, beryllium, manganese, selenium, and silver were detected in one or two surface soil samples out of ten total samples at concentrations that exceeded either their respective ESVs or background threshold values. HQ values for arsenic, beryllium, selenium, and silver were 7.13, 1.19, 3.52, and 1.37, respectively. The maximum detected concentration of manganese only slightly exceeded its BTV, indicating that the detected manganese may be naturally occurring. Butylbenzyl phthalate was detected in a single surface soil sample from the hillside along the southern boundary of Range 27 at a concentration that slightly exceeded its ESV ( $HQ_{\text{screen}} = 1.46$ ).

Based on the relative infrequency of detection and the fact that the maximum detected concentrations only slightly exceed the ESVs, it could be concluded that these constituents may not be COPECs in surface soil at Range 27.

#### **4.6.1.5 Surface Soil in Range Safety Fans**

Lead was detected in surface soil samples within the BGR range safety fans at concentrations that were within the range of lead concentrations for background soil at FTMC, except for two sample locations. The concentrations of lead in surface soil within the BGR range safety fans are not expected to be toxic to most ecological receptors. Japanese quail fed 5,000 mg/kg lead in their diets showed no effects on survival or food consumption (Hill and Camardese, 1986). Lambs fed 400 mg/kg lead in their diets showed some weight loss but were otherwise normal (Demayo et al., 1982). American kestrel fed 50 mg/kg lead in their diets showed no effects on survival, egg laying, fertility, or eggshell thickness (Pattee, 1984). Because the mean lead concentration in surface soil in the BGR range safety fans was determined to be 50.03 mg/kg (which is within the range of naturally occurring background concentrations of lead and also less than the potentially toxic levels discussed above), lead toxicity is not expected from exposure to surface soils in the BGR range safety fans. Therefore, the safety fans at the BGR ranges are not considered areas that have the potential to pose significant ecological risk.

#### **4.6.2 Surface Water**

Surface water from Cane Creek and its tributaries in the vicinity of the BGR ranges exhibited elevated concentrations of copper and lead in a number of samples (Table 4-9). In addition, three of 24 surface water samples exhibited elevated concentrations of cobalt with respect to the ESV ( $HQ_{\text{screen}} = 3.04$ ). Two of 24 surface water samples exhibited slightly elevated concentrations of thallium ( $HQ_{\text{screen}} = 1.88$ ). Based on the relative infrequency of detection of cobalt and thallium and the fact that the maximum detected concentrations of these two inorganic compounds only slightly exceeded their respective ESVs, it could be concluded that cobalt and thallium are not COPECs in surface water at the BGR ranges.

#### **4.6.3 Sediment**

Sediment from Cane Creek and its tributaries exhibited elevated concentrations of copper and lead in a number of samples (Table 4-10). Barium, iron, manganese, and thallium do not have ESVs, but they were detected in several samples at slightly elevated concentrations relative to their background threshold values. Barium was detected in four of 21 samples at elevated concentrations, iron was detected in two of 21 samples at elevated concentrations, manganese was detected in two of 21 samples at elevated concentrations, and thallium was detected in one of 21 samples at an elevated concentration, relative to background. Antimony was detected in one of 21 samples at a concentration that slightly exceeded its ESV ( $HQ_{\text{screen}} = 1.59$ ). 2,4-DNT was detected in one of 27 sediment samples at an estimated concentration that slightly exceeded the ESV ( $HQ_{\text{screen}} = 1.86$ ). Trichlorofluoromethane was detected in four sediment samples at estimated concentrations that slightly exceeded its ESV ( $HQ_{\text{screen}} = 1.53$ ).

Based on the relative infrequency of the detection of these constituents in sediment at elevated concentrations and the fact that the maximum detected concentrations only slightly exceed their respective ESVs, it could be concluded that these constituents may not be COPECs in sediment at the BGR ranges.

#### **4.6.4 Groundwater**

The rationale for assessing groundwater at the BGR ranges using surface water ESVs was to determine the potential for impacts to aquatic organisms from groundwater intrusion to Cane Creek and its tributaries. Based on the discussion of the groundwater/surface water interactions presented in Section 4.4, it is not clear that significant exchange of constituents between groundwater and surface water is taking place at the BGR ranges. However, for the sake of conservativeness, an assessment of the groundwater at the BGR ranges was conducted.

Several inorganic constituents were detected in groundwater at concentrations that exceeded the ESVs for surface water (Table 4-11). Aluminum, barium, copper, iron, and manganese were detected at concentrations in groundwater that exceeded their surface water ESVs. Aluminum was detected in two samples out of twelve at concentrations that exceeded the BTV. Barium, copper, and iron were detected in one of fifteen samples at concentrations that exceeded their respective BTVs. Manganese was detected in four of sixteen samples at concentrations that exceeded its BTV.

Based on the relative infrequency of detection of these inorganic constituents, the fact that the maximum detected concentrations only slightly exceeded their respective BTVs, and the fact that groundwater/surface water interactions do not appear to be significant at the BGR ranges, it could be concluded that these inorganic constituents are not COPECs in groundwater at the BGR ranges.

#### **4.6.5 Summary of COPECs**

In order to focus on the constituents that are most prevalent at the BGR 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  $HQ_{\text{screen}}$  value, association with Army activities, and bioaccumulation and toxicity potential. Based on these additional lines of evidence, the following COPECs have been identified at the BGR ranges:

- Surface Soil: antimony, copper, lead, and zinc
- Surface Water: copper and lead
- Sediment: antimony, copper, and lead.

#### **4.7 Uncertainty Analysis**

Uncertainties are inherent in any risk assessment and even more so in a SLERA due to the nature of the assessment process and the assumptions used in the process. A number of the major areas of uncertainty in this assessment are presented below.

Sampling strategies at the BGR ranges were designed to incorporate areas of the ranges (i.e., impact zones) where the greatest amount of contamination was suspected to occur based on site history, previous sampling results, and visual observation. Thus, the characterization of the environmental media at the BGR ranges included the areas of suspected maximum constituent concentrations at each of the ranges. Therefore, the analytical data for these ranges may be skewed towards overestimation of average exposures. Although a number of samples were collected at each of the ranges, many of the samples were analyzed for lead only. Only a limited

number of samples were analyzed for a full suite of constituents. Samples collected from the areas suspected of highest constituent concentrations (i.e., impact areas) were analyzed for a full suite of analyses; therefore, it is reasonable to assume that the maximum concentrations of all of the detected constituents were determined from these samples. It is possible that samples collected from other areas of the ranges (i.e., non-impact areas) may exhibit lower concentrations of constituents, but these constituents were not analyzed for.

An area of uncertainty that is inherent in a SLERA is the use of the maximum detected constituent concentration as the exposure point concentration for all receptors in a given medium. Most receptors have a home range large enough to preclude individuals from being exposed to the maximum constituent concentration for their entire lifetimes. Therefore, the actual exposure point concentration of a given constituent for many receptor species would be less than the maximum detected concentration. The use of the maximum detected constituent concentrations as the exposure point concentrations for all receptors may result in an overestimation of exposure for many receptors.

Another degree of uncertainty was introduced into the estimation of exposure point concentrations in soil by the removal of bullets and bullet fragments prior to analysis in the laboratory. The presence of bullets and bullet fragments in soil could act as a source of future contamination as these bullet fragments weather and release lead and other constituents to the soil. The removal of bullets prior to analysis may produce an underestimation of future exposure point concentrations for bullet-related constituents in soil. Bullet fragments may also be directly ingested by birds for grit in their crops. Therefore, removal of bullet fragments from soil samples prior to analysis may act to underestimate the potential exposure of birds and other animals to bullet fragments.

The precision and accuracy of the XRF analyses conducted in the range safety fans are less than those for the analytical data collected for the majority of the areas at the BGR ranges. The XRF analyses were intended as a screening tool to determine the presence or absence of site-related contamination in the range safety fans. Therefore, comparison of XRF data to background data, which was analyzed with a higher level of precision and accuracy, introduces a certain level of uncertainty.

Additionally, there is no consideration given to the bioavailability of COPECs to different organisms. In this SLERA it is assumed that all constituents are 100 percent bioavailable to all receptor organisms. It is known that many constituents (particularly inorganic compounds) have significantly lower bioavailabilities than the 100 percent that was assumed in this assessment

(e.g., 1 to 10 percent for some inorganics in soil). This assumption has the potential to greatly overestimate exposures to certain COPECs.

Several COPECs do not have ESVs. The lack of toxicity data for certain COPECs makes it impossible to determine the potential for ecological risk posed by those constituents. Risks may be underestimated due to this uncertainty.

The ESVs used in this assessment are all the most conservative values from the scientific literature, and many are based on the most sensitive endpoint (NOAEL values) for the most sensitive species tested. A less sensitive endpoint that is still protective of the ecological populations or communities of interest may be the LOAEL or some other endpoint. The use of NOAEL-based ESVs may overestimate potential for risks from certain COPECs. Additionally, certain ESVs may not be applicable to conditions at the BGR ranges. For instance, a number of the sediment ESVs are referenced from MacDonald (1994), which presents sediment benchmark values for coastal waters (saline) in Florida. The surface water of Cane Creek is fresh water and exhibits significantly different physical and chemical characteristics compared to those found in the coastal waters of Florida. Therefore, the use of sediment ESVs developed for the coastal waters of Florida, to determine risks in the freshwater streams of FTMC introduces a significant level of uncertainty. Also, the surface water and soil ESVs do not take into account site-specific conditions at the BGR ranges and thus, introduce a potentially significant level of uncertainty into the assessment.

Another area of uncertainty is the lack of consideration of synergisms and/or antagonisms between COPECs. Although it is widely accepted that synergisms and antagonisms occur between certain constituents under certain conditions, the SLERA process does not provide methods for assessing these potential synergisms/antagonisms.

#### **4.8 Summary and Conclusions**

The potential for ecological risks at the BGR ranges was determined through a SLERA. This ecological screening process consisted of a characterization of the ecological setting at the BGR ranges, development of an SCM, a description of the fate and transport of constituents detected in various environmental media, a description of the ecotoxicity of the various constituents detected at the BGR ranges, a description of the ecological receptors, a description of the complete exposure pathways, calculation of screening-level hazard quotients, identification of COPECs, and a description of the uncertainties within the process.

#### **4.8.1 SLERA Summary**

The following sections describe the findings of the SLERA process at each of the ranges at BGR and of Cane Creek in the vicinity of the BGR ranges.

##### **4.8.1.1 Range 21 Surface Soil**

The following constituents in surface soil at Range 21 exhibited maximum concentrations that exceeded their respective ESVs: antimony, arsenic, copper, lead, selenium, zinc, benzo(a)pyrene, fluoranthene, pyrene, and 4,4'-DDE. These ten constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.11 to 722.5. If mean hazard quotients were considered, arsenic and selenium would not be considered COPECs.

Antimony and arsenic were detected at elevated concentrations in two of nine surface soil samples, and selenium was detected at an elevated concentration in one sample out of nine. The PAHs were detected in only two of nine surface soil samples at Range 21, and the concentrations were relatively low ( $HQ_{\text{screen}}$  values ranged from 2.1 to 3.5). 4,4'-DDE was detected in only one of two surface soil samples at Range 21; the  $HQ_{\text{screen}}$  value was 1.32.

Additional lines of evidence (Section 4.6) suggest that only antimony, copper, lead, and zinc in surface soil have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in surface soil at Range 21.

##### **4.8.1.2 Range 22 Surface Soil**

Antimony, beryllium, copper, lead, zinc, fluoranthene, pyrene, and 4,4'-DDT were detected in surface soil at Range 22 at concentrations that exceeded their respective ESVs. These eight constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.1 to 2,280. If mean hazard quotients were considered, beryllium would not be considered a COPEC.

Beryllium was detected at a slightly elevated concentration in only one sample of a total of seven samples. Fluoranthene and pyrene were detected at slightly elevated concentrations in a single sample out of a total of eight samples. The  $HQ_{\text{screen}}$  values for these compounds ranged from 1.1 to 1.5. 4,4'-DDT was detected in the only surface soil sample that was analyzed for a full suite of analyses at Range 22, and the resultant  $HQ_{\text{screen}}$  value was 1.28.

Additional lines of evidence (Section 4.6) suggest that only antimony, copper, lead, and zinc in surface soil have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in surface soil at Range 22.

#### **4.8.1.3 Range 24 Upper Surface Soil**

The following constituents in surface soil at Range 24 Upper exhibited maximum concentrations that exceeded their respective ESVs: aluminum, antimony, barium, beryllium, chromium, copper, iron, lead, manganese, mercury, selenium, silver, zinc, MCP, aldrin, alpha-chlordane, and endrin. These seventeen constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.01 to 2,147. If mean hazard quotients were considered, barium, beryllium, mercury, and selenium would not be considered COPECs.

Barium, chromium, manganese, and mercury were detected at concentrations that exceeded either their ESV or BTV in a single sample, out of 38 samples collected. The chlorinated pesticides aldrin, alpha-BHC, and endrin were detected at concentrations that slightly exceeded their respective ESVs in a single sample out of five samples analyzed for pesticides.  $HQ_{\text{screen}}$  values for these pesticides ranged from 2.48 to 6.0.

Additional lines of evidence (Section 4.6) suggest that only antimony, copper, lead, and zinc in surface soil have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in surface soil at Range 24 Upper.

#### **4.8.1.4 Range 27 Surface Soil**

Antimony, arsenic, beryllium, copper, lead, manganese, selenium, silver, zinc, and butylbenzyl phthalate were detected in surface soil at Range 27 at concentrations that exceeded their respective ESVs. These ten constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.19 to 835. If mean hazard quotients were considered, beryllium, selenium, silver, and butylbenzyl phthalate would not be considered COPECs.

Arsenic, beryllium, manganese, selenium, and silver were detected at concentrations that exceeded either their respective ESVs or BTVs in one or two surface soil samples out of a total of ten samples collected. Additionally, butylbenzyl phthalate was only detected in a single surface soil sample out of ten samples collected at Range 27 at a concentration that slightly exceeded its ESV ( $HQ_{\text{screen}} = 1.46$ ).

Additional lines of evidence (Section 4.6) suggest that only antimony, copper, lead, and zinc in surface soil have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in surface soil at Range 27.

#### **4.8.1.5 Cane Creek Surface Water**

Cobalt, copper, lead, and thallium were detected in Cane Creek surface water at concentrations that exceeded their respective ESVs. These four constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.88 to 1,212. Two of 24 surface water samples exhibited slightly elevated concentrations of thallium, and three of 24 surface water samples exhibited slightly elevated concentrations of cobalt.

Additional lines of evidence (Section 4.6) suggest that only copper and lead in surface water have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in surface water in Cane Creek and its tributaries in the vicinity of the BGR ranges.

#### **4.8.1.6 Cane Creek Sediment**

The following constituents in sediment from Cane Creek and its tributaries exhibited maximum concentrations that exceeded their respective ESVs: antimony, copper, lead, 2,4-DNT, and trichlorofluoromethane. These five constituents exhibited maximum hazard quotients ( $HQ_{\text{screen}}$ ) ranging from 1.53 to 108.6. Hazard quotients could not be calculated for barium, iron, manganese, and thallium because ESVs were not available for these constituents. If mean hazard quotients were considered, antimony would not be considered a COPEC.

Barium was detected in four of 21 samples at elevated concentrations, iron was detected in two of 21 samples at elevated concentrations, manganese was detected in two of 21 samples at elevated concentrations, and thallium was detected in one of 21 samples at elevated concentrations, relative to background. Antimony was detected in one sample at a concentration that slightly exceeded the ESV. 2,4-DNT was detected in one of 27 samples at an estimated concentration that slightly exceeded the ESV. Trichlorofluoromethane was detected in four of 21 samples at estimated concentrations that exceeded its ESV.

Additional lines of evidence (Section 4.6) suggest that only antimony, copper, and lead in sediment have the potential to pose ecological risk; therefore, these constituents have been identified as COPECs in sediment in Cane Creek and its tributaries in the vicinity of the BGR ranges.

#### **4.8.1.7 Groundwater**

Aluminum, barium, copper, iron, and manganese were detected in groundwater at the BGR ranges at concentrations that exceeded their respective surface water ESVs. Aluminum was detected in two of twelve samples at concentrations that exceeded the BTV for aluminum. Barium, copper, and iron were detected in one of fifteen samples at concentrations that exceeded

their respective BTVs. Manganese was detected in four of sixteen samples at concentrations that exceeded the BTV. It is important to note that copper is the only constituent in groundwater at elevated concentrations that was also detected in surface water at elevated concentrations. However, copper was detected at an elevated concentration in groundwater in only one sample. Ecological receptors could potentially be exposed to constituents in groundwater only through surface water exposure pathways. Groundwater/surface water interchange does not appear to be significant in the upper reaches of Cane Creek but may be significant in the lower (westernmost) reaches of the creek. However, there does not appear to be a significant exchange of contaminants between the two media, based on the differences in the data collected from groundwater and surface water at the BGR ranges. Therefore, none of the constituents in groundwater have been identified as COPECs.

The COPECs at the BGR ranges (Table 4-12) have been identified through a very conservative screening process that uses ESVs based largely on NOAEL values from the scientific literature. If additional lines of evidence are considered, many of these constituents may not be considered COPECs. If, based on a risk management decision, the potential ecological risks at the BGR ranges are determined to be “unacceptable” at this screening-level stage, then a baseline ecological risk assessment is appropriate. The goal of the baseline ecological risk assessment, if deemed necessary, will be to reduce the levels of uncertainty and conservatism in the assessment process and to determine the potential for ecological risk at the BGR ranges through a number of lines of evidence.

#### **4.8.2 Conclusions of SLERA**

The SLERA for the BGR ranges determined that several inorganic constituents (i.e., antimony, copper, lead, and zinc) were routinely detected in surface soil at the BGR ranges at concentrations that exceeded their respective ESVs. Additionally, copper and lead were detected in a number of surface water and sediment samples from Cane Creek and its tributaries at concentrations that exceeded their respective ESVs. Antimony was also detected at elevated concentrations in sediment in Cane Creek. Therefore, the potential exists that these inorganic constituents may pose adverse ecological risks to the flora and/or fauna at the BGR ranges. Although several PAH compounds, chlorinated herbicides and pesticides, and VOCs were also detected in environmental media at the BGR ranges at concentrations that exceeded their respective ESVs, these constituents were detected sporadically and at relatively low concentrations; it could be concluded that these constituents do not pose significant risk to ecological receptors at the BGR ranges.

Because a number of constituents were detected in environmental media at the BGR ranges at concentrations that exceeded their respective ESVs, and because conservative assessment techniques were used in the SLERA process, a more thorough assessment is warranted to reduce uncertainties inherent in the SLERA process and to determine the potential for ecological risk at the BGR ranges.

## 5.0 References

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**ATTACHMENT 1**  
**LIST OF ABBREVIATIONS AND ACRONYMS**

## List of Abbreviations and Acronyms

2,4-D	2,4-dichlorophenoxyacetic acid	BCT	BRAC Cleanup Team	Cl	chlorinated
2,4,5-T	2,4,5-trichlorophenoxyacetic acid	BERA	baseline ecological risk assessment	CLP	Contract Laboratory Program
2,4,5-TP	silvex	BEHP	bis(2-ethylhexyl)phthalate	cm	centimeter
3D	3D International Environmental Group	BFB	bromofluorobenzene	CN	chloroacetophenone
AB	ambient blank	BFE	base flood elevation	CNB	chloroacetophenone, benzene, and carbon tetrachloride
AbB3	Anniston gravelly clay loam, 2 to 6 percent slopes, severely eroded	BG	Bacillus globigii	CNS	chloroacetophenone, chloropicrin, and chloroform
AbC3	Anniston gravelly clay loam, 6 to 10 percent slopes, severely eroded	bgs	below ground surface	CO	carbon monoxide
AbD3	Anniston and Allen gravelly clay loams, 10 to 15 percent slopes, eroded	BHC	betahexachlorocyclohexane	CO <sub>2</sub>	carbon dioxide
Abs	skin absorption	BHHRA	baseline human health risk assessment	Co-60	cobalt-60
ABS	dermal absorption factor	BIRTC	Branch Immaterial Replacement Training Center	CoA	Code of Alabama
AC	hydrogen cyanide	bkg	background	COC	chain of custody; contaminant of concern
ACAD	AutoCadd	bls	below land surface	COE	Corps of Engineers
AcB2	Anniston and Allen gravelly loams, 2 to 6 percent slopes, eroded	BOD	biological oxygen demand	Con	skin or eye contact
AcC2	Anniston and Allen gravelly loams, 6 to 10 percent slopes, eroded	Bp	soil-to-plant biotransfer factors	COPC	chemical(s) of potential concern
AcD2	Anniston and Allen gravelly loams, 10 to 15 percent slopes, eroded	BRAC	Base Realignment and Closure	COPEC	chemical(s) of potential ecological concern
AcE2	Anniston and Allen gravelly loams, 15 to 25 percent slopes, eroded	Braun	Braun Intertec Corporation	CPSS	chemicals present in site samples
ACGIH	American Conference of Governmental Industrial Hygienists	BSAF	biota-to-sediment accumulation factors	CQCSM	Contract Quality Control System Manager
AdE	Anniston and Allen stony loam, 10 to 25 percent slope	BSC	background screening criterion	CRDL	contract-required detection limit
ADEM	Alabama Department of Environmental Management	BTAG	Biological Technical Assistance Group	CRL	certified reporting limit
ADPH	Alabama Department of Public Health	BTEX	benzene, toluene, ethyl benzene, and xylenes	CRQL	contract-required quantitation limit
AEC	U.S. Army Environmental Center	BTOC	below top of casing	CRZ	contamination reduction zone
AEL	airborne exposure limit	BTV	background threshold value	Cs-137	cesium-137
AET	adverse effect threshold	BW	biological warfare; body weight	CS	ortho-chlorobenzylidene-malononitrile
AF	soil-to-skin adherence factor	BZ	breathing zone; 3-quinuclidinyl benzilate	CSEM	conceptual site exposure model
AHA	ammunition holding area	C	ceiling limit value	CSM	conceptual site model
AL	Alabama	Ca	carcinogen	CT	central tendency
ALAD	δ-aminolevulinic acid dehydratase	CAB	chemical warfare agent breakdown products	ctr.	container
amb.	Amber	CAMU	corrective action management unit	CWA	chemical warfare agent
amsl	above mean sea level	CBR	chemical, biological and radiological	CWM	chemical warfare material; clear, wide mouth
ANAD	Anniston Army Depot	CCAL	continuing calibration	CX	dichloroformoxime
AOC	area of concern	CCB	continuing calibration blank	'D'	duplicate; dilution
APEC	areas of potential ecological concern	CCV	continuing calibration verification	D&I	detection and identification
APT	armor-piercing tracer	CD	compact disc	DAAMS	depot area air monitoring system
AR	analysis request	CDTF	Chemical Defense Training Facility	DAF	dilution-attenuation factor
ARAR	applicable or relevant and appropriate requirement	CEHNC	U.S. Army Engineering and Support Center, Huntsville	DANC	decontamination agent, non-corrosive
AREE	area requiring environmental evaluation	CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act	°C	degrees Celsius
ASP	Ammunition Supply Point	CERFA	Community Environmental Response Facilitation Act	°F	degrees Fahrenheit
ASR	Archives Search Report	CESAS	Corps of Engineers South Atlantic Savannah	DCA	dichloroethane
AST	aboveground storage tank	CF	conversion factor	DCE	dichloroethene
ASTM	American Society for Testing and Materials	CFC	chlorofluorocarbon	DDD	dichlorodiphenyldichloroethane
AT	averaging time	CFDP	Center for Domestic Preparedness	DDE	dichlorodiphenyldichloroethene
ATSDR	Agency for Toxic Substances and Disease Registry	CFR	Code of Federal Regulations	DDT	dichlorodiphenyltrichloroethane
ATV	all-terrain vehicle	CG	carbonyl chloride (phosgene)	DEH	Directorate of Engineering and Housing
AWARE	Associated Water and Air Resources Engineers, Inc.	CGI	combustible gas indicator	DEP	depositional soil
AWWSB	Anniston Water Works and Sewer Board	ch	inorganic clays of high plasticity	DFTPP	decafluorotriphenylphosphine
'B'	Analyte detected in laboratory or field blank at concentration greater than the reporting limit (and greater than zero)	CHPPM	U.S. Army Center for Health Promotion and Preventive Medicine	DI	deionized
BCF	blank correction factor; bioconcentration factor	CK	cyanogen chloride	DID	data item description
		cl	inorganic clays of low to medium plasticity	DIMP	di-isopropylmethylphosphonate

## List of Abbreviations and Acronyms (Continued)

DM	dry matter; adamsite	EZ	exclusion zone	GS	ground scar
DMBA	dimethylbenz(a)anthracene	FAR	Federal Acquisition Regulations	GSA	General Services Administration; Geologic Survey of Alabama
DMMP	dimethylmethylphosphonate	FB	field blank	GSBP	Ground Scar Boiler Plant
DOD	U.S. Department of Defense	FD	field duplicate	GSSI	Geophysical Survey Systems, Inc.
DOJ	U.S. Department of Justice	FDA	U.S. Food and Drug Administration	GST	ground stain
DOT	U.S. Department of Transportation	Fe <sup>+3</sup>	ferric iron	GW	groundwater
DP	direct-push	Fe <sup>+2</sup>	ferrous iron	gw	well-graded gravels; gravel-sand mixtures
DPDO	Defense Property Disposal Office	FedEx	Federal Express, Inc.	H&S	health and safety
DPT	direct-push technology	FEMA	Federal Emergency Management Agency	HA	hand auger
DQO	data quality objective	FFCA	Federal Facilities Compliance Act	HCl	hydrochloric acid
DRMO	Defense Reutilization and Marketing Office	FFE	field flame expedient	HD	distilled mustard
DRO	diesel range organics	FFS	focused feasibility study	HDPE	high-density polyethylene
DS	deep (subsurface) soil	FI	fraction of exposure	HEAST	Health Effects Assessment Summary Tables
DS2	Decontamination Solution Number 2	Fil	filtered	Herb.	herbicides
DWEL	drinking water equivalent level	Flt	filtered	HHRA	human health risk assessment
E&E	Ecology and Environment, Inc.	FMDC	Fort McClellan Development Commission	HI	hazard index
EB	equipment blank	FML	flexible membrane liner	HPLC	high performance liquid chromatography
EBS	environmental baseline survey	FMP 1300	Former Motor Pool 1300	HNO <sub>3</sub>	nitric acid
EC <sub>50</sub>	effects concentration for 50 percent of a population	FOMRA	Former Ordnance Motor Repair Area	HQ	hazard quotient
ECBC	Edgewood Chemical/Biological Command	Foster Wheeler	Foster Wheeler Environmental Corporation	HQ <sub>screen</sub>	screening-level hazard quotient
ED	exposure duration	Frtn	fraction	hr	hour
EDD	electronic data deliverable	FS	field split; feasibility study	HRC	hydrogen release compound
EF	exposure frequency	FSP	field sampling plan	HSA	hollow-stem auger
EDQL	ecological data quality level	ft	feet	HTRW	hazardous, toxic, and radioactive waste
EE/CA	engineering evaluation and cost analysis	ft/ft	feet per foot	'I'	out of control, data rejected due to low recovery
Elev.	elevation	FTA	Fire Training Area	IATA	International Air Transport Authority
EM	electromagnetic	FTMC	Fort McClellan	ICAL	initial calibration
EMI	Environmental Management Inc.	FTRRA	FTMC Reuse & Redevelopment Authority	ICB	initial calibration blank
EM31	Geonics Limited EM31 Terrain Conductivity Meter	g	gram	ICP	inductively-coupled plasma
EM61	Geonics Limited EM61 High-Resolution Metal Detector	g/m <sup>3</sup>	gram per cubic meter	ICRP	International Commission on Radiological Protection
EOD	explosive ordnance disposal	G-856	Geometrics, Inc. G-856 magnetometer	ICS	interference check sample
EODT	explosive ordnance disposal team	G-858G	Geometrics, Inc. G-858G magnetic gradiometer	ID	inside diameter
EPA	U.S. Environmental Protection Agency	GAF	gastrointestinal absorption factor	IDL	instrument detection limit
EPC	exposure point concentration	gal	gallon	IDLH	immediately dangerous to life or health
EPIC	Environmental Photographic Interpretation Center	gal/min	gallons per minute	IDM	investigative-derived media
EPRI	Electrical Power Research Institute	GB	sarin	IDW	investigation-derived waste
ER	equipment rinsate	gc	clay gravels; gravel-sand-clay mixtures	IEUBK	Integrated Exposure Uptake Biokinetic
ERA	ecological risk assessment	GC	gas chromatograph	IF	ingestion factor; inhalation factor
ER-L	effects range-low	GCL	geosynthetic clay liner	ILCR	incremental lifetime cancer risk
ER-M	effects range-medium	GC/MS	gas chromatograph/mass spectrometer	IMPA	isopropylmethyl phosphonic acid
ESE	Environmental Science and Engineering, Inc.	GCR	geosynthetic clay liner	IMR	Iron Mountain Road
ESMP	Endangered Species Management Plan	GFAA	graphite furnace atomic absorption	in.	inch
ESN	Environmental Services Network, Inc.	GIS	Geographic Information System	Ing	ingestion
ESV	ecological screening value	gm	silty gravels; gravel-sand-silt mixtures	Inh	inhalation
ET	exposure time	gp	poorly graded gravels; gravel-sand mixtures	IP	ionization potential
EU	exposure unit	gpm	gallons per minute	IPS	International Pipe Standard
Exp.	explosives	GPR	ground-penetrating radar	IR	ingestion rate
E-W	east to west	GPS	global positioning system	IRDMIS	Installation Restoration Data Management Information System

## List of Abbreviations and Acronyms (Continued)

IRIS	Integrated Risk Information Service	μmhos/cm	micromhos per centimeter	NLM	National Library of Medicine
IRP	Installation Restoration Program	MeV	mega electron volt	NO <sub>3</sub> <sup>-</sup>	nitrate
IS	internal standard	min	minimum	NPDES	National Pollutant Discharge Elimination System
ISCP	Installation Spill Contingency Plan	MINICAMS	miniature continuous air monitoring system	NPW	net present worth
IT	IT Corporation	ml	inorganic silts and very fine sands	No.	number
ITEMS	IT Environmental Management System™	mL	milliliter	NOAA	National Oceanic and Atmospheric Administration
'J'	estimated concentration	mm	millimeter	NOAEL	no-observed-adverse-effects-level
JeB2	Jefferson gravelly fine sandy loam, 2 to 6 percent slopes, eroded	MM	mounded material	NR	not requested; not recorded; no risk
JeC2	Jefferson gravelly fine sandy loam, 6 to 10 percent slopes, eroded	MMBtu/hr	million Btu per hour	NRC	National Research Council
JfB	Jefferson stony fine sandy loam, 0 to 10 percent slopes have strong slopes	MOGAS	motor vehicle gasoline	NRCC	National Research Council of Canada
JPA	Joint Powers Authority	MP	Military Police	NRHP	National Register of Historic Places
K	conductivity	MPA	methyl phosphonic acid	ns	nanosecond
KeV	kilo electron volt	MPM	most probable munition	N-S	north to south
K <sub>ow</sub>	octonal-water partition coefficient	MQL	method quantitation limit	NS	not surveyed
L	lewisite; liter	MR	molasses residue	NSA	New South Associates, Inc.
l	liter	MRL	method reporting limit	nT	nanotesla
LBP	lead-based paint	MS	matrix spike	nT/m	nanoteslas per meter
LC	liquid chromatography	mS/cm	millisiemens per centimeter	NTU	nephelometric turbidity unit
LCS	laboratory control sample	mS/m	millisiemens per meter	nv	not validated
LC <sub>50</sub>	lethal concentration for 50 percent population tested	MSD	matrix spike duplicate	O <sub>2</sub>	oxygen
LD <sub>50</sub>	lethal dose for 50 percent population tested	MTBE	methyl tertiary butyl ether	O&G	oil and grease
LEL	lower explosive limit	msl	mean sea level	O&M	operation and maintenance
LOAEL	lowest-observed-adverse-effects-level	MtD3	Montevallo shaly, silty clay loam, 10 to 40 percent slopes , severely eroded	OB/OD	open burning/open detonation
LT	less than the certified reporting limit	mV	millivolts	OD	outside diameter
LUC	land-use control	MW	monitoring well	OE	ordnance and explosives
LUCAP	land-use control assurance plan	MWI&P	Monitoring Well Installation and Management Plan	oh	organic clays of medium to high plasticity
LUCIP	land-use control implementation plan	Na	sodium	ol	organic silts and organic silty clays of low plasticity
max	maximum	NA	not applicable; not available	OP	organophosphorus
MB	method blank	NAD	North American Datum	ORP	oxidation-reduction potential
MCL	maximum contaminant level	NAD83	North American Datum of 1983	OSHA	Occupational Safety and Health Administration
MCLG	maximum contaminant level goal	NAVD88	North American Vertical Datum of 1988	OSWER	Office of Solid Waste and Emergency Response
MCPA	4-chloro-2-methylphenoxyacetic acid	NAS	National Academy of Sciences	OVM-PID/FID	organic vapor meter-photoionization detector/flame ionization detector
MCS	media cleanup standard	NCEA	National Center for Environmental Assessment	OVS	oil/water separator
MD	matrix duplicate	NCP	National Contingency Plan	oz	ounce
MDC	maximum detected concentration	NCRP	National Council on Radiation Protection and Measurements	PA	preliminary assessment
MDCC	maximum detected constituent concentration	ND	not detected	PAH	polynuclear aromatic hydrocarbon
MDL	method detection limit	NE	no evidence; northeast	PARCCS	precision, accuracy, representativeness, comparability, completeness, and sensitivity
mg	milligrams	ne	not evaluated	Parsons	Parsons Engineering Science, Inc.
mg/kg	milligrams per kilogram	NEW	net explosive weight	Pb	lead
mg/kg/day	milligram per kilogram per day	NFA	No Further Action	PBMS	performance-based measurement system
mg/kgbw/day	milligrams per kilogram of body weight per day	NG	National Guard	PC	permeability coefficient
mg/L	milligrams per liter	NGP	National Guardsperson	PCB	polychlorinated biphenyl
mg/m <sup>3</sup>	milligrams per cubic meter	ng/L	nanograms per liter	PCDD	polychlorinated dibenzo-p-dioxins
mh	inorganic silts, micaceous or diatomaceous fine, sandy or silt soils	NGVD	National Geodetic Vertical Datum	PCDF	polychlorinated dibenzofurans
MHz	megahertz	Ni	nickel	PCE	perchloroethene
μg/g	micrograms per gram	NIC	notice of intended change	PCP	pentachlorophenol
μg/kg	micrograms per kilogram	NIOSH	National Institute for Occupational Safety and Health	PDS	Personnel Decontamination Station
μg/L	micrograms per liter	NIST	National Institute of Standards and Technology	PEF	particulate emission factor

## List of Abbreviations and Acronyms (Continued)

PEL	permissible exposure limit	RfD	reference dose	Ss	stony rough land, sandstone series
PERA	preliminary ecological risk assessment	RGO	remedial goal option	SS	surface soil
PES	potential explosive site	RI	remedial investigation	SSC	site-specific chemical
Pest.	pesticides	RL	reporting limit	SSHO	site safety and health officer
PETN	pentarey thritol tetranitrate	RME	reasonable maximum exposure	SSHP	site-specific safety and health plan
PFT	portable flamethrower	ROD	Record of Decision	SSL	soil screening level
PG	professional geologist	RPD	relative percent difference	SSSL	site-specific screening level
PID	photoionization detector	RRF	relative response factor	SSSSL	site-specific soil screening level
PkA	Philo and Stendal soils local alluvium, 0 to 2 percent slopes	RSD	relative standard deviation	STB	supertropical bleach
PM	project manager	RTC	Recruiting Training Center	STC	source-term concentration
POC	point of contact	RTECS	Registry of Toxic Effects of Chemical Substances	STD	standard deviation
POL	petroleum, oils, and lubricants	RTK	real-time kinematic	STEL	short-term exposure limit
POW	prisoner of war	SA	exposed skin surface area	STL	Severn-Trent Laboratories
PP	peristaltic pump; Proposed Plan	SAD	South Atlantic Division	STOLS	Surface Towed Ordnance Locator System®
ppb	parts per billion	SAE	Society of Automotive Engineers	Std. units	standard units
PPE	personal protective equipment	SAIC	Science Applications International Corporation	SU	standard unit
ppm	parts per million	SAP	installation-wide sampling and analysis plan	SUXOS	senior UXO supervisor
PPMP	Print Plant Motor Pool	sc	clayey sands; sand-clay mixtures	SVOC	semivolatile organic compound
ppt	parts per thousand	Sch.	Schedule	SW	surface water
PR	potential risk	SCM	site conceptual model	SW-846	U.S. EPA's <i>Test Methods for Evaluating Solid Waste: Physical/Chemical Methods</i>
PRA	preliminary risk assessment	SD	sediment	SWMU	solid waste management unit
PRG	preliminary remediation goal	SDG	sample delivery group	SWPP	storm water pollution prevention plan
PS	chloropicrin	SDZ	safe distance zone; surface danger zone	SZ	support zone
PSSC	potential site-specific chemical	SEMS	Southern Environmental Management & Specialties, Inc.	TAL	target analyte list
pt	peat or other highly organic silts	SF	cancer slope factor	TAT	turn around time
PVC	polyvinyl chloride	SFSP	site-specific field sampling plan	TB	trip blank
QA	quality assurance	SGF	standard grade fuels	TBC	to be considered
QA/QC	quality assurance/quality control	SHP	installation-wide safety and health plan	TCA	trichloroethane
QAM	quality assurance manual	SI	site investigation	TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
QAO	quality assurance officer	SINA	Special Interest Natural Area	TCDF	tetrachlorodibenzofurans
QAP	installation-wide quality assurance plan	SL	standing liquid	TCE	trichloroethene
QC	quality control	SLERA	screening-level ecological risk assessment	TCL	target compound list
QST	QST Environmental, Inc.	sm	silty sands; sand-silt mixtures	TCLP	toxicity characteristic leaching procedure
qty	quantity	SM	Serratia marcescens	TDEC	Tennessee Department of Environment and Conservation
Qual	qualifier	SMDP	Scientific Management Decision Point	TDGCL	thiodiglycol
'R'	rejected data; resample	s/n	signal-to-noise ratio	TDGCLA	thiodiglycol chloroacetic acid
R&A	relevant and appropriate	SO <sub>4</sub> <sup>-2</sup>	sulfate	TERC	Total Environmental Restoration Contract
RA	remedial action	SOP	standard operating procedure	THI	target hazard index
RAO	removal action objective	SOPQAM	U.S. EPA's <i>Standard Operating Procedure/Quality Assurance Manual</i>	TIC	tentatively identified compound
RBC	risk-based concentration	sp	poorly graded sands; gravelly sands	TLV	threshold limit value
RCRA	Resource Conservation and Recovery Act	SP	submersible pump	TN	Tennessee
RD	remedial design	SPCC	system performance calibration compound	TNT	trinitrotoluene
RDX	cyclonite	SPCS	State Plane Coordinate System	TOC	top of casing; total organic carbon
ReB3	Rarden silty clay loams	SPM	sample planning module	TPH	total petroleum hydrocarbons
REG	regular field sample	SQRT	screening quick reference tables	TR	target cancer risk
REL	recommended exposure limit	Sr-90	strontium-90	TRADOC	U.S. Army Training and Doctrine Command
RFA	request for analysis	SRA	streamlined human health risk assessment	TRPH	total recoverable petroleum hydrocarbons
RfC	reference concentration	SRM	standard reference material	TSCA	Toxic Substances Control Act

## List of Abbreviations and Acronyms (Continued)

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TSDF	treatment, storage, and disposal facility
TWA	time-weighted average
UBR	upper background range
UCL	upper confidence limit
UCR	upper certified range
'U'	not detected above reporting limit
UF	uncertainty 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
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 <sup>3</sup>	cubic yards