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Draft

Remedial Investigation Report
Baby Bains Gap Road Ranges

Range 18, Parcel 74Q; Range 20, Parcel 76Q-X; Range 23, Parcel 79Q; Range 26, Parcel 84Q-X; Range 25, Parcels 83Q and 118Q-X; Range 25 East, Parcel 223Q; Range 28, Parcel 86Q; and Area of Further Investigation for Ranges South of Range 25, Parcels 224Q and 226Q

Fort McClellan
Calhoun County, Alabama

Volume I of IV: Text, Tables, and Figures

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Executive Summary

In accordance with Contract Number DACA21-96-D-0018, Task Order CK11, Shaw Environmental, Inc. (Shaw) conducted remedial investigation (RI) activities at the Baby Bains Gap Road (BBGR) Ranges at Fort McClellan (FTMC) located in Calhoun County, Alabama. This report summarizes RI field activities, assesses the nature and extent of contamination at the BBGR Ranges, presents information on contaminant fate and transport, and evaluates risks to human health and the environment. The ranges and parcels that comprise the BBGR Ranges are:

- Range 18, Down Range Feedback (Known Distance) Range, Parcel 74Q
- Range 20, Infiltration Course, Parcel 76Q-X
- Range 23, Trainfire (Record) Range, Parcel 79Q
- Range 26, Live Fire and Maneuver Range, Parcel 84Q-X
- Range 25, Known Distance Range, Parcel 83Q; and Main Post Impact Area, Parcel 118Q-X
- Range 25 East, Parcel 223Q
- Range 28, Target Detection Range, Parcel 86Q
- Area of Further Investigation for Ranges South of Range 25, Parcels 224Q and 226Q.

The BBGR Ranges are located in the central section of the FTMC Main Post. The BBGR Ranges were primarily used for small-caliber weapons training and most were recently active prior to base closure. Several of the ranges were constructed on land that was previously used for other types of training. Numerous bullets and bullet fragments are present on the ground surface at many of the BBGR Range impact zones.

RI field activities were performed in several stages from July 2001 to September 2002. Environmental sampling included the collection and analysis of 339 surface and depositional soil samples, 144 subsurface soil samples, 19 groundwater samples, 12 surface water samples, and 12 sediment samples. Twenty-one groundwater monitoring wells were installed at the BBGR Ranges to facilitate groundwater sample collection and to provide site-specific geological and hydrogeological information. However, two of the wells did not produce sufficient water for sampling. In addition, x-ray fluorescence screening was performed to further delineate the

extent of metals contamination in surface soil and to evaluate potential contamination within the Range 25 safety fan.

The primary environmental concern at the BBGR Ranges is the prevalence of metals contamination (particularly lead, antimony, and copper) in surface soil. The source of contamination is bullets and bullet fragments from weapons training. Based on the subsurface soil sampling data, the depth of contamination typically does not extend more than one foot below ground surface. At the ranges that were active until base closure, the metals contamination is confined to the portion of the range that served as the impact zone (i.e., the areas adjacent to or downrange of the target line(s) or target mounds where bullets struck the surface). Copper and lead exceeded risk-based screening levels in sediments, although the lead and copper contamination was generally limited to intermittent stream beds within the boundaries of the area of investigation. Pesticides, polynuclear aromatic hydrocarbon (PAH) compounds, two explosive compounds, and one herbicide were also sporadically detected at low levels in surface soils. In groundwater, three pesticides and three explosive compounds were detected at concentrations exceeding risk-based screening values but these results appear to be isolated, low-level detections.

An evaluation of contaminant fate and transport concluded that lead, arsenic, antimony, and copper exceeded U.S. Environmental Protection Agency (EPA) default screening levels. These screening levels describe hydraulic conditions in soil between leachate and groundwater. These metals appear to be immobile in the subsurface environment. Antimony and lead concentrations in sediment indicate that erosion of surface soils is an active transport mechanism at the BBGR Ranges and continued transport can be expected unless mitigated. Although a few organic compounds (pesticides, PAHs, and explosives) were detected at levels above EPA screening values in surface soil, transport of these compounds is not considered a major concern because of their immobility in the soil system (i.e., pesticides, PAHs) or their susceptibility to microbial degradation (i.e., explosives).

A streamlined human health risk assessment (SRA) was performed to evaluate potential risks to human health from exposure to environmental media at the BBGR Ranges. Four plausible receptors were evaluated in the SRA: recreational site user, groundskeeper, construction worker, and resident. The SRA concluded that all of the ranges (in their current condition) are suitable for recreational site use. In the other receptor scenarios evaluated, lead in soil was identified as the primary risk driver at all of the BBGR Ranges, except Range 25 East and Range 28. No chemicals of concern were identified for Range 25 East and Range 28.

A screening-level ecological risk assessment (SLERA) was performed to evaluate potential risks to ecological receptors from exposure to environmental media at the BBGR Ranges. Antimony, copper, and lead in surface soil were the predominant constituents of potential ecological concern (COPEC) identified in the SLERA. These three metals were identified as COPECs at all of the BBGR Ranges except Range 25 East and Range 28. Beryllium and zinc were also identified as COPECs in surface soil at Range 18. Additionally, copper, lead, and gamma-chlordane were determined to be COPECs in Ingram Creek tributary sediments. No COPECs were identified in surface water or groundwater associated with the BBGR Ranges.

An important source of uncertainty associated with the risk assessments is the presence of bullets and lead fragments on the ground surface at most of the BBGR Ranges. Environmental degradation of bullets could result in increasing concentrations of lead in soil, thereby increasing risks to potential human and ecological receptors.

Based on the results of the RI, no further investigation is warranted to define the nature and extent of contamination at the BBGR Ranges. Shaw recommends conducting a baseline ecological risk assessment (BERA) to complete the ecological risk assessment process. Following the BERA, a feasibility study should be performed to evaluate and select a remedial alternative for the protection of human health and the environment.

1.2.2.2 Range 20, Infiltration Course, Parcel 76Q-X

Range 20 is discussed in the EBS and included in the ASR. For the this investigation, the study area (33.4 acres) extends over the entire range, including the M-60 impact zone area (1.7 acres) (Figure 1-9). The range safety fan extends to the southeast, covering an area of approximately 1,613 acres (Figure 1-10). The EBS states that Range 20 has been in use since 1980. Weapons fired at this range included M-60 (.308 caliber) machine gun and M-60 with tracer. The EBS further reports that dynamite, trinitrotoluene (TNT), and C4 were used in explosive pits for artillery impact simulation and may have been used in demolition training. The ASR states that the area near Range 20 was historically used as Old Range 27 (OA-35), which appeared on the 1958 FTMC range map as Close Combat 1 & 2. Old Range 27 included a safety fan that was similar to ranges where live-fire rifle and machine gun training was conducted. Old Range 27 was abandoned in 1967. The ASR also indicates that Range 20 was constructed within the WWI Artillery Impact Area (OA-29 and OA-39). Range 20 appears on Plates 6 and 7 in the ASR.

Aerial Photographs. Although the range history indicates Range 20 was constructed in 1980, the December 1982 aerial photograph and prior photographs do not show any development or activity in this area. Range 20 is evident on the February 1994 photograph and the March 1998 photograph (Figure 1-11). The M-60 machine gun impact area, a cleared area southeast of the range, is prominent in both photographs, as is the presence of 15 circular pits arranged in five staggered rows north of the semicircular access road.

The arrangement of the pits presents four separate paths leading northwest to the four machine gun firing point positions to the east and west of a central range tower. FTMC Regulation 350-2 (FTMC, 1991) describes two machine gun positions instead of the four positions visible in aerial photographs. The M-60 guns were used to provide overhead fire while explosives were used in the pits to simulate artillery impacts during maneuvers. No additional machine gun impact zones are evident in either 1994 or 1998 photographs.

Range Features. Appendix A contains site photographs that show various range features currently found at Range 20. Prior to November 1999, the entire area of Range 20 where the tower, machine gun firing points, and explosive pits were located was regarded, soil was removed, and all of the structures were demolished. Stockpiles of removed soil were sampled prior to disposal at various Calhoun County disposal sites. The material was disposed as nonhazardous waste. This area is now used by USACE-Huntsville for unexploded ordnance operations and has a 10-foot-high security fence around it; therefore, no sampling was conducted inside the firing point area.

1.0 Introduction

The U.S. Army has selected Fort McClellan (FTMC), located in Calhoun County, Alabama, for 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 (DOD) installations would be closed or realigned. The BRAC Environmental Restoration Program requires investigation and cleanup of federal properties prior to transfer to the public domain. The U.S. Army is conducting environmental studies of the impact of suspected contaminants at parcels at FTMC under the management of the U.S. Army Corps of Engineers (USACE)-Mobile District. The USACE contracted Shaw Environmental, Inc. (Shaw), formerly IT Corporation (IT), to conduct a remedial investigation (RI) of the Baby Bains Gap Road (BBGR) Ranges under Contract Number DACA21-96-D-0018, Task Order CK11.

The ranges and parcels that comprise the BBGR Ranges are:

- Range 18, Down Range Feedback (Known Distance) Range, Parcel 74Q
- Range 20, Infiltration Course, Parcel 76Q-X
- Range 23, Trainfire (Record) Range, Parcel 79Q
- Range 26, Live Fire and Maneuver Range, Parcel 84Q-X,
- Range 25, Known Distance Range, Parcel 83Q; and Main Post Impact Area, Parcel 118Q-X
- Former Range 25 East, Parcel 223Q
- Range 28, Target Detection Range, Parcel 86Q
- Area of Further Investigation for Ranges South of Range 25, Parcels 224Q and 226Q.

These ranges are located within the central portion of the FTMC Main Post in Anniston, Alabama (Figure 1-1).

The RI has been conducted in different stages at the BBGR Ranges. Sampling procedures followed methodology outlined in the installation-wide sampling and analysis plan (SAP) (IT, 2000a, 2002a). Three site-specific field sampling plans (SFSP) were prepared to direct the field investigation activities (IT, 2001a, 2001b, 2002b). The SFSPs were used as attachments to the

installation-wide work plan (IT, 1998, 2002c) and the SAP, which includes the installation-wide safety and health plan and quality assurance plan.

In July and August 2001, soil and groundwater sampling was conducted at the Ranges South of Range 25 (IT, 2001a). Soil sampling at Range 25 was performed in October and November 2001; groundwater sampling was conducted in February and May 2002 (IT, 2001b). Fieldwork began at the other BBGR Ranges in February 2002, and the first phase was completed in June 2002 (IT, 2002b). A second phase of soil sampling was conducted at Range 18 and Range 23 in August 2002. During this second phase, x-ray fluorescence (XRF) surveys were performed at Range 18, Range 20, Range 25, and Range 25 safety fan in August and September 2002.

These investigation activities conducted at the BBGR Ranges were summarized and reported in the *Data Evaluation Report and Screening Level Risk Assessment, Baby Bains Gap Road Ranges* (Shaw, 2003a). No additional sampling has been conducted since that report was issued.

1.1 Objectives

The primary objectives of this RI are to determine the nature and extent of contamination at the BBGR Ranges and to identify chemicals that pose an unacceptable risk to human health and the environment. Completion of these objectives under the Installation Restoration Program process enables preparation of a feasibility study for the evaluation and selection of a remedial alternative.

1.2 Site Background

The following sections provide site background information for both FTMC and the BBGR Ranges.

1.2.1 FTMC Site Description and History

FTMC is a U. S. Army facility under the control of the U. S. Army Training and Doctrine Command that was closed under the BRAC program in September 1999. Located in northeast Alabama, near the city of Anniston in Calhoun County, FTMC consisted of three tracts of land: the Main Post, Choccolocco Corridor, and Pelham Range. The majority of development at FTMC is in the northwest area of the Main Post. The City of Anniston is located to the south and west of the Main Post; adjoining the Main Post installation to the east are the Choccolocco Mountains of the Talladega National Forest.

The Main Post, consisting of 18,929 acres, was purchased by the federal government in March 1917 for the construction of a National Guard camp (Camp McClellan). Pistol and rifle ranges

were established north of the camp, automatic rifle and machine gun ranges were established southwest of the camp, and artillery firing ranges were established southeast of the camp toward the Choccolocco Mountains. Camp McClellan expanded throughout the 1920s and 1930s. The advent of World War II (WWII) in the 1940s brought continued growth for the installation. Most notably, the 22,245 acres of Pelham Range were purchased to the west of the Main Post in early 1940 for artillery, tank, and heavy mortar firing. Approximately 4,488 additional acres to the east of the Main Post (Choccolocco Corridor) were leased from the State of Alabama to connect the Main Post to the Talladega National Forest. Choccolocco Corridor was used for various range training activities. The lease was terminated in May 1998.

The post-war period initially brought a decline in operations at FTMC. A decrease in military spending placed the installation on inactive status. However, in 1950 the installation was reinstated to active status because of the Korean Conflict. The U.S. Army Chemical School was established at FTMC in 1951; the large outdoor training areas allowed for specialized chemical training involving chemical warfare protection, decontamination procedures, flame throwers, and the operation of smoke generators. The Base hospital was renovated to specialize in chest diseases. The first permanent Women's Army Corps (WAC) training facility was established in 1955, although two previous WAC detachments had been established at the installation during the 1940s. Radiological training was conducted in the mid-1950s at Iron Mountain, Alpha Field, and Bromine Field, all located on the Main Post, as well as at Rideout Field on Pelham Range.

The mission of FTMC was changed in 1966, and it became the U.S. Army School/Training Center. An Advanced Individual Training Infantry Brigade was activated in 1966 to meet requirements for the Vietnam War. The brigade was deactivated in 1970 due to continued force reduction in Vietnam.

In 1973, the Chemical Corps School closed, along with the U.S. Army Combat Developments Command Chemical/Biological Radiological Agency. Five years later, in 1978, the WAC was disbanded and the WAC school closed.

In 1979, the Military Police School was moved to FTMC. In the same year, the U.S. Army Chemical Corps school was re-established, along with a Brigade for Basic Training. U.S. Army Forces Command units, such as D Company, 46th Engineers, were also garrisoned at the post during the 1970s and 1980s.

The mid-1980s brought additional operations to Pelham Range, which is located approximately two miles northwest of Anniston. This area was used for maneuver training and a wide range of

activities from small-arms training to tank and artillery training. Pelham Range has also been used for chemical decontamination training and radiological training. FTMC operations were deactivated and missions completed with the installation closure on September 30, 1999.

1.2.2 BBGR Ranges Description and History

The sites that comprise the BBGR Ranges are located in the central section of the FTMC Main Post (Figure 1-1). Four ranges located in the study area are directly accessible by Baby Bains Gap Road: Range 20 is south, Range 25 and Range 28 are north, and the Former Range 25 East is divided north to south by Bains Gap Road (Figure 1-2). Snap Lane is an adjoining paved road that leads south from Baby Bains Gap Road, eventually intersecting Iron Mountain Road to the west. Range 23 and Range 26 are located east of Snap Lane, and the Ranges South of Range 25 site is located to the west. Range 18 lies approximately 2,000 feet west of Iron Mountain Road, south of Town Center Road.

An unnamed hill separates Range 20 from the Bains Gap Road Ranges (Range 27, Range 22, Range 21, and Range 24 Upper) to the northeast. This hill is near the terminus of a north-south ridgeline that includes another unnamed hill directly east of Range 23, Reeves Hill, and Holloway Hill to the east and south. This ridgeline forms the far eastern boundary of Range 20, Range 26, and Range 23; therefore, the BBGR Ranges generally slope to the north and west. The headwaters of Ingram Creek are located in this area; eleven small tributaries flow north and west from the surrounding hillsides, through the BBGR Ranges, and empty into Ingram Creek, which joins Cane Creek. Range 18 has Sunset Hill (1,050 feet above mean sea level [amsl]) to the west, and Ramsey Hill (1,280 feet amsl) and Jerry Hill (1,285 feet amsl) to the south. These features direct surface water to the northeast into South Branch of Cane Creek, which flows northwest.

Most ranges have been divided into two main areas: the study area and the safety fan. The study area includes the range firing line(s), target line(s), and impact zone(s). Study areas were originally defined in the SFSP (IT, 2002b) based on the historical range use and topography. However, the study areas were revised based on the results of the investigation activities; contaminated areas were added and uncontaminated areas were removed. The revised study areas are shown in this report. The safety fan is the downrange area where personnel were not allowed when training was in progress. Safety fan limits were originally defined by the Army and are based on the ballistics of the weapons used at the range and on the range orientation.

The Archives Search Report (ASR) indicates that these ranges are located within the impact area of the World War I (WWI) Artillery Impact Area and should be considered an area where unexploded ordnance may exist (USACE, 2001a). The ASR and the environmental baseline survey (EBS) (Environmental Science & Engineering, Inc. [ESE], 1998) summarize much of what is known about these ranges.

1.2.2.1 Range 18, Down Range Feedback (Known Distance) Range, Parcel 74Q

Range 18 is discussed in both the EBS and the ASR. For the purposes of this investigation, the study area encompasses the firing line, target lines, and impact zone and covers approximately 70 acres (Figure 1-3). The range safety fan extends to the south covering an area of approximately 3,304 acres (Figure 1-4). The EBS states that Range 18 was first shown on the 1956 U.S. Army Map Service map for FTMC. Through interviews with FTMC personnel, it was established in the EBS that the range was constructed around 1940 or 1941. Weapons fired at this range were limited to M-1903 Springfield and M-1 Garand rifles, and more recently M-16 automatic rifles (5.56 millimeter) used during day and M-16s with tracer (white phosphorus) for night fire training. The EBS further reports that the range was formerly used as a machine gun range but no fuzed ordnance was used at Range 18. The ASR identifies this range as the Washington Rifle Range and the Washington Known Distance Range (OA-44) and states that the range was built during WWII. The ASR reports that Range 18 was historically used as a rifle range and that there is no indication of explosive ordnance use on the range. Range 18 appears on the ASR maps including WWII to 1950 (Plate 5), 1950 to 1973 (Plate 6), and 1974 to present (Plate 7).

Aerial Photographs. From an aerial photograph taken in September 1940 (Figure 1-5), an area around Range 18 is in use as a training area and is marked by a clearing surrounded by dense woods. This photograph shows several parallel roads or lanes oriented from northwest to southeast. These lanes may be evidence of the machine gun training reported in the EBS for Range 18, although no recorded site use corresponds to this time period.

Later photographs, starting with July 1944 show the range under construction, and the Fall 1954 photograph (Figure 1-6) shows Range 18 in use. Both photographs show definitive bullet channels (scars on the ground's surface resulting from bullet impacts). Figure 1-3 depicts the site features shown on the 1954 photograph.

The November 1969 and March 1973 (Figure 1-7) aerial photographs show continued activity at Range 18, with possibly four firing points and several small structures housing pop-up targets evident. A row of four firing points is located to the north. A gravel- or concrete-lined ditch

appears on the 1973 photograph at the southern end of the impact area. The March 1998 aerial photograph (Figure 1-8) shows continued development with the construction of buildings to the north of the first target line. The March 1998 photograph shows 25- and 75-meter target lines very clearly.

Range Features. Appendix A contains site photographs that show various range features that are currently found at Range 18. Based on the aerial photographs, 35 firing points, possibly located in foxholes, were in the firing line area. In addition, the firing line also included a 15-foot wide, gravel firing lane approximately 10 feet downrange of the possible original foxholes. Currently, only the firing lane is present.

Multiple target lines at 25, 75, 150, 175, 250, and 300 meters (measured from the gravel firing lane north of the study area) are present on the range (Figure 1-3). Bullet channels and bullet fragments are present at all of these target lines. The 75-meter target line has concrete pads and a concrete wall 8 to 10 inches high. The main target line is the 300-meter target line, which consists of a soil berm, approximately 8 to 10 feet high and 12 feet wide, in front of a concrete wall 10 feet high and 3 feet wide. Metal target mounts are affixed to the top of the wall, which presumably provided cover for trainees who would maintain the targets during live fire exercises. A concrete-lined drainage ditch runs the length of the 300-meter target line, carrying runoff water east to a shallow depressed area on the eastern end of the target line area. Historical evidence indicates that the range orientation has always remained the same with the direction of fire to the southeast, away from the main cantonment and toward the hillsides.

Downrange of the 300-meter target line is an unimproved road that traverses Range 18. On the south side of this road is a drainage ditch that carries runoff water to the west from an approximately 50-foot hill, which forms the main impact zone. Additional visual surveys were conducted on this western hillside area to better define the extent of impact zone.

Site access is provided via an unimproved road that connects this area to Iron Mountain Road (east of the parcel) and Fox Road (north of the parcel). Surface water slowly flows from the impact zone area north and east through a series of shallow creeks and ditches that transect the range. This water eventually empties into South Branch of Cane Creek northeast of Range 18, flowing further north towards Cane Creek. The elevation of Range 18 ranges from approximately 800 to 875 feet amsl, with the highest elevation at the top of the hill in the impact zone. Ground surface is relatively flat in the firing line area, with a slight slope downward to the northeast.

1.2.2.2 Range 20, Infiltration Course, Parcel 76Q-X

Range 20 is discussed in the EBS and included in the ASR. For the this investigation, the study area (33.4 acres) extends over the entire range, including the M-60 impact zone area (1.7 acres) (Figure 1-9). The range safety fan extends to the southeast, covering an area of approximately 1,613 acres (Figure 1-10). The EBS states that Range 20 has been in use since 1980. Weapons fired at this range included M-60 (.308 caliber) machine gun and M-60 with tracer. The EBS further reports that dynamite, trinitrotoluene (TNT), and C4 were used in explosive pits for artillery impact simulation and may have been used in demolition training. The ASR states that the area near Range 20 was historically used as Old Range 27 (OA-35), which appeared on the 1958 FTMC range map as Close Combat 1 & 2. Old Range 27 included a safety fan that was similar to ranges where live-fire rifle and machine gun training was conducted. Old Range 27 was abandoned in 1967. The ASR also indicates that Range 20 was constructed within the WWI Artillery Impact Area (OA-29 and OA-39). Range 20 appears on Plates 6 and 7 in the ASR.

Aerial Photographs. Although the range history indicates Range 20 was constructed in 1980, the December 1982 aerial photograph and prior photographs do not show any development or activity in this area. Range 20 is evident on the February 1994 photograph and the March 1998 photograph (Figure 1-11). The M-60 machine gun impact area, a cleared area southeast of the range, is prominent in both photographs, as is the presence of 15 circular pits arranged in five staggered rows north of the semicircular access road.

The arrangement of the pits presents four separate paths leading northwest to the four machine gun firing point positions to the east and west of a central range tower. FTMC Regulation 350-2 (FTMC, 1991) describes two machine gun positions instead of the four positions visible in aerial photographs. The M-60 guns were used to provide overhead fire while explosives were used in the pits to simulate artillery impacts during maneuvers. No additional machine gun impact zones are evident in either 1994 or 1998 photographs.

Range Features. Appendix A contains site photographs that show various range features currently found at Range 20. Prior to November 1999, the entire area of Range 20 where the tower, machine gun firing points, and explosive pits were located was □egarded, soil was removed, and all of the structures were demolished. Stockpiles of removed soil were sampled prior to disposal at various Calhoun County disposal sites. The material was disposed as nonhazardous waste. This area is now used by USACE-Huntsville for unexploded ordnance operations and has a 10-foot-high security fence around it; therefore, no sampling was conducted inside the firing point area.

The main area of investigation for this range is the M-60 impact zone, a cleared area located approximately 1,200 feet southeast of the firing points on the northwestern face of a hillside (Figure 1-9). Historical evidence indicates that the Range 20 orientation has always been to the southeast, towards the northwestern slope of the unnamed hill. The impact zone is visible from the former firing line points and is bare of trees and vegetation. Bullet channels and numerous bullets and bullet fragments were found in this area. No evidence of additional target areas for the M-60 was found during the investigation.

Personnel used the area between the M-60 impact zone and the firing point area for "infiltration maneuver training." This area is moderately to densely wooded. Abandoned vehicles, concrete pillboxes, and a mock cemetery that were used for maneuver training are found in this area. According to former FTMC range control personnel, no live ammunition was fired in the infiltration maneuver training area, only blanks. No indication of bullet impact was noted in this area during investigation activities.

Access to the Range 20 firing line area is provided by a short gravel driveway that connects the range to Bains Gap Road. Two tributaries of Ingram Creek cut across the Range 20 infiltration training area, draining runoff water west towards Ingram Creek. The overall elevation of Range 20 ranges from approximately 925 to 1,200 feet amsl. The lowest elevation is found near the tributaries, and the highest elevation is at the top of the hill east of the M-60 impact zone. Ground surface is relatively flat in the M-60 firing point and explosive pit area, with a 75-foot descent to the tributary in the infiltration training area and a steep 1,025-foot rise towards the impact zone hillside southeast of the M-60 firing points.

1.2.2.3 Range 23, Trainfire (Record) Range, Parcel 79Q

Range 23 is discussed in both the EBS and the ASR. The study area encompasses the firing points, lanes, and target mounds and covers an area of approximately 59 acres (Figure 1-12). The range safety fan extends to the southeast and covers an area of approximately 4,566 acres (Figure 1-10). The EBS states that Range 23 has been in use since 1951. Weapons fired at this range included the M-16 automatic rifle and M-16 with tracer. The EBS further reports evidence of artillery ordnance impact at Range 23, as Base personnel have found shell fragments and an unexploded mortar round in the area. The ASR Range 23 (OA-41) site history states that this range started the Inter-War period as a pistol range and later became a rifle and machine gun range, with multiple orientations and layouts used during this period. Further, the ASR reports that a part of this area was used in Combat Range #1 (OA-43). The ASR also points out that Range 23 is constructed within the WWI Artillery Impact Area (OA-29 and OA-39). The

Range 26 is accessed via a gravel driveway that connects the range to Snap Lane near the intersection of Snap Lane and Bains Gap Road northwest of the parcel. Two tributaries of Ingram Creek collect runoff from the hillsides to the southeast, east, and northeast and flow in a northwesterly direction. The elevation of Range 26 gradually increases from approximately 865 to 975 feet amsl in the firing lanes to 985 feet amsl in the hillside.

1.2.2.5 Range 25, Known Distance Range, Parcel 83Q, and Main Post Impact Area, Parcel 118Q-X

Range 25, one of the oldest and most used ranges at FTMC, is discussed in both the EBS and the ASR. For the purposes of the investigation, the study area encompasses the firing lines, target line, and impact zone and covers approximately 103 acres (Figure 1-18). The range safety fan extends to the northeast, covering an area of approximately 1,700 acres (Figure 1-19). It was first documented on the General Map of Fort McClellan, Office of the Post Quartermaster, 1937, and may have been in use as early as the 1920s. Long-time FTMC personnel believe that Range 25 was constructed as a 600-yard known-distance range for training using M-1903 Springfield rifles (.30 caliber) and M-1 Garand rifles (.30 caliber). This range was also reportedly used as a machine gun range. FTMC Range Control records indicate that the range was used for M-14 training. Day-and-night-phase firing was practiced here. The EBS states that weapons fired at this range included various small arms, e.g., M-14 (7.62 millimeter [mm]), M-16 (5.56mm), M-1 (.30 caliber), and M-60 (7.62mm) with tracers (white phosphorus). The firing direction was to the northeast along the length of Range 25. The ASR points out that this area is within the WWI Artillery Impact Area (OA-29 and OA-39). The Bandholtz Rifle Range (OA-37) (Range 25) is shown in the ASR on Plates 4 through 7.

Parcel 118Q-X consists of an area approximately 3.9 acres that lies north of the Range 25 target berm. In this area, many ordnance items have been observed on the ground surface, including mortar and artillery rounds. Numerous 3-inch mortar rounds were observed behind the backstop at this range during the site visit by ESE personnel preparing the EBS. Personnel of the 142nd Explosive Ordnance Disposal unit accompanied ESE personnel to this site and reported that these 3-inch mortar rounds appeared to be sand-filled practice rounds or were otherwise inert.

Aerial Photographs. Available aerial photographs, beginning with the 1937 photograph (Figure 1-20), show activity in the Range 25 area. The 1937 photograph shows the firing lines and the main target berm in the same layout as the 1998 photograph (Figure 1-21). The range format has not changed significantly during the documented 61-year time period. These

Combat Range #1 appears on ASR maps, including Inter-War Period (Plate 4) and Plate 5. Range 23 is first observed on maps on Plates 6 and 7.

Aerial Photographs. Aerial photographs from 1940 through 1954 do not show any facilities or activity in the Range 23 area. In the November 1961 photograph (Figure 1-13), the outline of the range is first visible. A large clearing is present in the center of the range and four small buildings or targets are present in the center of the southernmost clearing. By the time the September 1964 photograph (Figure 1-14) was taken, six distinctive narrow roads oriented northeast to southwest are present. Presumably, the narrow interior range roads were used for support vehicles to access the target mounds during setup. These roads are even more well defined in the March 1973 photograph and in the March 1998 photograph (Figure 1-15). The orientation of the roads and firing lanes is consistent with the layout depicted on Figure 1-12.

Range Features. Appendix A contains site photographs that show various range features currently found at Range 23. Range 23 features included 16 separate firing lanes with each firing point consisting of a foxhole, sandbags, and a scorer's table (FTMC, 1991). Each firing lane has a series of six mounds with pop-up targets (Figure 1-12). The six mounds are located at 50, 100, 150, 200, 250, and 300 meters from each of the 16 firing points (a total of 96 mounds). These mounds and the area behind the mounds form the main impact area for the range. Recent historical evidence (since 1951) indicates that the range orientation has always remained to the southeast, away from the main cantonment and toward the hillside. The main area of investigation of this range consisted of the target mounds in each lane and the potential impact zone located beyond the 300-meter mounds and on the western slopes of Holloway Hill approximately 1,500 to 2,000 feet to the southeast (Figure 1-10). Additional visual surveys were conducted to better define the extent of impact in this direction.

Access to Range 23 is via Snap Lane just west of the parcel, adjacent to the range firing points. Three tributaries of Ingram Creek are present at the range, carrying hillside runoff water from the area southeast of the range through the firing lane area, northwest under Snap Lane and towards Ingram Creek. Two of these tributaries are seasonal, while the stream that flows from the central portion of Range 23 generally contains water all year. The elevation of Range 23 increases from approximately 850 to 900 feet amsl at the firing lane area, rising beyond the 300-meter mounds to 1,050 feet amsl atop the unnamed hill to the southeast.

1.2.2.4 Range 26, Live Fire and Maneuver Range, Parcel 84Q-X

Range 26 is discussed in both the EBS and the ASR. For the purposes of the investigation, the study area encompasses the firing lanes and impact zones and covers approximately 34.2 acres

(Figure 1-16). The range safety fan extends to the southeast, covering an area of approximately 1,267 acres (Figure 1-10). The EBS states that Range 26 was first displayed on the 1959 U.S. Army Map Service map. FTMC Range Control records show continuous range use from 1976 to Base closure, using M-16 rifles for day and night fire and maneuver training. According to the ASR, this area was previously used as the Infiltration Course (OA-58). The Infiltration Course was first shown on the 1949 aerial photograph map, and this area was labeled "R-26" on the 1958 and 1967 maps. This range was abandoned in 1974. The ASR also indicates that Range 26 was constructed within the WWI Artillery Impact Area (OA-29 and OA-39). The Infiltration Course appears on Plates 5, 6, and 7 in the ASR.

Aerial Photographs. Aerial photographs through 1973 show unimproved roads located in dense woods in this area but no obvious signs of training areas or development are evident. The December 1982 aerial photograph shows Range 26, but the image is not clear. Additional detail is provided on the February 1994 photograph and March 1998 photograph (Figure 1-17), which show the range divided into two sections: a partially wooded maneuver area to the northwest and an open, flat firing lane area with multiple firing points to the southeast. An improved road separates these areas and other roads further divide the firing lane area into four lanes.

Range Features. Appendix A contains site photographs that show various range features currently found at Range 26. In the Range 26 maneuver area, personnel would advance along four 30-meter-wide by 135-meter-long lanes with a 10-meter-wide buffer zone between the lanes (FTMC, 1991). At the end of the maneuver area, the trainees would cross the dividing road and encounter a series of fixed firing points as they advanced further downrange. Each firing point consisted of a gravel pit and stacked railroad ties. From behind these firing points, the trainees would direct their fire at a series of three to four pop-up targets located at the end of the lane (15 total pop-up target locations). Recent historical evidence (since 1976) indicates that the range orientation has always remained the same with the direction of fire to the southeast, away from the main cantonment and toward the hillside. The main areas of investigation at this range are the pop-up target locations at the end of each lane and the potential impact zone located on the hillside beyond the pop-up targets. No bullet channels were observed on the Range 26 pop-up target locations in recent site walks; however, bullet fragments and channels were evident further downrange.

A hillside rises from approximately 25 to 75 feet above the firing lane elevation about 600 to 1,200 feet downrange. Four vehicles with numerous bullet holes are located on this hillside. Bullet channels and fragments are present all around the vehicles. Additional visual surveys were conducted to better define the extent of impact zone in this direction.

photographs confirm the same orientation towards the hillside to the northeast as shown on Figure 1-18.

Range Features. Appendix A contains site photographs that show various range features currently found at Range 25. Historically, Range 25 was used as a known-distance range for rifle training, where personnel would be positioned at one of the five, known-distance firing lines and direct their fire towards a single target line. The firing line distances range from 100 yards to 600 yards. Currently, Range 25 consists of a large, relatively flat area where young pine trees have begun to grow. Each firing line is marked by a small earthen berm approximately 1 foot high. A single, concrete "bunker type" target line, similar to Range 18, is present. This target line has soil mounded in the forward-facing side, a concrete walk along the top of the berm, and metal supports to raise and lower targets during training.

The Range 25 impact zone is characterized on its northeast side by a partially cleared and moderately eroded hillside. The hillside impact area covers only about the eastern half of the target line length. At Parcel 118Q-X, the ground surface gradually slopes to the northwest towards St. Clair Road, approximately 600 feet north. Bullet impact channels and surface fragments are present north of the target berm in the area of Parcel 118Q-X and further east towards the top of the hillside. Expended mortars and other unexploded ordnance items are present throughout the Range 25 impact zone area. Except for the cleared area, which is sloped and eroded, the impact zone is moderately to densely wooded.

Range 25 is accessed via a gravel road that connects the range to Bains Gap Road to the south and by two gravel roads that connect the range to Iron Mountain Road to the west. The roads to Iron Mountain Road have been blocked by FTMC security since Base closure, using mounded soil topped with razor wire fencing. In the firing line area, two tributaries of Ingram Creek flow in a northwesterly direction, crossing Iron Mountain Road, and joining Ingram Creek west of Range 25. A third intermittent stream runs north from the hillside impact zone slope, crosses under St. Clair Road, and joins Cane Creek. Range 25 firing and target line and impact zone area elevation ranges from approximately 820 to over 900 feet amsl in the northeast corner of the impact area.

1.2.2.6 Former Range 25 East, Parcel 223Q; and Range 28, Parcel 86Q

Range 25 East is a historical range described in the EBS. Range 28 was an active range at Base closure in 1998. These ranges overlap along both sides of Bains Gap Road. For the purposes of this investigation, the study area encompasses the possible firing lines and impact zone and

covers approximately 44 acres. Additional ranges that overlap this area include Range 20, Range 26, Range 25, and Range 28 (Figure 1-22). The Range 28 safety fan is shown on Figure 1-19.

The EBS states that former Range 25 East is only shown on the General Map of FTMC, Office of the Post Quartermaster, 1937. According to the EBS, no other documentation for Range 25 East exists and the specific use and dates of operation for this range are unknown. It was presumed in the EBS that this range had the same layout as Range 25 and that the same caliber rifles were fired. The ASR discusses the Bandholtz Rifle Range (OA-37) and states that historical maps show that at one time Range 25 was larger and had a different orientation. This information may suggest the existence of Range 25 East.

Range 28 is discussed in the EBS as the Target Detection Range. The EBS states the range was in use from 1961 until Base closure. The ASR identifies Range 28 as the "Blank Fire and Maneuver Range," which was used from 1976 until Base closure. The ASR describes facilities at Range 28 that included bleachers, latrines, a target house, tower, and five firing lanes. Both the EBS and the ASR describe Range 28 as a range where only M-16 blanks were used. The ASR also states that "multiple 3-inch projectiles, multiple 3-inch stokes mortars, multiple M18 smoke grenades and fighting positions were identified in this area." While ordnance items were commonly found nearby in the Range 25 impact zone area (Parcel 118Q-X), no items were identified by Shaw personnel during investigations and site walks conducted at Range 28.

Other ranges have been built in this area (i.e., part of Old Range 27, Range 20, and Range 26). The ASR also states that this area is within the WWI Artillery Impact Area (OA-29 and OA-39). The Bandholtz Rifle Range (OA-37) (Range 25) is shown in the ASR on Plates 4 through 7. Range 25 East is not separately shown on Plate 4. Range 28 (OA-72) is shown only on Plate 7.

Aerial Photographs. No indication that former Range 25 East was in operation as a training range is evident on aerial photographs from 1937 or 1944. The 1937 photograph shows Range 25 to the west (Figure 1-20), as does the September 1940 photograph (Figure 1-23). In the September 1940 photograph, an opening with a dirt road is present in the woods immediately south of Range 28; however, it is unknown if this road was used to support any firing activities at Range 25 East. A dirt road clearing is visible along the northern boundary of Range 25 East on the 1940 photograph. Later photographs, including the July 1944 and the Fall 1954 photograph, show that Bains Gap Road has been constructed through the former Range 25 East impact zone. Range 28 is first visible in the 1973 photograph. No evidence of buildings or structures at Range 28 is visible until the 1998 photograph (Figure 1-21).

Range Features. Appendix A contains site photographs that show various range features currently found at Range 25 East and Range 28. If the EBS presumption is true that former Range 25 East was constructed in the same layout and orientation as Range 25, Range 25 East would have featured firing lines and a target line berm to the northeast. The historical evidence for Range 25 indicates that, if Range 25 East was ever actually constructed, the range orientation would have remained the same with regard to the direction of fire to the northeast, away from the main cantonment and toward the hillside. The main areas of investigation of this historical range were the possible former firing line areas (not currently reused or investigated for another range) and the possible impact zone located on the hillside northeast of the range. This hillside rises approximately 150 feet in elevation downrange of the original first firing line and forms the main range backstop. Additional visual surveys were conducted but no evidence of range use (e.g., bullets, impact zone) was found.

Range 28 is located near the southwestern base of the hillside immediately north of Bains Gap Road (Figure 1-22). The site is split between the Range 25 study area to the west and the Range 25 East study area on the east. It consists of a small cleared area adjacent to Bains Gap Road. The area now has few distinguishing features and no structures are present. No evidence of bullet impact was found at Range 28, supporting information in the EBS and ASR that only M-16 blanks were used at this range.

Former Range 25 East is bisected by Bains Gap Road, which runs through the impact zone area on the north side of the range. A tributary of Ingram Creek drains runoff water from the hillside to the northwest. Two other tributaries flow from the east through the Range 26 maneuver and firing lane areas. These tributaries converge near the parking area at Range 26, cross under Bains Gap Road, and continue flowing west towards Ingram Creek/Cane Creek. The elevation of Former Range 25 East varies from approximately 825 to 1,025 feet amsl at the top of the hillside at the potential impact zone. The ground surface rises gradually at the firing line area, with a steep rise at the potential impact zone.

1.2.2.7 Area of Further Investigation for Ranges South of Range 25, Parcels 224Q and 226Q

The Ranges South of Range 25 were originally investigated by Shaw during a site investigation (SI) in 2001. SI samples were collected over the area of investigation (34.5 acres) that included almost all of Parcel 224Q and the firing line areas of Parcels 226Q and 227Q. In addition to Parcels 224Q, 226Q, and 227Q described in the EBS, ten other ranges (areas) were shown on Plates 4, 5, 6, and 10 of the ASR within the study area. However, these ranges were not

described in the ASR or the EBS. ASR map plates show the additional ranges were present in the study area between the years of inter-war use (i.e., between WWI and WWII) through 1973.

The SI results for the Ranges South of Range 25 were presented to the FTMC Base Realignment and Closure (BRAC) Cleanup Team (BCT) in October 2001. During this meeting, the BCT decided that the original Ranges South of Range 25 area of investigation would be split into two areas based on the presence of soil contamination identified during the SI. An area of approximately 6 acres in the northeastern portion of the area of investigation (Figure 1-24), where elevated concentrations of lead in surface soil were detected, was moved into this investigation at the BBGR Ranges. The BCT agreed that the remainder of the original area of investigation for the Ranges South of Range 25 would be designated for "no further action." A final SI Report (IT, 2002d) and decision document (IT, 2002e) were published in 2002 in preparation for property transfer for redevelopment. The 6-acre area included in this investigation includes portions of Parcels 224Q and 226Q but not Parcel 227Q; therefore, only Parcels 224Q and 226Q are discussed further in this report.

Parcel 224Q was identified as a pistol range south of Range 25 on the 1937 General Map of FTMC. The direction of fire was likely to the east or south, although a range safety fan was not identified. Former Pistol Range South of Range 25 appears as a large clearing surrounded by sparse vegetation. Bains Gap Road borders the northern portion of the parcel and a westerly flowing tributary of Ingram Creek transects the parcel. The elevation of Parcel 224Q ranges from approximately 825 to 845 feet amsl. Ground surface slopes to the northwest. No other information was available regarding dates of use or training activities conducted at this range.

Former Machine Gun Range, Parcel 226Q, is identified on the 1946 Reservation Map south of Range 25. The direction of fire, based on the range fan presented in the EBS, was to the southeast. The 1946 Reservation Map is the only documentation of this range. The parcel boundary extends in a fan shape to the southeast toward current Range 23. The elevation of Former Machine Gun Range, Parcel 226Q, ranges from approximately 830 to 860 feet amsl in the study area. The impact area for Parcel 226Q overlaps the impact area for Range 23 and Range 26. No other information was available regarding dates of use or training activities conducted at this range.

Aerial Photographs. Available aerial photographs from 1937 to 1998 were reviewed for evidence of range use in the area of investigation for the Ranges South of Range 25. Several photographs referenced in the discussion of Range 23 also include the Ranges South of Range 25

study area. These include photographs from 1961 (Figure 1-13), 1964 (Figure 1-14), and 1998 (Figure 1-15).

The 1937 aerial photograph shows most of the study area, which is mostly wooded except for a cleared area in the vicinity of Parcel 224Q. A clearing extends the length of the range for Parcel 224Q. The 1944 aerial photograph (Figure 1-25) shows more activity in the study area and expanded land use in the area east of Parcel 224Q. The activity appears to be related to Parcel 226Q. Three buildings are located in the center of Parcel 224Q. (These building locations may correspond to the concrete slabs observed during the Shaw site walk in March 2001.) Perpendicular berms or target lines downrange of the probable firing line are observed in Parcel 226Q. These shapes also match the firing line area for Unnamed Range G shown on Plate 5 of the ASR (WWII to 1950 range use). The 1964 aerial photograph clearly shows Range 23, just east of the study area, as an active range.

Range Features. Site walks were conducted by Shaw personnel in March and June 2001. Most of the area was heavily wooded. Along the firing line for Former Machine Gun Range, Parcel 226Q, was a berm, approximately 10 to 18 feet wide, with 3-foot pipes and railroad ties running the length. The berm may have been the firing line for Parcel 226Q. A firebreak extends south from Bains Gap Road midway through Parcel 224Q and bends west towards Ingram Creek. Two concrete slabs were also observed: one measured 24 by 36 feet, and the other measured 25 feet by 36 feet with a block foundation. A mound with broken concrete was located about 50 feet to west of the second concrete slab. There is a 12-foot by 4-foot trench on the western side of Parcel 224Q.

A barren area with bullet fragments on the ground surface is near the center of the probable firing area for Parcel 226Q, just east of Parcel 224Q. Just southwest of this barren area, there is a metal rail possibly used to maneuver small-arms targets. An embankment that appears to have been used as an impact area is located southwest of the barren area.

1.2.3 Previous Investigations

An EBS was conducted by ESE to document current environmental conditions of all FTMC property (ESE, 1998). The study was to identify sites that, based on available information, have no history of contamination and comply with U.S. Department of Defense (DOD) guidance for fast-track cleanup at closing installations. The EBS also provides a baseline picture of FTMC properties by identifying and categorizing the properties by the following seven criteria:

1. Areas where no storage, release, or disposal of hazardous substances or petroleum products has occurred (including no migration of these substances from adjacent areas)

2. Areas where only release or disposal of petroleum products has occurred

3. Areas where release, disposal, and/or migration of hazardous substances has occurred, but at concentrations that do not require a removal or remedial response

4. Areas where release, disposal, and/or migration of hazardous substances has occurred, and all removal or remedial actions to protect human health and the environment have been taken

5. Areas where release, disposal, and/or migration of hazardous substances has occurred, and removal or remedial actions are underway, but all required remedial actions have not yet been taken

6. Areas where release, disposal, and/or migration of hazardous substances has occurred, but required actions have not yet been implemented

7. Areas that are not evaluated or require further evaluation.

For non-Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) environmental or safety issues, the parcel label includes the following components: a unique non-CERCLA issue number, the letter "Q" designating the parcel as a Community Environmental Response Facilitation Act (CERFA) Category 1 Qualified parcel, and the code for the specific non-CERCLA issue(s) present (ESE, 1998). The non-CERCLA issue codes used are:

- A = Asbestos (in buildings)
- L = Lead-based paint (in buildings)
- P = Polychlorinated biphenyls
- R = Radon (in buildings)
- RD = Radionuclides/radiological issues
- X = UXO
- CWM = Chemical warfare material.

The BBGR Ranges were classified as CERFA Category 1 Qualified parcels in the EBS. Category 1 parcels are areas where no storage, release, or disposal (including migration) has occurred; however, the parcels were qualified because chemicals of potential concern may be present as a result of historical range activities. Range 20 (Parcel 76Q-X), Range 26 (Parcel 84Q-X) and the Main Post Impact Area near Range 25 (Parcel 118Q-X) were also assigned the "X" designation for potential UXO.

The EBS was conducted in accordance with the CERFA protocols (CERFA-Public Law 102-426) and DOD policy regarding contamination assessment. Record searches and reviews were performed on all reasonably available documents from FTMC, ADEM, EPA Region 4, and Calhoun County, as well as a database search of CERCLA-regulated substances, petroleum products, and Resource Conservation and Recovery Act-regulated facilities. Available historical maps and aerial photographs were reviewed to document historical land uses. Personal and telephone interviews of past and present FTMC employees and military personnel were conducted. In addition, visual site inspections were conducted to verify conditions of specific property parcels.

Prior to the work performed by Shaw, no previous investigations have been conducted at the BBGR Ranges to determine their environmental condition.

1.3 Report Organization

This RI report is organized as follows:

- ***Chapter 1.0 – Introduction.*** This chapter provides site description and history information for FTMC and the BBGR Ranges.

- ***Chapter 2.0 – Study Area Investigation.*** This chapter summarizes the RI field activities conducted by Shaw at the BBGR Ranges, including environmental sampling and analysis, monitoring well installation, and XRF soil screening.

- ***Chapter 3.0 – Physical Characteristics of Study Area.*** This chapter describes the physical characteristics of the BBGR Ranges including demography and land reuse, meteorology, physiography, sensitive environments, soils, geology, and hydrogeology.

- ***Chapter 4.0 – Nature and Extent of Contamination.*** This chapter summarizes the analytical results and compares data with human health site-specific screening levels (SSSL), ecological screening values (ESV), and background screening values to assess the nature and extent of contamination and probable sources.

- ***Chapter 5.0 – Contaminant Fate and Transport.*** This chapter evaluates the chemical and physical properties of the site-related chemicals discussed in Chapter 4.0. It also describes potential migration mechanisms, contaminant persistence, and contaminant migration.

- ***Chapter 6.0 – Streamlined Human Health Risk Assessment.*** This chapter presents the streamlined human health risk assessment (SRA), including

the conceptual site exposure model, data evaluation, chemicals of potential concern, exposure and toxicity assessments, risk characterization, and uncertainty evaluation.

- ***Chapter 7.0 – Screening Level Ecological Risk Assessment.*** This chapter presents the screening-level ecological risk assessment (SLERA), including the environmental setting, site conceptual model, screening-level risk estimation, constituents of potential ecological concern, and uncertainty analysis.

- ***Chapter 8.0 – Summary, Conclusions, and Recommendations.*** This chapter summarizes the major conclusions of the RI report and provides recommendations for further action.

- ***Chapter 9.0 – References.*** This chapter lists the references cited in this RI report.

2.0 Study Area Investigation

This chapter summarizes RI field activities conducted by Shaw at the BBGR Ranges, including environmental sampling and analysis, and groundwater monitoring well installation.

2.1 UXO Avoidance

UXO avoidance was performed at the BBGR Ranges following methodology outlined in the SAP. Shaw UXO personnel used a low-sensitivity magnetometer to perform a surface sweep of the areas of investigation prior to site access. After the sites were cleared for access, sample locations were monitored by UXO personnel following procedures outlined in the SAP.

2.2 Environmental Sampling

The environmental sampling performed during the field investigation of the BBGR Ranges included the collection of surface soil and depositional soil samples, subsurface soil samples, groundwater samples, surface water samples, and sediment samples for chemical analysis. The sample locations were determined by observing physical characteristics on aerial photographs and based on information gathered during site reconnaissance, including a visual survey of surficial bullet fragments.

The fieldwork was conducted in stages from July 2001 through August 2002. In July and August 2001, soil and groundwater sampling was conducted at the Ranges South of Range 25 (IT, 2001a). Soil sampling at Range 25 was performed in October and November 2001; groundwater sampling was conducted in February and May 2002 (IT, 2001b). Fieldwork began at the other BBGR Ranges in February 2002, and the first phase was completed in June 2002 (IT, 2002b). A second phase of soil sampling was conducted at Range 18 and Range 23 in August and September 2002. During this second phase, XRF soil surveys were performed at Range 18, Range 20, Range 25, and within the Range 25 safety fan. Table 2-1 summarizes the number, type, and analyses performed on the samples collected at the BBGR Ranges.

As shown in Table 2-1, many samples were analyzed only for lead or target analyte list (TAL) metals, which are the primary suspected contaminants of concern at the BBGR Ranges. Select samples were submitted for additional analyses to assess other potential contaminants of concern, including target compound list (TCL) volatile organic compounds (VOC), TCL semivolatile organic compounds (SVOC), nitroaromatic and nitramine explosive compounds, chlorinated and organophosphorus pesticides, and polychlorinated biphenyls (PCB). Additional analyses were performed on samples from impact zone locations where high concentrations of

lead were suspected and in range support areas where organic solvents may have been used for weapon cleaning and maintenance.

2.2.1 Surface Soil and Depositional Soil Sampling

A total of 339 surface and depositional soil samples were collected at the BBGR Ranges, as summarized in Table 2-1 and shown on Figures 2-1 through 2-9. Sampling locations were determined in the field by the on-site geologist based on the work plan sampling rationale, presence of surficial bullet fragments, site features, and topography. Depositional soil samples were collected at proposed surface water/sediment sample locations if water was not present in the stream at the time of sample collection.

Sample Collection. Surface soil samples were collected from 0 to 1 foot below ground surface (bgs) using a stainless-steel hand auger, following the methodology specified in the SAP. Depositional soil samples were collected from the upper 6 inches of soil using a stainless-steel spoon or hand auger. Prior to collecting the surface and depositional soil samples, surface debris (e.g., rocks, vegetation) was removed from the immediate sample area. During sample collection, visible lead fragments were removed from the samples to improve their representativeness. The presence of bullet fragments was documented on the sample collection logs (Appendix B). The soil was screened with a photoionization detector (PID) to measure for volatile organic vapors in accordance with procedures outlined in the SAP. As necessary, samples for VOC analysis were collected directly from the sampler using three EnCore® samplers. The remaining soil was then transferred to a clean stainless-steel bowl, homogenized, and placed in the appropriate sample containers. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

2.2.2 Subsurface Soil Sampling

One hundred and forty-four subsurface soil samples were collected from 139 soil borings at the BBGR Ranges, as summarized in Table 2-1 and shown on Figures 2-1 through 2-9. Soil boring locations were determined in the field by the on-site geologist based on the work plan sampling rationale, presence of surficial bullet fragments, and site topography.

Sample Collection. Subsurface soil samples were collected from soil borings at depths greater than 1 foot bgs in the unsaturated zone. The soil borings were advanced and soil samples collected using either a direct-push technology (DPT) sampling system or a stainless-steel hand auger, following procedures specified in the SAP. Sample collection logs are included in

Appendix B. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

Subsurface soil samples were collected continuously to a maximum depth of 12 feet bgs or until sampler refusal was encountered. Samples were screened using a PID to measure for volatile organic vapors. The soil sample displaying the highest reading was selected and sent to the laboratory for analysis; however, at those locations where PID readings were below background, the deepest soil sample interval above the saturated zone was submitted for analysis. Samples for VOC analysis were collected directly from the sampler with three EnCore samplers. The remaining soil was then transferred to a clean stainless-steel bowl and homogenized before transfer into the appropriate sample containers. During sample collection, visible lead fragments were removed from the samples to improve their representativeness. The presence and description of the fragments present were documented on the sample collection logs (Appendix B). The on-site geologist constructed a detailed boring log for each soil boring (Appendix C).

At the completion of soil sampling, boreholes were abandoned with bentonite pellets and hydrated with potable water following borehole abandonment procedures summarized in the SAP.

2.2.3 Monitoring Well Installation

Twenty-one monitoring wells, including 17 residuum wells and four bedrock wells, were installed at the BBGR Ranges at the locations shown on Figures 2-1 through 2-9. Four residuum monitoring wells (HR-224Q-MW02, HR-226Q-MW01, HR-226Q-MW03 and HR-226Q-MW05) at the Ranges South of Range 25 were not installed because competent bedrock was encountered prior to reaching groundwater. Table 2-2 summarizes construction details of the wells installed at the BBGR Ranges. The lithologic logs and well construction diagrams are included in Appendix C.

2.2.3.1 Residuum Monitoring Wells

Shaw contracted Miller Drilling Company to install the 17 residuum monitoring wells using a hollow-stem auger rig. The wells were installed following procedures outlined in the SAP. The borehole at each well location was advanced with a 4¼-inch inside diameter (ID) hollow-stem auger from ground surface to the first water-bearing zone or until auger refusal was encountered. A 2-foot-long, 2-inch ID carbon steel split-spoon sampler was driven at 5-foot intervals to collect geologic materials for observing and describing lithology. The split-spoon samples and drill cuttings were logged to determine lithologic changes and the approximate depth of groundwater encountered during drilling. This information was used to determine the optimal placement of

the monitoring well screen interval and to provide site-specific geologic and hydrogeologic information. Soil characteristics were described using the "Burmeister Identification System" described in Hunt (1986) and the Unified Soil Classification System as outlined in the American Society for Testing and Materials (ASTM) Method D2488 (ASTM, 2000).

Upon reaching the target depth in each borehole, a 5- to 15-foot length of 2-inch ID, 0.010-inch continuous slot, Schedule 40 polyvinyl chloride (PVC) screen with a PVC end cap was placed through the auger to the bottom of the borehole. The screen and end cap were attached to 2-inch ID, flush-threaded Schedule 40 PVC casing. A filter pack consisting of number 1 filter sand (environmentally safe, clean fine sand, sieve size 20 to 40) was tremied around the well screen to approximately 5 feet above the top of the well screen as the augers were removed. A bentonite seal, consisting of approximately 3 feet of bentonite pellets, was placed immediately on top of the filter pack and hydrated with potable water. In wells where the bentonite seal was installed below the water table surface, the bentonite pellets were allowed to hydrate in the groundwater. Bentonite seal placement and hydration followed procedures in the SAP. Bentonite-cement grout was tremied into the remaining annular space of the well from the top of the bentonite seal to ground surface. A well cap was placed on the PVC casing. A locking protective steel casing was placed around the top of the PVC well casing and a concrete pad was constructed around the wellhead. Four protective steel posts were installed around the well pad.

2.2.3.2 Bedrock Monitoring Wells

Four bedrock monitoring wells (HR-74Q-MW04, HR-79Q-MW03, HR-84Q-MW03 and HR-118Q-MW02) were installed at the BBGR Ranges using a combination of hollow-stem auger, air-rotary, ODEX®, and PQ-3 wireline rock coring techniques. The borehole at each well location was advanced with a 4¼-inch ID hollow-stem auger from ground surface to the top of bedrock or until auger refusal was encountered. A 2-foot-long, 2-inch ID carbon steel split-spoon sampler was driven at 5-foot intervals to collect residuum for observing and describing lithology. The split-spoon samples were logged to determine lithologic changes and the approximate depth of groundwater encountered during drilling.

Bedrock monitoring well HR-74Q-MW04 was installed using an air-rotary rig with a 12¼-inch rotary bit or percussion bit. The borehole was advanced from ground surface to approximately 5 feet into competent bedrock. Eight-inch ID carbon steel International Pipe Standard (IPS) outer casing was installed from ground surface to the bottom of the borehole. A minimum 2-inch annular space was maintained between the outer casing and the borehole wall. The outer casing was grouted in place using a tremie pipe suspended in the annulus outside the casing.

At bedrock well locations HR-79Q-MW03 and HR-84Q-MW03, an air-rotary rig equipped with an 8-inch ODEX percussion bit was used to advance the 8-inch outer casing 5 feet into competent bedrock. At these locations, a minimum 2-inch annular space could not be maintained between the outer casing and borehole wall due to the ODEX drilling. After the grout cured a minimum of 48 hours, an all-terrain vehicle CME-550X drill rig equipped with a PQ-3 wireline was used to collect core samples continuously from the bottom of the 8-inch outer casings to the total depth of the boreholes. After completion of core sample collection, a 7⅝-inch air percussion or rotary bit was used to ream the borehole from the bottom of the outer casing to the total depth of the boring. Water was the only lubricant used during drilling operations.

Bedrock monitoring well HR-118Q-MW02 was installed using essentially the same techniques as previously described except 8-inch outer casing was not installed. A 7⅞-inch hammer bit was used to ream the borehole after core samples were collected.

Upon reaching the target depth in each borehole, a 20- or 30-foot length of 4-inch ID, threaded, flush joint, 0.010-inch continuous wrap Schedule 80 PVC well screen with a 3-inch PVC end cap was placed through the outer casing (or open borehole) to the bottom of the reamed borehole. The screen and end cap were attached to a 4-inch ID, flush threaded, Schedule 80 PVC well riser. A filter pack, consisting of number 1 filter sand (environmentally safe, clean fine sand, sieve size 20 to 40), was tremied in place from the bottom of the borehole to approximately 5 feet above the top of the screen. The filter pack also included a minimum 5-foot layer of extra fine filter sand (sieve size 30 to 70). A bentonite seal, consisting of approximately 5 to 15 feet of bentonite pellets, was placed immediately on top of the filter pack and hydrated with potable water. Bentonite-cement grout was tremied into the remaining annular space of the well from the top of the bentonite seal to ground surface. A well cap was placed on the PVC well casing. A locking protective steel casing was placed around the top of the PVC well casing, and a cement pad was constructed around the wellhead. Four protective steel posts were installed around the well pad.

2.2.3.3 Well Development

The wells were developed by surging and pumping with a submersible pump or using a Teflon bailer, in accordance with methodology outlined in the SAP. The submersible pump used for well development was moved up and down the water column to encourage any residual well installation materials to enter the well. These materials were then pumped out of the well to re-establish the natural hydraulic flow conditions. Development continued until the water turbidity was equal to or less than 20 nephelometric turbidity units (NTU), for a maximum of 8 hours (for

the residuum wells) or 12 hours (bedrock wells), or until the well was pumped dry and allowed to recharge three times. The well development logs are included in Appendix D.

2.2.4 Groundwater Sampling

Groundwater samples were collected from 19 of the 21 monitoring wells installed at the BBGR Ranges. Two residuum monitoring wells (HR-74Q-MW02 and HR-79Q-MW04) could not be sampled because they did not produce enough water. The well/groundwater sample locations are shown on Figures 2-1 through 2-9. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

Sample Collection. Groundwater sampling was performed following procedures outlined in the SAP. Groundwater was sampled after purging a minimum of three well volumes and after field parameters (i.e., temperature, pH, dissolved oxygen, specific conductivity, oxidation-reduction potential, and turbidity) stabilized. Purging and sampling were performed using a mechanical pump (i.e., peristaltic, submersible, or bladder pump) equipped with Teflon™ tubing. Samples for VOC analysis were collected via the "tube evacuation" method in instances where a peristaltic pump was used for sampling. Field parameters were measured using a calibrated water-quality meter, as summarized in Table 2-3. Sample collection logs are included in Appendix B. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

2.2.5 Water Level Measurements

The depth to groundwater was measured in the permanent wells at the BBGR Ranges and in select wells at surrounding parcels on December 17, 2002, following procedures outlined in the SAP. Depth to groundwater was measured with an electronic water-level meter. The meter probe and cable were cleaned before use at each well following decontamination methodology presented in the SAP. Measurements were referenced to the top of the PVC well casing, as summarized in Table 2-4.

2.2.6 Surface Water Sampling

Twelve surface water samples were collected in Ingram Creek, South Branch of Cane Creek, and their adjoining tributaries at the BBGR Ranges, as shown on Figures 2-1 through 2-9. Surface water samples were collected following methodology outlined in the SAP. The actual sampling locations were determined in the field, based on drainage pathways and field observations.

Sample Collection. The surface water samples were collected by dipping a stainless-steel pitcher in the water and pouring the water into the appropriate sample containers, following procedures in the SAP. The samples were collected after field parameters had been measured using a calibrated water quality meter. Surface water field parameters are summarized in Table 2-5. The sample collection logs are included in Appendix B. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

2.2.7 Sediment Sampling

Twelve sediment samples were collected at the same locations as the surface water samples, as shown on Figures 2-1 through 2-9. Sampling locations were determined in the field, based on drainage pathways and field observations.

Sample Collection. Samples were collected from the upper 6 inches of sediment with a stainless-steel spoon and placed in a stainless-steel bowl, following procedures outlined in the SAP. The sample fraction for VOC analysis was then immediately collected using three EnCore samplers. The remaining sample was homogenized and placed in the appropriate sample containers. Sample collection logs are included in Appendix B. The samples were analyzed for the parameters listed in Table 2-1 using methods outlined in Section 2.4.

2.2.8 XRF Soil Screening

A calibrated, field-portable XRF instrument was used at Range 18, Range 20, Range 25, and Range 25 safety fan to analyze soils collected at 123 locations. In the study areas of Range 18, Range 20, and Range 25, the XRF was used to further delineate the extent of contamination, usually around an existing sample location that contained elevated concentrations of lead and/or copper (Figures 2-1, 2-2, and 2-7). In the Range 25 safety fan, 25 soil samples were screened using XRF to evaluate potential metals contamination (Figure 2-10).

The XRF analysis was either performed at the sample location in the field or in an onsite laboratory. Samples analyzed in the onsite laboratory were further prepared by drying, crushing, and sieving before analysis. The XRF results were copied onto XRF laboratory results forms 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 BBGR Ranges site-specific sampling and analysis plan.

Fourteen percent of the XRF soil screening samples were selected for quality control (QC) confirmation analysis at a fixed-base laboratory (EMAX Laboratories, Inc.) using EPA Method

6010B for lead. Samples containing a range of lead concentrations were selected for confirmational analysis. Once selected, the archived portion of the field sample was air dried, crushed with a ceramic pestle, and passed through a standard No. 10 sieve (2 millimeter pore size). The prepared soil was transferred into a clean 120-milliliter glass sample container, labeled, and shipped to the laboratory under standard chain-of-custody procedures.

2.3 Surveying of Sample Locations

Monitoring well and sample locations were surveyed using GPS survey techniques and conventional civil survey techniques described in the SAP. Horizontal coordinates were referenced to the U.S. State Plane Coordinate System, Alabama East Zone, North American Datum of 1983. Elevations were referenced to the North American Vertical Datum of 1988. Horizontal coordinates and elevations are included in Appendix E.

2.4 Analytical Program

Samples collected during the field investigations at the BBGR Ranges were analyzed for various chemical parameters based on the potential site-specific chemicals and on EPA, ADEM, FTMC, and USACE requirements. Target analyses included the following parameters using EPA SW-846 methods, including Update III methods where applicable:

- TCL VOCs – EPA Method 8260B
- TCL SVOCs – EPA Method 8270C
- TAL metals (including lead) – EPA Method 6010B/7000
- Chlorinated pesticides – EPA Method 8081A
- Chlorinated herbicides – EPA Method 8151A
- Organophosphorous pesticides – EPA Method 8141A
- Nitroaromatic/nitramine explosives – EPA Method 8330
- PCBs – EPA Method 8082.

As shown in Table 2-1, not all samples were analyzed for the same parameters. Select samples were analyzed only for lead or metals and nitroaromatic explosives, while others were analyzed for a broader list that included organic compounds. In addition to chemical analyses, the sediment samples were analyzed for grain size (ASTM Method D422) and total organic carbon (EPA Method 9060).

2.5 Sample Preservation, Packaging, and Shipping

Sample preservation, packaging, and shipping followed requirements specified in the SAP. Sample containers, sample volumes, preservatives, and holding times for the analyses are listed in the SAP. Sample documentation and chain-of-custody records (Appendix B) were completed as specified in the SAP.

Completed analysis request and chain-of-custody records were included with each shipment of sample coolers to EMAX Laboratories, Inc. in Torrance, California. Geotechnical analyses were performed by Shaw Geotechnical Laboratory in Oak Ridge, Tennessee.

2.6 Investigation-Derived Waste Management and Disposal

Investigation-derived waste (IDW) was managed and disposed as outlined in the SAP. IDW generated during the field investigations at the BBGR Ranges was segregated as follows:

- Drill cuttings
- Purge water from well development, sampling activities, and decontamination fluids
- Spent well materials and personal protective equipment.

Solid IDW was stored in lined roll-off bins in a fenced area surrounding the FTMC Environmental Management office prior to characterization and disposal. Solid IDW was characterized using the toxicity characteristic leaching procedure (EPA Method 1311) in accordance with procedures described in the IDW Management Plan in the SAP. Based on the results, drill cuttings, spent well materials, and personal protective equipment generated during the RI were disposed as nonhazardous waste at either the Industrial Waste Landfill on the Main Post of FTMC (for waste disposed before September 30, 2001) or at the Three Corners Landfill in Piedmont, Alabama (for waste disposed after September 30, 2001).

Liquid IDW was contained in the sump associated with the former DEH Compound and/or in portable tanks. Liquid IDW was characterized by VOC, SVOC, and metals analyses using the ADEM-approved procedure described in the IDW Management Plan in the SAP. Based on the analytical results, liquid IDW was discharged on the ground surface as nonhazardous, nonregulated waste or sent to the City of Anniston wastewater treatment plant for disposal.

2.7 Variances/Nonconformances

Six variances to the SFSPs were recorded during completion of the RI at the BBGR Ranges. These variances did not alter the intent of the investigation or the sampling rationale presented in work plans. Copies of the variance reports are included in Appendix F.

No nonconformances were recorded during completion of field activities at the BBGR Ranges.

2.8 Data Quality

The field sample analytical data are presented in tabular form in Appendix G. The field samples were collected, documented, handled, analyzed, and reported in a manner consistent with the site-specific work plans; the FTMC SAP and quality assurance plan; and standard, accepted methods and procedures. Data were reported and evaluated in accordance with Corps of Engineers South Atlantic Savannah Level B criteria (USACE, 2001b) and the stipulated requirements for the generation of definitive data as described in the SAP. Chemical data were reported via hard-copy data packages by the laboratory using Contract Laboratory Program-like forms.

Data Validation. The reported analytical data were validated in accordance with EPA National Functional Guidelines by Level III criteria. The results of the data validation are summarized in the data validation summary report (Appendix H). Selected results were rejected or otherwise qualified based on the implementation of accepted data validation procedures and practices. These qualified parameters are highlighted in the report. The validation-assigned qualifiers were added to the FTMC ShawView™ database for tracking and reporting. The data presented in this report, except where qualified, meet the principle data quality objective for this investigation.

3.0 Physical Characteristics of Study Area

The physical characteristics of the BBGR Ranges are important to understanding the current nature and extent of contamination and the future transport of contaminants. These characteristics can be described in terms of demography and land reuse, meteorology, physiography, sensitive environments, soils, geology, hydrology, and hydrogeology.

3.1 Demography and Land Reuse

FTMC includes 45,679 acres of government-owned and formerly leased land situated in the foothills of the Appalachian Mountains of northeast Alabama. The post is located in Calhoun County (population 112,249), approximately 60 miles northeast of Birmingham (population 242,820), approximately 75 miles northwest of Auburn (population 42,987), and approximately 90 miles west of Atlanta, Georgia (population 416,474) (U.S. Census Bureau, 2000). The city of Anniston (population 24,276) adjoins the Main Post on the south and east. The city of Weaver (population 2,619) is located approximately 1 mile northwest of the Main Post, and the city of Oxford (population 14,592) is approximately 5 miles south of Anniston. In Calhoun County, 5 percent of the total labor force is in the armed forces. Of the civilian labor force, the top five industries in which people are employed are: retail trade (18.7 percent); manufacturing, durable goods (12.3 percent); public administration (12.0 percent); manufacturing, non-durable goods (10.8 percent); and educational services (9.0 percent) (SAIC, 2000).

Projected land reuse for FTMC is presented in the *Fort McClellan Comprehensive Reuse Plan* (EDAW, 1997). Almost all of the BBGR Ranges are included in the planned U.S. Fish and Wildlife Service (USFWS) refuge, a passive recreation reuse area. At Range 18, the northern portion of the area of investigation (firing line and target line areas) is planned for industrial reuse and the southern part (including the hillside impact zone) is planned for USFWS refuge.

3.2 Meteorology

FTMC is situated in a temperate, humid climate. Summers are long and hot, and winters are usually short and mild to moderately cold. The climate is influenced by frontal systems moving from northwest to southeast, and temperatures change rapidly from warm to cool due to the inflow of northern air. The average annual temperature is 63 degrees Fahrenheit (°F). Summer temperatures usually reach 90°F or higher about 70 days per year, but temperatures above 100°F are rare. Freezing temperatures are common in winter but are usually of short duration. The first frost may arrive by late October. Snowfall averages 0.5 to 1 inch. On rare occasions, several inches of snow accumulate from a single storm. At Anniston, the average date of the first 32°F

temperature is November 6, and the last is March 30. This provides a growing season of 221 days (ESE, 1998).

The average annual rainfall is approximately 53 inches and is well distributed throughout the year. The more intense rains usually occur during the warmer months, and some flooding occurs nearly every year. Drought conditions are rare, though the entire southeastern United States has been experiencing drought conditions for the three years previous to this writing. Approximately 80 percent of the flood-producing storms are of the frontal type and occur in the winter and spring, lasting from 2 to 4 days each. Summer storms are usually thunderstorms with intense precipitation over small areas, and these sometimes result in serious local floods. Occasionally, several wet years or dry years occur in series. Annual rainfall records indicate no characteristic order or pattern.

Winds in the FTMC area are seldom strong and frequently blow from the northwest. However, there is no truly persistent wind direction. Normally, only light breezes or calm prevails, except during passages of cyclonic disturbances, when destructive local wind storms develop, some into tornadoes, with winds of 100 miles per hour or more.

3.3 Physiography

All of FTMC except for the easternmost portion of the Choccolocco Corridor lies within the Valley and Ridge Physiographic Province. The easternmost portion of the Choccolocco Corridor lies within the Piedmont Physiographic Province. Local relief on the Main Post of FTMC is in excess of 1,320 feet. The lower elevations (700 feet above mean sea level [msl]) occur along Cane Creek, near Baltzell Gate Road, while the maximum elevations (2,063 feet above msl) occur on Choccolocco Mountain, with the steep easterly slopes grading abruptly into Choccolocco Valley. The western slopes are more continuous, with the southern extension near the western reservation boundary maintaining elevations up to 900 feet above msl. The northern extension decreases in elevation in the vicinity of Reilly Airfield. The central portion of FTMC is characterized by flat to gently sloping land (SAIC, 2000).

The BBGR Ranges are comprised of either large, flat, and open areas (Ranges 18, 25, and 26) or moderately to densely wooded and sometimes hilly (Ranges 20, 23, 25 East, 28, and the Ranges South of Range 25). Except for Range 18, the BBGR Ranges are bounded geographically by St. Clair Road to the north, the north-south ridgeline to the east, and Iron Mountain Road to the west. Elevations range from approximately 795 feet amsl (at Range 18) to 1,170 feet amsl (at Range 20). The ground surface slopes to the north and west towards Ingram Creek around Baby Bains Gap Road and to the north and east towards South Branch of Cane Creek near Range 18.

The following sections discuss the physical characteristics and settings of the BBGR Ranges that includes the sensitive environments, soil types, geology, surface water hydrology, and hydrogeology.

3.4 Sensitive Environments

3.4.1 Wetlands

A study conducted by Foster Wheeler in 1996 estimated over 3,000 acres of wetlands within FTMC. Major wetland communities were originally characterized and mapped in 1984. However, regulatory criteria for identifying wetlands have significantly changed since this original study was performed. Consequently, the USACE performed a supplementary study in 1992 to identify larger wetland complexes (Reisz Engineering [Reisz], 1998). The following are recognized wetland communities located within FTMC (Reisz, 1998): Bottomland Hardwoods, Depressions, Mixed Shrub Communities, Shrub Depression, and Herbaceous Wetlands.

Wetland habitats at FTMC are generally located in various topographical depressions, near stream seepage, and in valleys along creek floodplains (Weston, 1990; SAIC 1993). The indicator plant species that assist in defining a wetland include water oaks, sweet gum, bulrush, needlerush, and cattail. The Main Post, Pelham Range, and Choccolocco Corridor have an abundance of wetlands representing important habitats for a wide variety of plants and animals. The major wetland communities found on the Main Post are the Marcheta Hill Orchard Seep, Cane Creek Seep, South Branch of Cane Creek, and approximately 200 acres that comprise the tributary to Victoria Creek near Reilly Airfield (Garland, 1996; USACE, 1992). The major Pelham Range wetland communities occur along the banks of Cane Creek, Willett Spring, and Cabin Creek Spring (Alabama Department of Conservation and Natural Resources, 1994a and b). Additionally, wetland habitat potentially exists around the installation's lakes, namely, Reilly Lake, Lake Conteras, Yahou Lake, and Lake Willett, and along the nearly 10 miles of creeks, namely, Cane Creek and Cave Creek (Weston, 1990).

The only recognized wetland/seep habitat present in the vicinity of the BBGR Ranges is limited to the area east of Range 20. This area is known as the Marcheta Hill Orchid Seep Special Interest Natural Area (SINA). The spring seepage to the west of Marcheta Hill constitutes one of the more important SINAs on 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 approximately 250 feet east of Range 20. 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 National 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. No other wetlands have been identified in the vicinity of the BBGR Ranges.

3.4.2 Sensitive Habitats

FTMC operated under the guidelines of the Endangered Species Act of 1973, the regulations of the U.S. Fish and Wildlife Service (USFWS), Army Regulation 200-3, and the Endangered Species Management Plan (ESMP) (Garland, 1996). The overall objectives of the ESMP are to sustain the existing habitat that supports populations of species identified in the ESMP and to promote the augmentation of these species into unoccupied land that has similar habitats.

The ESMP identifies 11 SINAs on the Main Post. SINAs are locations where the habitat fosters one or more rare, threatened, or endangered species. Because these species are sensitive to environmental degradation, SINAs require management practices that promote the continued well being of these ecosystems. According to the ESMP, the 11 SINAs located on the Main Post include:

- Mountain Longleaf Community Complex
- Cave Creek Seep
- Moorman Hill Mountain Juniper
- Frederick Hill Aster Site
- Baby Bains Gap Seep
- Marcheta Hill Crow-Poison Seep
- Marcheta Hill Orchid Seep
- South Branch of Cane Creek Seep
- Stanley Hill Chestnut Oak Forest
- Reynolds Hill Turkey Oak
- Davis Hill Honeysuckle.

As stated in the previous section, the Marcheta Hill Orchid Seep SINA is located approximately 250 feet east of Range 20 (Figure 3-1). The Marcheta Hill Orchid Seep SINA is "upstream" of all of the BBGR Ranges. Additionally, there is a 100 to 250 feet high ridge that separates the closest BBGR range (Range 20) from the Marcheta Hill Orchid Seep SINA. Because there is a topographic divide between the BBGR Ranges and Marcheta Hill Orchid Seep SINA and this

SINA is upstream of all of the BBGR Ranges, it is highly unlikely that soil, surface water, or sediment contaminants from the BBGR Ranges could impact this SINA.

3.4.3 Threatened and Endangered Species

Rare species deserving unofficial protection and management measures in the State of Alabama are inventoried and ranked by the Alabama Natural Heritage Program. The sensitivity of these rare species to environmental degradation is used to gauge the well-being of the habitat as a whole. Four species listed as threatened or endangered by the U.S. Fish and Wildlife Service (USFWS) have been recorded at FTMC. These threatened and endangered species are as follows:

- Gray bat (*Myotis grisescens*)
- Blue shiner (*Cyprinella caerules*)
- Mohr's Barbara buttons (*Marshallia mohrii*)
- Tennessee yellow-eyed grass (*Xyris tennesseensis*).

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

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

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).

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 South Branch of Cane Creek or Ingram Creek in the vicinity of the BBGR 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 SINA located directly east of Range 20; 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 within 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).

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

3.5 Soils

The soil associations found at FTMC (U.S. Department of Agriculture [USDA], 1961), include:

- ***Anniston-Allen, Decatur-Cumberland.*** Alluvium, resulting from weathering of older residual soils developed from sandstone, shale, and quartzite; deep, well-drained, level to moderately steep soils in valleys underlain by limestone and shale. Subsoil is dark red sandy clay loam. Cumberland and Decatur soils are dark reddish brown gravelly loam developed from weathered limestone.

- ***Clarksville-Fullerton.*** Well-drained to moderately well-drained stony or cherty soils developed in the residuum of cherty limestone. This association is limited to Pelham Range. The soils are generally dark brown to dark gray-brown silt loam.

- ***Rarden-Montevallo-Lehew.*** Moderately deep or shallow soils on ridgetops and steep slopes and in local alluvium in draws. Soils are developed from the residuum of shale and fine-grained, micaceous sandstone; reddish brown to dark gray-brown to yellow-brown silt loam, clay, or silty clay.

- ***Stony Rough Land.*** Shallow, steep, and stony soils formed from the weathering of sandstone, limestone, and Talladega Slate. Infiltration is slow; the soils contain many boulders and fragments with clayey residuum. This association underlies a large portion of the Main Post at FTMC.

3.5.1 Site-Specific Soils

Seven soil types are mapped at the BBGR Ranges, as shown on Figure 3-2. The Anniston and Allen gravelly loam and the Stony Rough Land sandstone underlie the majority of the area of investigation, with the Philo and Stendal fine sandy loam, Montevallo shaly silty clay loam, the Anniston and Allen stony loam, the Anniston gravelly clay loam, and the Stony Rough Land limestone present within the remainder of the area of investigation (USDA, 1961).

The Anniston and Allen gravelly loam and Anniston and Allen stony 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 3 and 8 inches in diameter (USDA, 1961).

The Philo and Stendal fine sandy loam is found in the southwestern portion of Range 25. This soil is found in alluvium and is mainly derived from sandstone and shale, but some originates from limestone. The surface soil ranges from very dark grayish brown to dark brown. The subsoil consists of a slightly mottled, light to dark brown, fine sandy loam (USDA, 1961).

The Montevallo shaly silty clay loam is mapped along the northwestern boundary of Range 18. This soil develops in the residuum of interbedded shale and limestone. The surface soil ranges from very dark grayish brown and very dark brown to brown. The subsoil is yellowish-brown shaly silt loam. Fragments of shale are commonly found throughout the soil (USDA, 1961).

The Anniston gravelly clay loam is found only in the extreme southeastern corner of Range 18. This soil consists of areas that were formerly Anniston gravelly loam or Allen gravelly loam that has lost practically all of their original surface soil through erosion. The soil is a reddish-brown gravelly clay loam. Infiltration is moderately low, and the capacity for supplying available moisture is low (USDA, 1961).

The Stony Rough Land sandstone is found in the southern portion of Range 18, eastern and northern portions of Range 20, eastern portion of Range 25, and northern portion of Range 25

East. This land type is found in rugged areas with steep relief where outcrops of sandstone and quartzite bedrock are common. Runoff is high, infiltration is slow, and the capacity for available moisture is low. The soil material consists of only a thin veneer of loose rock fragments and scattered patches of sandy soil. It also includes rock escarpments on higher sections of Choccolocco and Coldwater Mountains where quartzite of the Weisner formation is common (USDA, 1961).

The Stony Rough Land limestone is located only in the southeastern section of Range 18. This miscellaneous land type is found at the base of slopes or along their lower edge. The land type is mostly boulders, fragments, and outcrops of limestone on the surface. In isolated areas, a clayey residuum fills the crevasses and spaces between rocks (USDA, 1961).

3.6 Geology

The regional geology in the vicinity of FTMC and site-specific geology at BBGR Ranges are discussed in the following sections.

3.6.1 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 the 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, and 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 Jacksonville 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). In Calhoun County, the Parkwood Formation 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 Parkwood Formation and 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 being 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. It 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 (approximately 5 to 20 miles wide) northeast-to-southwest-trending linear zone of complex structure (approximately 90 miles in length) 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).

3.6.2 Site-Specific Geology

Most of the BBGR Ranges are underlain by the Ordovician Little Oak and Newala Limestones Undifferentiated or the Mississippian/Ordovician Floyd and Athens Shale Undifferentiated, which are exposed in the FTMC geologic window of the Jacksonville thrust sheet. However, the western portion of Range 18, almost all of Range 20, the northern and eastern sections of Range 25, and the southeastern corner of Range 26 are underlain by the Cambrian-age Shady Dolomite and/or the Chilhowee Group that comprise the Jacksonville thrust sheet (Figure 3-3).

A series of asymmetric anticlinal folds strike northeast-southwest and plunge to the northeast within the area of investigation. One anticlinal fold partially crosses the western boundary of Range 25 and the second (southernmost) anticlinal fold crosses the southern portion of Range 23. A plunging synclinal fold parallels the two anticlinal folds and crosses the northwestern section of the Ranges South of Range 25. Two additional plunging anticlinal folds are located north of the area of investigation (Figure 3-3).

Besides the Jacksonville Fault, which frames the FTMC geologic window, two inferred thrust faults strike northeast-southwest in the vicinity of the BBGR Ranges. One thrust fault crosses the central portion of Range 20 and the second bisects the geologic window, traversing the northern portion of Range 18 (Figure 3-3). The second thrust fault approximately parallels the folds and adjoins the Jacksonville Fault north of the Range 25.

Two geologic cross sections (Figures 3-4 and 3-5) were constructed from hollow-stem auger, air rotary drilling, and bedrock coring data collected during the BBGR Ranges investigation. The geologic cross section locations are shown on Figure 3-3. The geologic data collected show alluvium and colluvium overlying the BBGR Ranges. The alluvial and colluvial materials generally

consisted of light brown to brown to reddish brown to yellowish orange cobbley, gravelly, sandy, silty clay. The gravel found within these soils generally consisted of rounded to subangular pieces of quartzite and sandstone.

Residuum, generally consisting of light brown to yellowish-orange silty clay or highly weathered limestone and shale, was typically present beneath the alluvium and colluvium. The residuum underlying the majority of the BBGR Ranges, except most of Range 20 and the eastern section of Range 25 and Range 26, was derived from the undifferentiated Little Oak and Newala Limestone and the undifferentiated Floyd and Athens Shale. This residuum was comprised primarily of yellowish-orange to light brown to brown silty clay, which graded to weathered shale or limestone.

Competent bedrock was penetrated at four monitoring well locations (HR-74Q-MW04, HR-79Q-MW03, HR-84Q-MW03, and HR-118Q-MW02). The bedrock encountered at monitoring wells HR-74Q-MW04, HR-79Q-MW03, and HR-118Q-MW02 was identified as limestone. The limestone was described as moderately hard to hard, slightly to moderately weathered, crystalline to very fine grained, moderately to highly fractured, dark to medium-dark gray limestone consistent with the Little Oak and Newala Limestone undifferentiated. The bedrock encountered at monitoring well HR-84Q-MW03 was light brown to medium gray quartzite and sandstone overlying moderately weathered, soft, highly to intensely fractured, dark gray to grayish black shale. Irregular, calcite-filled fractures and broken zones were noted on the shale core samples. This sequence is consistent with older (Cambrian age) members of the Undifferentiated Chilhowee Group thrust over younger (Mississippian/Ordovician age) Floyd and Athens Shale Undifferentiated (Figure 3-3).

3.7 Surface Water Hydrology

The regional surface water hydrology in the vicinity of FTMC and site-specific surface water hydrology at the BBGR Ranges are discussed in the following sections.

3.7.1 Regional Surface Water Hydrology

Portions of three drainage basins (Cane Creek, Choccolocco Creek, and Tallasseehatchee Creek) are found within the Main Post of FTMC. All three of these drainage basins eventually empty into the Coosa River, approximately 16 miles west of the Main Post. Figure 3-6 shows 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 and the eastern and southernmost portions of the Main Post. 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 Tallasseehatchee Creek drainage basins. 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 Creek, Remount Creek, South Branch of Cane Creek, and Ingram Creek. Although Cave Creek is a tributary to Cane Creek, it 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 and exit 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 Tallasseehatchee Creek drainage basin drains the northernmost portion of the Main Post. Most of the surface runoff from this portion of the Main Post collects in unnamed tributaries to Reilly Lake. Reilly Lake then empties into Dothard Creek, a tributary to the Tallasseehatchee, and exits the Main Post northwest of Reilly Airfield. The Tallasseehatchee drainage basin covers approximately 2.5 square miles of the Main Post.

Most surface water bodies are fed, at least in part, by fresh water 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 (SAIC, 2000).

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

3.7.2 Site-Specific Surface Water Hydrology

The BBGR Ranges are located within the Cane Creek Drainage Basin (Figure 3-6). The BBGR Ranges are drained by either Ingram Creek or South Branch of Cane Creek. Ingram Creek and its headwater tributaries drain Ranges 20, 23, 25, 26 and Ranges South of Range 25. Tributaries of South Branch of Cane Creek drain Range 18. Almost all of the tributaries to these creeks are intermittent streams which drain seasonal runoff from winter and spring rain events and generally dry during the summer and fall. Surface water in Ingram Creek, South Branch of Cane Creek, and their respective tributaries have a net flow direction to the northwest. Ingram Creek and South Branch of Cane Creek eventually empty into Cane Creek within the main cantonment of FTMC.

3.8 Hydrogeology

The regional hydrogeology in the vicinity of FTMC and site-specific hydrogeology of the BBGR Ranges are discussed in the following sections

3.8.1 Regional Hydrogeology

The hydrogeology of Calhoun County has been investigated by the Geologic Survey of Alabama (Moser and DeJarnette, 1992) and the U.S. Geological Survey in cooperation with the General Services Administration (Warman and Causey, 1962) and the Alabama Department of Environmental Management (Planert and Pritchette, 1989). Groundwater near FTMC occurs in residuum derived from bedrock decomposition along fault zones within fractured bedrock and from the development of karst frameworks. Groundwater flow may be estimated to be toward major surface water features. 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 sandstone, 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 enhances 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 one of the largest springs 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 buys 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. The residuum may yield adequate groundwater for domestic and livestock needs but may go dry during prolonged dry weather. 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). Neither the Knox-Shady aquifer nor the Tuscumbia-Fort Payne aquifer is mapped within the immediate vicinity of the BBGR Ranges. Furthermore, there are no potable water supply wells within a 1-mile radius of Parcel the BBGR Ranges.

Regional groundwater flow in the bedrock was approximated for the FTMC vicinity by the U.S. Geological Survey (Scott et al., 1987). Regional groundwater elevation ranged from 800 feet amsl on the main Base to about 600 feet amsl to the west on Pelham Range, based on water depths in wells completed across multiple formations. Groundwater elevation contours suggest that regional groundwater flow is from the Main Post northwest toward the city of Weaver; however, there is not enough groundwater data to support this interpretation. Scott et al. (1987) concluded that the groundwater surface broadly coincides with the surface topography and that the regional aquifers are hydraulically connected. Groundwater flow on a local scale may be more complex and may be affected by geologic structures such as shallow thrust faults, rock fracture systems, and karst development in soluble formations.

3.8.2 Site-Specific Hydrogeology

Static groundwater levels were measured in monitoring wells at the BBGR Ranges on December 17, 2002 (Table 2-4). A groundwater elevation map (Figure 3-7) was constructed for the residuum at the BBGR Ranges. Because of the limited number and wide spacing of the bedrock monitoring

wells installed at the site, a bedrock groundwater elevation map was not prepared. As shown on Figure 3-7, horizontal groundwater flow in the residuum follows the local topography and has a westerly and northwesterly flow direction across all of the ranges except Range 18. At Range 18, the groundwater flow direction is to the northeast. The horizontal hydraulic gradient within the residuum reflects the topography being relatively flat across Ranges 18, 23, and the Ranges South of Range 25 to relatively steep across Ranges 25 and 26. Calculated horizontal gradients ranged from 0.018 to 0.043 feet per foot (ft/ft), with an arithmetic mean of 0.031 ft/ft (Table 3-1).

Vertical hydraulic gradients for three well clusters are presented in Table 3-2. The gradients were calculated from hydraulic head differences for groundwater elevations measured in December 2002. The well clusters shown in Table 3-2 compare gradients between the residuum and bedrock wells. Relatively weak upward gradients (-0.007 to -0.061 ft/ft) were observed in all three well clusters. The upward hydraulic gradient within these wells supports general confined aquifer conditions.

Slug tests were not conducted during the RI. Therefore, hydraulic conductivity values could not be calculated for rising or falling head tests.

4.0 Nature and Extent of Contamination

This chapter discusses the nature and extent of contamination at the BBGR Ranges, including the results of the sampling and analysis activities conducted at the site. In addition, information is provided on potential sources of contamination.

4.1 Summary of Analytical Results

The results of the chemical analysis of samples collected at the BBGR Ranges indicate that metals, VOCs, SVOCs, nitroaromatic explosive compounds, pesticides, herbicides, and PCBs were detected in site media. To evaluate the nature and extent of the detected constituents, the analytical results were compared to the human health SSSLs and ESVs for FTMC. Specifically, residential SSSLs or recreational site user SSSLs (for surface water/sediment) were used. It should be noted that the assumptions for residential and recreational site user exposure to surface water/sediment are identical.

The SSSLs and ESVs were developed by Shaw as part of the human health and ecological risk evaluations associated with investigations performed under the BRAC Environmental Restoration Program at FTMC. The SSSLs, ESVs, and polynuclear aromatic hydrocarbon (PAH) background screening values are presented in the *Final Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000b). The PAH background screening values were developed by Shaw at the direction of the BRAC Cleanup Team to address the occurrence of PAH compounds in surface soils as a result of anthropogenic activities at FTMC. Background metals screening values are presented in the *Final Background Metals Survey Report, Fort McClellan, Alabama* (SAIC, 1998). Site metals data were also evaluated using statistical and geochemical methods to determine if the metals detected in site media are naturally occurring (Appendix I). Complete human health and ecological risk assessments are provided in Chapters 6.0 and 7.0, respectively, of this report.

The following sections and Tables 4-1 through 4-5 summarize the results of the comparison of detected constituents to the background screening values, SSSLs, and ESVs. Complete analytical results are presented in Appendix G. Lead from bullets constitutes the largest single source of environmental contamination at the BBGR Ranges; therefore, a separate discussion on lead contamination is presented in this section for each site medium.

4.1.1 Surface and Depositional Soil Analytical Results

A total of 339 surface soil and depositional soil samples were collected for chemical analysis at the BBGR Ranges. Surface and depositional soil samples were collected from the uppermost foot of soil at the locations shown on Figures 2-1 through 2-9. The analytical results were compared to residential SSSLs, ESVs, and background screening values, as presented in Table 4-1.

Lead. Lead was detected in all 323 surface and depositional soil samples analyzed for lead. Lead concentrations ranged from 7.13 mg/kg (at HR-83Q-GP08) to 23,300 mg/kg (at HR-74Q-GP42). Ninety-three lead results exceeded the SSSL (400 mg/kg) and background (40 mg/kg); 191 lead results exceeded the ESV (50 mg/kg) and background. The integrated statistical and geochemical evaluation (Appendix I) concluded that lead concentrations in 149 site surface soil samples were considered anomalous and indicative of site-related contamination.

Figure 4-1 is a computer-generated isocontour map showing lead concentrations in surface soils using both laboratory data and XRF data. The contour intervals shown on the figure are the ESV (50 mg/kg), 100 mg/kg, the residential SSSL (400 mg/kg), 1,000 mg/kg, and 10,000 mg/kg.

At Range 18, the lead contamination pattern was widespread with concentrations exceeding the ESV in the target line areas and in the hillside impact zone farther to the south (Figure 4-1).

Range 20 soils showed elevated lead concentrations in an area measuring approximately 750 by 400 feet around the M60 machine gun impact zone in the southeastern portion of the range (Figure 4-1).

At Range 23, elevated concentrations of lead are generally found in the pop-up target mounds, where nearly 80 percent of the samples that exceeded the ESV were collected (Figure 4-1). It should be noted that the contours shown on Figure 4-1 reflect the biased sampling approach conducted in the area of the pop-up targets and, thus, do not fully depict the lead contamination at this site. It is reasonable to assume that similar concentrations of lead are present at most or all of the target mounds, which cover the majority of the area of investigation. A second, smaller area of lead contamination is also present in the northeastern portion of the area of investigation, just east of the road that forms the eastern boundary of the range. This area is approximately 500 by 375 feet at its longest and widest extents.

At Range 25, lead-contaminated soils were mostly limited to the hillside impact zone in the northeastern portion of the range (Figure 4-1). This area is approximately 1,250 by 1,250 feet at its longest and widest extents.

At Range 26, two main areas of lead contamination are present, one in the area of the pop-up target locations located mid-range and the other in the area of vehicles used as targets along the hillside impact zone farther downrange (Figure 4-1). The area of contamination around the pop-up targets is approximately 700 by 400 feet; the other area measures approximately 375 by 175 feet.

The Ranges South of Range 25 area of investigation showed elevated lead concentrations throughout most of the 6-acre area of investigation (Figure 4-1).

No lead contamination was present in samples collected at Range 25 East. However, a small area of elevated lead concentrations is shown on Figure 4-1 in the southeastern portion of Range 25 East because this range partially overlaps Range 26.

Metals (Other than Lead). One hundred and forty-four surface and depositional soil samples were analyzed for TAL metals other than lead. A total of 21 metals were detected in the samples. The concentrations of ten metals exceeded SSSLs and background including:

- Aluminum (16,700 to 34,300 mg/kg) exceeded its SSSL (7,803 mg/kg) and background (16,306 mg/kg) in 40 samples.
- Antimony (4.03 to 86.4 mg/kg) exceeded its SSSL (3.11 mg/kg) and background (1.99 mg/kg) in 36 samples.
- Arsenic (14 to 36.9 mg/kg) exceeded its SSSL (0.43 mg/kg) and background (13.7 mg/kg) in 15 samples.
- Barium (570 to 1,440 mg/kg) exceeded its SSSL (547 mg/kg) and background (124 mg/kg) in three samples.
- Chromium (41.1 to 101 mg/kg) exceeded its SSSL (23.2 mg/kg) and background (37 mg/kg) in nine samples.
- Copper (321 to 15,000 mg/kg) exceeded its SSSL (313 mg/kg) and background (12.7 mg/kg) in eight samples.
- Iron (34,900 to 72,100 mg/kg) exceeded its SSSL (2,345 mg/kg) and background (34,154 mg/kg) in 23 samples.

- Manganese (1,660 to 6,120 mg/kg) exceeded its SSSL (363 mg/kg) and background (1,579 mg/kg) in 20 samples.

- Thallium (3.99 mg/kg) exceeded its SSSL (0.51 mg/kg) and background (3.43 mg/kg) at sample location HR-226Q-DEP02.

- Vanadium (69 to 131 mg/kg) exceeded its SSSL (53.1 mg/kg) and background (58.8 mg/kg) in five samples.

Seventeen metals were detected at concentrations exceeding ESVs and background including:

- Aluminum (16,700 to 34,300 mg/kg) exceeded its ESV (50 mg/kg) and background (16,306 mg/kg) in 40 samples.

- Antimony (4.03 to 86.4 mg/kg) exceeded its ESV (3.5 mg/kg) and background (1.99 mg/kg) in 36 samples.

- Barium (167 to 1,440 mg/kg) exceeded its ESV (165 mg/kg) and background (124 mg/kg) in 29 samples.

- Beryllium (1.1 to 5.95) exceeded its ESV (1.1 mg/kg) and background (0.8 mg/kg) in 32 samples.

- Chromium (41.1 to 101 mg/kg) exceeded its ESV (0.4 mg/kg) and background (37 mg/kg) in nine samples.

- Cobalt (20.8 to 140 mg/kg) exceeded its ESV (20 mg/kg) and background (15.2 mg/kg) in 21 samples.

- Copper (41.1 to 15,000 mg/kg) exceeded its ESV (40 mg/kg) and background (12.7 mg/kg) in 48 samples.

- Iron (34,900 to 72,100 mg/kg) exceeded its ESV (200 mg/kg) and background (34,154 mg/kg) in 23 samples.

- Manganese (1,660 to 6,120 mg/kg) exceeded its ESV (100 mg/kg) and background (1,579 mg/kg) in 20 samples.

- Mercury (0.14 mg/kg) exceeded its ESV (0.1 mg/kg) and background (0.08 mg/kg) at sample location HR-74Q-GP27.

- Nickel (33.8 and 65.9 mg/kg) exceeded its ESV (30 mg/kg) and background (10.3 mg/kg) at sample locations HR-74Q-GP27 and HR-74Q-GP30.

- Selenium (0.84 to 2.27 mg/kg) exceeded its ESV (0.81 mg/kg) and background (0.48 mg/kg) in 28 samples.

- Silver (2.17 to 4.72 mg/kg) exceeded its ESV (2 mg/kg) and background (0.36 mg/kg) in 10 samples.

- Thallium (3.99 mg/kg) exceeded its ESV (1 mg/kg) and background (3.43 mg/kg) at sample location HR-226Q-DEP02.

- Vanadium (69 to 131 mg/kg) exceeded its ESV (2 mg/kg) and background (58.8 mg/kg) in five samples.

- Zinc (50.3 to 1,250 mg/kg) exceeded its ESV (50 mg/kg) and background (40.6 mg/kg) in 32 samples.

The integrated statistical and geochemical evaluation (Appendix I) concluded that all detected concentrations of aluminum, barium, calcium, chromium, cobalt, iron, magnesium, manganese, potassium, selenium, silver and vanadium in the surface soil samples are naturally occurring. Antimony (34 samples), arsenic (HR-74Q-GP21), beryllium (HR-74Q-GP27), copper (48 samples), nickel (HR-74Q-GP27), and zinc (3 samples) were judged to be present at anomalously high concentrations indicating potential contamination.

In addition to lead, three other metals (antimony, copper, and zinc) have been identified as components of bullets; therefore, the elevated concentrations of these metals are frequently co-located in areas with high lead concentrations (Interstate Technology and Regulatory Council, 2003). For example, in the 36 samples where antimony exceeded its ESV and background, 31 locations (86 percent) also contained lead at concentrations exceeding its ESV and background. For copper, 46 of 48 results (96 percent) were co-located with samples that contained elevated lead. For zinc, 27 of 33 results (82 percent) were co-located with elevated lead concentrations.

Figure 4-2 shows antimony, copper, and zinc results exceeding ESVs and background in surface soil.

Volatile Organic Compounds. A total of 11 VOCs were detected in the 54 surface and depositional soil samples analyzed for VOCs. Several of the methylene chloride, acetone, and trichlorofluoromethane results were flagged with a "B" data qualifier, signifying that these compounds were also detected in an associated laboratory or field blank sample. VOC concentrations in the surface and depositional soil samples ranged from 0.00098 to 0.68 mg/kg. All results were less than SSSLs and ESVs except for one detection of trichloroethene (0.0018 mg/kg) at Range 23 sample location HR-79Q-GP17. This result slightly exceeded the ESV (0.001 mg/kg) but the result was "J" flagged indicating the concentration was estimated.

Semivolatile Organic Compounds. A total of 18 SVOCs, including 13 PAH compounds, were detected in the 54 surface and depositional samples analyzed for SVOCs. The concentrations of five PAHs and one non-PAH compound exceeded SSSLs in a limited number of samples:

- Benzo(a)anthracene (3.2 mg/kg) exceeded its SSSL (0.085 mg/kg) in one sample from Range 26 (location HR-84Q-GP06).

- Benzo(a)pyrene (0.1 to 2.5 mg/kg) exceeded its SSSL (0.085 mg/kg) in one sample from Range 18 (HR-74Q-GP02) and three samples from Range 26 (HR-84Q-GP06, HR-84Q-GP09, and HR-84Q-SB02).

- Benzo(b)fluoranthene (1.1 and 5.3 mg/kg) exceeded its SSSL (0.085 mg/kg) in one sample from Range 18 (HR-74Q-GP02) and one sample from Range 26 (HR-84Q-GP06).

- Dibenz(a,h)anthracene (0.15 and 0.37 mg/kg) exceeded its SSSL (0.086 mg/kg) in one sample from Range 18 (HR-74Q-GP02) and one sample from Range 26 (HR-84Q-GP06).

- Indeno(1,2,3-cd)pyrene (1.3 mg/kg) exceeded its SSSL (0.085 mg/kg) in one sample from Range 26 (HR-84Q-GP06).

- 2,4-Dinitrotoluene (1.9 mg/kg) exceeded its SSSL (0.93 mg/kg) in one sample from Range 23 (HR-79Q-GP04).

The concentrations of eight SVOCs, including five PAH compounds and three non-PAHs, exceeded ESVs in a limited number of samples:

- Anthracene (0.21 and 0.52 mg/kg) exceeded its ESV (0.1 mg/kg)) in one sample from Range 18 (HR-74Q-GP02) and one sample from Range 26 (HR-84Q-GP06).

- Benzo(a)pyrene (0.1 to 2.5 mg/kg) exceeded its ESV (0.1 mg/kg) in one sample from Range 18 (HR-74Q-GP02) and three samples from Range 26 (HR-84Q-GP06, HR-84Q-GP09, and HR-84Q-SB02).

- Fluoranthene (0.11 to 5.8 mg/kg) exceeded its ESV (0.1 mg/kg) in two samples from Range 18 (HR-74Q-GP02 and HR-74Q-GP44) and four samples from Range 26 (HR-84Q-GP06, HR-84Q-GP09, HR-84Q-SB02, and HR-84Q-SB04).

- Phenanthrene (0.76 mg/kg) exceeded its ESV (0.1 mg/kg) in one sample from Range 26 (HR-84Q-GP06).

- Pyrene (0.17 to 5 mg/kg) exceeded its ESV (0.1 mg/kg) in two samples from Range 18 (HR-74Q-GP02 and HR-74Q-GP44) and three samples from Range 26 (HR-84Q-GP06, HR-84Q-GP09, and HR-84Q-SB02).

- 2,4-Dinitrotoluene (1.9 mg/kg) exceeded its ESV (1.3 mg/kg) in one sample from Range 23 (HR-79Q-GP04).

- 2,6-Dinitrotoluene (0.086 mg/kg) exceeded its SSSL (0.033 mg/kg) in one sample from Range 23 (HR-79Q-GP04).

- Pentachlorophenol (0.081 mg/kg) exceeded its SSSL (0.002 mg/kg) in one sample from Range 26 (HR-84Q-DEP03).

Range 26 sample location HR-84Q-GP06 contained several PAHs that exceeded SSSLs/ESVs and PAH background values. Range 26 (Parcel 84Q) was constructed with creosote-treated wood railroad cross ties used to form firing points and they could be a possible source of the PAH contamination at this range.

Chlorinated Pesticides. Fifty-four surface and depositional soil samples were analyzed for chlorinated pesticides. A total of 18 compounds were detected in the samples. The majority of pesticide results were flagged with a "J" data qualifier, indicating that the concentrations were estimated. The pesticide concentrations ranged from 0.00033 to 0.043 mg/kg and all results were below SSSLs.

The concentrations of ten compounds exceeded ESVs:

- 4,4'-DDE (0.0033 mg/kg) at Range 23 sample location HR-79Q-DEP07 (ESV = 0.0025 mg/kg).

- 4,4'-DDT (0.0028 to 0.0057 mg/kg) at sample locations HR-74Q-GP02 (Range 18), HR-79Q-DEP07, (Range 23), HR-83Q-GP01 (Range 25), and HR-83Q-GP05 (Range 25) (ESV = 0.0025 mg/kg).

- Aldrin (0.013 to 0.0038 mg/kg) at Range 23 sample locations HR-79Q-GP04 and HR-79Q-GP05 (ESV = 0.0025 mg/kg).

- Beta-BHC (0.0011 to 0.0066 mg/kg) at sample locations HR-74Q-GP42 (Range 18), HR-79Q-DEP04 (Range 23), HR-79Q-GP17 (Range 23), HR-79Q-SB08 (Range 23), and HR-84Q-GP09 (Range 26) (ESV = 0.001 mg/kg).

- Dieldrin (0.00051 to 0.0039 mg/kg) at Range 23 sample locations HR-79Q-GP04, HR-79Q-SB03, HR-79Q-SB10, and Range 26 sample location HR-84Q-GP06 (ESV = 0.0005 mg/kg).

- Endosulfan sulfate (0.043 mg/kg) at Range 26 sample location HR-84Q-GP09 (ESV = 0.036 mg/kg).

- Endrin (0.0036 mg/kg) at Range 23 sample location HR-79Q-GP04 (ESV = 0.001 mg/kg).

- Endrin ketone (0.02 mg/kg) at Range 18 sample location HR-74Q-GP02 (ESV = 0.01 mg/kg).

- Gamma-BHC (0.021 and 0.0012 mg/kg) at Range 23 sample locations HR-79Q-GP20 and HR-79Q-SB10 (ESV = 0.00005 mg/kg).

- Methoxychlor (0.021 mg/kg) at Range 23 sample location HR-79Q-GP04 (ESV = 0.02 mg/kg).

The pesticide results that exceeded ESVs were sporadically distributed at the BBGR Ranges and showed no discernable pattern of contamination.

Organophosphorus Pesticides. Fifty-four surface and depositional soil samples were analyzed for organophosphorus pesticides. A total of six compounds were detected at only one sample location (HR-83Q-GP05). All of the results were flagged with a "J" data qualifier, indicating that the concentrations were estimated. The pesticide concentrations ranged from 0.0022 to 0.0052 mg/kg and all results were below SSSLs and ESVs.

Chlorinated Herbicides. Fifty-four surface and depositional soil samples were analyzed for chlorinated herbicides. A total of 18 compounds were detected in 10 of the samples. All of the herbicide results were flagged with a "J" data qualifier, indicating that the concentrations were estimated. The herbicide concentrations ranged from 0.0054 to 2.9 mg/kg and all results were below SSSLs.

MCPP concentrations (0.74 to 2.9 mg/kg) exceeded its ESV (0.1 mg/kg) in six samples: Range 18 sample locations HR-74Q-GP42 and HR-74Q-GP44; Range 23 sample locations HR-79Q-DEP07, HR-79Q-GP09, and HR-79Q-GP17; and Range 26 sample location HR-84Q-DEP02.

Explosives. One hundred and thirty-six surface and depositional soil samples were analyzed for explosive compounds. A total of two compounds were detected in one sample at Range 18 and in two samples at Range 23: 2,4-dinitrotoluene (at sample locations HR-74Q-GP02, HR-79Q-GP04, and HR-79Q-GP05) and 2,6-dinitrotoluene (HR-79Q-GP04).

The 2,4-dinitrotoluene results (1.1, 1.6, and 10 mg/kg) exceeded its SSSL (0.93 mg/kg) in all three samples. Two 2,4-dinitrotoluene results (1.6, and 10 mg/kg) and the 2,6-dinitrotoluene result (0.41 mg/kg) exceeded ESVs (1.28 and 0.033 mg/kg, respectively).

Polychlorinated Biphenyls. Forty-two surface and depositional soil samples were analyzed for PCBs. One Range 23 sample location (HR-79Q-SB10) contained an estimated concentration ("J" flagged) of PCB Aroclor 1260. The Aroclor 1260 concentration (0.022 mg/kg) was below its SSSL but slightly exceeded its ESV (0.02 mg/kg).

4.1.2 Subsurface Soil Analytical Results

A total of 144 subsurface soil samples were collected from 139 soil borings at the BBGR Ranges. Subsurface soil samples were collected at depths greater than 1 foot bgs at the locations shown on Figures 2-1 through 2-9. Analytical results were compared to residential SSSLs and background screening values, as presented in Table 4-2.

Lead. One hundred and twenty-eight subsurface soil samples were analyzed for lead. Lead was detected in each of the samples at concentrations ranging from 2.72 mg/kg (at HR-84Q-MW04) to 2,630 mg/kg at (HR-76Q-GP06). Six lead results exceeded the SSSL of 400 mg/kg:

- Two lead results were in samples from the Range 20 M60 impact zone (locations HR-76Q-GP06 [2 to 3 feet bgs] and HR-76Q-GP02 [2 to 3 feet bgs]).

- The other four lead results were in samples from Range 23: three target mound sample locations (HR-79Q-SB07 [3 to 4 feet bgs]; HR-79Q-SB04 [3 to 4 feet bgs]; and HR-79Q-SB01 [2 to 3 feet bgs]) and one at sample location HR-79Q-MW01 (2 to 4 feet bgs).

Figure 4-3 is a computer-generated isocontour map showing the areal extent of lead above the residential SSSL (400 mg/kg) in subsurface soil.

The integrated statistical and geochemical evaluation (Appendix I) concluded that lead concentrations in 14 subsurface soil samples were anomalous and indicative of site-related contamination. As noted above, however, only six lead results exceeded the residential SSSL of 400 mg/kg.

Metals (Other than Lead). Ninety-one subsurface soil samples were analyzed for TAL metals other than lead. A total of 21 metals were detected in the samples. The concentrations of eight metals exceeded SSSLs (and background) including:

- Aluminum (13,700 to 66,200 mg/kg) exceeded its SSSL (7,803 mg/kg) and background (13,591 mg/kg) in 67 samples.

- Antimony (3.83 to 27.2 mg/kg) exceeded its SSSL (3.11 mg/kg) and background (1.31 mg/kg) in seven samples.

- Arsenic (18.7 to 27.2 mg/kg) exceeded its SSSL (0.43 mg/kg) and background (18.3 mg/kg) in five samples.

- Chromium (41.5 to 92.3 mg/kg) exceeded its SSSL (23.2 mg/kg) and background (38.3 mg/kg) in ten samples.

- Iron (45,400 to 68,600 mg/kg) exceeded its SSSL (2,345 mg/kg) and background (44,817 mg/kg) in 12 samples.

- Manganese (1,470 to 4,690 mg/kg) exceeded its SSSL (363 mg/kg) and background (1,355 mg/kg) in nine samples.

- Thallium (1.46 to 1.76 mg/kg) exceeded its SSSL (0.51 mg/kg) and background (1.4 mg/kg) in three samples.

- Vanadium (65.2 to 109 mg/kg) exceeded its SSSL (53.1 mg/kg) and background (64.9 mg/kg) in seven samples.

The integrated statistical and geochemical evaluation (Appendix I) concluded that all detected concentrations of aluminum, arsenic, barium, calcium, chromium, cobalt, iron, magnesium, manganese, nickel, potassium, selenium, silver and vanadium in the subsurface soil samples are naturally occurring. Antimony (at seven locations), beryllium (HR-226Q-MW02), copper (14 samples), mercury (HR-224Q-MW03), and zinc (HR-74Q-GP30) each have anomalously high concentrations in one or more subsurface soil samples and may be indicative of contamination. As indicated above, however, these metals results were below their respective SSSLs except for antimony in five samples. Figure 4-3 shows the antimony results exceeding the SSSL and background in subsurface soil.

Volatile Organic Compounds. Twenty-six subsurface soil samples were analyzed for VOCs. A total of four VOCs (2-butanone, acetone, methylene chloride, and p-cymene) were detected in the samples. One acetone result and four methylene chloride results were flagged with a "B" data qualifier, signifying that these compounds were also detected in an associated laboratory or field blank. VOC concentrations in subsurface soil ranged from 0.0013 to 0.28 mg/kg and all results were below SSSLs.

Semivolatile Organic Compounds. Twenty-six subsurface soil samples were analyzed for SVOCs. Three SVOCs (2,4-dinitrotoluene, di-n-butyl phthalate, and n-nitrosodiphenylamine) were detected at Range 23 sample location HR-79Q-GP04 only. All results were below SSSLs.

Chlorinated Pesticides. Twenty-six subsurface soil samples were analyzed for chlorinated pesticides. A total of six compounds were detected in five of the samples. The majority of pesticide results were flagged with a "J" data qualifier, indicating that the concentrations were estimated. The pesticide concentrations ranged from 0.00042 to 0.019 mg/kg and all results were below SSSLs.

Organophosphorus Pesticides. Organophosphorus pesticides were not detected in the 26 subsurface soil samples analyzed for organophosphorus pesticides.

Chlorinated Herbicides. Twenty-six subsurface soil samples were analyzed for chlorinated herbicides. One compound (MCPP) was detected at one Range 18 sample location (HR-74Q-MW03) at an estimated concentration (0.53 mg/kg) below its SSSL (7.77 mg/kg).

Explosives. Ninety-five subsurface soil samples were analyzed for explosive compounds. One compound (2,4-dinitrotoluene) was detected at one Range 23 sample location (HR-79Q-GP04) at a concentration (0.44 mg/kg) below its SSSL (0.93 mg/kg).

Polychlorinated Biphenyls. PCBs were not detected in the 16 subsurface soil samples analyzed for PCBs.

4.1.3 Groundwater Analytical Results

Nineteen groundwater samples were collected for chemical analysis at the BBGR Ranges. The well/groundwater sample locations are shown on Figures 2-1 through 2-9. The analytical results were compared to residential SSSLs and metals background screening values, as presented in Table 4-3.

Total Metals (Including Lead). All of the groundwater samples were analyzed for total metals. A total of 15 metals were detected in the samples. The concentrations of six metals (aluminum, arsenic, barium, iron, manganese, and thallium) exceeded SSSLs. Of these, barium (four samples), manganese (three samples), and thallium (one sample) also exceeded their respective background concentrations.

The integrated statistical and geochemical evaluation (Appendix I) concluded that the metals detected in groundwater are naturally occurring, except for calcium in the sample from well HR-118Q-MW01. The one anomalous calcium result may have been caused by chemical interference from the grout used during well installation.

Dissolved Metals (Including Lead). One groundwater sample (location HR-79Q-MW02) was analyzed for dissolved metals. Six metals were detected in the sample: barium, calcium, iron, magnesium, manganese, and sodium. The concentrations of two metals (barium and manganese) exceeded SSSLs. The barium result was below its background value but the manganese result (1.5 milligrams per liter [mg/L]) exceeded its background value (0.58 mg/L).

Volatile Organic Compounds. Five groundwater samples (locations HR-74Q-MW03, HR-79Q-MW01, HR-83Q-MW02, HR-84Q-MW02, and HR-118Q-MW01) were analyzed for VOCs. One VOC (carbon disulfide) was detected in one Range 23 sample (location HR-79Q-MW01) at an estimated concentration (0.00068 mg/L) below its SSSL (0.15 mg/L).

Semivolatile Organic Compounds. SVOCs were not detected in the five groundwater samples (locations HR-74Q-MW03, HR-79Q-MW01, HR-83Q-MW02, HR-84Q-MW02, and HR-118Q-MW01) analyzed for SVOCs.

Chlorinated Pesticides. Five groundwater samples (locations HR-74Q-MW03, HR-79Q-MW01, HR-83Q-MW02, HR-84Q-MW02, and HR-118Q-MW01) were analyzed for chlorinated pesticides. A total of eight compounds were detected in three of the samples. Each of the pesticides was detected in only one sample except for alpha-chlordane (two samples) and beta-BHC (three samples). Pesticide concentrations in groundwater ranged from 0.000011 to 0.00013 mg/L and all results were "J" flagged, indicating that the concentrations were estimated. The concentrations of three compounds exceeded their respective SSSLs in one well each:

- Beta-BHC (0.000091 mg/L) exceeded its SSSL (0.000036 mg/L) in Range 25 well HR-83Q-MW02.

- Gamma-BHC (0.00013 mg/L) exceeded its SSSL (0.00005 mg/L) in Range 18 well HR-74Q-MW03.

- Heptachlor (0.00012 mg/L) exceeded its SSSL (0.000015 mg/L) in Range 25 well HR-83Q-MW02.

Figure 4-4 shows the well locations with pesticide results exceeding SSSLs in groundwater. Based on the historical use of the BBGR Ranges as weapons training ranges, it is likely that the pesticides are isolated, low-level detections.

Organophosphorus Pesticides. Organophosphorus pesticides were not detected in the five groundwater samples (locations HR-74Q-MW03, HR-79Q-MW01, HR-83Q-MW02, HR-84Q-MW02, and HR-118Q-MW01) analyzed for organophosphorus pesticides.

Chlorinated Herbicides. Chlorinated herbicides were not detected in the five groundwater samples (locations HR-74Q-MW03, HR-79Q-MW01, HR-83Q-MW02, HR-84Q-MW02, and HR-118Q-MW01) analyzed for chlorinated herbicides.

Explosives. All 19 groundwater samples were analyzed for explosives. A total of five explosives (2,4-dinitrotoluene, 2,6-dinitrotoluene, 2-nitrotoluene, 3-nitrotoluene, and 4-amino-2,6-dinitrotoluene) were detected in one sample each. Explosives were detected in only three wells (HR-79Q-MW02, HR-83Q-MW01, and HR-83Q-MW04) at Ranges 23 and 25. Three of the five detections were flagged with a "J" data qualifier, signifying that the compounds were positively identified, but the concentrations were estimated. The concentrations of three compounds exceeded their respective SSSLs in one well each:

- 2,4-Dinitrotoluene (0.0011 mg/L) in Range 25 well HR-83Q-MW04 (SSSL = 0.000098 mg/L).
- 2,6-Dinitrotoluene (0.00037 mg/L) in Range 25 well HR-83Q-MW04 (SSSL = 0.000098 mg/L).
- 4-Amino-2,6-dinitrotoluene (0.0004 mg/L) in Range 23 well HR-79Q-MW02 (SSSL = 0.000094 mg/L).

Figure 4-4 shows the spatial distribution of explosive compound concentrations exceeding SSSLs in groundwater. As shown on the figure, the explosive compound results that exceeded SSSLs were restricted to one well at Range 23 and one well at Range 25, located several thousand feet from one another. These compounds were not detected in any other wells. Based on the existing well locations and available data, the explosive compounds in groundwater appear to be isolated, low-level detections.

Polychlorinated Biphenyls. PCBs were not detected in the three groundwater samples analyzed for PCBs.

4.1.4 Surface Water Analytical Results

Twelve surface water samples were collected for chemical analysis at the BBGR Ranges. The surface water sample locations are shown on Figures 2-1 through 2-9. The analytical results were compared to the recreational site user SSSLs, ESVs, and background screening values, as presented in Table 4-4.

Lead. All of the surface water samples were analyzed for lead. Lead was detected in three samples (HR-76Q-SW/SD03 at Range 20, HR-74Q-SW/SD01 at Range 18, and HR-79Q-SW/SD05 at Range 25) at concentrations ranging from 0.0016 mg/L to 0.004 mg/L. All results were below the SSSL. All three results exceeded the ESV but were below background.

Metals (Other than Lead). All of the surface water samples were analyzed for TAL metals other than lead. A total of 10 metals were detected in the samples. Only arsenic (0.0032 mg/L) exceeded its SSSL (0.00073 mg/L) in one Range 20 sample HR-76Q-SW/SD03. The arsenic result also exceeded its background value (0.0022 mg/L). However, the result was "B" flagged indicating that arsenic was also detected in an associated laboratory or field blank sample.

The concentrations of four metals (aluminum, barium, beryllium, and manganese) exceeded ESVs but were below background except for the following:

- Barium (0.092 and 0.093 mg/L) exceeded its ESV (0.0039 mg/L) and background (0.075 mg/L) at sample locations HR-74Q-SW/SD01 (Range 18) and HR-79Q-SW/SD02 (Range 23).

- Beryllium (0.001 mg/L) exceeded its ESV (0.00053 mg/L) and background (0.00039 mg/L) at sample location HR-74Q-SW/SD01 (Range 18). The result was flagged with a "B" data qualifier, indicating that the metal was also detected in an associated laboratory or field blank sample.

The integrated statistical and geochemical evaluation (Appendix I) concluded that all of the metals detected in the site surface water samples are naturally occurring.

Volatile Organic Compounds. Ten surface water samples were analyzed for VOCs. One VOC (methylene chloride) was detected in five of the samples. All of the results were flagged with a "B" data qualifier, indicating that the compound was also detected in an associated laboratory or field blank sample. Methylene chloride concentrations ranged from 0.00021 to 0.00032 mg/L and all results were below the SSSL and ESV.

Semivolatile Organic Compounds. SVOCs were not detected in the 10 surface water samples analyzed for SVOCs.

Chlorinated Pesticides. Chlorinated pesticides were not detected in the 10 surface water samples analyzed for chlorinated pesticides.

Organophosphorus Pesticides. Organophosphorus pesticides were not detected in the 10 surface water samples analyzed for organophosphorus pesticides.

Chlorinated Herbicides. Chlorinated herbicides were not detected in the 10 surface water samples analyzed for chlorinated herbicides.

Explosives. Explosive compounds were not detected in the 10 surface water samples analyzed for explosives.

Polychlorinated Biphenyls. PCBs were not detected in the nine surface water samples analyzed for PCBs.

4.1.5 Sediment Analytical Results

Twelve sediment samples were collected for chemical and physical analyses at the BBGR Ranges. The sediment sample locations are shown on Figures 2-1 through 2-9. The analytical results were compared to recreational site user SSSLs, ESVs, and background screening values, as presented in Table 4-5.

Lead. Lead was detected in all 12 sediment samples at concentrations ranging from 16.1 to 533 mg/kg. Only one result (533 mg/kg), at Range 23 sample location HR-79Q-SW/SD02, exceeded the SSSL of 400 mg/kg.

Eight of the 12 lead results exceeded the ESV (30.2 mg/kg) and four of these eight also exceeded background (37.8 mg/kg): HR-74Q-SW/SD01 (129 mg/kg) at Range 18; HR-76Q-SW/SD03 (56.8 mg/kg) at Range 20; HR-79Q-SW/SD02 (533 mg/kg); and HR-79Q-SW/SD05 (38.1 mg/kg) at Range 23.

The integrated statistical and geochemical evaluation (Appendix I) concluded that the lead concentration at sample location HR-79Q-SW/SD02 was anomalously high, indicating potential contamination.

Figure 4-5 shows the lead results exceeding the ESV and background in sediment. Two of the samples with elevated lead concentrations (HR-79Q-SW/SD02 and HR-76Q-SW/SD03) were collected from intermittent streams within the study areas. Sediment samples collected downstream of these locations, however, showed no lead contamination above background (Figure 4-5). Another elevated lead result (38.1 mg/kg) was in a sample (HR-79Q-SW/SD05) collected upstream of Range 23. However, the lead concentration in a sediment sample (HR-79Q-SW/SD04) collected downstream of this location was below background. Therefore, the lead concentration at sample location HR-79Q-SW/SD05, which only marginally exceeded the background value (37.8 mg/kg), does not appear to be related to Range 23.

At Range 18, the one sediment sample collected from South Branch of Cane Creek adjacent to the site contained an elevated lead concentration. However, several additional sediment samples, collected upstream and downstream of Range 18 as part of other investigations, showed lead at levels below the ESV and background (Figure 7-2). Thus, impacts to South Branch of Cane Creek sediments attributable to Range 18 appear to be very limited.

Metals (Other than Lead). Ten sediment samples were analyzed for TAL metals other than lead. A total of 21 metals were detected in the samples. None of these metals concentrations exceeded SSSLs.

The concentrations of four metals (arsenic, chromium, copper, and silver) exceeded ESVs. Of these, three metals also exceeded their respective background values:

- Chromium (136 mg/kg) exceeded its ESV (52.3 mg/kg) and background (31.2 mg/kg) at HR-83Q-SW/SD01 (Range 25).

- Copper (19.1 to 220 mg/kg) exceeded its ESV (18.7 mg/kg) and background (17.1 mg/kg) at HR-74Q-SW/SD01 (Range 18), HR-76Q-SW/SD03 (Range 20), HR-79Q-SW/SD02 (Range 23), and HR-79Q-SW/SD03 (Range 23).

- Silver (4.69 mg/kg) exceeded its ESV (2 mg/kg) and background (0.32 mg/kg) at HR-83Q-SW/SD01 (Range 25).

The integrated statistical and geochemical evaluation (Appendix I) concluded that the metals detected in the sediment samples are naturally occurring except for copper in one sample. The copper concentration at Range 23 sample location HR-79Q-SW/SD03 is anomalously high and may contain a component of contamination. Copper detected in the other site sediment samples is naturally occurring.

Figure 4-5 shows the copper results exceeding the ESV and background in sediment. With one exception (sample location HR-79Q-SW/SD03 within the Range 23 study area), the elevated copper results are co-located with the elevated lead results discussed previously.

Volatile Organic Compounds. Ten sediment samples were analyzed for VOCs. A total of five VOCs (2-butanone, acetone, methylene chloride, toluene, and carbon disulfide) were detected in the samples at concentrations ranging from 0.0013 to 0.52 mg/kg. All results were below SSSLs and ESVs except for acetone (0.52 mg/kg), which exceeded its ESV (0.45 mg/kg), but not its SSSL, in one Range 23 sediment sample (location HR-79Q-SW/SD04).

Semivolatile Organic Compounds. SVOCs were not detected in the 10 sediment samples analyzed for SVOCs.

Chlorinated Pesticides. Ten sediment samples were analyzed for chlorinated pesticides. A total of four compounds (4,4'-DDT, gamma-chlordane, alpha-BHC, and heptachlor) were detected in the samples. The pesticide concentrations ranged from 0.00095 to 0.089 mg/kg and all but two of the results were "J" flagged, indicating the concentrations were estimated. All results were below SSSLs. Each of the detected compounds exceeded its ESV in one or two samples each as follows:

- 4,4'-DDT (0.032 mg/kg) at Range 20 sample location HR-76Q-SW/SD03 (ESV = 0.0033 mg/kg).
- Alpha-BHC (0.0087 mg/kg) at Range 20 sample location HR-76Q-SW/SD03 (ESV = 0.006 mg/kg).
- Gamma chlordane (0.089 and 0.0042 mg/kg) at Range 20 sample location HR-76Q-SW/SD03 and Range 25 East sample location HR-223Q-SW/SD01 (ESV = 0.0017 mg/kg).
- Heptachlor (0.00099 and 0.0012 mg/kg) at Range 23 sample locations HR-79Q-SW/SD03 and HR-79Q-SW/SD04 (ESV = 0.0006 mg/kg).

Organophosphorus Pesticides. Organophosphorus pesticides were not detected in the 10 sediment samples analyzed for organophosphorus pesticides.

Chlorinated Herbicides. Ten sediment samples were analyzed for chlorinated herbicides. One compound (MCPP) was detected at estimated concentrations in two samples at Range 20

(locations HR-76Q-SW/SD01 and HR-76Q-SW/SD03). The MCPP results were below the SSSL (note: an ESV was not available for MCPP).

Explosives. Explosive compounds were not detected in the 10 sediment samples analyzed for explosives.

Polychlorinated Biphenyls. PCBs were not detected in the nine sediment samples analyzed for PCBs.

Total Organic Carbon. All of the sediment samples were analyzed for TOC. TOC concentrations ranged from 22.3 to 8,380 mg/kg.

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

4.2 XRF Surveys

Range 25 Safety Fan. An XRF survey of lead and copper concentrations at 25 surface soil locations was conducted within the Range 25 safety fan (Table 4-6, Figure 2-10). The XRF data indicate that lead concentrations within the Range 25 safety fan are generally within the range of background (40 to 83 mg/kg) (SAIC, 1998). The sample containing the highest lead concentration (195 mg/kg, sample no. R25X03) was located on the bank of Cane Creek, just north of the Range 25 impact zone, and its results may not be representative of the Range 25 safety fan.

Supplemental Metals Delineation at Range 18, Range 20, and Range 23. Additional XRF lead and copper data were collected in impact zone hotspots at Range 18 (56 locations), Range 20 (23 locations), and Range 25 (19 locations) to further delineate metals contamination. These XRF data are included on the lead-in-surface soil isocontour map (Figure 4-1). At Range 18 (Figure 2-1) and Range 25 (Figure 2-7) these supplemental XRF measurements were made relative to a previous sample location in either downrange or cross-range directions at increments of 50, 100, or 200 feet from the original location until the XRF reading for lead was less than 400 mg/kg. At Range 20 (Figure 2-2), additional surface and subsurface soil samples were collected using a hand auger and these samples were analyzed by the XRF. The XRF lead and copper results were used to supplement the laboratory-measured data and help further delineate the extent of contamination. However, the XRF data were not included in the risk assessments. Table 4-6 summarizes the XRF results for lead and copper.

Seventeen XRF soil samples (14 percent) were selected for QC confirmational analysis at an offsite laboratory. QC samples were sent to EMAX Laboratories, Inc. for lead and copper analysis using EPA Method 6010B. As shown in Table 4-7, the fixed-laboratory data generally agree with the XRF results. Only two of the 17 split sample results (locations 74Q-GP55 and 74Q-GP57) showed significant variation from the XRF results. In each of these cases, the laboratory results were considerably higher than the XRF results. These two samples were collected in areas at Range 18 where bullet fragments were present in the sampled interval and may have biased the QC sample results.

4.3 Sources of Contamination

The obvious source of metals contamination present at BBGR Ranges is ammunition (e.g., bullets, bullet fragments) from historical use of the ranges in weapons training. Ranges 18, 20, 25, and 26 were oriented such that the direction of fire was toward an impact zone that was typically defined by a natural hillside. Range 18 had a single firing line with multiple target zones resulting in a widespread metals contamination pattern (Figure 4-1). At Range 20, M-60 machine guns were mounted in a fixed position and fired at a small, hillside impact zone that contained numerous bullets and metals contamination. Range 25 had a single target line and multiple firing lines so the metals contamination was almost entirely present in the downrange hillside impact zone.

Range 26 had multiple firing lanes and firing points, leading to a series of fixed pop-up targets with a hillside impact zone forming the backdrop farther downrange. Therefore, at Range 26 the bulk of the metals contamination is concentrated in the pop-up target mounds and in the hillside impact area. Range 23 is somewhat similar to Range 26 in that pop-up target mounds were used; however, at Range 23 multiple pop-up targets were fired upon from a single firing point in each lane. Range 23 metals contamination is largely contained in and around these target mounds. Ranges South of Range 25 metals contamination exists from historical use as a training range within the area defined by the previous investigation. Within the Range 25 East and Range 28 parcels, no metals contamination was found. Range 25 East most likely was never constructed or used as a training range, as no range-related features were observed in the field and no evidence of range use exists on historical aerial photographs dating to the 1920s. Range 28 was used exclusively for training with M16 blanks, which did not contain a lead projectile and, thus, did not produce metals contamination.

5.0 Contaminant Fate and Transport

The fate and transport of contaminants in the environment depends on a variety of chemical, physical, and biological processes. This chapter describes the parameters that influence the fate and transport of contaminants found in soils at the BBGR Ranges. Soils, surface water, sediments, and groundwater were sampled to identify contaminants present at the BBGR Ranges, and the data are summarized in Chapter 4.0. Based on the watersheds present in the vicinity, the BBGR Ranges are divided into two groups. The first group consists of Ranges 20, 23, 25, 25 East, 26, and the Ranges South of Range 25, which are all adjoining (i.e., contiguous) and share confluent watersheds. Range 18 is separate and the watershed in which this range is located does not join with Ingram or Cane Creek. Therefore, where possible, contaminants in soils of those ranges that may impact Ingram and Cane Creeks (referred to in this chapter as the Contiguous Ranges) will be distinguished from the soils at Range 18.

5.1 Inclusion Criteria and Initial Screening

Not all constituents detected in samples are contaminants or related to site activities, and not all contaminants identified as site related will be mobile in the environment. Contamination of soils at the BBGR Ranges results from weapons testing or training and mainly includes metals (e.g., lead, antimony, copper, and zinc). Some detectable concentrations of metals may be naturally occurring, and low levels of organic compounds may not pose a threat to human health or the environment. Therefore, several means of screening the detected constituents have been applied and are described in the following sections.

Metal concentrations were statistically evaluated to distinguish naturally occurring concentrations and statistically insignificant outlier concentrations from concentrations that are attributable to site activities. However, most metals failed statistical comparison to background and required geochemical evaluation. Only cadmium, sodium, and thallium passed statistical tests. Geochemical evaluation indicated that all detected concentrations of aluminum, barium, calcium, chromium, cobalt, iron, magnesium, manganese, potassium, selenium, silver, and vanadium in the surface and subsurface soil samples are naturally occurring. In addition, the detected concentrations of arsenic and nickel in the subsurface soil samples are naturally occurring; and the detected concentrations of mercury in the surface soil samples are naturally occurring.

Table 5-1 provides a summary of the occurrences of these metals in environmental media at the BBGR Ranges. The metals determined to be present at anomalous concentrations in a single sample and all other naturally occurring metals will not be considered further.

Concentrations of metals that may not be naturally occurring, and that may represent a contamination at a site, may not pose a threat to human or environmental receptors. Threats to human or ecological receptors due to direct contact or injection are discussed in Chapters 6.0 and 7.0, and are not considered in this chapter, but the potential for site contaminants to leach to groundwater is of concern. The EPA has developed a method for screening site contaminants for protection of groundwater and has issued a list of generic soil screening levels (SSL) that may be used to determine if contaminants at a site pose a threat to groundwater (EPA, 1996a). These SSLs are soil concentrations of potential contaminants below which the soils should pose no threat to groundwater at the site. Generic SSLs may be applied where site soil, chemical, and hydraulic properties are not available. The development of the SSLs is described below.

5.1.1 Dilution Attenuation Factors and Soil Screening Levels

A process for establishing SSLs that are protective of groundwater from impact by potentially contaminated soils was developed by EPA (EPA 1996a; 1996b). This process provides a framework for screening contaminated soils, either by comparison to default generic SSLs or by developing site-specific SSLs where site-specific properties are available or known. There are two steps in determining SSLs:

- Derive a dilution-attenuation factor (DAF) that describes the hydraulic conditions representative of mixing between leachate and groundwater.
- Use the soil/water partitioning to obtain a soil contaminant concentration that is in equilibrium with an acceptable target leachate concentration (e.g., maximum contaminant level, preliminary remediation goal) and multiply it by the DAF.

The resulting soil concentration represents the SSL for that corresponding DAF. Soil concentrations in excess of this site-specific SSL could potentially result in a groundwater concentration that exceeds the target concentration and presents a risk to receptors.

Dilution-Attenuation Factor. In the SSL technical background document (EPA, 1996a), data from sites across the nation, representing a wide spectrum of hydrogeologic settings, were reviewed and generic values for the hydrogeologic conditions and soil properties considered representative of most site settings were established as default values for the SSL and DAF computations. Generic SSLs were calculated for DAF_1 (i.e., no dilution or attenuation) and

DAF_{20} (i.e., a 20-times dilution) and are presented in EPA (1996a) and in EPA preliminary remediation goal tables (EPA, 2002a; 2003a).

The DAF is calculated as follows:

$$DAF = 1 + ((K*dh/dl*b)/(I*L)) \qquad \text{Eq. 5-1}$$

where:

K	=	Hydraulic conductivity (meters per year)
dh/dl	=	Hydraulic gradient (unitless)
b	=	Mixing zone depth (meters).

given by:

$$b = (0.0112L^2)^{0.5} + b_A*\{1-\exp[(-L*I)/(K*dh/dl*b_A)]\} \qquad \text{Eq. 5-2}$$

b_A	=	Thickness of the aquifer (meters)
I	=	Infiltration rate (meters per year)
L	=	Length of flowpath (meters).

The generic DAF SSLs are based on a number of default assumptions chosen to be protective of human health for most site conditions, and are expected to be more conservative than site-specific levels (EPA, 1996a). The DAF_{20} SSLs are used in this evaluation. The default values used by EPA in deriving the DAF_1 site-specific SSLs are:

	EPA Default
f_{oc}	0.002
n_W	0.30
n_A	0.134
p_b	1.50
p_s	2.65
pH	6.8

where:

f_{oc}	=	Organic carbon content
n_W	=	Water-saturated porosity
n_A	=	Air-saturated porosity
p_b	=	Bulk density
p_s	=	Soil particle density.

Table 5-2 provides generic SSLs for metals that were determined not to be naturally occurring in more than one soil sample. Table 5-3 provides generic SSLs for organic compounds identified in soils.

The EPA process for screening soils for protection of groundwater is based on the following simplifying assumptions:

- There is an infinite source term (steady state over time).
- The contaminants are uniformly distributed throughout the vadose zone, extending to the top of the water table (essentially, the highest soil contamination is located at the water table, regardless of where it is encountered in the subsurface).
- There is no contaminant attenuation via adsorption, biodegradation, or chemical degradation in the vadose zone.
- Soil/water equilibrium partitioning is instantaneous.
- The "receptor" well is located at the immediate downgradient edge of the source, such that there is not additional dilution afforded by distance from the source.
- No contaminant attenuation, apart from dilution and dispersion, is occurring in the aquifer.

Contaminant Travel Times. Because of these assumptions, the site-specific SSLs are very conservative screening values and may overestimate impact by contaminated soils. Therefore, should a contaminant concentration exceed the DAF_{20} SSL, further evaluation may be conducted to assess whether there is sufficient source term strength to create a current or potential future release to groundwater at a concentration that would pose a risk to a receptor. Because soil contamination in excess of DAF_{20} SSLs may be observed at shallow depths, far from the underlying water table, it is possible that such contamination will not reach the aquifer below within a reasonable timeframe. Alternatively, given how long many of the sites have been in use, the contaminants may have already leached to groundwater. The driving factor in contaminant leaching in the vadose zone is vertical infiltration. For the BBGR Ranges, travel time is based on the annual recharge rate of the water table, calculated as:

$$I = P-(Ro+ET) \qquad \text{Eq. 5-3}$$

where:

I = Recharge
P = Annual precipitation (53 inches per year)

Ro = Runoff; using the methods in USDA (1986)
ET = Estimated evapotranspiration, (assumed as 60 percent of P).

Recharge is estimated to be about 17.6 inches per year (0.45 meters per year). The time required for a contaminant to leach to the water table is proportional to the infiltration rate and the depth to the water table from the depth where the contamination was detected. However, various processes (e.g., mineral precipitation, cation exchange, sorption, and degradation) act to further retard migration. A preliminary estimate of the time required for a contaminant to leach to the water table is given by:

$$T = (z/I)*R \qquad \text{Eq. 5-4}$$

where:

T = Transport time (years)
z = Vertical distance to water table (meters)
I = Infiltration rate (0.45 meters per year)
R = Retardation factor ($[1 + (p_b/n] * K_d$).

Some of the BBGR Ranges were active as early as the 1940s. Thus, contaminants with calculated transport times less than 70 years should be present or evident in the groundwater. Where the travel time is between 70 and 100 years, an impact in the near future may be expected. Where the travel time is more than 100 years, the impact of the contaminant to groundwater does not pose a significant risk to human health, and the only action warranted is monitoring. Tables 5-2 and 5-3 give the travel times for inorganic and organic constituents to migrate from the surface soils to the water table. Based on water levels measured in December 2002, the depth to groundwater at the BBGR Ranges varies from less than 1 foot (Range 23) to about 22 feet bgs (Range 25). Travel time estimates are based on an average depth to water of 6.5 feet.

5.1.2 Comparison to Soil Screening Levels

The maximum concentrations of metals that failed comparison to background and the maximum detected concentrations of organic compounds in soils and sediment are given in Table 5-4, and indicate the following:

- The maximum arsenic concentration exceeds the DAF_{20} SSL in one surface soil sample at the Contiguous Ranges only. However, the result was judged to be naturally occurring in the geochemical evaluation.

- Maximum antimony and lead concentrations exceed DAF_{20} SSLs in surface soil, subsurface soil, and sediment within the Contiguous Ranges, and in surface soil at Range 18.

- Copper concentrations exceed the DAF_{20} SSL in surface soil within the Contiguous Ranges and at Range 18.

These findings indicate that only these metals potentially pose a threat to groundwater at the BBGR Ranges due to leaching. However, none of the metals that were present at levels above DAF_{20} SSLs were found to be elevated in groundwater, thus, active leaching of these metals may not be occurring. The large number of lead and antimony concentrations in both surface and subsurface soils and the occurrence of elevated concentrations of these metals in sediment (Table 5-1) indicate that transport via erosion may be a concern at this site. Table 5-1 shows that elevated concentrations of antimony, lead, and copper that could not be attributed to natural sources were found in many surface soil samples. Therefore, while these metals are not a threat to groundwater, an action may be considered for protection of sediments and surface waters at the site.

Organic compounds detected in surface soils, subsurface soils, or sediment that exceeded the generic DAF_{20} SSLs included:

- 2,4-DNT in three surface soil samples and one subsurface soil sample
- 2,6-DNT in one surface soil sample
- Benzo(a)anthracene and benzo(b)fluoranthene in one surface soil sample
- Alpha-BHC in four surface soil samples
- Beta-BHC in two surface soil samples
- Dieldrin in four surface soil samples and one subsurface soil sample.

As shown in Tables 5-1 and 5-4, the number of detected contaminants and the frequency of detection for individual contaminants decrease greatly from surface to subsurface soils, consistent with the expected mobility of organic compounds in site soils (Table 5-3). Except for the dinitrotoluene compounds, travel times for the SVOCs and pesticides listed above to reach groundwater are more than 700 years. Therefore, these compounds are nearly immobile in the soil-soil water-groundwater system.

5.2 Migration Mechanisms

Potential routes of contaminant migration at the BBGR Ranges include:

- Transport of soils by wind or water
- Contaminants leaching from soil to groundwater
- Discharge of contaminated groundwater to surface water
- Transport of contaminants by migration with groundwater.

5.2.1 Physical Transport by Erosion

Metals and organic compounds may adhere to soil particles and be transported via erosion. Erosion of contaminated soils will occur where the surface soil is exposed and vegetation cover is poor or absent. Erosion of soil by wind is not likely at the majority of BBGR Ranges due to the presence of grassy areas and wooded cover. Where water erosion is important, sediment and surface water in receiving streams should exhibit contamination by the constituents identified for surface soils at the site. At the BBGR Ranges, sediments exhibit impact by antimony, copper, lead, zinc, and a number of VOCs, SVOCs, and pesticides (Tables 5-1 and 5-4) that can be attributed to surface soils. Thus, erosion is an active transport mechanism, and continued downstream transport can be expected.

5.2.2 Transport in Soil Waters

For metals, the primary migration pathway in soil waters will be via migration to groundwater as ionic or complex species dissolved in infiltrating soil waters. Organic compounds will migrate in soil waters mainly in the dissolved phase; however, the dissolved concentration will be strongly affected by adsorption to organic matter in the soils.

Inorganic constituents exist in soils in several forms, and can derive from naturally occurring minerals and ore deposits, or from wastes. Contamination may result from improper disposal of solid or liquid wastes from municipal or industrial facilities, from mine tailings, or, in the case of the BBGR Ranges, from bullets or shot used in weapons training.

Metals transport in the subsurface generally occurs in the ionic form and is influenced by various processes. These include oxidation-reduction reactions that change ionic state, adsorption of an ionic species, reactions that form insoluble precipitates, and formation of organic complexes or ionic radicals (Davis et al., 1993). The degree to which these processes affect a metal in soil is dependent on the mineral composition of the soils, the pH and oxidation-reduction potential (Eh) of the soil waters, and the presence or absence of ligands or anions with which to form complexes or ionic radicals.

Concentrations of inorganic ions in soil are usually controlled by adsorption-desorption reactions, most commonly to oxide, carbonate, or clay minerals (Davis and Kent, 1990). Adsorption behavior of metal ions is dependent on the ionic charge and size. Oxide and hydrous oxide minerals are the most important substrates for adsorption (Davis et al., 1993). Clays and aluminosilicate minerals appear to be weak absorbance substrates. Metal ion adsorption can be significantly decreased under conditions of low pH; however, the sorption behavior is highly

dependent on both ionic speciation and mineral types present in the soil. In addition, if organic materials are present in the soils, metal-ligand complexes may form that prevent adsorption entirely (Davis et al., 1993)

Oxidation and reduction reactions may occur in the soil system that change the valence of the metal ion. When these metal ions combine with anions present in the soil waters or with components of the soil materials, an insoluble precipitate may form. These reactions are most common under reducing conditions, producing insoluble sulfides. Because the soil environment commonly contains organic matter, the soil Eh may be low and mildly reducing (less than 500 millivolts), and oxides, nitrates, and sulfates may be reduced.

Models for solute transport of inorganic compounds focus mainly on metals. Most solute transport models emphasize sorption processes, oxidation-reduction reactions, and mineral solubility. Adsorption reactions are modeled in a similar way as sorption of organic molecules, using partition coefficients and a variety of isotherm equations; however, the derivation of the adsorption coefficients is different and is dependent on site-specific factors, including:

- Metal type and ionic species
- Mineralogical composition of the soils
- Soil moisture content, pH, and Eh environment
- Concentration of ligands that can form complexes with metal ions.

Adsorption to soil is a function of the abundance of sorption sites. Generally, metals adsorb to clay minerals in the soil, but organic material may also provide sorption sites for some metallic ions. Adsorption is governed by the metal's specific adsorption coefficient (K_d). A compound with a higher K_d value will be more strongly partitioned to the soil than compounds with a lower K_d value.

However, the distribution coefficient is variable and depends on the pH of the environment. The K_d values used for this report are the default values used by the EPA (EPA, 1996a; 1996b) given in Table 5-2.

Metals in BBGR Range Soils. Lead, antimony, and copper are the only contaminants in soils that may pose a threat to human and environmental receptors at the BBGR Ranges. All lead, antimony, and copper concentrations in groundwater were found to be within the range of background. Thus, while the threat to environmental receptors attributable to the BBGR Ranges is due to metals in soils, no measurable impact is evident in groundwater. Lead and antimony

concentrations in sediment are above the DAF_{20} SSLs; however, these metals are either not detected or are at concentrations below background values in surface water and, thus, may not be mobile in the surface water system.

Organic Compounds in BBGR Range Soils. Detectable concentrations of eight pesticides were found in groundwater samples (Table 5-1); however, five of the detected pesticides are restricted to a single well (HR-83Q-MW02), and two are restricted to two locations (HR-83Q-MW02 and HR-79Q-MW01). In addition, methylene chloride was the only detected organic compound in surface water. Moreover, both wells are located in areas where groundwater is in contact with surface soil, and HR-89Q-MW02 is in a spring area. Thus, contaminants are probably partitioning to groundwater at these locations as it is moving into and through the surface soils. Further, the compounds detected in these samples were not detected in surface water samples or in any other groundwater sample, thus, there is no apparent impact to surface water, or migration in groundwater. Because of the near immobility of the organic compounds detected in surface and subsurface soils, and their infrequency of detection in groundwater associated with the site, transport of organic compounds to or within groundwater does not appear to be occurring, and will not be considered further. Organic and nitroaromatic compounds detected in soils were not detected in associated groundwater samples, thus impact to groundwater from metals and organic compounds in surface and subsurface soils should not be expected to result in groundwater concentrations above regulatory limits.

5.3 Compound Descriptions

Metals are not degraded or readily detoxified by microbial activities and, thus, can pose a long-term environmental hazard. Downward movement of inorganic constituents from surface soil will occur if the capacity of the soil to retain them is exceeded or if changes in site conditions, such as pH or oxidation state, transform them to a mobile form. If the soil loses its ability to retain the inorganic constituents, metals will migrate with the leaching waters. The inorganic constituents may rebind to soils in lower layers if reversals in the conditions that caused the mobility are encountered, creating chemical fronts in the soil column. The presence of such fronts is an indicator that the constituents are, in fact, mobile in the subsurface environment.

Surface transport via dust and erosion of soils is also a common transport mechanism. Surface transport normally does not involve changes in the chemical environment, and is driven mainly by physical forces such as wind or water erosion.

Properties and behavior of specific metals are discussed below. The information provided in this section was obtained from several sources including:

- Toxicological profiles published by the Agency for Toxic Substances and Disease Registry (ATSDR) (1996a, 1999a, 2002, 2003).
- World Health Organization (1996) Guidelines for Drinking Water Quality.
- EPA Drinking Water Contaminant Fact Sheets (EPA, 2002b).

Lead. Lead occurs as sulfide ores in natural deposits. Lead is produced as a primary product of smelting lead and zinc ores (galena and sphalerite). Where primary sulfide ore deposits intersect the water table, secondary supergene deposits containing lead carbonate, sulfate, and phosphate ores develop. Metallic lead is malleable, ductile, has a low melting point, is very reactive in acids, and is very dense. Because of these properties, there are many commercial uses for lead, primarily in small-arms ammunition, lead-acid batteries, solder, and alloys. The organolead compounds tetraethyl and tetramethyl lead have also been used extensively as antiknock and lubricating agents in gasoline. Because of its ductile and malleable nature, lead has been used to manufacture pipes for water supplies (World Health Organization, 1996). Lead is also used extensively in manufacturing shot and sinkers because of its high density and ductility. Because of the serious health effects of lead, EPA banned the use of lead pipes in 1998 and established a maximum soil contamination level of 400 mg/L.

Lead readily forms oxides and carbonates. Exposure to environmental conditions and acidic conditions (pH less than 6.0) tends to accelerate the weathering process of lead. At high pH levels and under reducing conditions, lead becomes nearly immobile (Scheuhammer and Norris, 1995).

Antimony. Only one antimony ore, stibnite, is abundant enough for commercial mining, but antimony is economically recovered from copper and lead ores when they are smelted. Antimony is used as an alloy or an oxide. Antimony is alloyed with lead as a hardening agent for use in small-arms ammunition and in lead-acid batteries and may be alloyed with tin or copper for use in machine bearings (EPA, 2002b). Antimony released to the environment has a strong affinity for soil particles containing iron, manganese, or aluminum (EPA, 2002b). Adsorption of antimony to soil and sediments is dependent on the soil and soil-water pH, organic matter content of the soil, and ionic charge. Antimony may be highly mobile or may be strongly adsorbed to soil, depending on site conditions. In water, it usually binds to sediments. There is

no evidence of bioconcentration of most antimony compounds, though one report states that the tribromide salt can be concentrated by certain forms of marine life to over 300 times its concentration in water (EPA, 2002b).

Copper. Copper is a widely naturally occurring metal. Copper occurs in economically exploitable ore deposits and at trace levels in many common sulfide, oxide, and carbonate minerals. Copper is used widely for a variety of purposes: in electronic and electrical applications, in heat exchangers, in motors, for plumbing fittings, in building construction and roofing, in chemical and marine equipment, for cooking utensils, in wood preservatives, as an antifouling agent in paints, and as a trace nutrient in livestock feeds. Primary sources of copper releases to the environment include coal-fired power stations, metal production, waste incinerators, sewage treatment processes, and the application of agricultural chemicals. Excess copper in soils is toxic to some microorganisms, disrupting nutrient cycling and inhibiting the metabolic use of essential nutrients such as nitrogen and phosphorus. Some species accumulate copper. Toxic effects on fish and other aquatic organisms have also been observed.

Dinitrotoluenes. The isomers 2,4- and 2,6-DNT are primarily used as chemical intermediaries for toluenediamines and as a gelatinizing and waterproofing agent in explosives. DNT has also has been used in the manufacturing of some dyes. The primary use has been by the munitions industry, as a plasticizer for smokeless powders, as an explosives intermediate, and in the manufacture of rubber chemicals and plastics. DNT may enter the environment from spent bullet casings or unexploded ordnance. In soil, DNT is slightly mobile. Based on aqueous biodegradation tests, DNT may biodegrade in both aerobic and anaerobic zones of soil. DNT is subject to destruction by ultraviolet light; thus, photolysis will probably be the most important removal process for DNT in surface waters. Photolytic half-lives for DNT in surface water bodies are relatively short, ranging from 2.7 to 9.6 years. Bioaccumulation of DNT appears to be negligible.

Hexachlorcyclohexane (BHC) Isomers. Alpha-BHC and beta-BHC are immobile, long-lived organochlorine isomers of lindane (gamma-BHC). The alpha and beta isomers of BHC are derived by transformation of lindane during degradation. Like lindane, these isomers are strongly adsorbed to organic matter and, thus, the organic matter content of soils is an important factor in controlling their mobility; as organic matter content of the soils increases, the residence time of BHC increases. Lindane and other BHC isomers appear to be resistant to biodegradation pathways, and chemical processes nearly exclusively control their persistence in soils. It appears that the order of resistance to degradation or loss from soil compartments is

beta>gamma>delta>alpha (Wagenet et al., 2004). Persistence of BHC is influenced mainly by increased organic matter and higher pH.

Polynuclear Aromatic Hydrocarbons. Benzo(a)anthracene and benzo(b)fluoranthene are both PAHs, hydrocarbon compounds with multiple benzene rings. Both are known carcinogens. These compounds are typical components of asphalt, fuels, oils, and may be formed during the incomplete burning of coal, oil, gas, wood, garbage, or other organic substances, such as tobacco. PAHs generally occur as complex mixtures (e.g., combustion products) and not as single compounds. PAHs usually occur naturally, but they can be manufactured as individual compounds to be used in medicines and to make dyes, plastics, and pesticides. PAHs may occur throughout the environment in the air, water, and soil. When present in air, PAHs will generally be adsorbed to dust particles or as solids in soil or sediment, and may be transported suspended in air currents. Some PAHs volatize to the atmosphere from surface waters or soils, but are usually strongly adsorbed to solid particles. In water bodies they settle to the bottoms of rivers or lakes on the matrix to which they are adsorbed. In soils, PAHs are strongly adsorbed to soil particles, but may exhibit limited partition to soil moisture or groundwater. PAHs strongly bioaccumulate and the PAH content of plants and animals living on the land or in water can be many times higher than the content of PAHs in soil or water. PAHs can be degraded via photolysis (reactions induced by the UV in sunlight), by oxidization in the atmosphere, or by microbial activity in soils. Photolytic and oxidizing reactions are rapid, on the order of days to weeks, however, biodegradation is slower, on the order of weeks to months.

Dieldrin. Dieldrin is a pesticide that was widely used for crops like corn and cotton. Dieldrin is chemically similar to aldrin, which is altered to dieldrin by the action of sunlight and bacteria. Because of concerns about damage to the environment and, potentially, to human health, EPA banned the use of aldrin and dieldrin in 1974, except to control termites. In 1987, EPA banned all uses of these compounds. Dieldrin is immobile in the soil system and is resistant to degradation. Dieldrin is strongly adsorbed to organic matter and, thus, the organic matter content of soils is an important factor in controlling its mobility; as organic matter content increases, the mobility of dieldrin decreases.

5.4 Contaminant Transport and Persistence

Constituents identified in soils that exceeded the site-specific DAF_{20} SSLs and background levels included:

- Antimony
- Arsenic
- Copper
- Lead
- Benzo(a)anthracene and benzo(b)fluoranthene
- 2,4- and 2,6-DNT
- Alpha and beta BHC isomers
- Dieldrin.

Table 5-4 summarizes the maximum concentrations of these contaminants in surface and subsurface soils and sediments at the BBGR Ranges.

Lead was detected at concentrations above the DAF_{20} SSL (400 mg/kg) as follows:

- Seventeen surface soil samples at Range 18
- Seventy-five surface soil samples at the Contiguous Ranges
- Six subsurface soil samples at the Contiguous Ranges
- One sediment sample at Range 23.

Antimony was detected at concentrations above the DAF_{20} SSL (5 mg/kg) as follows:

- Seven surface soil samples at Range 18
- Ten surface soil samples within the Contiguous Ranges
- Four subsurface soil samples at four ranges within the Contiguous Ranges
- One sediment sample at Range 25.

Antimony was detected in surface and subsurface soil samples, and in sediment but was not detected in groundwater or surface water. Therefore, the primary mode of antimony transport appears to be via erosion of surface soils and with the bedload in streams.

Arsenic was detected at a concentration above the DAF_{20} SSL (29 mg/kg) in one surface soil sample at Range 25; however, the detected concentration was determined to be naturally occurring. Arsenic was either not detected or was found to be naturally occurring in both surface water and groundwater. Therefore, arsenic appears to be immobile at the BBGR Ranges.

Copper was detected at concentrations above the DAF_{20} SSL (2,070 mg/kg) as follows:

- One surface soil sample at Range 25
- One surface soil sample at Range 18.

Copper was not detected in subsurface soils or sediments at concentrations above the DAF_{20} SSL, and was not detected in groundwater or surface water. Therefore, copper is not mobile at the BBGR Ranges.

DNT was detected at concentrations above DAF_{20} SSLs (0.0008 and 0.0007 mg/kg for 2,4-DNT and 2,6-DNT, respectively) in a total of three soil samples at Range 23 and one surface soil sample at Range 18. DNT appears to be relatively mobile in soils (Table 5-3), however, this compound is degraded by microorganisms in the soil under aerobic conditions, and is not present in sediment, surface water, or groundwater. Thus, it is not expected to impact other environmental media.

Numerous PAHs were detected in a few samples, however, only benzo(a)anthracene and benzo(b)fluoranthene were present at concentrations above DAF_{20} SSLs in a single surface soil sample from Range 26. The PAHs are strongly bound to soil, and are essentially immobile in soils. Furthermore, these compounds were not detected in sediment, surface water, or groundwater. Therefore, PAHs (including benzo(a)anthracene and benzo(b)fluoranthene) should be considered immobile at the BBGR Ranges.

Alpha- and beta-BHC isomers were detected at concentrations above DAF_{20} SSLs (0.0005 and 0.003 mg/kg, respectively) in five surface soil samples from the Contiguous Ranges, and alpha-BHC exceeded the DAF_{20} SSL in one surface soil sample from Range 18. Alpha-BHC was also detected in four sediment samples at concentrations above the DAF_{20} SSL. Beta-BHC was detected in two subsurface soil samples at the Contiguous Ranges but at concentrations below the DAF_{20} SSL. Beta-BHC was detected in groundwater, however, the samples are from wells where the groundwater is in contact with surface soils and are potentially in spring areas. No pesticides were detected in surface waters. While there is limited impact to groundwater, these compounds bind strongly with the soil matrix and the transport of these compounds is expected to be limited. Therefore, BHC isomers are considered to be immobile at the BBGR Ranges.

Dieldrin was detected at concentrations above the DAF_{20} SSL (0.0002 mg/kg) in four surface soil samples and one subsurface soil sample collected at 2 to 4 feet bgs. Dieldrin was not detected in the sediment, surface water, or groundwater samples collected at the BBGR Ranges. Therefore, this compound has not impacted these media, and appears to be immobile at the BBGR Ranges.

5.5 Contaminant Fate and Transport Evaluation Findings

- Formation of leachate and transport in soil waters is a viable transport mechanism for organic compounds at the BBGR Ranges. However, concentrations of most compounds are below their DAF_{20} SSLs, and groundwater concentrations are expected to remain below their regulatory limits.

- Lead, arsenic, antimony, and copper are the only metals at the BBGR Ranges that exceed the DAF_{20} SSLs; however, these metals appear to be nearly immobile in the subsurface environment. Concentrations of antimony and lead in sediment indicate that erosion of surface soils is an active transport mechanism at the BBGR Ranges, and continued transport can be expected unless some action is taken.

- Benzo(a)anthracene, benzo(b)fluoranthene, 2,4-DNT, 2,6-DNT, alpha-BHC, beta-BHC, and dieldrin were detected at concentrations above DAF_{20} SSLs for surface soils; however, the impact of these compounds is largely limited to the surface soils. The BHC isomers are nearly immobile in the soil system, and DNT compounds may be removed via microbial activity; thus, impact to the groundwater is not expected. Concentrations of alpha-BHC in sediment indicate that erosion of surface soils may be transporting contaminated soils.

6.0 Streamlined Human Health Risk Assessment

Due to the large size and complexity of the FTMC installation, a streamlined human health risk assessment (SRA) protocol has been developed to standardize risk assessments across sites and to simplify their presentation, resulting in a considerable economy of scale. The heart of the SRA protocol is the development of site-specific screening levels (SSSL), which are medium-specific and receptor-specific, risk-based screening concentrations. The SSSLs are used to quickly and efficiently screen the site for potential cancer risk and noncancer hazard from residual chemicals in environmental media. They address all significant exposure pathways and are sufficiently site-specific regarding exposure assumptions that they are used to estimate risk with as much precision as a typical CERCLA baseline risk assessment. The exposure assumptions and SSSL methodology are described in detail in the ADEM- and EPA-approved *Installation-Wide Work Plan* (IT, 2002a), hereafter referred to as the IWWP. It is assumed that the reader is familiar with the FTMC facility, with risk assessment in general, and with the IWWP.

The SSSLs were recently updated with the most current toxicity values, and the SSSLs, toxicity values and chemical-specific physical properties used to generate the SSSLs are compiled in the *Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000b). IT (2000b) also presents toxicity profiles, which are brief descriptions of the physical and toxicological properties of the chemicals that may be identified as contaminants at FTMC sites. The toxicity values for all chemicals detected at the BBGR Ranges were examined for currency before the SRA was performed. For example, the chronic oral reference dose (RfD) for the aminodinitrotoluenes (ADNT), and the RfDs and cancer evaluation of the nitrotoluenes have been recently updated and the SSSLs were revised accordingly.

The SSSLs are based on the most highly exposed receptor or receptor scenario for each of several uses to which an individual site on FTMC may be subject. For example, it was determined that current or future land-use of most FTMC sites may be categorized broadly as follows:

- ***Residential:*** restricted to living areas, such as single- or multiple-dwelling homes and retirement centers.

- ***Commercial/Industrial:*** including schools, parks, playgrounds, golf courses, employment areas, office buildings, stores, malls, medical facilities, research facilities, motor pools and garages, transportation facilities, military facilities,

training fields, landfills, dumps, disposal sites, and all other areas and activities other than residential.

- ***Open Space:*** including "unused" land or buffer space, wetlands, wooded or meadow areas, and all other areas not used for residential or industrial activities. Hiking, hunting, fishing or other recreational use may occur.

The most highly exposed receptor for each land-use category was selected and SSSLs were developed for that receptor alone, rather than for each receptor that might be included in a land-use category. For example, an exposure scenario for an on-site resident consisting of a family with children was developed as a surrogate for all who may be exposed in a residential setting, and separate SSSLs were not developed for residents in a children's home, boarding school, retirement or assisted living home, etc. The on-site resident would be more highly and frequently exposed to site media, such as soil and groundwater and would represent the upper bound on residential risk. Most sites are evaluated with the residential scenario even if the site is not slated for residential development, because the resident generally represents the upper bound on long-term risk, and the information provided by this evaluation is useful to site managers. Sites that "pass" the residential evaluation generally can be released for unrestricted use with no further action. Sites that "fail" the residential evaluation do not necessarily require remediation, but the information may be useful for determining future site use, site-use restrictions, or other site management decisions.

An exposure scenario for an on-site groundskeeper was developed as a surrogate for all site workers. The groundskeeper would be more intensely exposed to site media than an indoor worker, delivery person or part-time maintenance personnel, and would represent the upper bound on worker risk. Similarly, an exposure scenario was developed for a recreational site user as a surrogate for all recreational site use. The original EDAW (1997) *Fort McClellan Comprehensive Reuse Plan, Implementation Strategy* distinguished active recreation (e.g., parks, playgrounds) from passive recreation (e.g., open space, hiking or camping facilities), but the distinctions were insufficient for developing separate exposure assumptions with confidence. Therefore, the exposure assumptions for the recreational site user are intended to be sufficiently protective for either active or passive recreation.

Please refer to the IWWP for more detail regarding the rationale and protocol for development of the SSSLs.

It should be noted that the IWWP has been updated from the 1998 version, improving the manner in which source-term concentrations (STC) are estimated for soil, surface water and

sediment. Also, the protocol for comparing site and background data for the identification of site-related chemicals has been revised since the updated IWWP (Shaw, 2003b). Changes in protocol are discussed in more detail in their appropriate sections in the remainder of this chapter. The reuse plan for the FTMC facility as a whole also has been updated (EDAW, 1997).

6.1 Streamlined Risk Assessment Protocol

An SRA consists of the following steps:

- Develop a conceptual site exposure model (CSEM)
- Evaluate data
- Assess exposure
- Assess toxicity
- Characterize risk
- Discuss sources of uncertainty

The SRA protocol is briefly described below. The results for the individual ranges are discussed in Section 6.2.

6.1.1 Conceptual Site Exposure Model

Each of the BBGR Ranges consists of a smaller "study area," generally incorporating the area of the firing lines, targets, berms and backstops, and a larger "safety fan," which includes areas from which personnel were excluded during firing exercises. The seven ranges or combinations included in this evaluation and the order in which they are discussed are as follows:

- ***Range 18, Down Range Feedback (Known Distance) Range, Parcel 74Q.*** Historical use as a rifle range (assorted calibers) and machine gun range. Topography scarred with bullet channels and littered with bullet fragments. No known or expected use of fused ordnance on this site, reducing concern for UXO. Study area comprises approximately 70 acres. Range safety fan extends to the south covering an area of approximately 3,304 acres. Located to the southwest of the South Branch of Cane Creek, towards which runoff flows. Range 18 (Parcel 74Q) is evaluated in the SRA as an individual unit designated "Range 18."

- ***Range 20, Infiltration Course, Parcel 76Q-X.*** Weapons fired included machine gun with tracer. Dynamite, trinitrotoluene (TNT), and C4 (military explosive) were used in pits for artillery impact simulation and may have been used in demolition training. Range 20 was constructed within the WWI Artillery Impact Area. UXO is a concern. The Range 20 study area includes 33 acres; the range safety fan extends to the southeast an area of approximately 1,613 acres. Two tributaries cut across Range 20 draining runoff water west towards Ingram Creek. Range 20 (Parcel 76Q-X) is evaluated in the SRA as an individual unit designated "Range 20."

- ***Range 23, Trainfire (Record) Range, Parcel 79Q.*** Was used as a pistol range before being used as a rifle range and machine gun range. Has had multiple layouts and orientations. Constructed within WWI Artillery Impact area. The range study area consists of approximately 59 acres. The range safety fan extends to the southeast an area of approximately 4,566 acres. Three tributaries of Ingram Creek are present at the range. Two are seasonal; the stream that flows from the central portion of the range generally contains water all year. Range 23 (Parcel 79Q) is evaluated in the SRA as an individual unit designated "Range 23."

- ***Range 26, Live Fire and Maneuver Range, Parcel 84Q-X.*** Was used for day and night rifle fire and maneuver training. Was constructed within the WWI Artillery Impact Area. Bullet fragments and channels were evident. The study area is defined as approximately 34 acres; the range safety fan extends to the southeast an area of approximately 1,267 acres. Two tributaries of Ingram Creek collect runoff from the hillsides to the south and northeast (in the possible impact area) and flow toward the northwest. However, water is generally present only during and shortly after storm events. Range 26 (Parcel 84Q-X) is evaluated in the SRA as an individual unit designated "Range 26."

- ***Range 25, Known Distance Range, Parcel 83Q; and Main Post Impact Area, Parcel 118Q-X (the latter contained entirely within the former).*** Weapons fired included various small arms and machine guns with tracers. Bullet impact channels, surface fragments and expended mortars and other unexploded ordnance are present. The study area is defined as approximately 103.1 acres; the safety fan extends to the northeast approximately 1,713 acres. Parcel 118Q-X consists of approximately 4 acres. Two tributaries of Ingram Creek flow west. A third drainage feature runs north to Cane Creek. Soil data from Range 25 and the Main Post Impact Area are combined for the purposes of the SRA. Data from Range 25 and the Main Post Impact Area are combined and designated as "Range 25" for this evaluation.

- ***Former Range 25 East, Parcel 223Q; and Range 28, Target Detection Range, Parcel 86Q.*** Range 25 East is a historical range; documentation regarding its location and uses is limited. Range 28 was an active range in use until Base closure. Range 28 partially overlaps Range 25 East. Specific weapons used here probably limited to various rifle calibers, although the area is within the WWI Artillery Impact Area. Weapons used on Range 28 may have been limited to M-16 blanks. No evidence of bullet impact was found at Range 28 and soil samples were not taken specifically from Range 28. Three tributaries of Ingram Creek cross Range 25 East. The study area is approximately 44 acres. Data from Range 25 East and Range 28 are combined and designated as "Range 25E/28" for this evaluation.

- ***Area of Further Investigation for Ranges South of Range 25, including the Former Pistol Range south of Range 25 (Parcel 224Q) and Former Machine Gun Range (Parcel 226Q).*** Little information is available for the Former Pistol Range, except that fire was probably directed to the east or south.

The impact area for Parcel 226Q appears to be within the impact area for Range 23. The study area covers approximately 6 acres. Data from the Former Pistol Range and the Former Machine Gun Range are combined and designated as "Ranges South of Range 25" for this evaluation.

The history and physical features of these seven ranges are presented in greater detail in Chapter 1.0. The information summarized above, however, is sufficient to support the SRA.

These seven ranges have been grouped together for investigation and possible remediation because they were all small arms ranges and are located in the vicinity of the southern branch of Baby Bains Gap Road that was generally referred to as "Baby Bains" before Base closure. Most were either still in use by the Army or had been recently deactivated at the time of Base closure.

There is considerable overlap of several of the BBGR Range study areas (Figure 1-2) and even greater overlap of the range safety fans (Figure 1-1). Each range, however, is large and is evaluated separately (except for the combinations noted in the bulleted paragraphs above) in the SRA to minimize the dilution effect of incorporating all data for a given medium into a single data set.

6.1.1.1 Contaminant Sources, Release and Transport

The CSEM identifies the potential contaminant sources and release mechanisms, the originally contaminated environmental media ("source" media), and contaminant migration pathways for the BBGR Ranges (Figure 6-1). The BBGR Ranges are sufficiently similar that they can be described as one for the purposes of the CSEM.

Briefly, the discharge of various weapons and firearms caused a variety of bullets and shot to affect surface soil and, to a lesser extent, subsurface soil, releasing into soil the metals, explosives, and propellants associated with bullets, shot, and projectiles. The impact of metals was most likely to have occurred at target locations, backstops, or berms. Soil at the firing lines may have been affected by chemicals associated with gunpowder, explosive or propellants, or chemicals used to clean, lubricate, or maintain firearms.

Chemicals in soil may leach to groundwater and remain in the groundwater beneath the ranges, or migrate by groundwater flow to downgradient and possibly off site locations. Seeps may discharge groundwater to the surface. Chemicals in surface soil may enter surface water and sediment via runoff and erosion, and migrate downgradient through various drainage features or tributaries to South Branch of Cane Creek (at Range 18) or to Ingram Creek (all the other ranges

evaluated in this report). Chemicals in surface soil also may be taken up by vegetation that is subsequently browsed by deer, volatilize into the air, or enter the air in the form of dust. Contaminant sources and their release and transport mechanisms are displayed graphically on Figure 6-1.

6.1.1.2 Receptors, Exposure Media and Exposure Routes

The BBGR Ranges are currently unused. Future site use is projected to be almost entirely recreation except for the northeastern part of Range 18, which is slated for mixed business. Therefore, it is accurate to conclude, for the purposes of CSEM development, that the future site use of the BBGR Ranges is projected to be recreational or commercial.

Four receptor scenarios were chosen from the IWWP to evaluate potential human exposure to the environmental media at the BBGR Ranges. The receptor scenarios shown on Figure 6-1 are explained more fully in Table 6-1.

The recreational site user is included as the only plausible receptor for the site under the current site-use scenario. The groundskeeper is evaluated as the upper bound on risk for long-term commercial site use in the future. The construction worker is evaluated as the upper bound on risk for intense, short-term exposure. Presumably, construction activity would be required should the ranges be developed for any use other than passive recreation. The resident is included for the additional information provided by the theoretical upper-bound on long-term risk. Residential development is very unlikely for the BBGR Ranges.

Human receptors may come into contact with the chemicals in soil, groundwater, surface water and sediment. Another plausible exposure pathway involves consumption of game animals harvested from the ranges. The game is assumed to be venison because deer are the game species most widely hunted and most likely to contribute significantly to the diet. Game consumption is evaluated only for the recreational site user under the current site-use scenario. It is assumed that development for future site use would remove deer habitat or close the area to hunting. Consumption of sport-caught fish was also considered, but most of the drainage features are ephemeral, and Ingram Creek and the South Branch of Cane Creek in the vicinity of the BBGR Ranges are not sufficient to support sport fishing. Therefore, consumption of sport-caught fish is not evaluated. The receptor scenarios and relevant exposure routes are summarized on Figure 6-1 and Table 6-1.

6.1.2 Data Evaluation

Data evaluation consists of the following steps as described in the IWWP.

6.1.2.1 Establish Data Sets

The exposure unit (EU) concept must be introduced at this point. An EU is the area over which exposure is assumed to be random. The concept applies to soil, surface water and sediment. For example, a reasonable EU for soil for an on-site resident or a construction worker may be as small as 0.5 acre or possibly smaller (i.e., the size of a building lot, or a small area on which a construction project is performed). An EU for a groundskeeper or recreational site user, on the other hand, may be much greater, perhaps the size of a given range, because these receptors are expected to work or play over a larger area.

Each of the ranges is much larger than a reasonable EU for construction worker or residential exposure. Future site use, however, is not known in sufficient detail to identify realistic EUs. Therefore, the approach taken herein is to treat each of the seven ranges as a single EU, but to identify the sample locations where chemical of potential concern (COPC) concentrations exceed their cleanup levels. This reduces the likelihood that analytical data from lightly or uncontaminated areas or ranges would "dilute" the data from the more heavily contaminated areas or ranges. This approach may also be more useful to decision makers, because it facilitates focusing further action, if necessary, on the more heavily contaminated areas, returning greater benefit per unit cost.

The EU concept applies also to surface water bodies. It is assumed that the recreational site users that play in surface water would be randomly exposed to the drainage features or tributaries of creeks associated with any given range.

Separate data sets were established for each of the environmental media of interest – surface soil, subsurface soil, total soil, groundwater, surface water and sediment (see Section 6.1.2.3 for an explanation of total soil). More surface soil samples were taken than subsurface soil samples because most of the contamination from bullets and shot is expected to occur on or close to the surface. Also, most subsurface soil samples were restricted to 4 feet bgs because contamination is not expected to reach greater depths.

6.1.2.2 Select Site-Related Chemicals

Recently the procedure for comparing site and background metals data sets and for selecting site-related chemicals has changed, making better use of background and site data, applying more precise statistical comparisons, and employing geochemical analysis to help resolve the site-related question as necessary (Shaw, 2003b). Briefly, the new procedure consists of the following three-tiered process:

- ***Tier 1.*** The maximum detected concentration (MDC) of the metal is compared with the Background Screening Criterion (BSC), which is two-times the mean background concentration as estimated by SAIC (1998). If the MDC does not exceed the BSC, concentrations of that metal at all points within the geographic area represented by the data set are considered to be comparable to background and that metal is not evaluated further. Metals whose MDCs exceed their BSCs are carried to Tier 2.

- ***Tier 2.*** Two different statistical tests described in detail in Appendix I are used to determine whether the site concentrations are comparable to background concentrations. One test is the slippage test, which is used as a test of upper tails. The other is the Wilcoxon Rank Sum Test (WRST), which evaluates the likelihood that the site and background data sets are drawn from different populations. Metals that pass statistical testing are ordinarily not evaluated further, unless knowledge about site activities or the nature of the data set suggest that the metal could be site related. Such chemicals, as well as those that fail their statistical tests are carried to Tier 3.

- ***Tier 3.*** Geochemical analysis is based on the natural association between the metal of interest and one or more specific soil-forming minerals that concentrate that metal. The correlation of the metal of interest with a major element representing the abundance of the specific mineral that concentrates the trace element is evaluated. Metals that pass the geochemical evaluation are not evaluated further in the SRA. Metals that fail are selected as site-related chemicals and are subjected to COPC selection as described below.

Metals eliminated from COPC selection by the three-tiered system described above are termed "background metals" to differentiate them from "site-related metals," which are subjected to COPC selection as described below. Data for each medium were combined across all seven ranges to produce the largest and most robust data sets for the statistical (Tier 2) evaluation.

6.1.2.3 Identify Chemicals of Potential Concern

The MDCs of the site-related chemicals were compared with their respective SSSLs to identify COPCs, as explained in the IWWP. Cancer-based SSSLs correspond to an incremental lifetime cancer risk (ILCR) of 1E-6, and noncancer-based SSSLs correspond to a hazard index (HI) of 0.1 to provide additional protection for simultaneous exposure to multiple chemicals. The COPCs identified in this manner are receptor- and medium-specific risk-based concentrations.

An important clarification of the IT (2002a) IWWP involves how COPCs are identified and STCs are estimated for total soil, which is a hypothetical soil data set designed to address the potential for exposure to both surface and subsurface soil as a result of excavation and grading. A former commonly used approach involves combining the surface and subsurface soil data sets

into one data set for COPC identification. This approach implies that perfect blending of surface and subsurface soil over the entire site occurs during excavation and grading. More realistically, soil at the surface to which a receptor may be exposed is likely to be predominantly surface soil or predominately subsurface soil from one place to another within an EU. Therefore, the surface and subsurface soil data sets are not combined in this manner for the total soil evaluation. Instead, COPCs are selected separately for surface and subsurface soil. Any chemical selected as a COPC for either surface or subsurface soil is selected as a COPC for total soil. The STC selected for a given COPC in total soil is the larger of the STCs estimated for surface or subsurface soil. This revision is somewhat more health-protective than the earlier protocol, because data from the more lightly contaminated soil level do not "dilute" the data from the more heavily contaminated soil level. Furthermore, maintaining separate data sets for surface and subsurface soil facilitates identifying the depth at which contamination is present at unacceptable concentrations, which permits more efficient and cost-effective cleanup.

6.1.2.4 Estimate Source-Term Concentrations

Ideally, the STC of a given COPC in soil, sediment and surface water is a single value that reflects a conservative estimate of average over the area represented by the entire data set. The 95th upper confidence limit (UCL) on the arithmetic mean is usually estimated for this purpose. The first step in UCL estimation is application of the Shapiro-Wilks test to characterize the nature of the data distribution for each COPC. Distribution testing is limited to data sets with 5 or more samples. The Student's *t* test was used to estimate UCLs for COPCs with normal distributions, and the Chebyshev Inequality method was used to estimate UCLs for COPCs with lognormal and nonparametric distributions as explained in the IWWP and recommended by EPA (2002c). A minor change from previous SRAs for other sites is use of one-half the method detection limit (MDL) rather than one-half the reporting limit (RL) as a surrogate concentration for non-detects in the data sets of detected chemicals. Use of one-half the MDL yields a more reasonable UCL because the region between the MDL and RL is not double-counted in estimating a surrogate concentration.

The MDC is selected as the STC when it is lower than the UCL on the mean, or when a UCL is not estimated (e.g., if the data set contains fewer than five samples). The arithmetic mean is selected as the STC for lead as explained in Section 6.1.5.2.

EPA (2002d) recommends that the arithmetic mean of the most concentrated part of the plume should be selected as the STC for COPCs in groundwater. However, discrete plumes cannot be identified in most groundwater data sets, in which case the MDC is conservatively selected as the STC.

6.1.3 Exposure Assessment

The exposure assessment involves developing the receptor scenarios, estimating exposure-point concentrations, and estimating exposure route-specific intake or uptake rates for all receptors. The receptor scenarios included in this evaluation were identified in the CSEM and previously developed in the IWWP. The SSSLs incorporate the models and variable values that relate exposure-media concentrations to STCs, as well as the models and variable values for COPC intake and uptake, which obviates estimating exposure route-specific intake or uptake rates as a separate task in the risk assessment.

The EU concept was introduced in Section 6.1.2.1. Underestimating the size of a reasonable EU may impart a conservative bias to a risk assessment, largely because the size of the data set is reduced, which usually results in increased STC estimates. Overestimating the size of the EU, on the other hand, may impart a non-conservative bias to the risk assessment. This could occur because the analytical results from smaller, more heavily contaminated areas may be diluted by analytical data from larger, less heavily contaminated areas. In other words, the STC estimated from the entire data set may be less than the STC estimated from the data for a smaller area. It is generally impractical to divide the area of a site into reasonable EUs for the risk assessment, in part because the future site use is not that precisely known. The usual approach, therefore, is to include all of the soil data into a single data set, or, as was done herein, treat each of the seven ranges as a single EU. It is acknowledged, however, that each of the ranges constitutes an unrealistically large EU for receptors such as the construction worker and on-site resident. The implications to the risk assessment are discussed in the results sections below for each of the seven ranges.

6.1.4 Toxicity Assessment

The toxicity assessment involves identifying the nature of the adverse health effects and quantifying the dose-response relationship for the chemicals selected as COPCs. The dose-response relationship is generally quantified in the form of a cancer slope factor (SF) or unit risk factor (URF) for estimating cancer risk, and a threshold level called an RfD or reference concentration (RfC) for estimating the hazard of noncancer effects. These toxicity values are usually found in various EPA sources, or they may be developed from the primary literature or by analogy to more completely studied chemicals that behave similarly.

The toxicity values are incorporated into the SSSLs. They are not developed or presented separately herein; rather, they were located or developed in the IT (2000b) report. However, the toxicity values of all detected chemicals were checked to ensure they were current before the

SRA was initiated. As mentioned above, it was noted that the oral RfD for the ADNTs and RfD and cancer evaluation of the nitrotoluenes were recently revised; therefore, the SSSLs for these chemicals were updated accordingly.

6.1.5 Risk Characterization

6.1.5.1 Protocol and Policy: Chemicals Other than Lead

Up to this point, the terms "risk" or "risk assessment" have been used generically to refer to the association of adverse human health effects with exposure to hazardous chemicals. In the risk characterization discussion, however, it is often helpful to distinguish cancer risk and noncancer hazard, because the two are estimated separately and the estimates are not combined. Cancer risk is more precisely called ILCR to reflect the assumption that risk above background levels is directly and linearly correlated with cumulative dose averaged over the receptor's lifetime. Cancer is generally assumed to be a non-threshold phenomenon (i.e., exposure to a potential carcinogen in any amount, however small, leads to some measurable risk).

The non-threshold assumption requires the application of policy to interpretation of ILCR estimates. EPA (1990) considers a cancer risk of 1E-6 to be a point of departure below which risks are generally considered to be insignificant. Therefore, chemicals that yield ILCR estimates less than 1E-6 are not considered in ILCR sums across COPCs or media. An ILCR between 1E-6 and 1E-4 is considered to fall within the risk management range, and is interpreted herein as not posing an unacceptable cancer risk. The regulators may exercise their prerogative to order further action, but they are not required to do so, particularly if further action is impractical. ILCR estimates that exceed 1E-4 are generally considered to be unacceptable.

The potential for the occurrence of noncancer effects is termed noncancer hazard, and is measured as a HI. Noncancer effects are generally considered to be threshold phenomena; therefore, the HI is the ratio of the ambient concentration of the COPC to the threshold level at which adverse effects are not expected to occur. This elevates interpretation to a science rather than policy. By definition, an HI less than or equal to 1 is interpreted to mean that noncancer adverse health effects are unlikely. An HI greater than 1 is interpreted to mean that the threshold has been exceeded, raising concern that adverse effects might occur.

Generally the first step in characterizing noncancer hazard is to sum the HI values across chemicals and media to estimate a total HI for the receptor. Total HI values that do not exceed the threshold level of 1 indicate that adverse noncancer health effects are unlikely, and noncancer

effects are not investigated further. Total HI values that exceed the threshold level of 1 are subject to further refinement.

EPA (1989) acknowledges that adding all HI values may overestimate hazard, because the assumption of additivity is probably appropriate only for those chemicals that exert their toxicity by the same mechanism. However, sufficient data describing mechanisms of toxicity for use in predicting additivity with a high level of confidence are available for very few chemicals or chemical mixtures. In the absence of such data, EPA (1989) assumes that chemicals that act on the same target organ or that exert the same critical effect may do so by the same mechanism of toxicity. In other words, target organ or critical effect serves as a surrogate for mechanism of toxicity. When the sum of HI values for a receptor exceeds 1 due to the contributions of several chemicals, it may be appropriate to refine the HI estimate by segregating the chemicals by mechanism of toxicity (i.e., target organ or critical effect) and estimating separate HI values for each target organ.

As a practical matter, since human environmental exposures are likely to involve near- or sub-threshold dose rates, the target organ(s) chosen for a given chemical are those associated with dose rates at or near the critical effect. The target organ selected is also based on duration of exposure; i.e., the target organ for chronic or subchronic exposure to low or moderate dose rates is selected rather than the target organ for acute exposure to high dose rates.

Risk characterization involves combining the results of the exposure assessment and toxicity assessment to estimate cancer risk and noncancer hazard. As noted above, both the exposure assumptions and toxicity assessment are included in the SSSLs. Therefore, all that remains for risk estimation in the SRA is to divide the STC by the appropriate cancer or noncancer SSSL and adjust for the basis of the SSSL. Specifically, ILCR for a given receptor exposed to a given COPC in a given medium is estimated as:

$$ILCR = \frac{STC \bullet TR}{SSSLc} \qquad \text{Eq. 6.1}$$

where:

ILCR	=	incremental lifetime cancer risk (unitless, calculated)
STC	=	source-term concentration (mg/kg for solid media, mg/L for water)
TR	=	target cancer risk on which SSSLc is based (unitless, 1E-6)
SSSLc	=	site-specific screening level based on cancer risk (same units as STC).

Similarly, the HI for a given receptor exposed to a given COPC in a given medium is estimated as:

$$HI = \frac{STC \bullet THI}{SSSLnc} \qquad \text{Eq. 6.2}$$

where:

HI	=	hazard index (unitless, calculated)
STC	=	source-term concentration (mg/kg for solid media, mg/L for water)
THI	=	target HI on which SSSLnc is based (unitless, 1E-1)
SSSLnc	=	site-specific screening level based on noncancer hazard (same units as STC).

EPA (1989, 2002d) suggests rounding risk and hazard estimates to one significant figure to reflect the uncertainty inherent in their estimation. Numerical estimates in the tables and text are presented in scientific notation with one integer and two digits to the right of the decimal. This presentation facilitates document review and replication of the calculations. Rounding of risk estimates was done only in the text, and only when it was helpful for interpretation. For example, an HI of 1.49E+0 would be rounded to 1 and interpreted as equal to, but not exceeding, the threshold level of 1.

6.1.5.2 Protocol and Policy: Lead

Lead is not evaluated in the same manner as other chemicals for two reasons. One reason is that health effects appear to occur at levels so low as to have no practical threshold. The second reason is that lead is ubiquitous so that exposure simultaneously involves multiple routes and sources. This means that toxicity values (reference doses and cancer slope factors), which generally apply only to one exposure route and source, may not be sufficiently protective. Therefore, neither a cancer SF nor noncancer RfD are available or developed. Instead, the EPA (2003b) Integrated Exposure Uptake Biokinetic model (IEUBK) was developed to estimate blood lead levels of very young children (0 to 7 years old) exposed to lead from multiple sources. The unborn or very young child is the human receptor most sensitive to exposure to lead, and delayed or impaired neurological development is the critical endpoint. The EPA (1996c) adult blood lead model (ABLM) was developed to estimate blood lead levels of women of child-bearing age, assuming that the unborn child is the most sensitive receptor in an adult exposure context. The ABLM was designed primarily to address exposure to lead in soil, but can be applied to other media as well.

The most appropriate way to evaluate lead is to compare the *average* ambient concentration with receptor- and medium-specific cleanup levels. Average concentration is used as the STC instead of the more conservative UCL on the mean because the blood lead models contain modules that

statistically adjust for variation in exposure. Application of the EU concept, however, is important to ensure that data from relatively clean areas do not dilute the data from the more heavily contaminated areas.

EPA (1998) recommends using the IEUBK to back-calculate site-specific cleanup levels for lead in soil to which children might be regularly exposed. This recommendation was followed to develop the SSSL for residential soil of 400 mg/kg. A soil lead concentration of 350 mg/kg (calculations not shown) was found to comply with EPA (1998) standards. The concentration of 350 mg/kg, rounded to one significant figure to reflect the uncertainty about the estimation, is 400 mg/kg. The SSSL of 400 mg/kg is accepted as the cleanup level for lead in soil for residential exposure scenarios that may include children. The net effect of this exercise is to elevate the status of the 400 mg/kg soil lead concentration from an SSSL to a cleanup level. The new designation of 400 mg/kg in residential soil as a cleanup level is consistent with the recent EPA (1998, 2001) evaluation that describes 400 mg/kg as a "level of public health concern (that) may pose a health risk to children through elevated blood lead levels." The ABLM was used in a similar manner to estimate the SSSL of 880 mg/kg for groundskeeper and construction worker exposure to soil. The SSSL of 880 mg/kg is adopted as the cleanup level for soil for these receptors.

The SSSL for residential soil of 400 mg/kg was adopted as the SSSL for lead in soil for the recreational site user. This approach is very conservative because recreational site use is expected to result in far less intense exposure to soil than residential use, so a more realistic cleanup level for lead in soil was developed for this scenario. Unsupervised young children are unlikely to roam repeatedly and predictably over the BBGR Ranges used for passive recreation; therefore, a woman of child-bearing age was selected as the most sensitive receptor for lead in soil. The recreational site user is visualized as a woman – perhaps an avid wildlife/birdwatcher-hiker – who visits the site regularly for two days on each weekend of the year. (The site-specific exposure parameters for the recreational site user scenario were developed and defended in the IWWP). A cleanup level for lead in soil of 7.60E+3 mg/kg was derived for areas designated for passive recreation by applying the site-specific exposure variables developed in the IWWP to the EPA (1996c) ABLM. The details of the calculation are presented in Appendix J.

The SSSL for residential soil of 400 mg/kg was adopted as the SSSL for lead in soil for the indirect exposure venison consumption pathway. The intention of this extremely conservative approach is to cause lead to be selected as a COPC if its MDC in soil exceeds its residential screening level. Selection as a COPC generally means that further investigation is in order. Further investigation consists of clarifying the potential for lead to participate significantly to

human health through food-chain pathways. The biotransfer of lead has been studied extensively. Although minimal biotransfer occurs, biomagnification does not (ATSDR, 1999b). Furthermore, lead in terrestrial animals (such as deer and other game animals) partitions largely to the bones, which are generally considered to be inedible. It is concluded that lead in soil is unlikely to participate significantly in terrestrial food-chain pathways and further evaluation is not required.

The residential soil SSSL of 400 mg/kg was adopted as the SSSL for lead in sediment for the recreational site user. This approach is very conservative because exposure to sediment in a recreational site-use scenario would be far less than exposure to soil in a residential scenario. A somewhat more realistic approach, adopted herein, is to adopt the cleanup level for lead in soil of 7.60E+3 mg/kg for the recreational site user described in the previous paragraph as the cleanup level for lead in sediment for the recreational site user.

The EPA (2004) action level for lead in tap water of 1.50E-2 mg/L was adopted as the SSSL for lead in groundwater for residential use. The same value was also adopted as the SSSL for lead in surface water for recreational site-user exposure, to cover for the plausible but infrequent intentional consumption of surface water (1 liter [L]/day) by a thirsty individual playing far from a potable water source. A more realistic approach, however, would be to adjust the SSSL by the differences in drinking water consumption and exposure frequency between a resident and a recreational site user. The adult resident is assumed to consume 2 L of drinking water from the site on 350 days/year; the recreational site user is assumed to consume 1 L of surface water on the site on 104 days/year. In other words, exposure is expected to be 0.15 times as great for the recreational site user as for the on-site resident. Therefore, the EPA (2004) action level for lead of 1.5E-2 mg/L is divided by 0.15 to yield a cleanup level of 1.0E-1 mg/L for lead in surface water for recreational site user exposure.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. EPA (2003c) developed an acute criterion for lead in soil of 6.50E-3 mg/kg, intended to be protective of a two-year-old child ingesting 10 grams of soil as a one-time event. The acute criterion of 6.50E-3 mg/kg should be applied as a not-to-exceed value for lead in residential soil.

6.1.6 Uncertainty Evaluation

This section discusses some of the more significant uncertainties inherent in the risk assessment for the BBGR Ranges as a whole that may affect the results or their interpretation. Uncertainty is a factor in each step of the SRA. Uncertainties associated with the earlier stages of the SRA become magnified when they are concatenated with uncertainties in the latter stages of the

process. It is not possible to eliminate all uncertainty; however, a recognition of the more significant uncertainties is fundamental to the understanding and reasonable use of risk assessment results. Significant sources of uncertainty specific to a given range are addressed in the discussion for that range.

Generally, risk assessments carry two types of uncertainty. Measurement uncertainty refers to the usual variance that accompanies the scientific measurements, e.g., instrument uncertainty (accuracy and precision) associated with contaminant concentrations. A different kind of uncertainty stems from data gaps (i.e., additional information needed to complete the database for the assessment). Often the data gap is significant, such as the absence of information on the effects of human exposure to a chemical (e.g., 4-ADNT) or the frequency with which a certain receptor (e.g., recreational site user) may visit the site.

Adequacy of the sampling program to identify the areas of greatest contamination is a source of uncertainty in any risk assessment. However, the former use and layout of the ranges under investigation is fairly well known, facilitating identification of areas most likely to be contaminated. Also, investigation of the BBGR Ranges has undergone several iterations, including screening to identify areas most likely to yield significant contaminant concentrations. Therefore, uncertainty about the adequacy of the sampling program is considered to be minimal for the BBGR Ranges. Exceptions are noted in the discussions of uncertainty for the individual ranges.

Variations in sampling and laboratory analytical procedures introduce uncertainty; it is unclear whether this imparts a positive or negative bias to the risk assessment. Along with this is the method detection limit (MDL) achieved by the laboratory. For example, antimony was detected in 3 of 22 surface soil samples at Range 23. The MDL for antimony ranged from 5.66 to 6.77 mg/kg. Therefore, 85 percent of the samples were nondetects, for which one-half the MDL was used as a surrogate concentration. However, the value of one-half the MDL exceeds the background screening criterion of 1.99 mg/kg, which suggests that its use as a surrogate concentration for nondetects artificially increases the mean concentration of antimony as a site-related metal in surface soil.

Another source of uncertainty is the manner in which the data sets for soil are compiled for the various receptors. It is reasonable to assume that a groundskeeper or recreational site user may be exposed randomly over an entire parcel or range in the course of his activities, in which case estimating a single STC for a given COPC from the entire data set for that range is appropriate. However, a construction project or a residential site may be restricted to an EU as small as a

quarter acre, in which case estimating a single STC from the entire data set for the range may underestimate risk. Inability at this time to identify actual future EUs adds to the uncertainty. This source of uncertainty was addressed by identifying the sample locations where site-related concentrations exceed their cleanup levels.

Uncertainty arises by excluding background metals from the quantitative risk assessment. A non-conservative bias could have been imparted to the results and interpretation if these chemicals had been excluded in error (i.e., if in fact their ambient concentrations reflect site-related releases rather than background conditions). The likelihood of error, however, is less for weapons ranges than for other kinds of sites (e.g., landfills) because the activities that took place at the ranges are well known and the metals associated with the discharge of weapons are generally limited to arsenic, antimony, copper, lead, nickel and zinc (EPA, 2003d).

As noted above, the exclusion of background metals is performed as a three-tiered process. Tier 1 – comparison of the MDC with the BSC – is generally considered to be sufficiently conservative so that the uncertainty associated with metals excluded at this tier is minimal. Also, only metals present at concentrations greater than their SSSL are considered to potentially contribute significant risk. Therefore, only metals excluded at Tier 2 or Tier 3, *and* that have MDCs that exceed their SSSLs are considered to impart significant uncertainty. These metals are addressed in the uncertainty discussions for the individual ranges.

Iron was designated a background metal at several ranges. The toxicity evaluation for iron, however, is controversial and is probably overly conservative for environmental exposures in the United States, and generally is not used in risk assessments (see toxicity profile for iron in IT, 2000b). Therefore, the exclusion of iron as a COPC in total soil is judged to contribute minimal uncertainty to the SRA.

Probably the most important source of uncertainty regarding the SRA for the BBGR Ranges is the presence of lead fragments or nuggets from bullets and shot on and near the surface of the ground of several ranges. Depending on their size, lead fragments are unlikely to be incidentally ingested along with soil or sediment (EPA, 2000a). Also, the bioavailability of ingested lead fragments is expected to be low in humans (ATSDR, 1999b). Empirical evidence, however, suggests that lead fragments in soil degrade, albeit very slowly, releasing more soluble and bioavailable forms of lead to the soil (EPA, 2000a, 2000b, 2003d). This has the potential to increase the risk from exposure to lead in soil over current levels; however, it is not possible to predict the rate or the concentrations of lead in soil that may occur.

The purpose of the uncertainty discussion is not to disparage the results of the risk assessment, but to provide perspective to risk managers who may use this evaluation to make decisions regarding future use or management of the site.

6.2 Site-Specific Results

6.2.1 Range 18

6.2.1.1 Data Sets

The surface soil data set consists of 62 samples taken from 62 locations across Range 18 (Table 6-2). A total of 55 were analyzed for lead, 16 were analyzed for other metals, 18 were analyzed for nitroaromatic compounds, and 11 were analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of 23 samples taken from 23 locations across Range 18 (Table 6-3). A total of 16 were analyzed for lead, 7 were analyzed for other metals, 12 were analyzed for nitroaromatic compounds, and 5 were analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The surface water (Table 6-4) and sediment (Table 6-5) data sets consist of single co-located samples analyzed for metals, chlorinated herbicides, organophosphate pesticides, organochlorine pesticides and PCBs, SVOCs and VOCs.

The groundwater data set consists of three samples taken from three monitoring wells (Table 6-6). All samples were analyzed for metals and nitroaromatic compounds; one sample was analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

6.2.1.2 Chemicals of Potential Concern

Chemicals detected in Range 18 soil include metals, infrequent detections of a chlorinated herbicide, a single detection of a nitroaromatic compound, low concentrations of VOCs, and several organochlorine pesticides and SVOCs (predominantly PAHs) – the latter two classes of compounds limited to surface soil (Tables 6-7 and 6-8). Site-related metals in surface soil include those associated with bullets (antimony, arsenic, copper, lead, zinc) as well as beryllium and nickel (Table 6-7). No explanation is apparent for beryllium and nickel. Copper, lead, and zinc were selected as site-related metals in subsurface soil (Table 6-8). All the organic chemicals were selected as site-related.

COPCs in surface soil were limited to antimony, arsenic, copper, lead, 2,4-dinitrotoluene (2,4-DNT) and three PAHs (Table 6-9). The PAHs and 2,4-DNT were selected as COPCs only for the on-site resident. Chemical concentrations in subsurface soil were sufficiently low so that none were selected as COPCs (Table 6-10). STCs selected for total soil reflect those of the COPCs identified in surface soil (Table 6-11).

Chemicals detected in surface water were limited to metals (Table 6-12). None appear to be present at concentrations greater than background; therefore none were identified as site-related chemicals.

Chemicals detected in sediment include metals and two VOCs (Table 6-13). The metals appear to be present at concentrations comparable to background; site-related chemicals were limited to the two VOCs. Neither were selected as COPCs (Table 6-14).

Chemicals detected in groundwater are limited to metals and low concentrations of two organochlorine pesticides (Table 6-15). The metals appear to be present at concentrations comparable to background; therefore metals were not identified as site-related chemicals. COPCs in groundwater are limited to gamma-lindane (only for the on-site resident) (Table 6-16). The MDC was selected as the STC because no discernable plume was apparent.

6.2.1.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-17. Documentation is provided in the form of receptor- and medium-specific tables (Tables 6-18 through 6-24) that detail the risk from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil, surface water, and sediment. In addition he is assumed to consume venison, representing an indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user (Tables 6-18, 6-19) because no chemicals in the relevant media except lead in surface soil (Table 6-9) were selected as COPCs. It is concluded that exposure to COPCs other than lead in surface soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-17) because no chemicals in the relevant media except lead in total soil (Table 6-11) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because Range 18 is not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are expected to be sporadic and of short duration because they are not related to the groundskeeper's regular duties. Therefore, exposure to surface water and sediment is not quantified for this receptor.

COPCs were not identified in groundwater for the groundskeeper scenario (Table 6-16). The total ILCR of 1.07E-5 (Table 6-17) is due almost entirely to arsenic in total soil (Table 6-21). The total HI of 2.39E-1 (Table 6-17) is due to antimony and copper in total soil (Table 6-21). The total HI does not exceed the threshold level of 1. It is concluded that exposure to COPCs other than lead in total soil is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are expected to be sporadic and of short duration because they are not related to the construction worker's regular duties as explained for the groundskeeper. Therefore, exposure to surface water and sediment is not quantified.

COPCs were not identified in groundwater for the construction worker (Table 6-16). The total ILCR of 8.06E-7 (Table 6-17) is due entirely to arsenic in total soil (Table 6-22). The total HI of 5.88E-1 (Table 6-17) is due to antimony, arsenic and copper in total soil (Table 6-22). The HI does not exceed the threshold level of 1. It is concluded that exposure to COPCs other than lead

in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker, as long as exposure occurs randomly across Range 18.

Construction projects, however, may involve much smaller EUs than represented by the data set for Range 18. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for all of Range 18. The EU issue for the construction worker is discussed in detail in Section 6.2.1.4 on lead.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water, and sediment under the future site-use scenario. The assumptions for exposure to surface water and sediment are identical to those for the recreational site user; therefore, the risk estimates for recreational site user exposure to these media, had they been calculated, would be applied also to the on-site resident.

The total ILCR summed across all relevant media of 4.49E-5 (Table 6-17) is due largely to arsenic in total soil, with smaller contributions from benzo(a)pyrene in total soil (Table 6-23), and gamma-BHC in groundwater (Table 6-24). The total HI summed across all relevant media of 3.81E+0 (Table 6-17), due to antimony, arsenic and copper in total soil (Table 6-23) exceeds the threshold level of 1.

The total HI of 3.81E+0 was refined by segregating the contribution to each target organ as described in Section 6.1.5.1 (Table 6-25). Several target organs have total HI values rounded to one significant figure exceeding the threshold of 1, due to copper in surface soil. It is concluded that exposure to COPCs other than lead in total soil and groundwater does not pose an unacceptable cancer risk, but raises concern for adverse noncancer health effects for the future on-site resident. Exposure to COPCs other than lead in sediment and surface water poses no significant cancer risk or concern for adverse noncancer effects.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of Range 18. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in the following section on lead.

6.2.1.4 Risk Characterization: Lead

Ideally, the evaluation of lead would be accomplished by comparing the lead cleanup levels with the average concentration of lead over an appropriate EU. However, data are not available with

which to identify future EUs. Therefore, the foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it was determined to be site related. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Included in the discussion for the construction worker and on-site resident is the EU concept applied to the COPCs evaluated in the previous section. Therefore, the comparison tables for these receptors may include all relevant COPCs in the media of interest. The cleanup levels for COPCs other than lead are receptor-specific and range-specific and are described in the appropriate subsections below. Lead is the most prevalent COPC in the environmental media at the BBGR Ranges. Therefore, the comparison tables list lead as the first COPC, with concentrations ranked from highest to lowest. Cleanup levels of COPCs other than lead should be applied as the maximum *conservative estimate of average* concentration over the EU.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil, surface water and sediment under the current site-use scenario. The entire data set for each medium was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 18 area. Lead was identified as a COPC for the recreational site user in surface soil (Table 6-9). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg.

The average concentration of lead in surface soil of 1.79E+3 mg/kg (Table 6-7) is below the cleanup level for soil of 7.60E+3 mg/kg. It should be noted, however, that the concentrations of lead in 6 surface soil samples exceed the cleanup level (Table 6-26). It is concluded that lead in surface soil at Range 18 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. The entire data set for each medium was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in total soil (Table 6-11), but not in subsurface soil (Table 6-10). The identification of surface soil samples with lead concentrations that exceed the cleanup level was discussed above. It is concluded that lead in total soil at Range 18 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil and groundwater under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 18 area. Lead was identified as a COPC for the groundskeeper in surface soil (Table 6-9). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentration of lead in surface soil (1.79E+3 mg/kg, Table 6-7) exceeds the cleanup level of 8.80E+2 mg/kg; the average concentration of lead in subsurface soil (2.86E+1 mg/kg, Table 6-8) does not exceed the cleanup level. Specifically, the concentrations of lead in 13 surface soil samples exceed the cleanup level (Table 6-27). It is concluded that lead in surface soil raises concern for adverse health effects for the future groundskeeper at Range 18.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater under the future site-use scenario. COPCs for the construction worker included lead, antimony, arsenic and copper in surface soil (Table 6-9).

The cleanup level for lead in soil for the construction worker is 8.80E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in surface soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. The cleanup level can be estimated from the SSSL as follows:

$$\left(\frac{SSSL}{THI_{SSSL}}\right)=\left(\frac{CL}{THI_{CL}}\right) \qquad \text{Eq. 6.3}$$

where:

CL	=	cleanup level (mg/kg, calculated)
SSSL	=	site-specific screening level (mg/kg)
THI_{SSSL}	=	target HI on which SSSL is based (0.1, unitless)
THI_{CL}	=	target HI for the cleanup level (unitless).

Application of a value for THI_{CL} of 1.49E+0 and the SSSL of 1.99E+1 mg/kg for construction worker exposure to antimony in soil to Equation 6.3 yields a cleanup level for antimony of 2.96E+2 mg/kg (Table 6-28).

Arsenic was selected as a COPC in surface soil both for cancer risk and noncancer hazard. Arsenic in surface soil was the only contributor to the total ILCR of 8.06E-7. An ILCR of

1.23E-6 can be estimated for arsenic by applying the MDC of 2.43E+1 mg/kg to Equation 6.1. The cancer risk of 1.23E-6 does not exceed the risk management range. Applying the MDC of arsenic to Equation 6.1 is a worst-case evaluation, which shows that cancer risk will not exceed the risk management range regardless of how EUs are established. It also permits establishing a cleanup level for arsenic based on noncancer effects alone. Since arsenic shares no target organ with the other COPCs in soil, the cleanup level can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. Application of a value for THI_{CL} of 1.49E+0 and the noncancer SSSL of 1.53E+1 mg/kg for construction worker exposure to arsenic in soil to Equation 6.3 yields a cleanup level for arsenic of 2.28E+2 mg/kg (Table 6-28).

Copper shares no target organ with the other COPCs in surface soil, so the cleanup level for copper can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. Application of a value for THI_{CL} of 1.49E+0 and the SSSL of 2.04E+3 mg/kg for construction worker exposure to copper in soil to Equation 6.3 yields a cleanup level for copper of 3.04E+4 mg/kg (Table 6-28).

As noted above, an EU for the construction worker would probably be much less than the entire area of Range 18. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-28). Concentrations of lead in 13 surface soil samples exceed the cleanup level. However, no concentrations of antimony, arsenic or copper exceed their cleanup levels. It is concluded that lead in surface soil raises concern for noncancer health effects for the construction worker exposed to total soil and groundwater at Range 18.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water, and sediment under the future site-use scenario. COPCs for the on-site resident include lead, antimony, arsenic, copper, 2,4-DNT, benzo(a)pyrene, benzo(b)fluoranthene, and dibenz(a,h)anthracene in surface soil (Table 6-9), and gamma-BHC in groundwater (Table 6-16). 2,4-DNT, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene and gamma-BHC were selected as COPCs based only on cancer risk. Arsenic was selected as a COPC based on cancer risk and noncancer effects. The other metals were selected as COPCs based only on noncancer effects.

The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in soil, so the cleanup level for antimony

can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-29) was estimated by applying the SSSL of 3.11E+0 mg/kg and a THI_{CL} of 1.49 to Equation 6.3, as was done for the construction worker.

Several chemicals in surface soil and one in groundwater were selected as COPCs based on cancer risk. A worst-case estimation of total cancer risk can be calculated by applying the MDCs and SSSLs of these chemicals to Equation 6.1. These calculations are presented below:

Chemical	MDC	SSSL	ILCR
Surface Soil (Table 6-9)			
Arsenic	2.43E+1	4.26E-1	5.70E-5
2,4-Dinitrotoluene	1.60E+0	9.27E-1	1.73E-6
Benzo(a)pyrene	4.40E-1	8.51E-2	5.17E-6
Benzo(b)fluoranthene	1.10E+0	8.51E-1	1.29E-6
Dibenzo(a,h)anthracene	1.50E-1	8.61E-2	1.74E-6
Groundwater (Table 6-16)			
gamma-BHC	1.30E-4	5.03E-5	2.58E-6
Total ILCR			6.96E-5

MDC = maximum detected concentration (mg/kg in soil, mg/L in groundwater); SSSL = site-specific screening level (same units as MDC); ILCR = incremental lifetime cancer risk.

The total cancer risk of 6.96E-5 does not exceed the risk management range. This shows that cancer risk will not exceed the risk management range regardless of how EUs are established. It also permits establishing a cleanup level for arsenic based on noncancer effects alone. Since arsenic shares no target organ with the other COPCs in soil, the cleanup level can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. Application of a value for THI_{CL} of 1.49E+0 and the noncancer SSSL of 2.34E+0 mg/kg for residential exposure to arsenic in soil to Equation 6.3 yields a cleanup level for arsenic of 3.49E+1 mg/kg (Table 6-29).

Copper shares no target organ with the other COPCs in soil, so the cleanup level for copper can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for copper of 4.67E+3 mg/kg (Table 6-29) was estimated by applying the SSSL of 3.13E+2 mg/kg and a THI_{CL} of 1.49 to Equation 6.3.

As noted above, an EU for the on-site resident would probably be much less than the entire area of Range 18. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-29). Concentrations of lead in 17 surface soil samples exceed the cleanup level. Also, the concentrations of antimony in

three surface soil samples and copper in one surface soil sample exceed their cleanup levels. However, no concentrations of arsenic exceed their cleanup levels.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. It should be noted that the lead concentrations in six surface soil samples exceed the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-29).

It is concluded that lead, antimony, and copper in surface soil raise concern for noncancer health effects in the on-site resident at Range 18.

6.2.1.5 Uncertainty Analysis

Probably the most important source of uncertainty regarding Range 18 pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. Several metals in soil were excluded from COPC selection based on their Tier 2 or Tier 3 analysis (Tables 6-7 and 6-8). None of these metals, however, would be subject to evaluation for cancer risk (i.e., none have cancer-based SSSLs). Those with MDCs that exceed their residential noncancer SSSLs are limited to aluminum, chromium, manganese (subsurface soil only) and iron. In all cases except iron the exceedance was less than a factor of 10, indicating that HI values calculated for these chemicals would fall below the threshold level of 1. As noted above, the oral RfD for iron is controversial and most likely unnecessarily conservative, and its selection as a background chemical is judged not to impart significant uncertainty to the assessment. Some level of uncertainty remains about these chemicals, however, because they could contribute to target organ HI values summed across the chemicals that exceed the threshold of 1. This issue is addressed below. It should be noted that none of these metal MDCs exceed their recreational site user SSSLs; therefore, excluding background metals in soil imparts no uncertainty to the risk estimates for the recreational site user.

Surface Water. No metals in surface water excluded at the Tier 2 or Tier 3 stage had an MDC that exceeded its residential SSSL (Table 6-12). Excluding background metals in surface water imparts no uncertainty to the risk estimates for the recreational site user or resident.

Sediment. No background metals in sediment excluded at the Tier 2 or Tier 3 stage had MDCs exceeding their recreational site user SSSLs (Table 6-13). The exclusion of background metals is judged to impart no significant uncertainty to the SRA.

Groundwater. Barium and manganese were the only background metals in groundwater excluded at the Tier 2 or Tier 3 stage with MDCs that exceeded their residential SSSLs (Table 6-15). The barium exceedance was less than a factor of 10. The MDC of manganese exceeded its SSSL by about 14-fold. Manganese, however, is not a component of bullets and no rationale is apparent for its presence as a site-related chemical. The designation of manganese as a background chemical in groundwater is judged to impart little uncertainty to the Range 18 assessment.

Background Metals in All Media. As discussed above, HI values for chemicals that share a target organ are generally summed. The nervous system is a common target organ for aluminum and manganese (see Toxicity Profiles in IT [2000b]), both of which were judged to be present in soil and groundwater at concentrations comparable to background. Concentrations of both metals were higher in subsurface soil (Table 6-8) than in surface soil (Table 6-7); therefore, only their concentrations in subsurface soil is considered further. The MDC of aluminum in groundwater does not exceed the BSC, which is considered a very conservative basis for background comparison. Therefore, aluminum in groundwater is not considered further. HI values of 3.95E-1, 6.69E-1 and 1.40E+0 could be estimated for residential exposure to aluminum and manganese in subsurface soil, and manganese in groundwater, respectively (calculations not shown). The sum, 2.47E+0, exceeds the threshold value of 1. Aluminum and manganese, however, are not ordinarily associated with shooting ranges. The designation of aluminum and manganese in soil and manganese in groundwater as background metals at Range 18 is judged to impart a small level of uncertainty to the assessment.

6.2.1.6 Conclusions, Range 18

Range 18 is expected to be used for active recreation and mixed business (EDAW, 1997), for which the recreational site user, groundskeeper, and construction worker are the relevant receptor scenarios. The recreational site user is assumed to be exposed randomly to surface soil or total soil, surface water and sediment over the entire area of Range 18. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 18 area, and to groundwater developed as a potable source – an unlikely possibility.

The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Range 18 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the construction worker must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, and to surface water and sediment in the same manner as the recreational site user. Home sites are expected to involve exposure to soil in an area much smaller than the entire range. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat. Exposure to surface water and sediment, however, is expected to occur randomly.

COPCs at Range 18 include antimony, arsenic, copper, lead, 2,4-DNT, and three PAHs in soil, and one organochlorine pesticide in groundwater. COPC concentrations are sufficiently low so that the recreational site user is not expected to experience adverse health effects from exposure to the site in its current condition. Concentrations of lead in surface soil, however, are sufficiently high to raise concern for adverse health effects for the groundskeeper and construction worker.

The on-site resident, exposed to an area much smaller than the entirety of Range 18, may experience adverse noncancer health effects from lead, antimony or copper in surface soil.

It must be understood that a major source of uncertainty regarding Range 18 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil, although the potential for this is unclear.

6.2.2 Range 20

6.2.2.1 Data Sets

The surface soil data set consists of 15 samples taken from 15 locations across Range 20 (Table 6-30). All were analyzed for lead, four were analyzed for other metals, and three were analyzed for nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of three samples taken from three locations (Table 6-31). All samples were analyzed for lead; one was analyzed for other metals, nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The surface water (Table 6-32) and sediment (Table 6-33) data sets consist of three co-located samples analyzed for metals, nitroaromatic compounds, chlorinated herbicides, organophosphate pesticides, organochlorine pesticides and PCBs, SVOCs, and VOCs.

It is noteworthy that groundwater samples were not taken at Range 20. The main impact zone is on a steep hillside on the northeast corner of the parcel were it would be very difficult to drill and install monitoring wells. It was thought that groundwater contamination from Range 20 would be captured in the four monitoring wells located on Range 26, which is immediately downgradient of Range 20. Therefore, receptors scenarios for Range 20 do not include exposure to groundwater.

6.2.2.2 Chemicals of Potential Concern

Chemicals detected in Range 20 soil include metals, a single occurrence of a chlorinated herbicide and an organochlorine pesticide (surface soil only), and low levels of acetone (Tables 6-34 and 6-35). Site-related metals in soil include those associated with bullets (antimony, copper, and lead). All the organic chemicals were selected as site-related.

COPCs in soil were limited to antimony and lead (Tables 6-36 and 6-37). Antimony was selected as a COPC in both surface and subsurface soil only for the on-site resident. The total soil STC for lead was the higher STC estimated for surface soil; the total soil STC for antimony was the higher STC estimated for subsurface soil (Table 6-38).

Chemicals detected in surface water were limited to metals (Table 6-39). None appear to be present at concentrations greater than background; therefore, none were identified as site-related chemicals.

Chemicals detected in sediment include metals, one chlorinated herbicide, three organochlorine pesticides, and one VOC (Table 6-40). The metals appear to be present at concentrations comparable to background; site-related chemicals were limited to the organic compounds. No chemicals in sediment were selected as COPCs (Table 6-41).

6.2.2.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-42. Documentation is provided in the form of receptor- and medium-specific tables (Tables 6-43 through 6-48) that detail the risk from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil, surface water, and sediment. In addition, he is assumed to consume venison, representing an indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user (Tables 6-43 and 6-44) because no chemicals in the relevant media except lead in surface soil (Table 6-36) were selected as COPCs. It is concluded that exposure to COPCs other than lead in surface soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-45) because no chemicals in the relevant media except lead in total soil (Table 6-38) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because Range 20 is not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are expected to be sporadic and of short duration because they are not related to the groundskeeper's regular duties. Therefore, exposure to surface water and sediment is not quantified for this receptor.

COPCs other than lead were not identified in total soil for the groundskeeper scenario (Table 6-38). Therefore, neither HI nor ILCR were estimated for groundskeeper exposure to total soil (Table 6-46). It is concluded that exposure to COPCs other than lead in total soil is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are expected to be sporadic and of short duration because they are

not related to the construction worker's regular duties as explained for the groundskeeper. Therefore, exposure to surface water and sediment is not quantified.

COPCs based on cancer risk were not identified in any media for the construction worker; therefore, ILCR was not estimated. The total HI summed across all relevant media of 1.37E-1 (Table 6-42), due to antimony in total soil (Table 6-47) does not exceed the threshold level of 1.

Construction projects, however, may involve much smaller EUs than represented by the data set for Range 20. This could result in underestimating potential risks, because the STC of a given COPC for a small area may be greater than the STC for all of Range 20. The EU issue for the construction worker is discussed in detail in Section 6.2.2.4 on lead.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. The assumptions for exposure to surface water and sediment are identical to those for the recreational site user; therefore, the risk estimates for recreational site user exposure to these media, had they been calculated, would be applied also to the on-site resident.

COPCs based on cancer risk were not identified in any media for the on-site resident; therefore, ILCR was not estimated. The total HI summed across all relevant media of 8.75E-1 (Table 6-42), due to antimony in total soil (Table 6-48) does not exceed the threshold level of 1.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of Range 20. This could result in underestimating potential risks, because the STC of a given COPC for a small area may be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in Section 6.2.2.4 on lead.

6.2.2.4 Risk Characterization: Lead

The foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it was determined to be site related, as explained for Range 18. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil, surface water and sediment under the current site-use scenario. The entire data set for each medium was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 20 area. Lead was identified as a COPC for the recreational site user in surface and subsurface soil (Tables 6-36 and 6-37). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg.

Both the average concentration (1.42E+3 mg/kg) and MDC (4.47E+3 mg/kg) of lead in surface soil (Table 6-34) are below the cleanup level of 7.60E+3 mg/kg. It is concluded that lead in surface soil at Range 20 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. The entire data set for each medium was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in both surface and subsurface soil (Tables 6-36 and 6-37). As noted above, both the average concentration and MDC of lead in surface soil fall below the cleanup level. Also, both the average concentration (1.08E+3 mg/kg) and MDC (2.63E+3 mg/kg) of lead in subsurface soil (Table 6-35) are below the cleanup level. It is concluded that lead in surface and subsurface soil at Range 20 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 20 area. Lead was identified as a COPC for the groundskeeper in surface soil (Table 6-36) and subsurface soil (Table 6-37). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentration of lead in surface soil (1.42E+3 mg/kg, Table 6-34) and the average concentration of lead in subsurface soil ((1.08E+3 mg/kg, Table 6-35) both exceed the cleanup level of 8.80E+2 mg/kg. Specifically, the concentrations of lead in eight surface soil samples and one subsurface soil sample exceed the cleanup level (Table 6-49). It is concluded that lead in surface and subsurface soil raises concern for adverse health effects for the future groundskeeper at Range 20.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil under the future site-use scenario. COPCs for the construction worker were limited to lead in surface soil (Table 6-36), and antimony and lead in subsurface soil (Table 6-37). The cleanup level for lead in soil for the construction worker is 8.80E+2 mg/kg as described above. The cleanup level for antimony in soil of 2.96E+2 mg/kg is the same as that developed for Range 18 (see Section 6.2.1.4).

An EU for the construction worker would probably be much less than the entire area of Range 20. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-50). Concentrations of lead in eight surface soil samples and one subsurface soil sample exceed the cleanup level. However, no concentrations of antimony exceed the cleanup level. It is concluded that lead in surface and subsurface soil raises concern for noncancer health effects in the construction worker exposed to total soil at Range 20.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. COPCs for the on-site resident are limited to antimony and lead in surface soil (Table 6-36) and subsurface soil (Table 6-37). The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-51) was estimated as described for Range 18.

As noted above, an EU for the on-site resident would probably be much less than the entire area of Range 20. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-51). Concentrations of lead in eight surface soil samples and two subsurface soil samples exceed the cleanup level. No concentrations of antimony exceed the cleanup level.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. Lead concentrations in surface and subsurface soil samples do not exceed the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-51).

It is concluded that lead in surface and subsurface soil raises concern for noncancer health effects for the on-site resident at Range 20.

6.2.2.5 Uncertainty Analysis

A possible source of uncertainty regarding Range 20 arises from the lack of groundwater data. However, groundwater data are available from Range 26, which is immediately downgradient. As discussed in Section 6.2.4.2, no chemicals in Range 26 groundwater were selected as COPCs. Therefore, it is concluded that the lack of groundwater data specifically for Range 20 does not represent a significant source of uncertainty.

Another source of uncertainty pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. A few metals in surface or subsurface soil were excluded from COPC selection based on their Tier 3 analysis (barium, beryllium, zinc) (Tables 6-34 and 6-35). None of these metals, however, have MDCs that exceed their residential noncancer SSSLs. Therefore, it is concluded that excluding background metals in soil imparts no uncertainty to the risk estimates for any receptor scenario.

Surface Water. No metals in surface water were excluded at the Tier 2 or Tier 3 stage of the background evaluation (Table 6-39). Excluding background metals in surface water imparts no uncertainty to the risk estimates for the recreational site user or resident.

Sediment. Several metals in sediment were excluded from COPC selection based on their Tier 3 analysis (Table 6-40). However, none had MDCs exceeding their recreational site user SSSLs. The exclusion of background metals is judged to impart no significant uncertainty to the SRA.

6.2.2.6 Conclusions, Range 20

Range 20 is expected to be used for recreation (EDAW, 1997), for which the recreational site user is the relevant receptor scenario. The recreational site user is assumed to be exposed randomly to surface soil or total soil, surface water and sediment over the entire area of Range 20. The groundskeeper and construction worker were included in the event that the site might be developed for commercial use at some time in the future. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 20 area. The construction worker is assumed to be exposed to total soil. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Range 20 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that exceed the cleanup level must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil, and to surface water and sediment in the same manner as the recreational site user. Home sites are expected to involve exposure to soil in an area much smaller than the entire range. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat. Exposure to surface water and sediment, however, is expected to occur randomly.

COPCs at Range 20 are limited to antimony and lead in surface and subsurface soil. Concentrations of the COPCs are sufficiently low that they raise no concern for health effects for the recreational site user. However, concentrations of lead are sufficiently high that there is concern for adverse health effects for the groundskeeper, construction worker and on-site resident.

It must be understood that a major source of uncertainty regarding Range 20 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil, although the potential for this is unclear.

6.2.3 Range 23

6.2.3.1 Data Sets

The surface soil data set consists of 103 samples taken from 103 locations across Range 23 (Table 6-52). A total of 94 were analyzed for lead, 22 were analyzed for other metals and nitroaromatic compounds, and 13 were analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of 36 samples taken from 31 locations across Range 23 (Table 6-53). A total of 13 were analyzed for lead, 9 were analyzed for other metals, 12 were analyzed for nitroaromatic compounds, and 4 were analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The surface water (Table 6-54) and sediment (Table 6-55) data sets consist of six co-located samples. Four samples of each medium were analyzed for metals, chlorinated herbicides, organophosphate pesticides, organochlorine pesticides and PCBs, SVOCs and VOCs. The remaining two samples were analyzed for lead only.

The groundwater data set consists of three samples taken from three monitoring wells (Table 6-56). All samples were analyzed for metals and nitroaromatic compounds; one was analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

6.2.3.2 Chemicals of Potential Concern

Chemicals detected in Range 23 soil include metals, infrequent detections of a chlorinated herbicide, several organochlorine pesticides, several SVOCs, 2 nitroaromatic compounds, and several VOCs, particularly in surface soil (Tables 6-57 and 6-58). Site-related metals in soil are limited to those associated with bullets (antimony, copper, lead). All the organic chemicals were selected as site-related.

COPCs in surface soil were limited to antimony, lead, and 2,4-DNT (Table 6-59). 2,4-DNT was selected as a COPC only for the groundskeeper and on-site resident. COPCs in subsurface soil were limited to antimony and lead (Table 6-60). STCs selected for total soil reflect those of the COPCs identified in surface soil because COPC concentrations were higher in surface soil (Table 6-61).

Chemicals detected in surface water were limited to metals (Table 6-62). None appear to be present at concentrations greater than background; therefore none were identified as site-related chemicals or selected as COPCs.

Chemicals detected in sediment include metals, two organochlorine pesticides, and several VOCs (Table 6-63). Copper and lead appear to be site-related, suggesting that erosion is the major contributor to sediment contamination. Lead was selected as the only COPC in sediment (Table 6-64).

Chemicals detected in groundwater are limited to metals and low concentrations of two organochlorine pesticides, one nitroaromatic compound, and one VOC (Table 6-65). The metals appear to be present at concentrations comparable to background; therefore, metals were not identified as site-related chemicals. COPCs in groundwater are limited to 4-ADNT (on-site resident only) (Table 6-66). The MDC was selected as the STC because the data are inadequate to identify a plume.

6.2.3.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-67. Documentation is provided in the

form of receptor- and medium-specific tables (Tables 6-68 through 6-75) that detail the risk from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil, surface water and sediment. In addition he is assumed to consume venison, representing an indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user (Tables 6-68 and 6-69) because no chemicals in the relevant media except lead in surface soil (Table 6-59) and sediment (Table 6-64) were selected as COPCs. It is concluded that exposure to COPCs other than lead in surface soil and sediment does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-67) because no chemicals in the relevant media except lead in total soil (Table 6-61) and sediment (Table 6-64) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil and sediment does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because Range 23 is not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. Exposure to surface water and sediment is possible but is not quantified as explained for Range 18.

COPCs were not identified in groundwater for the groundskeeper scenario (Table 6-66). The total ILCR of 6.14E-7 (Table 6-67) is due entirely to 2,4-DNT in total soil (Table 6-72). An HI was not estimated because no chemicals were selected as noncancer-based COPCs. It is concluded that exposure to COPCs other than lead in total soil is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are not quantified as explained above.

COPCs were not identified in groundwater for the construction worker (Table 6-66). Lead was identified as the only COPC in total soil for the construction worker (Table 6-61); therefore, neither an ILCR nor an HI were estimated (Tables 6-67 and 6-73). It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker, as long as exposure occurs randomly across Range 23.

Construction projects, however, may involve much smaller EUs than represented by the data set for Range 23. This could result in underestimating potential risks, had COPCs other than lead been identified in soil, because the STC of a given COPC for a small area is likely to be greater than the STC for all of Range 23. The absence of COPCs, however, means that the size or location of an actual EU is not an issue for the construction worker. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water and sediment under the future site-use scenario. The assumptions for exposure to surface water and sediment are identical to those for the recreational site user; therefore, the risk estimates for recreational site user exposure to these media, had they been calculated, would be applied also to the on-site resident.

The total ILCR summed across all relevant media of 2.77E-6 (Table 6-67) is due entirely to 2,4-DNT in total soil (Table 6-74). The total HI summed across all relevant media of 2.77E-1 (Table 6-67), due to antimony in total soil (Table 6-74) and 4-ADNT in groundwater (Table 6-75) does not exceed the threshold level of 1.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of Range 23. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in the following section on lead.

6.2.3.4 Risk Characterization: Lead

Ideally, the evaluation of lead would be accomplished by comparing the lead cleanup levels with the average concentration of lead over an appropriate EU. However, data are not available with which to identify future EUs. Therefore, the foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it was determined to be site related. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Included in the discussion for the on-site resident is the EU concept applied to the COPCs evaluated in the previous section. Therefore, the comparison table for the on-site resident includes all relevant COPCs in the media of interest. The cleanup levels for COPCs other than lead are receptor-specific and range-specific and are described in the appropriate subsections below. Lead is the most prevalent COPC in the environmental media at the BBGR Ranges. Therefore, the comparison tables list lead as the first COPC, with concentrations ranked from highest to lowest. Cleanup levels of COPCs other than lead should be applied as the maximum *conservative estimate of average* concentration over the EU.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil, surface water and sediment under the current site-use scenario. The entire data set for each medium was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 23 area. Lead was identified as a COPC for the recreational site user in surface soil (Table 6-59) and sediment (Table 6-64). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg. The concentration of 7.60E+3 mg/kg was also adopted as a conservative cleanup level for lead in sediment.

The average concentration of lead in surface soil of 5.10E+2 mg/kg (Table 6-57) is below the cleanup level for soil of 7.60E+3 mg/kg. Also, the average concentration of lead in sediment of 1.16E+2 mg/kg (Table 6-63) is below the cleanup level for sediment of 7.60E+3 mg/kg. It should be noted, however, that the concentration of lead in one surface soil sample exceeds the cleanup level (Table 6-76). No lead concentrations in subsurface soil or sediment exceed the cleanup level. It is concluded that lead in soil and sediment at Range 23 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. The entire data set for each medium was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in total soil (Table 6-61), including surface soil (Table 6-59) and subsurface soil (Table 6-60), as well as in sediment (Table 6-64). The identification of surface soil samples with lead concentrations that exceed the cleanup level was discussed above. No lead concentrations in subsurface soil or sediment exceed the cleanup level. It is concluded that lead in total soil, sediment and surface water at Range 23 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil and groundwater under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 23 area. Lead was identified as a COPC for the groundskeeper in total soil (Table 6-61) including both surface soil (Table 6-59) and subsurface soil (Table 6-60). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentrations of lead in surface soil (5.10E+2 mg/kg, Table 6-57) and subsurface soil (2.15E+2, Table 6-58) fall below the cleanup level of 8.80E+2 mg/kg. However, concentrations of lead in 14 surface soil and 2 subsurface soil samples exceed the cleanup level (Table 6-77). It is concluded that lead in soil at Range 23 raises no concern for adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater under the future site-use scenario. COPCs for the construction worker were limited to lead in total soil (Table 6-61), including both surface soil (Table 6-59) and subsurface soil (Table 6-60). The cleanup level for lead in soil for the construction worker is 8.80E+2 mg/kg as described above.

An EU for the construction worker would probably be much less than the entire area of Range 23. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-78). Concentrations of lead in 14 surface soil samples and 2 surface soil samples exceed the cleanup level. It is concluded that lead in surface and subsurface soil raises concern for noncancer health effects for the construction worker exposed to total soil at Range 23.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water and sediment under the future site-use scenario. COPCs for the on-site resident include lead, antimony and 2,4-DNT in surface soil (Table 6-59), antimony and lead in subsurface soil (Table 6-60) and 4-ADNT in groundwater (Table 6-66). 2,4-DNT was the only chemical selected as a COPC based on cancer risk.

The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-79) was estimated as described in Section 6.2.1.4.

No chemicals other than 2,4-DNT in surface soil were selected as cancer-based COPCs. Therefore, a cleanup level for 2,4-DNT can be estimated as the highest concentration that would yield an ILCR of 1E-4 when rounded to one significant figure. The cleanup level can be estimated from the SSSL as follows:

$$\left(\frac{SSSL}{TILCR_{SSSL}}\right)=\left(\frac{CL}{TILCR_{CL}}\right) \qquad \text{Eq. 6.4}$$

where:

CL	=	cleanup level (mg/kg, calculated)
SSSL	=	site-specific screening level (mg/kg)
$TILCR_{SSSL}$	=	target ILCR on which SSSL is based (1E-6, unitless)
$TILCR_{CL}$	=	target ILCR for the cleanup level (unitless).

Application of a value for $TILCR_{CL}$ of 1.49E-4 and the SSSL of 9.27E-1 mg/kg for on-site resident exposure to 2,4-DNT in soil to Equation 6.4 yields a cleanup level for 2,4-DNT of 1.38E+2 mg/kg (Table 6-79).

As noted above, an EU for the on-site resident would probably be much less than the entire area of Range 23. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-79). Concentrations of lead in 30 surface soil samples and 4 subsurface soil samples exceed the cleanup level. However, no concentrations of antimony or 2,4-DNT exceeded their cleanup levels.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. It should be noted that the lead concentration in one surface soil sample exceeded the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-79).

It is concluded that lead in surface and subsurface soil raises concern for noncancer health effects in the on-site resident at Range 23.

6.2.3.5 Uncertainty Analysis

Probably the most important source of uncertainty regarding Range 23 pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. Several metals in soil were excluded from COPC selection based on their Tier 2 or Tier 3 analysis (Tables 6-57 and 6-58). Of these, the only metal that would be subject to evaluation for cancer risk is arsenic in surface soil. Substituting the MDC for arsenic of 1.75E+1 mg/kg and the cancer-based SSSL for residential exposure to arsenic in soil of 4.26E-1 mg/kg into Equation 6.1 yields an ILCR of 4.11E-5, which is within the EPA risk management range. Furthermore, the ILCR total across chemicals (4.11E-5 for arsenic and 2.77E-6 from Table 6-67) of 4.38E-5 falls within the risk management range. It is concluded that the uncertainty imparted to the cancer assessment by designating arsenic a background chemical is small.

Metals in soil with MDCs that exceed their residential noncancer SSSLs include aluminum, arsenic and barium (surface soil only), chromium (subsurface soil only), iron, manganese, and thallium (subsurface soil only). In all cases except iron, and manganese in surface soil, the exceedance was less than a factor of 10, indicating that HI values calculated for these chemicals would fall below the threshold level of 1. As noted above, the oral RfD for iron is controversial and most likely unnecessarily conservative. Selection of iron as a background metal is judged not to impart significant uncertainty to the assessment. The uncertainty regarding manganese in surface soil is considered low because the MDC of site data (4.58E+3 mg/kg, Table 6-57) is below the MDC of background data (6.85E+3 mg/kg, [IT, 2002a]) and there is no rationale to explain manganese as a site-related metal at shooting ranges because it is not associated with weapons or bullets. Some level of uncertainty remains about these metals, however, because they could contribute to target organ HI values summed across the chemicals that exceed the threshold of 1. This issue is addressed below. It should be noted that none of these metal MDCs exceed their recreational site user SSSLs; therefore, excluding background metals in soil imparts no uncertainty to the risk estimates for the recreational site user.

Surface Water. No metal in surface water excluded at the Tier 2 or Tier 3 stage had an MDC that exceeded its residential SSSL (Table 6-62). Excluding background metals in surface water imparts no uncertainty to the risk estimates for the recreational site user or resident.

Sediment. No background metals in sediment excluded at the Tier 2 or Tier 3 stage had MDCs exceeding their recreational site user SSSLs (Table 6-63). The exclusion of background metals is judged to impart no significant uncertainty to the SRA.

Groundwater. Barium was the only background metal in groundwater excluded at the Tier 2 or Tier 3 stage with an MDC that exceeded its residential SSSL (Table 6-65). The barium exceedance was less than a factor of 10, indicating that its designation as a background chemical in groundwater is judged to impart no significant uncertainty to the Range 23 assessment.

Background Metals in All Media. As discussed above, HI values for chemicals that share a target organ are generally summed. The nervous system is a common target organ for aluminum and manganese (see Toxicity Profiles in IT [2000b]), both of which were judged to be present in soil and groundwater at concentrations comparable to background. Concentrations of both metals were higher in surface soil (Table 6-57) than in subsurface soil (Table 6-58); therefore, only surface soil is considered further. The MDC of aluminum in groundwater does not exceed the BSC, which is considered a very conservative basis for background comparison. Therefore, aluminum in groundwater is not considered further. HI values of 3.00E-1, 1.26E+0 and 4.41E+0 could be estimated for residential exposure by applying the MDCs of aluminum and manganese in surface soil, and manganese in groundwater, respectively to Equation 6.2 (calculations not shown). The sum, 5.97E+0, exceeds the threshold value of 1. Aluminum and manganese, however, are not ordinarily associated with shooting ranges. The designation of aluminum and manganese in soil and manganese in groundwater as background metals at Range 23 imparts a small level of uncertainty to the assessment.

6.2.3.6 Conclusions, Range 23

Range 23 is expected to be used for recreation, for which the recreational site user is the most relevant receptor scenario. The groundskeeper and construction worker are relevant receptor scenarios should the site be developed for any commercial purpose. The recreational site user is assumed to be exposed randomly to surface soil or total soil, surface water and sediment over the entire area of Range 23. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 23 area, and to groundwater developed as a potable source – an unlikely possibility.

The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Range 23 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the construction worker must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, and to surface water and sediment in the same manner as the recreational site user. Home sites are expected to involve exposure to soil in an area much smaller than the entire range. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat. Exposure to surface water and sediment, however, is expected to occur randomly.

COPCs at Range 23 include antimony, lead, and 2,4-DNT in soil, lead in sediment, and 4-ADNT in groundwater. COPC concentrations are sufficiently low so that the recreational site user and groundskeeper are not expected to experience adverse health effects from exposure to the site in its current condition. Concentrations of lead in soil, however, are sufficiently high to raise concern for adverse health effects for the construction worker and on-site resident.

It must be understood that a major source of uncertainty regarding Range 23 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil, although the potential for this is unclear.

6.2.4 Range 26

6.2.4.1 Data Sets

The surface soil data set consists of 67 samples taken from 67 locations across Range 26 (Table 6-80). All samples were analyzed for lead, 19 were analyzed for other metals, and 11 were analyzed for nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of 14 samples taken from 14 locations (Table 6-81). All samples were analyzed for lead, seven were analyzed for other metals, and three were analyzed for nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

Surface water and sediment samples were not taken because surface water was not present at the time of sampling. Although two tributaries of Ingram Creek collect runoff from the possible impact area, water is generally present only during and shortly after storm events. Two depositional soil samples from the tributaries, however, are included in the surface soil data set.

The groundwater data set consists of four samples taken from four monitoring wells (Table 6-82). All samples were analyzed for metals and nitroaromatic compounds; one sample was analyzed for organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

6.2.4.2 Chemicals of Potential Concern

Chemicals detected in Range 26 surface and subsurface soil include metals and low detections of two VOCs (Tables 6-83 and 6-84). Chlorinated herbicides, organochlorine pesticides and SVOCs (largely PAHs) were detected only in surface soil. Site-related metals in surface soil (Table 6-83) and subsurface soil (Table 6-84) are limited to those associated with bullets (antimony, copper, lead). All the organic chemicals were selected as site-related.

COPCs in surface soil were limited to antimony, lead and five PAHs (Table 6-85). PAHs were selected as COPCs only for the groundskeeper and on-site resident. Antimony was the only COPC selected in subsurface soil (only for the on-site resident) (Table 6-86). STCs selected for total soil reflect those of the COPCs identified in surface soil (Table 6-87).

Chemicals detected in groundwater were limited to metals (Table 6-88). None appear to be present at concentrations greater than background; therefore none were identified as site-related chemicals.

6.2.4.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-89. Documentation is provided in the form of receptor- and medium-specific tables (Tables 6-90 through 6-95) that detail the risk from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil. In addition, he is assumed to consume venison, representing an

indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-89) because no chemicals in the relevant media except lead in surface soil (Table 6-90, 6-91) were selected as COPCs. It is concluded that exposure to COPCs other than lead in surface soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-89) because no chemicals in the relevant media except lead in total soil (Table 6-92) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because Range 26 is not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs were not identified in groundwater (Table 6-88). The total ILCR of 4.00E-6 (Table 6-89) is due entirely to two PAHs in total soil (Table 6-93). No HI was estimated because no chemicals were selected as noncancer-based COPCs. It is concluded that exposure to COPCs other than lead in total soil is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs were not identified in groundwater (Table 6-88). Neither ILCR nor HI values were estimated for the future construction worker (Table 6-89) because no chemicals in the relevant media except lead in total soil (Table 6-94) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker.

Construction sites, however, may involve much smaller EUs than represented by the data set for all of Range 26. This could result in underestimating potential risks, because the STC of a given

COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the construction worker is discussed in detail in the following section on lead.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source. The total ILCR summed across all relevant media of 2.30E-5 (Table 6-89) is due entirely to five PAHs in total soil (Table 6-95). The total HI summed across all relevant media of 2.28E-1 (Table 6-89), due entirely to antimony in total soil (Table 6-95) falls below the threshold level of 1. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk, or raise concern for adverse noncancer health effects for the future on-site resident.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of Range 26. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in the following section on lead.

6.2.4.4 Risk Characterization: Lead

Ideally, the evaluation of lead would be accomplished by comparing the lead cleanup levels with the average concentration of lead over an appropriate EU. However, data are not available with which to identify future EUs. Therefore, the foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it was determined to be site related. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Included in the discussion for the construction worker and on-site resident is the EU concept applied to lead as well as to the COPCs evaluated in the previous section. Therefore, the comparison tables for these receptors include all relevant COPCs in the media of interest. The cleanup levels for COPCs other than lead are receptor-specific and range-specific and are described in the appropriate subsections below. Lead is the most prevalent COPC in the environmental media at the BGR ranges. Therefore, the comparison tables list lead as the first COPC, with concentrations ranked from highest to lowest. Cleanup levels of COPCs other than lead should be applied as the maximum *conservative estimate of average* concentration over the EU.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil under the current site-use scenario. The entire data set for surface soil was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 26 area. Lead was identified as a COPC for the recreational site user in surface soil (Table 6-85). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg.

The average concentration of lead in surface soil of 5.73E+2 mg/kg (Table 6-83) is below the cleanup level for soil of 7.60E+3 mg/kg. It should be noted, however, that the concentrations of lead in 1 surface soil sample exceeds the cleanup level (Table 6-96). It is concluded that lead in surface soil at Range 26 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil under the future site-use scenario. The entire data set was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in total soil (Table 6-87), but not in subsurface soil (Table 6-86). The identification of surface soil samples with lead concentrations that exceed the cleanup level was discussed above. It is concluded that lead in total soil at Range 26 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil and groundwater under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 26 area. Lead was identified as a COPC for the groundskeeper in total soil (Table 6-87), but not in subsurface soil (Table 6-86). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentration of lead in surface soil (5.73E+2 mg/kg, Table 6-83) falls below the cleanup level of 8.80E+2 mg/kg. The concentrations of lead in nine surface soil samples exceed the cleanup level (Table 6-97). It is concluded that lead in surface soil raises no concern for adverse health effects for the future groundskeeper at Range 26.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater under the future site-use scenario. COPCs for the construction worker are limited to lead in surface soil (Table 6-85). The cleanup level for lead in soil for the construction worker is 8.80E+2 mg/kg as described above.

As noted above, an EU for the construction worker would probably be much less than the entire area of Range 26. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-98). Concentrations of lead in nine surface soil samples exceed the cleanup level. It is concluded that lead in surface soil raises concern for noncancer health effects for the construction worker exposed to total soil at Range 26.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs for the on-site resident include lead, antimony, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene in surface soil (Table 6-85), and antimony in subsurface soil (Table 6-86). The PAHs were selected as COPCs based only on cancer risk. Antimony was selected as a COPC based on noncancer effects.

The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-99) was estimated as described above.

The PAHs were selected as COPCs based on cancer risk. A worst-case estimation of total cancer risk can be calculated by the MDCs and SSSLs of these chemicals to Equation 6.1. These calculations are presented below:

Chemical	MDC	SSSL	ILCR
Surface Soil (Table 6-85)			
Benzo(a)anthracene	3.20E+0	8.51E-1	3.76E-6
Benzo(a)pyrene	2.50E+0	8.51E-2	2.94E-5
Benzo(b)fluoranthene	5.30E+0	8.51E-1	6.23E-6
Dibenzo(a,h)anthracene	3.70E-1	8.61E-2	4.30E-6
Indeno(1,2,3-cd)pyrene	1.30E+0	8.51E-1	1.53E-6
Total ILCR			4.52E-5

MDC = maximum detected concentration (mg/kg in soil); SSSL = site-specific screening level (same units as MDC); ILCR = incremental lifetime cancer risk.

The total cancer risk of 4.25E-5 does not exceed the risk management range. This shows that cancer risk will not exceed the risk management range regardless of how EUs are established, and the PAHs are considered no further.

As noted above, an EU for the on-site resident would probably be much less than the entire area of Range 26. Lacking more definitive data for EU identification, lead and antimony concentrations in individual samples were compared with their cleanup levels (Table 6-99). Concentrations of lead in 19 surface soil samples exceed the cleanup level. However, no concentrations of antimony exceed the cleanup level.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. It should be noted that the lead concentrations in one surface soil sample exceeds the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-99).

It is concluded that lead in surface soil raises concern for noncancer health effects for the on-site resident at Range 26.

6.2.4.5 Uncertainty Analysis

Probably the most important source of uncertainty regarding Range 26 pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. Several metals in soil were excluded from COPC selection based on their Tier 3 analysis (Tables 6-83 and 6-84). None of these metals, however, would be subject to evaluation for cancer risk (i.e., none have cancer-based SSSLs). Those with MDCs that exceed their residential noncancer SSSLs are limited to aluminum, barium, iron and manganese – all in surface soil only. In all cases except iron and manganese the exceedance was less than a factor of 10, indicating that HI values calculated for these chemicals would fall below the threshold level of 1. As noted above, the oral RfD for iron is controversial and most likely unnecessarily conservative; selection of iron as a background metal is judged not to impart significant uncertainty to the assessment. An HI of 1.09E+0 could be estimated by applying the MDC of 3.94E+3 mg/kg for manganese in surface soil to Equation 6.2. Manganese, however, is not associated with bullets or shooting ranges; therefore the uncertainty associated with designating manganese as a background metal is judged to be very small.

Some level of uncertainty remains about these metals because they could contribute to target organ HI values summed across the chemicals that exceed the threshold of 1. This issue is

addressed below. It should be noted that none of these metal MDCs exceed their recreational site user SSSLs; therefore, excluding background metals in soil imparts no uncertainty to the risk estimates for the recreational site user.

Groundwater. Thallium was the only background metal in groundwater excluded at the Tier 2 or Tier 3 stage with MDCs that exceed their residential SSSLs (Table 6-88). The MDC of thallium exceeded its SSSL by nearly 40-fold. Thallium, however, is not a component of bullets and no rationale is apparent for its presence as a site-related chemical. The designation of thallium as a background chemical in groundwater is judged to impart little uncertainty to the Range 26 assessment.

Background Metals in All Media. As discussed above, HI values for chemicals that share a target organ are generally summed. The nervous system is a common target organ for aluminum and manganese (see Toxicity Profiles in IT [2000b]), both of which were judged to be present in soil at concentrations comparable to background. HI values of 2.22E-1 and 1.09E+0 could be estimated for residential exposure to aluminum and manganese in surface soil, respectively, by applying their MDCs to Equation 6.2 (calculations not shown). The sum, 1.31E+0, equivalent to 1 when rounded to 1 significant figure, does not exceed the threshold value of 1. The designation of aluminum and manganese in soil as background metals at Range 26 imparts a small level of uncertainty to the assessment.

6.2.4.6 Conclusions, Range 26

Range 26 is expected to be used for recreation (EDAW, 1997), for which the recreational site user is the most relevant receptor scenario. The groundskeeper and construction worker are relevant should the site be developed for any use. The recreational site user is assumed to be exposed randomly to surface soil or total soil over the entire area of Range 26. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 26 area, and to groundwater developed as a potable source – an unlikely possibility.

The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Range 26 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the construction worker must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source. Home sites are expected to involve exposure to soil in an area much smaller than the entire range. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat.

COPCs at Range 26 are limited to antimony, lead, and five PAHs in surface soil, and antimony in subsurface soil. COPC concentrations are sufficiently low so that the recreational site user and groundskeeper are not expected to experience adverse health effects from exposure to the site in its current condition. Concentrations of lead, however, are sufficiently high to raise concern for adverse health effects for the construction worker and on-site resident.

It must be understood that a major source of uncertainty regarding Range 26 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil in the future, although the potential for this is unclear.

6.2.5 Range 25

6.2.5.1 Data Sets

The surface soil data set consists of 62 samples taken from 62 locations across Range 25 (Table 6-100). All samples were analyzed for metals and nitroaromatic compounds. Twelve samples were analyzed for organochlorine pesticides, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of 53 samples taken from 53 locations across Range 25 (Table 6-101). All samples were analyzed for metals and nitroaromatic compounds. Ten samples were analyzed for organochlorine pesticides, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The surface water (Table 6-102) and sediment (Table 6-103) data sets consist of single co-located samples analyzed for metals, chlorinated herbicides, organophosphate pesticides, organochlorine pesticides, nitroaromatic compounds, SVOCs, and VOCs.

The groundwater data set consists of six samples taken from six monitoring wells (Table 6-104). All samples were analyzed for metals and nitroaromatic compounds; two were analyzed for

organochlorine pesticides, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

6.2.5.2 Chemicals of Potential Concern

Chemicals detected in Range 25 soil include metals, organochlorine pesticides and chlorinated herbicides (particularly in surface soil), and a few VOCs (Tables 6-105 and 6-106). Site-related metals in surface soil include those associated with bullets (antimony, copper, lead, and zinc) (Table 6-105). Site-related metals in subsurface soil include antimony, copper, lead, and mercury (Table 6-106). No explanation is apparent for mercury, which is not a usual component of bullets. All the organic chemicals were selected as site-related.

COPCs in surface soil were limited to antimony, copper, and lead (Table 6-107). Antimony and copper were selected as COPCs only for the construction worker and on-site resident. Antimony was selected as the only COPC in subsurface soil (only for the on-site resident) (Table 6-108). STCs selected for total soil reflect those of the COPCs identified in surface soil (Table 6-109).

Chemicals detected in surface water were limited to 4 metals (Table 6-110). None appear to be present at concentrations greater than background or were essential nutrients; therefore none were identified as site-related chemicals or selected as COPCs.

Chemicals detected in sediment include metals and three VOCs (Table 6-111). The metals appear to be present at concentrations comparable to background or were essential nutrients; site-related chemicals were limited to the three VOCs. None were selected as COPCs (Table 6-112).

Chemicals detected in groundwater include metals, organochlorine pesticides, and nitroaromatic compounds (Table 6-113). The metals appear to be present at concentrations comparable to background or were essential nutrient; therefore, metals were not identified as site-related chemicals. COPCs in groundwater are limited to beta-BHC, heptachlor, 2,4-DNT, and 2,6-dinitrotoluene (2,6-DNT) (Table 6-114). The MDCs were selected as the STCs because plumes could not be discerned. The source of the nitroaromatic compounds is unclear because none were detected in site soil.

6.2.5.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-115. Documentation is provided in the form of receptor- and medium-specific tables (Tables 6-116 through 6-123) that detail the risk

from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil, surface water, and sediment. In addition, he is assumed to consume venison, representing an indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user (Tables 6-115) because no chemicals in the relevant media except lead in surface soil were selected as COPCs (Tables 6-116 and 6-117). It is concluded that exposure to COPCs other than lead in surface soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-115) because no chemicals in the relevant media except lead in total soil (Table 6-118) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because Range 25 is not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. Exposure to surface water and sediment is possible, but such exposures are not quantified as explained previously.

The total ILCR of 4.64E-6 (Table 6-115) is due entirely to heptachlor and 2,4-DNT in groundwater (Table 6-120). An HI was not estimated because no noncancer-based chemicals were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil and groundwater is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario.

Exposure to surface water and sediment is possible, but such exposures are not quantified as explained above.

An ILCR was not estimated (Table 6-115) because no cancer-based chemicals were selected as COPCs. The total HI of 3.71E-2 (Table 6-115) is due entirely to antimony and copper in total soil (Table 6-121). The HI does not exceed the threshold level of 1. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker, as long as exposure occurs randomly across Range 25.

Construction projects, however, may involve much smaller EUs than represented by the entire data set for Range 25. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for all of Range 25. The EU issue for the construction worker is discussed in detail in Section 6.2.5.4 on lead.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water and sediment under the future site-use scenario. The assumptions for exposure to surface water and sediment are identical to those for the recreational site user; therefore, the risk estimates for recreational site user exposure to these media, had they been calculated, would be applied also to the on-site resident.

The total ILCR summed across all relevant media of 2.58E-5 (Table 6-115) is due to beta-BHC, heptachlor, 2,4-DNT and 2,6-DNT in groundwater (Table 6-123). The total HI summed across all relevant media of 2.39E-1 (Table 6-115), due to antimony and copper in total soil (Table 6-122) falls below the threshold level of 1. Exposure to COPCs other than lead in surface soil poses no significant cancer risk or concern for adverse effects.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of Range 25. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in the following section on lead.

6.2.5.4 Risk Characterization: Lead

Ideally, the evaluation of lead would be accomplished by comparing the lead cleanup levels with the average concentration of lead over an appropriate EU. However, data are not available with which to identify future EUs. Therefore, the foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it

was determined to be site related. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Included in the discussion for the construction worker and on-site resident is the EU concept applied to the COPCs evaluated in the previous section. Therefore, the comparison tables for these receptors include all relevant COPCs in the media of interest. The cleanup levels for COPCs other than lead are receptor-specific and range-specific and are described in the appropriate subsections below. Lead is the most prevalent COPC in the environmental media at the BBGR Ranges. Therefore, the comparison tables list lead as the first COPC, with concentrations ranked from highest to lowest. Cleanup levels of COPCs other than lead should be applied as the maximum *conservative estimate of average* concentration over the EU.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil, surface water and sediment under the current site-use scenario. The entire data set for each medium was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 25 area. Lead was identified as a COPC for the recreational site user in surface soil (Table 6-107). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg.

The average concentration of lead in surface soil of 6.34E+2 mg/kg (Table 6-105) is below the cleanup level for soil of 7.60E+3 mg/kg. It should be noted, however, that the concentration of lead in 1 surface soil sample exceeds the cleanup level (Table 6-124). It is concluded that lead in surface soil at Range 25 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil, surface water and sediment under the future site-use scenario. The entire data set for each medium was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in total soil (Table 6-109), but not in subsurface soil (Table 6-108). The identification of surface soil samples with lead concentrations that exceed the cleanup level was discussed above. It is concluded that lead in total soil at Range 25 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil and groundwater under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Range 25 area. Lead was identified as a COPC for the groundskeeper in total soil (Table 6-109) but not subsurface soil (Table 6-107). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentration of lead in surface soil (6.34E+2 mg/kg, Table 6-105) falls below the cleanup level of 8.80E+2 mg/kg. The concentrations of lead in seven surface soil samples exceed the cleanup level (Table 6-125). It is concluded that lead in surface soil raises no concern for adverse health effects for the future groundskeeper at Range 25.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater under the future site-use scenario. COPCs for the construction worker included lead, antimony and copper in total soil (Table 6-109) but not subsurface soil (Table 6-108).

The cleanup level for lead in soil for the construction worker is 8.80E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in surface soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. The cleanup level for antimony of 2.96E+2 mg/kg (Table 6-126) was described above.

Copper shares no target organ with the other COPCs in surface soil, so the cleanup level for copper can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. The cleanup level for copper of 3.04E+4 mg/kg (Table 6-126) was described above.

As noted above, an EU for the construction worker would probably be much less than the entire area of Range 25. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-126). Concentrations of lead in seven surface soil samples exceed the cleanup level. However, no concentrations of antimony or copper exceed their cleanup levels. It is concluded that lead in surface soil raises concern for noncancer health effects for the construction worker exposed to total soil and groundwater at Range 25.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, surface water and sediment under the future site-use scenario. COPCs for the on-site resident include lead, antimony and copper in surface soil (Table 6-107), antimony in subsurface soil (Table 6-108), and beta-BHC, heptachlor, 2,4-DNT and 2,6-DNT in groundwater (Table 6-114). Beta-BHC, heptachlor, 2,4-DNT, and 2,6-DNT were selected as COPCs based only on cancer risk. The metals in soil were selected as COPCs based only on noncancer effects.

The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. Antimony shares no target organ with the other COPCs in soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-127) was estimated as described above.

Copper shares no target organ with the other COPCs in soil, so the cleanup level for copper can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for copper of 4.67E+3 mg/kg (Table 6-127) was estimated as described above.

Several chemicals in groundwater were selected as COPCs based on cancer risk. In all cases, however, the MDC was selected as the STC because data were available from only one sample. Therefore, the total ILCR reported in Table 6-115 (2.58E-5) can be considered a worst-case estimation. The total cancer risk of 2.58E-5 does not exceed the risk management range. This shows that cancer risk will not exceed the risk management range regardless of how EUs are established.

As noted above, an EU for the on-site resident would probably be much less than the entire area of Range 25. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-127). Concentrations of lead in nine surface soil samples exceed the cleanup level. However, no concentrations of antimony in surface or subsurface soil or copper in surface soil exceed their cleanup levels.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. It should be noted that the lead concentrations in one surface soil sample exceeds the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-127).

It is concluded that lead in surface soil raises concern for noncancer health effects in the on-site resident at Range 25.

6.2.5.5 Uncertainty Analysis

Probably the most important source of uncertainty regarding Range 25 pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. Several metals in soil were designated as background chemicals and excluded from COPC selection based on their Tier 2 or Tier 3 analysis (Tables 6-105 and 6-106). Those with MDCs that exceed their residential SSSLs include aluminum, arsenic, chromium, iron, manganese, and vanadium. An ILCR of 8.66E-5 could be estimated for arsenic by applying the MDC in surface soil of 3.69E+1 mg/kg to Equation 6.1. The total ILCR summed across chemicals of 1.12E-4 (8.66E-5 for arsenic and 2.58E-5 from Table 6-115), when rounded to 1 significant figure does not exceed the EPA risk management range, indicating that designation of arsenic as a background metal imparts a small degree of uncertainty to the overall cancer assessment of Range 25. All the metals (including arsenic) are subject to noncancer assessment. In all cases except arsenic, iron and manganese the MDC exceeds the SSSL by less than a factor of 10, indicating that HI values calculated for these metals would fall below the threshold level of 1. As noted above, the oral RfD for iron is controversial and most likely unnecessarily conservative, and the selection of iron as a background chemical is judged to impart no significant uncertainty to the assessment. HI values of 1.58 and 1.32 could be estimated for the MDCs of arsenic and manganese respectively, by applying their respective MDCs to Equation 6.2. The HI for arsenic rounded to 1 significant figure exceeds the threshold level of 1; therefore, designating arsenic as a background metal imparts a small degree of uncertainty to the noncancer assessment. Aluminum and manganese share the nervous system as a common target organ. An HI of 8.49E-1 could be estimated for the MDC of aluminum in subsurface soil from Equation 6.2. The HI for aluminum summed with the HI for manganese of 1.32 yields a total HI for the nervous system of 2.17, which exceeds the threshold level of 1. The designation of aluminum and manganese as background metals in soil imparts a small degree of uncertainty to the noncancer evaluation.

Surface Water. No metals in surface water were excluded at the Tier 2 or Tier 3 stage (Table 6-110). Excluding background metals in surface water imparts no uncertainty to the risk estimates for the recreational site user or resident.

Sediment. No background metals in sediment excluded at the Tier 2 or Tier 3 stage had MDCs exceeding their recreational site user SSSLs (Table 6-111). The exclusion of background metals is judged to impart no significant uncertainty to the SRA.

Groundwater. No metals in groundwater were excluded at the Tier 2 or Tier 3 stage (Table 6-113). Excluding background metals in groundwater imparts no uncertainty to the risk estimates for the recreational site user or resident.

6.2.5.6 Conclusions, Range 25

Range 25 is expected to be used for recreation (EDAW, 1997), for which the recreational site user is the most relevant receptor scenario. The groundskeeper and construction worker would be relevant receptor scenarios should the site be developed for other industrial or commercial uses. The recreational site user is assumed to be exposed randomly to surface soil or total soil, surface water, and sediment over the entire area of Range 25. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 25 area, and to groundwater developed as a potable source – an unlikely possibility.

The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Range 25 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the construction worker must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil, groundwater developed as a potable source, and to surface water and sediment in the same manner as the recreational site user. Home sites are expected to involve exposure to soil in an area much smaller than the entire range. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat. Exposure to surface water and sediment, however, is expected to occur randomly.

COPCs at Range 25 include antimony, copper and lead in surface soil, antimony in subsurface soil, and two organochlorine pesticides and two nitroaromatic compounds in groundwater. COPC concentrations are sufficiently low so that the recreational site user and groundskeeper are not expected to experience adverse health effects from exposure to the site in its current

condition. Concentrations of lead, however, are sufficiently high to raise concern for adverse health effects for the construction worker and on-site resident.

It must be understood that a major source of uncertainty regarding Range 25 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil in the future, although the potential for this is unclear.

6.2.6 Range 25E/28

6.2.6.1 Data Sets

The surface soil data set consists of 14 samples taken from 14 locations across Range 25E/28 (Table 6-128). All samples were analyzed for lead, five were analyzed for other metals, and four were analyzed for nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The subsurface soil data set consists of five samples taken from five locations (Table 6-129). All samples were analyzed for lead, four were analyzed for other metals, and three were analyzed for nitroaromatic compounds, organochlorine pesticides and PCBs, SVOCs, VOCs, chlorinated herbicides, and organophosphate pesticides.

The surface water (Table 6-130) and sediment (Table 6-131) data sets consist of a single co-located sample analyzed for metals, nitroaromatic compounds, chlorinated herbicides, organophosphate pesticides, organochlorine pesticides and PCBs, SVOCs, and VOCs.

6.2.6.2 Chemicals of Potential Concern

Chemicals detected in Range 25E/28 soil include metals, two organochlorine pesticides and a few VOCs (Tables 6-132 and 6-133). All metals in soil were determined to be present at background concentrations or were identified as essential nutrients. All the organic chemicals were selected as site-related, but their concentrations were sufficiently low that none were selected as COPCs (Tables 6-134 and 6-135).

Chemicals detected in surface water were limited to metals (Table 6-136). All appeared to be present at concentrations comparable to background or were essential nutrients; therefore, none were identified as site-related chemicals and subjected to COPC selection.

Chemicals detected in sediment include metals, one organochlorine pesticide, and one VOC (Table 6-137). The metals appear to be present at concentrations comparable to background or are essential nutrients; site-related chemicals were limited to the organic compounds. No chemicals in sediment were selected as COPCs (Table 6-138).

6.2.6.3 Risk Characterization: Chemicals Other than Lead

No chemicals in soil, surface water or sediment were selected as COPCs for any of the 4 receptor scenarios evaluated herein. Therefore, ILCR and HI values were not estimated. It is concluded that exposure to the media of interest does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current or future recreational site user, or the future groundskeeper, future construction worker or future on-site resident.

6.2.2.4 Risk Characterization: Lead

Lead was present in all media of interest at Range 25E/28 at concentrations comparable to background. Therefore, lead was not selected as a COPC and it is concluded that exposure to lead at Range 25E/28 raises no concern for adverse health effects for the current or future recreational site user, future groundskeeper, future construction worker, or future on-site resident.

6.2.6.5 Uncertainty Analysis

Possibly the most important source of uncertainty regarding Range 25E/28 arises from the lack of groundwater data. However, there is no evidence that ammunition other than blanks were ever discharged on these parcels. Also, there is considerable overlap between Range 25E and Ranges 26, 20 and 25, and between Range 28 and Ranges 25 and 26. Thus, it was judged that groundwater data from Ranges 26 and 25 would reflect contamination from Range 25E/28. Finally, the lack of metal concentrations in soil, surface water, or sediment at Range 25E/28 that exceed background concentrations assuages concern that groundwater contamination may have been overlooked. Therefore, it is concluded that the lack of groundwater data specifically for Range 25E/28 does not represent a significant source of uncertainty.

Another source of uncertainty pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. A few metals in surface or subsurface soil were excluded from COPC selection exposure based on their Tier 3 analysis (Tables 6-132 and 6-133). None except aluminum in subsurface soil have MDCs that exceed their residential noncancer SSSLs. The MDC of aluminum, however, exceeds its SSSL by less than a factor of 10, indicating that the associated HI would be

less than the threshold level of 1. Therefore, it is concluded that excluding background metals in soil imparts no significant uncertainty to the risk assessment for any receptor scenario.

Surface Water. No metals in surface water were excluded at the Tier 2 or Tier 3 stage of the background evaluation (Table 6-136). Excluding background metals in surface water imparts no uncertainty to the risk estimates for the recreational site user or resident.

Sediment. No metals in sediment were excluded at the Tier 2 or Tier 3 stage of the background evaluation (Table 6-137). The exclusion of background metals is judged to impart no uncertainty to the SRA.

6.2.6.6 Conclusions, Range 25E/28

Range 25E/28 is expected to be used for recreation (EDAW, 1997), for which the recreational site user is the most relevant receptor scenarios. The recreational site user is assumed to be exposed randomly to surface soil or total soil, surface water, and sediment over the entire area of Range 25E/28. The groundskeeper and construction worker were included in the event that the site might be developed for commercial use at some time in the future. The groundskeeper is assumed to be randomly exposed to total soil over the entire Range 25E/28 area. The construction worker is assumed to be exposed to total soil. The on-site resident is assumed to be exposed to total soil, and to surface water and sediment in the same manner as the recreational site user.

No chemicals in the media of interest were identified as COPCs at Range 25E/28. It is concluded that exposure to Range 25E/28 poses no threat of unacceptable cancer risk or adverse noncancer health effects to the current or future recreational site user, future groundskeeper, future construction worker or future on-site resident.

6.2.7 Ranges South of Range 25

6.2.7.1 Data Sets

The surface soil data set consists of 16 samples taken from 16 locations across the Ranges South of Range 25 (Table 6-139). All samples were analyzed for metals and nitroaromatic compounds.

The subsurface soil data set consists of 10 samples taken from 10 locations (Table 6-140). All samples were analyzed for metals and nitroaromatic compounds.

Surface water and sediment samples were not taken because surface water was not present at the time of sampling. Surface water is generally present only during and shortly after storm events. Three depositional soil samples, however, are included in the surface soil data set.

The groundwater data set consists of three samples taken from three monitoring wells (Table 6-141). All samples were analyzed for metals and nitroaromatic compounds.

6.2.7.2 Chemicals of Potential Concern

Chemicals detected in the Ranges South of Range 25 surface and subsurface soil are limited to metals; in other words, nitroaromatic compounds were not detected (Tables 6-142 and 6-143). Site-related metals in surface soil (Table 6-142) are limited to those associated with bullets (antimony, copper, and lead). Site-related metals in subsurface soil (Table 6-143) include antimony, which is associated with bullets, and also beryllium and mercury, which are not generally associated with bullets. Lead was not identified as a site-related chemical. Geochemical evaluation showed that lead concentrations in subsurface soil are present as naturally occurring background. All lead concentrations fell well within the range of naturally occurring background (9.62E-1 to 5.00E+2 mg/kg, IT [2002a]). COPCs in soil were limited to antimony and lead in surface soil (Table 6-144) and antimony in subsurface soil (only for the on-site resident) (Table 6-145). The STCs of antimony and lead in total soil reflect those of surface soil (Table 6-146).

Chemicals detected in groundwater were limited to six metals (Table 6-147). None appear to be present at concentrations greater than background; therefore, none were identified as site-related chemicals or subjected to COPC selection.

6.2.7.3 Risk Characterization: Chemicals Other than Lead

ILCR and HI estimates for all receptors under current and future site-use scenarios are compiled for each medium and summed across media in Table 6-148. Documentation is provided in the form of receptor- and medium-specific tables (Tables 6-149 through 6-154) that detail the risk from exposure to each COPC. The numerical results and their interpretation are discussed by receptor and site-use scenario.

Recreational Site User, Current Site Use: The recreational site user is evaluated as the only plausible receptor under the current site use scenario. The recreational site user is assumed to be exposed to surface soil. In addition he is assumed to consume venison, representing an indirect or food-chain pathway for exposure to surface soil, from an annual successful deer-hunting expedition. Neither ILCR nor HI were estimated for the current recreational site user

(Table 6-148) because no chemicals in the relevant media were selected as COPCs except lead in surface soil (Tables 6-149 and 6-150). It is concluded that exposure to COPCs other than lead in surface soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the current recreational site user.

Recreational Site User, Future Site Use: The future site-use scenario for the recreational site user is identical to the current site-use scenario, except that he is assumed to be exposed to total soil to reflect the possibility that construction or development for future use may bring subsurface soil to the surface. Also, the recreational site user is assumed not to hunt deer or consume venison under the future site-use scenario. Neither ILCR nor HI were estimated for the current recreational site user (Table 6-148) because no chemicals in the relevant media except lead in total soil (Table 6-146) but not subsurface soil (Table 6-145) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is not evaluated as a plausible receptor for current site use because the Ranges South of Range 25 are not currently used for any purpose. The groundskeeper is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs were not identified in groundwater (Table 6-147). No ILCR or HI was estimated (Table 6-148) because no chemicals in total soil except lead (Table 6-152) were selected as COPCs for this receptor. It is concluded that exposure to COPCs other than lead in total soil is unlikely to induce adverse health effects for the future groundskeeper.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs were not identified in groundwater (Table 6-147). Neither ILCR nor HI were estimated for the future construction worker (Table 6-148) because no chemicals in the relevant media except lead in total soil (Table 6-146) were selected as COPCs. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk and is unlikely to induce adverse noncancer health effects for the future construction worker.

Construction sites, however, may involve much smaller EUs than represented by the data set for all of the Ranges South of Range 25. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the

entire data set. The EU issue for the construction worker is discussed in detail in the following section on lead.

On-Site Resident, Future Site Use: The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source. No ILCR was estimated because no chemicals in total soil or groundwater were selected as cancer-based COPCs (Tables 6-146 and 6-147). The total HI summed across all relevant media of 2.06E-1 (Table 6-148), due entirely to antimony in total soil (Table 6-154) falls below the threshold level of 1. It is concluded that exposure to COPCs other than lead in total soil does not pose an unacceptable cancer risk, or raise concern for adverse noncancer health effects for the future on-site resident.

Residential home sites, however, may involve much smaller EUs than represented by the data set for all of the Ranges South of Range 25. This could result in underestimating potential risks, because the STC of a given COPC for a small area is likely to be greater than the STC for the entire data set. The EU issue for the on-site resident is discussed in detail in the following section on lead.

6.2.7.4 Risk Characterization: Lead

Ideally, the evaluation of lead would be accomplished by comparing the lead cleanup levels with the average concentration of lead over an appropriate EU. However, data are not available with which to identify future EUs. Therefore, the foundation for this discussion for each receptor is a table that compares the cleanup level of lead with its concentration in each sample in which it was determined to be site related. Site-related concentrations of lead were ranked from highest to lowest to facilitate identifying sample locations that might require further action. The cleanup levels for lead are the receptor-specific cleanup levels described above. The cleanup level should be applied as the maximum *average* concentration over the EU. The comparisons are discussed below for each receptor scenario.

Included in the discussion for the construction worker and on-site resident is the EU concept applied to lead as well as to the COPCs evaluated in the previous section. Therefore, the comparison tables for these receptors include all relevant COPCs in the media of interest. The cleanup levels for COPCs other than lead are receptor-specific and range-specific and are described in the appropriate subsections below. Lead is the most prevalent COPC in the environmental media at the BBGR Ranges. Therefore, the comparison tables list lead as the first COPC, with concentrations ranked from highest to lowest. Cleanup levels of COPCs other than lead should be applied as the maximum *conservative estimate of average* concentration over the EU.

Recreational Site User, Current Site Use: The recreational site user is assumed to be exposed to surface soil under the current site-use scenario. The entire data set for surface soil was included in the evaluation because it was assumed that exposure would occur randomly over the entire Ranges South of Range 25 area. Lead was identified as a COPC for the recreational site user in surface soil (Table 6-144). A cleanup level for lead in soil of 7.60E+3 mg/kg was developed as a refinement to the SSSL of 400 mg/kg.

The average concentration of lead in surface soil of 8.96E+2 mg/kg (Table 6-142) is below the cleanup level for soil of 7.60E+3 mg/kg. Furthermore, no concentrations of lead in any surface soil sample exceed the cleanup level (Table 6-155). It is concluded that lead in surface soil at the Ranges South of Range 25 raises no concern for adverse health effects for the current recreational site user.

Recreational Site User, Future Site Use: The recreational site user is assumed to be exposed to total soil under the future site-use scenario. The entire data set was included in the evaluation as explained above. Lead was identified as a COPC for the recreational site user in total soil (Table 6-146), but not in subsurface soil (Table 6-145). As discussed above, no surface soil samples had lead concentrations exceeding the cleanup level. It is concluded that lead in total soil at the Ranges South of Range 25 raises no concern for adverse health effects for the future recreational site user.

Groundskeeper, Future Site Use: The groundskeeper is assumed to be exposed to total soil and groundwater under the future site-use scenario. The entire total soil data set was included in the evaluation because it was assumed that exposure would occur randomly over the entire Ranges South of Range 25 area. Lead was identified as a COPC for the groundskeeper in total soil (Table 6-146), but not in subsurface soil (Table 6-145). A cleanup level for lead in soil of 8.80E+2 mg/kg was developed for the groundskeeper.

The average concentration of lead in surface soil (8.96E+2 mg/kg, Table 6-142) slightly exceeds the cleanup level of 8.80E+2 mg/kg. Specifically, concentrations of lead in six surface soil samples exceed the cleanup level (Table 6-156). It is concluded that lead in surface soil raises concern for adverse health effects for the future groundskeeper at the Ranges South of Range 25.

Construction Worker, Future Site Use: The construction worker is assumed to be exposed to total soil and groundwater under the future site-use scenario. COPCs for the construction worker are limited to lead in surface soil (Table 6-144). The cleanup level for lead in soil for the

construction worker is 8.80E+2 mg/kg as described above. The average concentration of lead in surface soil (8.96E+2 mg/kg, Table 6-142) slightly exceeds the cleanup level of 8.80E+2 mg/kg.

As noted above, an EU for the construction worker would probably be much less than the entire area of the Ranges South of Range 25. Lacking more definitive data for EU identification, COPC concentrations in individual samples were compared with their cleanup levels (Table 6-157). Concentrations of lead in six surface soil samples exceed the cleanup level. It is concluded that lead in surface soil raises concern for noncancer health effects for the construction worker exposed to total soil at the Ranges South of Range 25.

On-Site Resident, Future Site Use: Although residential use is very unlikely for this site, the on-site resident was evaluated for the additional information provided. The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source under the future site-use scenario. COPCs for the on-site resident include lead and antimony in surface soil (Table 6-144) and antimony in subsurface soil (Table 6-145). No chemicals in groundwater were selected as COPCs (Table 6-147). Antimony was selected as a COPC based on noncancer effects.

The cleanup level for lead in soil for the on-site resident is 4.00E+2 mg/kg as described above. The average concentration of lead in surface soil (8.96E+2 mg/kg, Table 6-142) exceeds the cleanup level of 4.00E+2 mg/kg. Antimony shares no target organ with the other COPCs in soil, so the cleanup level for antimony can be estimated as the highest concentration that would yield an HI of 1 when rounded to one significant figure. A cleanup level for antimony of 4.63E+1 mg/kg (Table 6-158) was estimated as described above.

As noted above, an EU for the on-site resident would probably be much less than the entire area of the Ranges South of Range 25. Lacking more definitive data for EU identification, lead and antimony concentrations in individual samples were compared with their cleanup levels (Table 6-158). Concentrations of lead in nine surface soil samples exceed the cleanup level. However, no concentrations of antimony exceed the cleanup level.

An additional consideration for lead in soil at residential sites is the potential for harm from acute exposure. None of the lead concentrations in surface soil exceed the acute criterion for lead in soil of 6.50E+3 mg/kg (Table 6-158).

It is concluded that lead in surface soil raises concern for noncancer health effects for the on-site resident at the Ranges South of Range 25.

6.2.7.5 Uncertainty Analysis

Probably the most important source of uncertainty regarding the Ranges South of Range 25 pertains to the exclusion of background metals from the quantitative risk assessment. The background metals are discussed by medium.

Soil. Several metals in soil were excluded from COPC selection based on their Tier 3 analysis (Tables 6-142 and 6-143). Arsenic in surface and subsurface soil is the only metals subject to cancer evaluation. An ILCR of 4.67E-5 could be estimated for arsenic by applying the MDC in subsurface soil of 1.99E+1 mg/kg to Equation 6.1. The ILCR of 4.67E-5, which reflects a worst-case estimate, falls within the risk management range, indicating that no significant uncertainty is imparted to the cancer assessment by designating arsenic as a background metal. Metals with MDCs that exceed their residential noncancer SSSLs include aluminum (subsurface soil only), barium, chromium, and manganese (surface soil only), and arsenic and iron (surface and subsurface soil). In all cases except iron and manganese the exceedance was less than a factor of 10, indicating that HI values calculated for these chemicals would fall below the threshold level of 1. The uncertainty imparted to the noncancer assessment by designating these metals as background metals is insignificant. As noted above, the oral RfD for iron is controversial and most likely unnecessarily conservative; selection of iron as a background metal is judged not to impart significant uncertainty to the assessment. An HI of 1.69E+0 could be estimated by applying the MDC of 6.12E+3 mg/kg for manganese in surface soil to Equation 6.2. Manganese, however, is not associated with bullets or shooting ranges; therefore, the uncertainty associated with designating manganese as a background metal is judged to be small.

Some level of uncertainty remains about these metals because they could contribute to target organ HI values summed across the chemicals that exceed the threshold of 1. This issue is addressed below. It should be noted that none of these metal MDCs exceed their recreational site user SSSLs; therefore, excluding background metals in soil imparts no uncertainty to the risk estimates for the recreational site user.

Groundwater. No metals were excluded at the Tier 2 or Tier 3 stage (Table 6-147); therefore, the designation of groundwater metals as background is judged to impart no uncertainty to the Ranges South of Range 25 assessment.

Background Metals in All Media. As discussed above, HI values for chemicals that share a target organ are generally summed. The nervous system is a common target organ for aluminum and manganese (see Toxicity Profiles in IT [2000b]), both of which were judged to be present in

soil at concentrations comparable to background. HI values of 2.55E-1 and 1.69E+0 could be estimated for residential exposure to aluminum in subsurface soil and manganese in surface soil, respectively, by applying their MDCs to Equation 6.2 (calculations not shown). The sum, 1.95+0, exceeds the threshold value of 1. The designation of aluminum and manganese in soil as background metals at the Ranges South of Range 25 imparts a small level of uncertainty to the assessment.

6.2.7.6 Conclusions, Ranges South of Range 25

The Ranges South of Range 25 (EDAW, 1997) are expected to be used for recreation, for which the recreational site user is the most relevant receptor scenario. The groundskeeper and construction worker are relevant receptors should the site be developed for any use. The recreational site user is assumed to be exposed randomly to surface soil or total soil over the entire area of the Ranges South of Range 25. The groundskeeper is assumed to be randomly exposed to total soil over the entire area, and to groundwater developed as a potable source – an unlikely possibility.

The construction worker is assumed to be exposed to total soil and groundwater developed as a potable source. Construction projects, however, are expected to involve exposure to soil in an area much smaller than the entire Ranges South of Range 25 area. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the construction worker must be interpreted as posing a health threat.

The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source. Home sites are expected to involve exposure to soil in an area much smaller than the entire Ranges South of Range 25. Lacking definitive EU information, COPC concentrations at one or more soil sample locations that would yield a total ILCR that exceeds 1E-4 or a target organ HI that exceeds the threshold level of 1 for the on-site resident must be interpreted as posing a health threat.

COPCs at the Ranges South of Range 25 are limited to antimony and lead in surface soil and antimony in subsurface soil. COPC concentrations are sufficiently low so that the recreational site user is not expected to experience adverse health effects from exposure to the site in its current condition. Concentrations of lead, however, are sufficiently high to raise concern for adverse health effects for the groundskeeper, construction worker and on-site resident.

It must be understood that a major source of uncertainty regarding the Ranges South of Range 25 is the observation of metallic lead fragments on the ground. These fragments could increase the concentration and bioavailability of finely divided forms of lead in soil in the future, although the potential for this is unclear.

6.3 Final SRA Summary and Conclusions

The seven BBGR Ranges (or range combinations) were evaluated in an SRA. The SRA provides all the precision and accuracy of a CERCLA baseline risk assessment, but the presentation has been streamlined to capture the economy of scale associated with several similar sites. The heart of the SRA is the development of SSSLs, which are medium-specific and receptor-specific, risk-based screening concentrations used to quickly and efficiently screen the site for potential cancer risk and noncancer hazard from site-related chemicals in environmental media. It is assumed that the reader is familiar with the FTMC facility, with risk assessment in general, and with the IWWP, in which the SRA is described in detail.

The BBGR Ranges are currently unused. Plausible receptors under the current site-use scenario are limited to a recreational site user (e.g., hiker, sight-seer) who may also be a hunter. The current recreational site user is assumed to be exposed to surface soil, and to surface water and sediment (wading scenario) if present on the site. In addition, the current recreational site user is assumed to consume venison from deer that browse vegetation growing on the site, representing an indirect food-chain pathway for exposure to surface soil.

Future site use is projected to be recreational, but it is possible that a site could be developed for active recreation, or commercial or industrial uses. Therefore, plausible receptors under the future site-use scenario include a recreational site user, groundskeeper and construction worker. The future recreational site user is assumed to be exposed to total soil – a hypothetical medium created from the surface and subsurface soil data sets to reflect the possibility that site development may bring subsurface soil to the surface where contact or exposure is possible. Site development is assumed to remove the habitat favorable for deer hunting, or to result in restrictions against hunting, so indirect exposure to soil via venison consumption is not included. The future recreational site user is also assumed to be exposed to surface water and sediment as described above.

The groundskeeper and construction worker are assumed to be exposed to total soil, and also to groundwater developed as a potable source – a very unlikely scenario. Exposure to surface water and sediment is possible but is likely to be sporadic and toxicologically insignificant. Therefore, exposure to surface water and sediment is not quantified for these receptors.

An on-site resident is also included, although residential development is very unlikely, as generally the upper-bound on exposure and risk. The additional information provided by evaluating residential exposure is often helpful for future site management, even though residential exposure is not plausible for these ranges. The on-site resident is assumed to be exposed to total soil and groundwater developed as a potable source. He is also assumed to be exposed to surface water and sediment in the same manner as the recreational site user.

It is important to understand the EU concept, which is defined as the area over which a given receptor is assumed to be randomly exposed. For example, a recreational site user may hike or roam over a large area – likely the entire area of a given range. Similarly, a groundskeeper may work over the entire area of a range. To simplify the risk assessment, the entire range was assumed to be a reasonable EU for these receptors. Construction projects, on the other hand, are likely to involve areas much smaller than an entire range. Similarly, housing sites on which an on-site resident would be exposed are expected to be much smaller than an entire range. It is likely that contamination in some areas may be heavier than that which is represented by the entire data set. Information is not available to predict or identify future EUs. Therefore, the potential for exposure to contamination heavier than represented by the entire data set for a range is addressed by identifying specific sample locations within the range where COPC concentrations exceed cleanup levels.

The analytical data were evaluated, metals recognized as essential nutrients and those present at background concentrations were removed from evaluation, and COPCs – those chemicals that may contribute significantly to risk – were identified. Metals associated with bullets and shot – predominantly lead, but also antimony and copper – were the primary COPCs. Some nitroaromatic compounds, possibly from gunpowder, and some organochlorine pesticides, probably used for insect control, were sporadically detected and selected as COPCs. STCs, representative chemical concentrations in each medium, were estimated for each COPC.

The SSSLs were applied to the STCs to estimate the ILCR and HI for each COPC. The risk estimates are rounded to one significant figure when necessary for interpretation. ILCR estimates below 1E-6 are considered insignificant; ILCR estimates between 1E-6 and 1E-4 are considered to fall within a risk management range. ILCR estimates that exceed 1E-4 are generally considered unacceptable. HI estimates that exceed the threshold level of 1 for any target organ raise concern for noncancer health effects. Lead is assessed by comparing average ambient concentrations with cleanup levels established to ensure that blood lead levels remain within an acceptable range.

The results of the SRA for each range are summarized qualitatively in Table 6-159.

7.0 Screening Level Ecological Risk Assessment

This chapter presents the screening-level ecological risk assessment (SLERA) conducted to determine the potential for ecological risks posed by site-related chemicals at the BBGR Ranges.

7.1 Introduction

The BBGR Ranges include Range 18 (Parcel 74Q), Range 20 (Parcel 76 Q-X), Range 23 (Parcel 79Q), Range 25 (Parcels 83Q and 118Q-X), Range 25-East (Parcel 223Q)/Range 28 (Parcel 86Q), Range 26 (Parcel 84Q-X), and the Ranges South of Range 25. This SLERA consists of a description of the habitats in and around the BBGR Ranges, a discussion of the constituents detected in samples collected from environmental media at the BBGR Ranges, a discussion of the conceptual site model, an estimation of the screening-level risk, the identification of the constituents of potential ecological concern, an uncertainty analysis, a discussion of the different lines of evidence used to identify the constituents of potential ecological concern, and a summary of the results and conclusions.

7.2 Environmental Setting

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

The forested areas outside of the "cleared" areas are best characterized as mixed deciduous/coniferous forest. The canopy species typically found in the forested areas surrounding the BBGR Ranges include yellow poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), white oak (*Quercus alba*), and northern red oak (*Quercus rubra*). The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this habitat type.

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

Terrestrial species that may inhabit the area of the BBGR Ranges include opossum, short-tailed shrew, raccoon, white-tail deer, red fox, coyote, gray squirrel, striped skunk, a number of species of mice and rats (e.g., white-footed mouse, eastern harvest mouse, cotton mouse, eastern woodrat, and hispid cotton rat), and eastern cottontail. Approximately 200 avian species reside at FTMC at least part of the year (USACE, 1998). Common species expected to occur in the vicinity of the BBGR Ranges include northern cardinal (*Cardinalis cardinalis*), northern mockingbird (*Mimus polyglottus*), warblers (*Dendroica spp.*), indigo bunting (*Passerina cyanea*), red-eyed vireo (*Vireo olivaceus*), American crow (*Corvus brachyrhynchos*), bluejay

(*Cyanocitta cristata*), several species of woodpeckers (*Melanerpes spp., Picoices spp.*), and Carolina chickadee (*Pocile carolinensis*). Game birds present in the vicinity of the BBGR Ranges may include northern bobwhite (*Colinus virginianus*), mourning dove (*Zenaida macroura*), and eastern wild turkey (*Meleagris gallopavo*). Woodland hawks (e.g., sharp-shinned hawk) were observed in this area during the ecological investigation (December 2002) and are expected to use this area for a hunting ground. A variety of other raptors (e.g., red-tailed hawk, barred owl, and great horned owl) could also use portions of this area for a hunting ground, particularly the fringe areas where the forested areas abut roads and cleared areas. Although the small size and ephemeral nature of the tributaries at the BBGR Ranges limits the potential for the occurrence of piscivorous birds, they may be present during limited periods of the year when precipitation is heavy and the tributaries exhibit flowing water. The piscivorous bird species that may be present in the vicinity of the BBGR Ranges include great blue heron (*Ardea herodias*), green-backed heron (*Butorides striatus*), and belted kingfisher (*Ceryle alcyon*).

In general, the terrain at FTMC supports large numbers of amphibians and reptiles. A Jacksonville State University report entitled *Amphibians and Reptiles of Fort McClellan, Calhoun County, Alabama* 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 (Cline and Adams, 1997). Typical inhabitants of the area surrounding the BBGR Ranges are copperhead (*Agkistrodon contortix*), king snake (*Lampropeltis getulus*), black racer (*Coluber constrictor*), fence lizard (*Sceloporour undulatus*), and six-lined racerunner (*Cnemidophorous sexlineatus*).

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

7.2.1 Range 18 Habitat

Range 18 is comprised of two main habitat types: "cleared" and forested areas. The cleared area comprises the vast majority of Range 18. The entire area of Range 18, including the extensive safety fan is approximately 3,304 acres. The study area of Range 18 is approximately 69.6 acres in size. The overall elevation of Range 18 ranges from approximately 800 to 875 feet amsl, with the highest elevation at the top of the hill which forms the impact zone in the southern portion of the study area. Ground surface is flat in the firing line area with a slight downward slope to the northeast.

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

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

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

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

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

7.2.2 Range 20 Habitat

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

A small, ephemeral tributary of Ingram Creek cuts across Range 20 draining surface water runoff west towards Ingram Creek. The overall elevation of Range 20 ranges from approximately 925 to 1,200 feet amsl. The lowest elevation is found near the Ingram Creek tributary and the highest elevation is at the top of the hill in the impact zone. Ground surface is relatively flat in the M-60 firing point and explosive pit area with a 75-foot decent to the tributary in the infiltration training area and a steep 1,025-foot rise towards the impact zone hillside southeast of the M-60 firing points.

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

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

Ingram Creek and its tributaries in the vicinity of Range 20 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the

ephemeral nature of the small tributary at Range 20, aquatic insects would not be expected to occur in this area for significant periods during any given year. Additionally, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

7.2.3 Range 23 Habitat

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

Snap Lane forms the western boundary of the Range 23 study area. Three tributaries of Ingram Creek are present at the range, carrying runoff water from the hillside east of the range, through the firing lane area, under Snap Lane, and northwards towards Ingram Creek. Two of these tributaries are ephemeral, while the tributary that flows from the central portion of Range 23 generally contains water throughout the year. The elevation of Range 23 increases from approximately 850 to 900 feet amsl at the firing lane area in the western portion of the range, steeply rising beyond the 300-meter target mounds to an elevation of 1,050 feet amsl at the top of the unnamed hill east of the main study area.

Range 23 exhibits both "cleared" and forested areas. The 16 firing lanes are characteristic of cleared areas. The firing lanes are separated from each other by narrow strips of forest. The formerly cleared firing lanes are characteristic of old field, early successional habitat. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these areas. There are significant areas within the firing lanes that are unvegetated, with large areas of bare soil. The narrow strips of forest between the firing lanes can be characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. These forested strips also exhibit dense understory and herbaceous layers. The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis rboretu*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum rboretum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*)

vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

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

Ingram Creek and its tributaries in the vicinity of Range 23 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the ephemeral nature of two of the tributaries at Range 23, aquatic insects would not be expected to occur in these tributaries for significant periods during any given year. Aquatic insects may be present in the central tributary since it is more perennial in nature. However, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

7.2.4 Range 25 Habitat

Range 25 is comprised of two main habitat types: "cleared" and forested areas. The cleared area comprises the vast majority of Range 25. The total area of Range 25, including the extensive safety fan, comprises 1,713 acres. The main study area of Range 25 is approximately 103 acres and is topographically relatively flat. A hillside in the northern portion of the study area forms one of the major impact zones for this range. The Range 25 study area is comprised 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. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. Significant portions of Range 25 remain unvegetated, with large areas of bare soil.

The forested area in the northern portion of Range 25 is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak

dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

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

The small ephemeral tributaries that cross Range 25 are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

7.2.5 Range 25-East Habitat

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

The majority of Range 25 East is forested with mixed deciduous/coniferous forest. The southeastern corner of the study area (which overlaps the firing line area of Range 26) is characteristic of old field, early successional habitat. Shortleaf pine, loblolly pine, white oak,

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

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

7.2.6 Range 26 Habitat

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

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

On the periphery of the study area the habitat is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak

dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

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

7.2.7 Ranges South of Range 25 Habitat

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

The entire study area is forested with a somewhat immature mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate the forest habitat. The study area exhibits dense understory and herbaceous layers. The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

A small ephemeral tributary of Ingram Creek runs through the center of the study area from east-to-west. This drainage features exhibits vegetation that is characteristic of the surrounding upland vegetation, indicating that this ditch only transmits water during periods of significant

precipitation. The small ephemeral tributary that crosses the study area is not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

7.2.8 Ingram Creek Tributaries Habitat

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

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

Although the Ingram Creek tributaries at the BBGR Ranges are ephemeral in nature, they have the potential to support a variety of amphibious species and some small, drought-tolerant fish species. Bullfrog *(Rana catesbeiana)* and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in these small ephemeral tributaries in the vicinity of the BBGR Ranges. Fish species that may be found in the Ingram Creek tributaries in the vicinity of the BBGR Ranges include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*),

and various darters (*Etheostoma spp.*). The shallow nature of these ephemeral tributaries limits their ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the Ingram Creek tributaries in the vicinity of the BBGR Ranges due to the fact that they hold water only during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

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

7.2.9 South Branch Tributaries Habitat

A single tributary to South Branch of Cane Creek occurs within the BBGR Ranges and only at Range 18. This small, ephemeral tributary runs along the western boundary of Range 18 and then transects the range from west-to-east in the northern portion of the range. This small tributary then turns north along the eastern boundary of Range 18 and joins South Branch of Cane Creek immediately north of the Range 18 study area. Surface water runoff from the hills west and south of Range 18 is the major source of water for this tributary. As such, it only conveys water during periods of significant precipitation. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution from groundwater varies according to the amount of precipitation, with an increase when precipitation raises the potentiometric surface.

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

upland vegetation, indicating that this tributary only transmits water during periods of significant precipitation.

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

The small ephemeral tributary to South Branch of Cane Creek within the Range 18 study area is classified as low-quality gray bat foraging habitat. South Branch of Cane Creek, located just east and north of the Range 18 study area, is classified as moderate-quality gray bat foraging habitat. The tributary that is classified as low-quality gray bat foraging habitat is classified as such because it has the potential to provide habitat for aquatic insects, which may be fed upon by the gray bat. However, this drainage feature contains water only during periods of significant precipitation and is dry during the majority of the year. Thus, aquatic insects would not be expected to occur in this ditch for the majority of the year. Additionally, gray bats prefer continuous cover while traveling to and from their foraging habitats and while foraging. Due to historical maintenance activities at Range 18, the forest canopy has been eliminated and only grasses and weeds remain over the majority of Range 18. Thus, the currently existing vegetation at Range 18 does not provide the cover favored by gray bats. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

7.3 Constituents Detected On-Site

The sampling and analysis programs conducted at the BBGR 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 BBGR Ranges are described in Chapter 2.0 of this report. Constituents detected in surface soil at the BBGR Ranges, and sediment and surface water in the drainage features at the BBGR Ranges are summarized in Chapter 4.0 of this report.

In general, inorganic constituents were commonly detected in soils throughout the BBGR Ranges, but organic compounds (i.e., SVOCs and chlorinated pesticides) were less frequently detected and at relatively low concentrations. Antimony, copper, and lead were the most commonly detected inorganic compounds that were detected at elevated concentrations (with respect to ESVs) in surface soil at the BBGR Ranges.

Antimony, copper, and lead were frequently detected at elevated concentrations in surface soil at Range 18. The highest concentrations of these metals were found in soil samples collected from the hillside in the southern portion of the range which acts as an impact zone for Range 18. Arsenic was detected in one out of 16 samples at an elevated concentration, beryllium was detected in three out of 16 samples at elevated concentrations, nickel was detected in one out of 16 samples at elevated concentrations, and zinc was detected in two out of 16 samples at elevated concentrations relative to naturally occurring levels. The maximum concentrations of arsenic, beryllium, nickel, and zinc in surface soil at Range 18 were only slightly elevated relative to ESVs and naturally occurring levels.

The herbicide MCPP was detected in two samples out of 11 at elevated concentrations. The pesticides 4,4'-DDT, beta-BHC, and endrin ketone were all detected in one sample out of 11 samples at elevated concentrations. The explosive compound 2,4-DNT was detected at an elevated concentration in one out of 18 samples. The elevated concentration of 2,4-DNT was from the same sample as the elevated pesticide concentrations. The polynuclear aromatic hydrocarbon (PAH) compounds anthracene and benzo(a)pyrene were also detected at elevated concentrations in one out of 11 samples at the same location (HR-74Q-GP02) as the elevated pesticides and 2,4-DNT. Fluoranthene and pyrene were detected in two out of 11 samples at elevated concentrations with respect to their ESVs. These compounds were infrequently detected at elevated concentrations and detected concentrations were relatively low. Additionally, there was no apparent contamination pattern for these organic constituents.

Antimony, copper, and lead were frequently detected in surface soil at Range 20 at elevated concentrations with respect to their ESVs and naturally occurring levels. No other constituents were detected in surface soil at Range 20 at elevated concentrations.

Antimony, copper, and lead were frequently detected in surface soil at Range 23 at elevated concentrations. The highest concentrations of copper and lead were found in the target mounds within the firing lanes and in samples from the hillside east of the range. This pattern of elevated concentrations is consistent with the use of the site as a firing range.

A number of pesticides and herbicides were detected at elevated concentrations in soil from Range 23, including: 4,4'-DDE, 4,4'-DDT, aldrin, beta-BHC, dieldrin, endrin, gamma-BHC, methoxychlor, and MCPP. These pesticides and herbicides were relatively infrequently detected at elevated concentrations and their detected concentrations only marginally exceeded their respective ESVs. Aroclor-1260 was detected in one out of 13 surface soil samples and the detected concentration only marginally exceeded the ESV. The explosive compounds 2,4-DNT and 2,6-DNT were detected in one sample out of 22 at elevated concentrations. Trichloroethene was also detected in one surface soil sample at an elevated concentration, albeit at only a slightly elevated concentration. All of these organic constituents were infrequently detected at elevated concentrations and their detected concentrations only marginally exceeded their respective ESVs.

Several inorganic constituents in surface soil at Range 25 were frequently detected at elevated concentrations, including: antimony, copper, lead, vanadium, and zinc. The highest concentrations of copper and lead were found in samples collected from the hillside in the northeast portion of the site which acts as an impact area. There was no apparent pattern to the elevated concentrations of antimony, vanadium, and zinc. 4,4'-DDT was detected in two surface soil samples out of 12, albeit at relatively low concentrations.

No constituents in surface soil samples from Range 25-East were detected at elevated concentrations relative to ESVs and naturally occurring levels.

Antimony, copper, and lead were frequently detected in surface soil samples from Range 26 at elevated concentrations relative to ESVs and naturally occurring levels. There was no apparent pattern to the elevated antimony concentrations. The highest concentrations of copper and lead were found in the target mounds and samples from the hillside on the eastern end of the range

that acts as an impact zone. This pattern of elevated concentrations is consistent with the use of the site as a firing range.

The pesticides beta-BHC, dieldrin, and endosulfan sulfate were detected in one out of 11 surface soil samples from Range 26 at elevated concentrations. The herbicide MCPP was detected in one out of 11 surface soil samples. The concentrations of these pesticides and herbicides were only slightly elevated with respect to ESVs. There was no apparent pattern to the elevated concentrations of pesticides or herbicides in surface soil at Range 26.

The PAHs anthracene, benzo(a)pyrene, fluoranthene, pentachlorophenol, phenanthrene, and pyrene were also detected at elevated concentrations in surface soil from Range 26. There was no apparent pattern to the elevated concentrations of PAHs in surface soil at Range 26.

Surface soil samples from the Ranges South of Range 25 exhibited elevated concentrations of antimony, copper, and lead. There was no apparent pattern to the elevated concentrations of these inorganic compounds in surface soil at the Ranges South of Range 25.

No constituents were found at elevated concentrations in surface water in either the South Branch of Cane Creek tributaries or the Ingram Creek tributaries.

No constituents were detected at elevated concentrations in sediment in the South Branch of Cane Creek tributaries.

Sediment from the Ingram Creek tributaries exhibited elevated concentrations of copper and lead. Several organic constituents were also detected at elevated concentrations in Ingram Creek tributary sediments. MCPP, 4,4'-DDT, alpha-BHC, gamma-chlordane, and heptachlor were detected in one or two sediment samples out of nine at elevated concentrations. Acetone was also detected in one out of nine sediment samples at an elevated concentration.

Although ecological receptors are generally not directly exposed to groundwater, groundwater data were compared to surface water screening levels in order to assess potential future ecological impacts due to potential groundwater intrusion into surface water bodies at the BBGR Ranges. No inorganic constituents were detected in groundwater samples from the BBGR Ranges at concentrations that exceeded their respective surface water ESVs and naturally occurring levels. The pesticides 4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, and heptachlor were all detected in a single groundwater sample from Range 25 at concentrations that exceeded their

respective ESVs. Gamma-BHC was detected in a single groundwater sample from Range 18 at an elevated concentration, and alpha-chlordane was detected in two groundwater samples at elevated concentrations. It is important to note that none of these pesticides were detected in any of the surface water samples collected from the South Branch of Cane Creek or Ingram Creek tributaries at the BBGR Ranges.

7.4 Site Conceptual Model

The ecological site conceptual model (SCM) is a schematic diagram of possible exposure pathways and the means by which contaminants are transported from the primary contaminant source(s) to ecological receptors (Figure 7-1). The exposure scenarios include the sources, environmental transport, partitioning of the contaminants 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 BBGR Ranges, and the complete exposure pathways expected to exist at the BBGR Ranges.

7.4.1 Constituent Fate and Transport

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

7.4.1.1 Fate and Transport in Soil

Contaminants in surface soil at the BBGR 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.

Most of the metals and SVOCs in the surface soil at the BBGR 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 BBGR Ranges are closely associated with particulate matter and could be transported from their source areas by fugitive dust generation and entrainment by the wind. Subsequent dispersion by

atmospheric mixing could transport particulate-associated contaminants to other parts of the BBGR 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 BBGR Ranges, based on the presence of unvegetated areas and areas of sparse vegetation within certain areas of these ranges (i.e., impact areas and soil berms).

The transport of surface soil-associated contaminants by surface runoff is another potentially significant transport mechanism. Soil-related contaminants may become entrained by surface runoff during storm events and be transported to drainage ditches, low-lying areas, and the tributaries to South Branch of Cane Creek and Ingram Creek. This transport mechanism appears to be a significant transport mechanism for soil contaminants at the BBGR Ranges because several of the constituents found at elevated concentrations in surface soil (e.g., copper and lead) were also detected at elevated concentrations in sediment within depositional areas in the Ingram Creek tributaries. This evidence points to the probability that surface soil constituents at the BBGR Ranges are being entrained in stormwater runoff and are being transported to drainage features during storm events.

Surface soil contaminants may also be solubilized by rainwater and subsequently transported either to surface water drainage features (e.g., ditches, tributaries) dissolved in stormwater runoff or vertically to subsurface soils and groundwater. Contaminant transport in this manner is dependent upon contaminant solubility and frequency of rainfall. This potential transport mechanism is expected to be minimal for the less soluble surface soil constituents (e.g., metals and SVOCs) because constituents detected at elevated concentrations in surface soil are not routinely found at elevated concentrations in either subsurface soil or groundwater. Also, the affinity of SVOCs to adhere to soil particles minimizes the potential for these compounds to be solubilized and transported off-site.

The more soluble constituents in surface soil (e.g., pesticides and herbicides) could be expected to solubilize more readily and be transported from the surface soil source to groundwater and/or surface water. The soil types (sand, stone, and gravel) in the vicinity of the BBGR Ranges are expected to promote relatively rapid infiltration of rainwater. The groundwater data from the BBGR Ranges show that this transport mechanism may in fact be operational at these ranges. Sporadic, low-level detection of several herbicides and pesticides in groundwater may indicate that these compounds are being solubilized from the soil and transported vertically to the groundwater. Although these more soluble compounds have been detected sporadically in groundwater, they have not been detected in surface water at the BBGR Ranges. This may be

the result of significant dilution in the surface water bodies or it may indicate that groundwater discharge to surface water bodies is insignificant at the BBGR Ranges.

The transfer of contaminants in surface soil to terrestrial plants through root uptake and 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 (ATSDR, 1996b, 1999c). Many of the SVOCs have the potential to bioaccumulate in lower trophic level organisms (i.e., 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 BBGR 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 contaminants (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.

As discussed in the geochemical evaluation (Appendix I), most of the detected element concentrations in BBGR Ranges soil are naturally occurring. The exceptions include lead, antimony, copper, and zinc, which are expected contaminants at the site due to their presence in bullets, shell casings, and jackets.

The distribution coefficients (K_d) of most metals are relatively high – for example, typical K_d values for clays are 550 cubic centimeters per gram (cm^3/g) for cobalt and lead; 650 cm^3/g for nickel; 1,500 cm^3/g for chromium; and 2,400 cm^3/g for zinc (VOCs such as benzene and toluene, in contrast, have K_d values that are three to four orders of magnitude lower) (Sheppard and Thibault, 1990). As a result, these metals are likely to adsorb to site soils.

The BBGR Ranges soil boring logs note that many of the sampled intervals consist primarily of brown, reddish brown, red-orange, and yellowish orange clay. The brown, red, and orange colors of the site soils are imparted by iron oxides, which are highly pigmented and occur as discrete mineral grains or as coatings on silicate minerals (Cornell and Schwertmann, 2003). Clays and iron oxides tend to exist as very fine particles, so both aluminum and iron are enriched in samples with finer grain sizes. Although site-specific mineralogical analyses are not

available, a comparison of aluminum versus iron concentrations can be used to qualitatively assess the relative abundance of clays and iron oxides in site soil (see Appendix I). Although the site samples have higher aluminum and iron concentrations than many of the background samples, they exhibit aluminum/iron ratios that are similar to the background aluminum/iron ratios. These observations indicate that aluminum and iron in the site samples have a natural source, and that many of the site samples are preferentially enriched in clays and iron oxides relative to background.

Clay, iron oxide, and manganese oxide minerals are characterized by large surface areas and strong surface charges, and thus they have an affinity to adsorb specific trace elements (see Appendix I). The prevalence of clays and iron oxides in the site soils, as discussed above, suggests that adsorption is a key factor limiting element mobility at the site.

7.4.1.2 Fate and Transport in Surface Water

In general, constituents present in the surface water associated with the BBGR Ranges (South Branch of Cane Creek and Ingram Creek tributaries) are the result of erosion and runoff from the ranges. Constituents in surface water at the BBGR 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. No VOCs were detected in surface water from the BBGR Ranges; therefore, volatilization is not an important mechanism at these ranges.

Water in the South Branch of Cane Creek and Ingram Creek tributaries originates mainly from overland flow from the surrounding watershed. There also appears to be localized contributions to creek flow from groundwater where the potentiometric surface exceeds the bed surface of the ephemeral tributaries. The flow contribution in these ephemeral tributaries 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 these tributaries. This transport mechanism appears to be relatively insignificant compared to other transport mechanisms at these ranges (i.e., surface runoff).

Constituent transfer to sediments represents another potentially significant transfer mechanism, especially where constituents are in the form of suspended solids, or are hydrophobic substances (i.e., PAHs) that can become adsorbed to organic matter in the sediments. Metals in surface water have the potential to associate with suspended particulate matter. Sequestration of

constituents in sediment is expected to be minimal at the BBGR Ranges due to the lack of organic matter in the sediments of the small ephemeral tributaries. As presented previously, the substrate of the small ephemeral tributaries at the BBGR Ranges is made up mostly of sand and gravel, with very little evidence of fine-grained organic sediment, thus the binding capacity of these sediments is minimal.

Constituents in surface water can be transported to other ranges or off-site during heavy precipitation events via the small ephemeral tributaries. Transfer of contaminants in surface water to aquatic organisms is a potential transfer mechanism; however, due to the ephemeral nature of the drainage features at the BBGR Ranges, this mechanism is expected to be minimal. Because no constituents were detected in surface water samples from the BBGR Ranges at elevated concentrations relative to ESVs and naturally occurring levels, fate and transport of constituents via surface water is not considered significant at the BBGR Ranges.

7.4.1.3 Fate and Transport in Sediment

Constituent transport via suspended sediment in the drainage features at the BBGR Ranges during precipitation events is a potentially significant transport mechanism. Particulates may become entrained in surface runoff and be transported within a given range or they may be transported to other ranges and off-site locations. Transfer between sediment and surface water is a potential transfer mechanism; however, due to the ephemeral nature of the drainage features at the BBGR Ranges, surface water is only present during periods of significant precipitation and this transfer mechanism is expected to be minimal in relation to other transfer mechanisms (i.e., particulate entrainment). Sorbed or settled constituents can be transported with the sediment to downstream locations.

Although transfer of sediment-associated constituents to bottom-dwelling biota represents a potential transfer mechanism, it is not expected to be a major mechanism at the BBGR Ranges. The ephemeral nature of many of the drainage features at the BBGR Ranges limits their ability to support aquatic biota. The vegetation in the majority of these drainage features is characteristic of the upland vegetation in the surrounding area, indicating that these drainage features are dry for the majority of most years. Therefore, the contaminant transfer mechanisms most likely to occur in these drainage features are the same mechanisms that are active in the terrestrial habitats. Due to the perennial nature of South Branch of Cane Creek at the BBGR Ranges, sediment-to-biota transfer is a potentially significant transfer mechanism. Most of the inorganics detected in sediment are not bioaccumulative. 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. Pesticides and herbicides in sediment may be accumulated and magnified through the food chain.

7.4.2 Ecotoxicity

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

7.4.2.1 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).

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

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). Difference in the toxicity of aluminum to plants is closely linked to the 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 Efroymson et al. (1997a) 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). A derived chronic no observable adverse effect level (NOAEL) of 0.043 mg/kg/day has been reported for laboratory rats exposed to aluminum (EPA, 2003e). Laboratory-derived toxicity data from studies conducted with mice fed aluminum (as $AlCl_3$) in their drinking water over three generations were used to derive a NOAEL value of 1.93 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 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). Ringed doves fed aluminum (as $Al_2[SO_4]_3$) for four months in their diet showed no adverse effects at a dose level of 1,000 ppm. This dosage was used to derive a NOAEL value of 109.7 mg/kg/day (Carreire et al., 1986). Reproduction was the critical endpoint in this study.

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 μg/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 μg/L, respectively (Suter and Tsao, 1996). The test 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 EC_{20} value for aluminum is 4,700 μg/L (Suter and Tsao, 1996). The EC_{20} value for aluminum with respect to daphnids is 540 μg/L (Suter and Tsao, 1996).

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

Terrestrial Invertebrates. Three studies were identified that assessed the effects of antimony in soil on invertebrate reproduction (Kuperman et al., 2002; Phillips et al., 2002; and Simini et al., 2002). Kuperman et al. (2002) studied the pot worm (*Enchytraeus crypticus*), Phillips et al. (2002) studied the springtail (*Folsomia candida*), and Simini et al. (2002) studied the earthworm (*Eisenia fetida*). The concentrations of antimony in soil that elicited an adverse effect in 20 percent of the tested populations (EC_{20}) in these three studies were 194, 81, and 30 mg/kg. The geometric mean of these three EC_{20} values is 78 mg/kg, which is the value the EPA (2003f) recommends for use as the ecological soil screening level for antimony.

Mammals. Female mice exposed to 5.0 milligrams per liter (mg/L) antimony (as antimony potassium tartrate) in their drinking water showed a reduction in their lifespan. This dose was equivalent to a lowest-observed-adverse-effects-level (LOAEL) of 1.25 milligrams per kilogram per day (mg/kg/day), which can be converted to a no-observed-adverse-effects-level (NOAEL) of 0.125 mg/kg/day (EPA, 2003e).

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

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

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

7.4.2.3 Arsenic

Arsenic naturally occurs 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 to parts per million (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). In abundance of elements, arsenic ranks 20th 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 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, 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 Efroymson et al. (1997a) 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).

Terrestrial Invertebrates. Fischer and Koszorus (1992) tested the earthworm *Eisenia fetida* for effects on growth and reproduction by exposing five-week old worms to 68 mg/kg arsenic (as potassium arsenate) in soil. Reduced cocoon production was reported at this exposure level. Based on the results of this test, a soil benchmark value of 60 mg/kg has been proposed by Efroymson et al. (1997b).

Mammals. Arsenic is more toxic to wildlife in the 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 arsenic 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 LD_{50}s 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_{50} 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).

Brown-headed cowbirds fed arsenic in their diets for seven months experienced mortality at doses as low as 75 ppm. These data were used to derive a NOAEL value of 2.46 mg/kg/day (USFWS, 1969). Data from a study of mallard ducks fed arsenic in their diets for 128 days were used to derive a NOAEL value of 5.14 mg/kg/day (USFWS, 1964). Mortality was the endpoint for both of these studies.

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 Weaks,

1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The LC_{50} values, for example, are markedly affected by water temperature, pH, Eh, organic content, phosphate concentration, suspended solids, and presence of other substances and toxicants, as well as arsenic speciation, and duration of exposure. In general, inorganic arsenicals are more toxic than organoarsenicals to aquatic biota, 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 machrochirus*) exhibited reduced survival after sixteen weeks when exposed to a single treatment of trivalent arsenic at 0.69 mg/L (EPA, 1980a; 1985a). The 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 cladaceran *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).

7.4.2.4 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 Efroymson et al. (1997a) as a benchmark screening value for barium phytotoxicity.

Terrestrial Invertebrates. Three studies were identified that assessed the effects of barium in soil on invertebrate reproduction (Kuperman et al., 2002; Phillips et al., 2002; and Simini et al., 2002). Kuperman et al. (2002) studied the pot worm (*Enchytraeus crypticus*), Phillips et al. (2002) studied the springtail (*Folsomia candida*), and Simini et al. (2002) studied the earthworm (*Eisenia fetida*). The concentrations of barium in soil that elicited an adverse effect in 20 percent of the tested populations (EC_{20}) in these three studies were 585, 165, and 370 mg/kg. The geometric mean of these three EC_{20} values is 330 mg/kg, which is the value the EPA (2003f) recommends for use as the ecological soil screening level for barium.

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) (EPA, 2003e). Laboratory rat toxicity data for barium chloride in drinking water were used to calculate a NOAEL value of 5.1 mg/kg/day (Perry et al., 1983). Growth and hypertension were the test endpoints. Rats administered barium through oral gavage showed no adverse effects at dosages up to 300 mg/kg/day. These data were used to derive a NOAEL value of 19.8 mg/kg/day (Borzelleca et al., 1988). Mortality was the endpoint for this study.

Birds. A study of one-day-old chicks fed barium in their diet was used to determine the amount of barium to cause mortality. While barium exposures up to 2,000 ppm produced no mortality, chicks in the 4,000 to 32,000 ppm exposure groups experienced 5 to 100 percent mortality. Because 2,000 ppm was the highest nonlethal dose, this dose was considered to be the subchronic NOAEL (Johnson et al., 1960). A NOAEL value of 20.8 mg/kg/day was derived from these data.

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 *Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 110 and 4.0 μg/L, respectively.

7.4.2.5 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, 1985b).

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).

Terrestrial Invertebrates. Three studies were identified that assessed the effects of beryllium in soil on invertebrate reproduction (Kuperman et al., 2002; Phillips et al., 2002; and Simini et al., 2002). Kuperman et al. (2002) studied the pot worm (*Enchytraeus crypticus*), Phillips et al. (2002) studied the springtail (*Folsomia candida*), and Simini et al. (2002) studied the earthworm (*Eisenia fetida*). The concentrations of beryllium in soil that elicited an adverse effect in 20 percent of the tested populations (EC_{20}) in these three studies were 45, 28, and 52 mg/kg. The geometric mean of these three EC_{20} values is 40 mg/kg, which is the value the EPA (2003f) recommends for use as the ecological soil screening level for beryllium.

Mammals. The major exposure route for mammals is inhalation. Beryllium is poorly absorbed from the gastrointestinal tract, and is not absorbed through intact skin to any significant degree. Mammals exposed via inhalation exhibit pulmonary effects which may last long after exposure ceases. Based on animal studies, beryllium is poorly absorbed through both the gastrointestinal tract and the skin. The most important route of exposure for beryllium is inhalation, although absorption by this route does not appear to be extensive. 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).

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, 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 *Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 35 and 0.66 μg/L, respectively.

The 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 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_{20} 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_{20} for daphnids is 3.8 μg/L (Suter and Tsao, 1996).

7.4.2.6 Chromium

Chromium in the trivalent form is required for normal glucose metabolism, 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).

Terrestrial Invertebrates. Abbasi and Soni (1983) exposed the earthworm *Octochaetus pattoni* to chromium (as $K_2Cr_2O_7$) in soil for 60 days to assess the effects on survival and reproduction. Survival was the most sensitive endpoint with a 75 percent decrease resulting from exposure to 2.0 ppm chromium, the lowest concentration tested. The number of cocoons produced was not reduced until the concentration reached 20 ppm chromium (the highest concentration tested). The number of juveniles produced was not affected.

It is difficult to establish a benchmark concentration for chromium based on earthworm toxicity because the relative toxicity of Cr^{+3} and Cr^{+6} is not clear from the available data. Cr^{+6} ions can pass through cell membranes with much greater ease than Cr^{+3} ions. However, it is thought that Cr^{+6} is reduced to Cr^{+3} inside the cell (Molnar et al., 1989). Without a better understanding of chromium transformations in the soil, transport across earthworm cell membranes, and reactions within the cell, it is difficult to separate the effects of the two different forms of chromium. These difficulties not withstanding, a soil benchmark value of 0.4 mg/kg has been suggested by Efroymson et al. (1997b), based on the work of Abbasi and Soni (1983).

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, 1986a).

Laboratory data based on rats exposed to chromium (as Cr_2O_3) 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 $K_2Cr_2O_4$) 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, 1986a).

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 $CrK[SO_4]_2$). 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 have been 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.

7.4.2.7 Cobalt

Cobalt is a natural element and is widely distributed in the earths crust at 0.001 to 0.002 percent (Merck Index, 1983). Small amounts of cobalt are found in rocks, soil, 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 concentration 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 Efroymson et al., (1997a) 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 and white leaf margins and damaged root tips (Kabata-Pendias and Pendias, 1992).

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

Mammals. Cobalt is a component of vitamin B_{12} and, therefore, is an essential micro-nutrient 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 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 were noted (National Research Council [NRC], 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 (NRC, 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.

7.4.2.8 Copper

Copper is ubiquitously distributed in nature in the free state and in sulfides, arsenides, chlorides, and carbonates. Several copper containing proteins have been identified in biological systems as

oxygen binding hemomcyanin, 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 parts per million (ppm) (Moore and Ramamoorthy, 1984).

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

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

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

Terrestrial Invertebrates. The work of Streit and Jaggy (1983) and others shows that the organic carbon content of the soil is a strong determinant of bioavailability and toxicity of copper. Low pH has a compounding effect, with an increase in copper availability resulting from more acid soil conditions. Overall, reproduction is more sensitive than mortality, and there

is no consistent evidence that one genus of earthworms is any less tolerant to copper under a given set of conditions than another genus.

Neuhauser et al. (1984) evaluated the effects of soluble forms of copper on growth and reproduction in *Eisenia fetida*. After six weeks, both growth and cocoon production were decreased by 2,000 ppm copper, while 1,000 ppm copper had no effect. Spurgeon and Hopkin (1995) investigated the effect of copper on survival, growth, and cocoon viability of *Eisenia fetida*. The lowest EC_{50}, 601 ppm, was for growth.

Using artificial soil and a 21-day test procedure, van Gestel et al. (1989) looked at the effects of copper on reproductive parameters of adult *Eisenia andrei*. After 21 days, cocoon production was decreased 36 percent by the addition of 180 ppm copper, while 120 ppm had no effect. Cocoon hatchability and number of juveniles per cocoon were not affected. Based on these studies and a number of others, Efroymson et al. (1997b) have proposed a soil screening level for copper of 60 mg/kg.

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, 1985c). 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 10 species of herbivorous, omnivorous, and carnivorous animals in Donana National Park in Spain (Hernandez et al., 1985). Copper concentrations in small mammals collected from various uncontaminated sites ranged from 8.3 to 13.4 mg/kg (whole-body concentrations) (Talmage and Walton, 1991). Highest concentrations of copper tend to be in hair, followed in decreasing concentration by liver, kidney, and whole body (Hunter and Johnson, 1982). Among the small mammals collected, Hunter and Johnson (1982) found shrews (*Sorex araneus*) to contain the highest concentrations of copper. Mice were found to contain the lowest copper concentrations. Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg-day of copper as copper sulfate (Lecyk, 1980). Increased mortality rates in mink offspring have been observed at levels above 3.21 mg/kg-day (Aulerich et al., 1982).

Laboratory toxicity data for mink exposed to copper sulfate in their diet were used to estimate a NOAEL value of 11.7 mg/kg/day (Aulerich et al., 1982). Reproduction was the endpoint studied. Symptoms of acute copper poisoning in mammals include vomiting, hypotension,

melena, coma, jaundice, and death (Klaassen et al., 1991). Selenium can act as an antidote for copper poisoning.

Birds. Laboratory toxicity data for one-day old chicks exposed to copper oxide in their diets were used to estimate a NOAEL value of 47 mg/kg/day (Mehring et al., 1960). Growth and mortality were the endpoints studied. Reduced growth and mortality were seen at copper concentrations as low as 749 ppm over the 10-week exposure period.

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 that chironomid larval growth was reduced 50 percent (EC_{50}) was 1,602 mg/kg. These researchers found that deformities in larval mouth parts were observed in elevated concentrations, and adult emergence was inhibited when the sediment concentration exceeded 1,800 mg/kg. Carins et al. (1984) reported copper toxicity in sediment for several chironomus midges and cladocerans with LC_{50}s ranging from 681 to 2,296 mg/kg.

7.4.2.9 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 $FeSO_4$) 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 $FeSO_4$ 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 non-heme proteins and is concentrated in chloroplasts. Organic iron complexes are

involved in phtotosynthetic electron transfer. Plant symptoms of toxicity are not specific and differ among plant species and growth stages (Foy et al., 1978).

Terrestrial Invertebrates. The main concern from an ecological risk perspective for iron is not direct chemical toxicity per se, but the effect of iron as a mediator in the geochemistry of other potentially toxic metals (EPA, 2003f). No information was found regarding the toxicity of iron to terrestrial invertebrates.

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. 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 (1992) prepared provincial sediment quality guidelines using the screening-level concentration approach, which 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.

7.4.2.10 Lead

Global production of lead from both smelter and mining operations has been high throughout this century. Lead is commonly used in storage batteries as well as 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 of lead in soil to plants 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 by 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).

Terrestrial Invertebrates. A number of studies have been conducted to assess the potential toxicity of lead to terrestrial invertebrates. Bengtsson et al. (1986) examined the effects of lead on *Dendrobaena rubida* at different soil acidities. After 4 months at pH 4.5, the number of cocoons produced per worm, hatchlings per cocoon, and percent hatched cocoons were reduced 75, 100 and 100 percent, respectively, by 500 mg/kg lead, while 100 mg/kg lead had no effect. At pH 5.5 and 6.5, lead had no adverse effects at any of the lead concentrations.

Spurgeon et al. (1994) kept adult *Eisenia fetida* in contaminated soil for 8 weeks to examine the effects of lead (as $Pb[NO_3]_2$) on survival and growth of the earthworms. After 56 days, the calculated LC_{50} was 3,760 mg/kg and the EC_{50} for cocoon production was 1,940 mg/kg.

Efroymson et al. (1997b) have established a benchmark value of 500 mg/kg for earthworms based on the work of Bengtsson et al. (1986) which showed inhibition of reproduction at this concentration.

Sandifer and Hopkin (1996 and 1997) conducted four studies using Collembola (*Folsomia candida*) to assess the effects of lead on reproduction. The maximum acceptable toxicant concentration (MATC) for each of these studies was 3,162, 3,162, 894, and 894 mg/kg. The geometric mean of these four values (1,682 mg/kg) is the value suggested for use as the ecological soil screening level for lead by EPA (2003f).

Mammals. As with plants, lead is not considered an essential nutrient for mammalian life. Ingestion is the major route of exposure for wildlife. Lead tends to accumulate in bone, hair, and teeth. Biomagnification of lead is negligible (Eisler, 1988b). 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 Mitchner, 1971). Exposure of rats in this same experiment resulted in many early deaths and runts. Blood δ-aminolevulinic acid dehydratase (ALAD) 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 tolerant to moderate lead exposure, including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2,000 ppm (Hoffman et al., 1985).

Toxicity of lead to birds is dependent upon the form of lead, the route of exposure and exposure duration, and the species and age of the bird. 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 is 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 mat 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_{50}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_{50} of 4.89 ppm.

7.4.2.11 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 manganese 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 foliar uptake (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. (1997a) 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).

Terrestrial Invertebrates. No information was found regarding the potential toxicity to terrestrial invertebrates from exposure to manganese.

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 mg/m^3 for five hours per day, five days a week, for 100 days (Klaassen et al., 1991).

Birds. Japanese quail were studied by Laskey and Edens (1985) to determine the toxic effects of 5,000 ppm manganese fed to one-day old chicks. Growth and aggressive behavior were studied throughout the 75-day exposure period. While no reduction in growth was observed, aggressive behavior was reduced in 25 to 50 percent of the chicks relative to controls. However, reduced aggressive behavior was not considered a significant adverse affect. A NOAEL value of 977 mg/kg/day was derived from this study.

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_{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 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_{20} 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_{20} for daphnids is less than 1,100 µg/L (Suter and Tsao, 1996).

7.4.2.12 Mercury

Mercury is a toxic compound with no known natural biological function. Mercury exists in three valence states: mercuric (Hg^{2+}), mercurous (Hg^{1+}), and elemental (Hg^{0+}) mercury. It is present in the environment in inorganic and organic forms. Inorganic mercury compounds are less toxic than 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. (1997a) as a benchmark screening value for mercury phytotoxicity. General symptoms of mercury toxicity in plants include severe stunting of seedlings and roots and leaf chlorosis and browning of leaf points (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Abbasi and Soni (1983) exposed earthworms (*Octochaetus pattoni*) to mercury (as HgCl) to assess the effect on reproduction and growth. Survival and cocoon production were reduced at 0.5 ppm mercury, the lowest concentration tested. The

number of juveniles produced was not affected. Based on these test data, a benchmark value for mercury in soil of 0.1 mg/kg has been proposed by Efroymson et al. (1997b).

Mammals. Mercury is not an essential element for animal life. Background mercury concentrations in wildlife tend to be less than 1.0 mg/kg (wet weight) (Eisler, 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 et al., 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).

Japanese quail were fed mercury in their diet for one year to study the effects on reproduction. Egg production increased with increasing mercury dose, while fertility and hatchability

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

Aquatic Life. Concentrations of mercury in freshwater fish collected from 12 monitoring stations in the United States from 1978 to 1981 ranged from 0.1 to 1.1 mg/kg (wet weight), with an average of 0.11 mg/kg (Lowe et al., 1985). Elevated concentrations of mercury in fish have often been associated with low pH, low calcium concentrations in the water, and low water hardness (Eisler, 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 μg/L, respectively (EPA, 1999a). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} value for methylmercury is less than 0.03 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids has been determined to be 0.87 μg/L (Suter and Tsao, 1996).

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

7.4.2.13 Nickel

Nickel is a naturally occurring silvery metal that is found in the Earth's crust in the form of various nickel minerals. Exposure of organisms to nickel and its compounds results from

breathing air, ingesting water and food that contain nickel and compounds, and skin contact with a media contaminated with nickel.

Plants. Nickel is not believed to be an essential element for plant growth; however, beneficial effects of nickel have been reported on the growth of legumes. Background concentrations of nickel in grasses and clovers collected in the United States averaged 0.13 and 1.5 mg/kg, respectively (Kabata-Pendias and Pendias, 1992). The concentration of nickel in plants is positively correlated with nickel concentrations in soil.

Concentrations of nickel in leaf tissue that are excessive or toxic to plant species range from 10 to 100 m/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 30 mg/kg has been proposed by Efroymson, et al. (1997a) as a benchmark screening value for nickel phytotoxicity. General symptoms of nickel toxicity in plants include the presence of interveinal chlorosis in new leaves, gray-green leaves, and brown and stunted root and plant growth. The uptake of nutrients and minerals, especially iron, can be substantially reduced as a consequence of nickel toxicity in plants (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. The effects of nickel on *Eisenia fetida* were investigated by Malecki et al. (1982). The most sensitive endpoint was cocoon production. In the 8-week test, 300 mg/kg nickel caused a 41 percent decrease in cocoon production, while 200 mg/kg had no adverse effect. In the 20-week test, 200 mg/kg nickel caused a 23 percent decrease in cocoon production, while 100 mg/kg had no effect.

Neuhauser et al. (1984) evaluated the effects of soluble forms of nickel on growth and reproduction in *Eisenia fetida*. After 6 weeks cocoon production was decreased 33 percent by 250 mg/kg nickel, the lowest concentration tested. Growth was not affected until 500 mg/kg nickel was added to the substrate.

Efroymson et al. (1997b) have established a soil benchmark value for nickel of 200 mg/kg based on the work of Malecki et al. (1982), which showed inhibition of reproduction at this concentration.

Mammals. Nickel is a nonessential element for animal life. Nickel concentrations within the whole bodies of small mammals from uncontaminated sites were reported to range from 2.2 to 6.2 mg/kg (dry weight) (Talmage and Walton, 1991). Highest concentrations were measured in the deer mouse (*Peromyscus maniculatus*). Highest tissue concentrations of nickel are usually

found in the liver of mammals (Schroeder et al., 1964). Because nickel is poorly absorbed by the gastrointestinal tract, ingested nickel is generally not of great toxicological concern. Inhaled nickel, however, is relatively toxic. Rats fed nickel in their diet as nickel sulfate hexahydrate over three generations were studied for effects on reproduction. They were fed three dose levels (250, 500, and 1,000 ppm Ni) in their diet, and only the highest dose level caused reduced offspring body weights. No adverse effects were observed at the other dose levels. Because this study considered exposures over multiple generations, the 500 ppm dose was considered to be the chronic NOAEL, and the 1,000 ppm dose was considered to be the chronic LOAEL (EPA, 1999b). These data were used to derive a NOAEL value of 40 mg/kg/day (Ambrose et al, 1976). Reproduction was the endpoint studied.

Birds. Mallard ducklings were fed nickel as nickel sulfate in their diet for a duration of 90 days to study the effects on mortality, growth, and behavior. They were fed three dose levels (176, 774, and 1,069 ppm Ni), and only the highest dose reduced growth and resulted in 70 percent mortality. Because the study considered exposure over 90 days, the 774 ppm dose was considered to be the chronic NOAEL, and the 1,069 dose was considered to be the chronic LOAEL (Cain and Pafford, 1981). A NOAEL value of 77.4 mg/kg/day was derived from this study based on mortality, growth, and behavior as endpoints.

Aquatic Life. The bioavailability and toxicity of nickel to aquatic biota is influenced by the pH of the water (Schubauer-Berigan et al., 1993). The national recommended water quality criteria for the protection of aquatic life for acute and chronic exposure are 470 and 52 μg/L, respectively (EPA, 1999a). Background concentrations of nickel in adult anurans ranged between 0.9 and 2.9 mg/kg (dry weight) (Hall and Mulhern, 1984).

The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} value for nickel is 62 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids has been determined to be 45 μg/L (Suter and Tsao, 1996).

7.4.2.14 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, Eh, 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).

Terrestrial Invertebrates. Fischer and Koszorus (1992) tested the effects of 77 ppm of selenium on growth and reproduction of *Eisenia fetida* when added to a combination of peaty marshland soil and horse manure (1:1). The number of survivors and their live mass and number of cocoons produced were measured. The number of cocoons produced per worm showed the highest sensitivity to selenium with a 69 percent reduction at a selenium concentration of 77 ppm. Efroymson et al. (1997b) have proposed a soil benchmark value of 70 ppm based on this study.

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 (SeO_4) 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. While consumption of 1, 5, or 10 ppm selenium in the diet of mallard ducks had no effect on weight or survival of adults, 100 ppm selenium reduced adult survival and 25 ppm selenium reduced duckling survival. Consumption of 10 or 25 ppm selenium in the diet resulted in significantly larger frequency of lethally deformed embryos compared to the 1 or 5 ppm selenium exposures. These data were used to derive a NOAEL value of 0.5 mg/kg/day, with reproduction as the critical endpoint (Heinz et al., 1987).

Screech owls fed selenium in their diets were studied for the effects on reproduction. While exposure of owls to 0.44 mg/kg/day of selenomethionine had no adverse effects on reproduction, exposure to 1.5 mg/kg/day reduced egg production, hatchability, and nestling survival (Wiemeyer and Hoffman, 1996 as cited in Sample et al., 1996). A NOAEL value of 0.44 mg/kg/day was derived from these data based on reproductive effects.

Black-crowned night herons fed selenium their diets for 94 days were studied for the effects on reproduction. Exposure to 1.8 mg/kg/day selenium had no adverse effects on reproduction.

These data were used to derive a NOAEL value of 1.8 mg/kg/day based on reproduction as the critical endpoint (Smith et al., 1988 as cited in Sample et al., 1996).

Aquatic Life. Selenium is an essential micro-nutrient 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_{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 the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} 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_{20} benchmark for daphnids is 25 μg/L selenium (Suter and Tsao, 1996).

7.4.2.15 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 mater 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., (1997a) as a benchmark screening value for silver phytotoxicity.

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

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 lowest observable adverse effect level (chronic) for rats (EPA, 2003e).

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_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} value for silver is 0.2 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and

survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids has been determined to be less than 0.56 µg/L (Suter and Tsao, 1996).

7.4.2.16 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).

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

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 for 60 days were used to estimate a NOAEL value of 0.0074 mg/kg/day (Formigli et al., 1986). Reduced sperm motility in males 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 beryllium, as calculated by the method described in the EPA's *Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 110 and 12 μg/L, respectively.

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

7.4.2.17 Vanadium

Vanadium, a white to gray metal, occurs naturally in fuel oils and coal. It is used as a catalyst in the production of various chemicals including sulfuric acid. It is also used in the hardening of steel, the manufacture of pigments, and photography. The general population and many ecological receptors are exposed to background levels of vanadium primarily through ingestion of food.

Plants. There is some controversy over whether vanadium is an essential element for plants (Kabata-Pendias and Pendias, 1992). It appears to be required by some algal species and may be required by nitrogen-fixing bacteria. Mean background concentrations of vanadium in plants are 1.6 mg/kg for angiosperms, 0.69 mg/kg for gymnosperms, and 0.67 mg/kg for fungi (Waters, 1977). The availability of vanadium to plants is highly dependent on soil pH. Elevated levels of vanadium in soil can reduce the uptake of manganese, copper, calcium, and phosphorus (NRCC, 1980).

Concentrations of vanadium 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. (1997a) as a benchmark screening value for vanadium phytotoxicity.

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

Mammals. Vanadium has been shown to be essential in the diets of rats (Waters, 1977). Background concentrations of vanadium in the kidneys and livers of wild mammals have been reported to range from 0 to 2.07 mg/kg, and from 0 to 0.94 mg/kg, respectively (Waters, 1977). Liver and skeletal tissues usually contain the highest concentrations of vanadium (Waters, 1977). Experimental animal investigations have suggested that the liver, adrenal, and bone marrow may be adversely affected by subacute exposure to high levels of vanadium (ATSDR, 1992c; Klaassen et al., 1991). Vanadium fed to rats prior to gestation, during gestation, and through delivery and lactation was studied for effects on reproduction. The rats were fed three dose levels (5, 10, and 20 mg $NaVO_3$/kg/day or 2.1 mg V/kg/day). Significant differences in reproductive parameters (e.g., number of dead young per litter, size and weight of offspring) were observed at all dose levels. Therefore, the lowest dose was considered to be a chronic LOAEL. A chronic NOAEL was estimated by applying an uncertainty factor of 0.1 (chronic NOAEL = 0.21 mg V/kg/day) (EPA, 1999b).

Based on oral intubation exposure of rats to sodium metavanadate by Domingo et al. (1986), an estimated NOAEL value of 0.21 mg/kg/day has been derived. Reproduction was the endpoint for this study. Signs of acute toxicity in animals include alterations in nervous system responses, gastrointestinal distress, hemorrhaging, paralysis, convulsions, and respiratory depression (Klaassen et al., 1991).

Birds. Mallard ducks were fed vanadium as vanadyl sulfate in their diet for 12 weeks and observed for effects on mortality, body weight, and blood chemistry. The ducks were fed three different doses (2.84, 10.36, and 110 ppm V). No effects were observed at any of the dose levels. Because this study was greater than ten weeks in duration and did not consider a critical lifestage, the maximum dose was considered to be a chronic NOAEL (White and Dieter, 1978). From these data a NOAEL value of 11.4 mg/kg/day has been estimated.

Aquatic Life. Background concentrations of vanadium in freshwater fish are usually less than 2.5 mg/kg (wet weight) (Jenkins, 1980). A bioconcentration factor of 3,000 has been reported for aquatic invertebrates exposed to vanadium (Neumann, 1976). No federal ambient water quality criteria exist for the protection of freshwater biota (EPA, 1999a). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC_{20} value for vanadium is 41 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_{20} benchmark for daphnids has been determined to be 430 μg/L (Suter and Tsao, 1996).

7.4.2.18 Zinc

Zinc is a naturally occurring element which 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. 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).

Terrestrial Invertebrates. Spurgeon and Hopkin (1996) exposed the earthworm *Eisenia fetida* to zinc in soils with differing organic matter content and soil pH. The EC_{50} concentrations for cocoon production in soils with a pH of 7.0 and 5 percent, 10 percent, and 15 percent organic matter were 136, 462, and 592 mg/kg, respectively. The EC_{50} concentrations for cocoon production in soils of pH 6.0 and 5 percent, 10 percent, and 15 percent organic matter were 199, 343, and 548 mg/kg, respectively. The EC_{50} concentrations for cocoon production in soils of pH 5.0 and 5 percent, 10 percent, and 15 percent organic matter were 142, 189, and 230 mg/kg, respectively. Mortality was observed at higher zinc concentrations. A decrease in pH and/or organic matter content in soil led to a lower toxic concentration of zinc.

Neuhauser et al. (1985) determined an LC_{50} for zinc in soil using adult *Eisenia fetida* exposed for 14 days. The calculated LC_{50} was 662 mg/kg. Data from the preceding studies were used to derive a soil benchmark value for zinc in soil of 100 mg/kg (Efroymson et al., 1997b).

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 to 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 for 44 weeks 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 accumulation (Moore and Ramamoorthy, 1984). Rainbow trout (*Oncorhyncus mykiss*) have the ability to detect and avoid

areas of water containing 5.6 ppb zinc (Sprague, 1968). Cairns and Scheier (1968) reported 96-hour LC_{50}s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC_{50}s ranging from 2.86 to 3.78 ppm in soft water. These results demonstrate that water hardness affects the toxicity of zinc to fish. Chronic toxicity tests have been conducted with five species of freshwater fish. Chronic values ranged from 47 µg/L for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinallis*) (EPA, 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_{50} for the cladaceran *Daphnia hyalina* was 0.055 mg/L, and 5.5 mg/L for the copepod *Cyclops abyssorum*. Four chronic toxicity tests are reported for *Daphnia magna*, with chronic values ranging from 47 µg/L to 136 µg/L (EPA, 1980c). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1980c).

7.4.2.19 Acetone

Acetone is a highly volatile organic compound. It exists almost exclusively in the vapor phase in the atmosphere. Volatilization and biodegradation are the major fate processes affecting acetone released to soil, surface water, or sediment. Routes of exposure or wildlife include ingestion, inhalation, and dermal uptake. Acetone is not bioconcentrated in aquatic organisms, and is not bioaccumulated by mammals or birds. Therefore, it does not bioaccumulate or bioconcentrate in aquatic or terrestrial food chains (EPA, 1999b).

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

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

Mammals. Acetone is a highly volatile compound and may be inhaled in large quantities. Acetone is very water soluble, so it is quickly absorbed following inhalation into the blood stream and dispersed throughout the body. A large portion of acetone is excreted primarily unchanged through the lungs and urine, with only a small portion reduced and excreted as carbon dioxide. Because acetone is quickly eliminated, wildlife receptors will not accumulate it in tissues (EPA, 1999b).

Rats exposed to 100, 500, and 2,500 mg/kg/day acetone via oral intubation exhibited significant liver and kidney damage at 500 and 2,500 mg/kg dose levels. Because no significant differences were observed at the 100 mg/kg/day dose level and the study considered exposure for 90 days and did not include critical life stages (i.e., reproduction), this dose was considered to be the subchronic NOAEL. The chronic NOAEL was estimated to be 10 mg/kg/day based on an uncertainty factor of 0.1 (EPA, 1986).

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

Aquatic Life. In most aquatic systems, acetone will exist in water rather than sediment, due to acetone's high water solubility and low sediment adsorption coefficient. Bioaccumulation does not occur in aquatic organisms as suggested by its low log K_{ow} value (Rathburn et al., 1982). Adult haddock tested under static conditions at 7.9 °C showed a bioconcentration factor of 1.0 for acetone (Rustung et al., 1931). Biomagnification along the aquatic food chain is also considered insignificant for acetone as suggested by its low K_{ow} value.

The Cladoceran, *Daphnia magna* exposed to acetone concentrations of 15,072 mg/L exhibited reduced survival, narcosis, and death (Pawlisz and Peters, 1993). The Tier II Secondary Acute and Chronic Values for acetone calculated using the methods proposed in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995) were 28,000 and 1,500 µg/L, respectively. The lowest test EC_{20} for fish (which has been extrapolated from 96-hour LC_{50} values using equations from Suter and Tsao, 1996) has been estimated to be 161,867 µg/L. The secondary chronic value of 1,500 µg/L has been proposed as a preliminary remediation goal for acetone in surface water (Efroymson et al., 1997a).

7.4.2.20 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.

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

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

Mammals. Oral LD_{50} 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 6 weeks exhibited an increase in mortality at doses of aldrin of 8 mg/kg/day (National Cancer Institute, 1978). When exposed for 2 years of more, rats exhibited decreased survival at doses of 2.5 – 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 1 – 5 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.

Data from a laboratory study using rats fed aldrin in their diet over three generations were used to derive a NOAEL value of 0.2 mg/kg/day (Treon and Cleveland, 1955). Reproduction was the critical 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 μg/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 μg/L for carp to 2,735 μg/L for mosquito fish (Verschueren, 1983).

7.4.2.21 Hexachlorocyclohexane

Hexachlorocyclohexane (HCH), also known as benzene hexachloride (BHC), is a synthetic chemical that exists in eight different isomers. Lindane is an organochlorine insecticide and fumigant which has been used on a wide range of soil-dwelling and plant-eating insects. Technical grade lindane is comprised of the gamma-isomer of HCH. Five other isomers of HCH are commonly found in technical grade lindane, but the gamma isomer is the predominant one, comprising at least 99 percent of the mixture of isomers. Gamma-HCH has been shown to be the insecticidally effective isomer (Extension Toxicology Network [EXTOXNET], 2003).

Plants. No data on the toxicity of HCH to plants was found in the literature.

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

Mammals. Lindane is readily absorbed through the gastrointestinal tract, skin and lungs. Lindane is a moderately toxic compound via oral exposure, with a reported oral LD_{50} of 88 to 190 mg/kg in rats. Other reported LD_{50} values are 59 to 562 mg/kg in mice, 100 to 127 mg/kg in guinea pigs, and 200 mg/kg in rabbits. Gamma-HCH is generally considered to be the most acutely toxic of the isomers following single administration. It is moderately toxic via the dermal route as well, with reported dermal LD_{50} values of 500 to 1,000 mg/kg in rats, 300 mg/kg in mice, 400 mg/kg in guinea pigs, and 300 mg/kg in rabbits. Effects of acute exposure to lindane may include central nervous system stimulation, mental/motor impairment, excitation, clonic (intermittent) and tonic (continuous) convulsions, increased respiratory rate and/or failure, pulmonary edema, and dermatitis. Chronic exposures of animals to lindane have caused kidney, pancreas, testes, and nasal mucous membrane damage (EXTOXNET, 2003).

A NOAEL value of 0.4 mg/kg/day has been derived for beta-BHC based on a study conducted with rats fed beta-BHC in their diets (Van Velsen et al., 1986). The critical endpoints in this study were growth, blood chemistry, and organ histology. Studies using mink and rats fed mixed isomers of BHC in their diets were used to derive NOAEL values of 0.014 and 1.6 mg/kg/day, respectively (Bleavins et al., 1984; and Grant et al., 1977).

Birds. Lindane is moderately toxic to bird species, with a reported LD_{50} of more than 2,000 mg/kg for the mallard duck. The 5-day dietary LC_{50} of lindane in Japanese quail is 490 ppm. The LC_{50} values for lindane in pheasant and bobwhite quail are 561 and 882 ppm, respectively. Egg-shell thinning and reduced egg production has occurred in birds exposed to lindane.

Lindane can be stored in the fat of birds. Residues can also find their way into egg yolks at measurable concentrations for 32 days after dosing (EXTOXNET, 2003).

A NOAEL value of 0.56 mg/kg/day has been derived for mixed isomers of BHC based on a study with Japanese quail fed BHC in their diet (Vos et al., 1971). The critical endpoint for this study was reproduction.

Mallard ducks were exposed to 20 mg/kg/day lindane via oral intubation for 8 weeks and studied for adverse effects on reproduction (Chakravarty and Lahiri, 1986). Mallards exposed to 20 mg/kg/day displayed reduced eggshell thickness, laid fewer eggs, and had longer time intervals between eggs. Based on these data, a NOAEL value of 2.0 mg/kg/day was derived.

Aquatic Life. Lindane is highly toxic to fish and aquatic invertebrate species. Reported 96-hour LC_{50} values range from 1.7 to 90 μg/L in trout (rainbow, brown, and lake), coho salmon, carp, fathead minnow, bluegill, largemouth bass, and yellow perch. Reported 96-hour LC_{50} values for aquatic invertebrates were: Daphnia, 460 μg/L; scuds, 10-88 μg/L; stone flies, 4.5 μg/L. The bioconcentration factor for lindane is 1,400 times ambient water concentrations, indicating significant bioaccumulation (EXTOXNET, 2003).

The National Recommended Water Quality Criterion maximum concentration (CMC) for gamma-BHC is 0.95 μg/L (EPA, 1999a). The Tier II secondary acute and secondary chronic values for mixed isomers of BHC are 39 and 2.2 μg/L, respectively (Suter and Tsao, 1996). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of BHC 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_{20} value for gamma-BHC for fish is less than 1.1 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration of BHC causing less than 20 percent reduction in the growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids is 11 μg/L (Suter and Tsao, 1996).

7.4.2.22 Chlordane

Commercial use of chlordane, a chlorinated pesticide, has been banned in the United States since 1988 (EPA, 1988). Technical chlordane is a mixture of some 50 different compounds, the major constituents being alpha-, beta, and gamma-chlordane (Howard, 1991). Chlordane is generally immobile in soil; however, movement to groundwater can occur (Howard, 1991). The

compound is biodegraded at a very slow rate (Howard, 1991). In aquatic systems, chlordane is not expected to undergo significant hydrolysis, oxidation, or direct photolysis (Howard, 1991). Because of its widespread use in the past and its low biodegradation rate, chlordane is commonly measured in low concentrations in environmental samples (Eisler, 1990). Chlordane, with an organic carbon-sediment partition coefficient (K_{oc}) of between 15,500 and 24,600, strongly adsorbs to sediment (Howard, 1991).

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

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

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

Acute oral LD_{50} values for exposure of sensitive mammalian species to technical grade chlordane range from 25 to 50 mg/kg body weight. Chlordane fed to rats at 2.5 mg/kg caused slight liver damage (NRC, 1977). Growth retardation and liver damage were observed in rats fed 150 and 300 mg/kg chlordane in their diets over a two-year period (Clayton and Clayton, 1982). No measurable effects were found in Cynomolgus monkeys (*Macaca spp.*) exposed to technical grade chlordane concentrations of 10 µg/L in air for 90 days (Khasawinah et al., 1989). Based on a study of mice fed technical grade chlordane in their diets, a NOAEL value of 4.6 mg/kg/day has been derived (Keplinger et al., 1968). The critical endpoint for this study was reproduction.

Symptoms of acute chlordane poisoning in mammals include diarrhea, avoidance of food and water, hair loss, hunched appearance, abdominal distension, labored respiration, salivation, muscle tremors, incoordination, convulsions, and death in some cases (Eisler, 1990). Chlordane

has been reported to be carcinogenic in mice (Eisler, 1990) and teratogenic in rats (NRC, 1977). Chlordane potentiates aldrin, endrin, and parathion toxicity in mice (Jones et al., 1977).

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

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

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

Variations in the response of aquatic organisms to chlordane are related to both biological factors of the biota and physical factors of chlordane and the aquatic environment. Younger life-stages are more sensitive to chlordane toxicity than older life-stages (Eisler, 1990). In addition, the health and lipid content of the organism can have an impact on the response (Eisler, 1990).

Water temperature, salinity, and sediment loading are abiotic factors that influence the bioavailability of chlordane (NRCC, 1975a). Cis-chlordane appears to be more toxic than trans-chlordane, and the photoisomers seem to be more toxic than the parent compounds (Eisler, 1990).

The EPA's National Ambient Water Quality Criteria for chlordane is 2.4 μg/L for acute exposure and 0.0043 μg/L for chronic exposure of aquatic life (EPA, 2003e). The water quality criteria for chlordane represent the sum of alpha-chlordane, gamma-chlordane, alpha-chlordene, gamma-chlordene, alpha-nonachlor, gamma-nonachlor, and oxy-chlordane. The lowest chronic toxicity values for chlordane reported in the literature for fish and daphnids are 1.6 and 16 μg/L, respectively (Suter and Tsao, 1996).

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

7.4.2.23 4,4'-DDT and Metabolites

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

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

Terrestrial Invertebrates. No information was found regarding the toxicity of DDT, DDD, or DDE to terrestrial invertebrates.

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

The most significant exposure route for vertebrates to DDT and its metabolites is oral exposure. Dermal exposure is believed to be very limited, and inhaled DDT and associated particulates are believed to be deposited in the upper respiratory tract and eventually swallowed. The toxicity of DDT and its metabolites to mammalian and avian wildlife is dependent on the fat content within the animal. In both mammals and birds the storage of DDT and DDE in fat is protective, because it decreases the amount of chemical in circulation that may reach the brain, which is the site of toxic action (NLM, 1996).

Data from a study of rats fed DDT in their diets for 2 years were used to derive a NOAEL value of 0.8 mg/kg/day (Fitzhugh, 1948). The critical endpoint in this study was reproduction. Signs of acute DDT poisoning in animals include paresthesia of the tongue, lips and face; apprehension; dizziness; tremor; disturbed equilibrium; and convulsions (Klaassen et al., 1991).

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

Aquatic Life. DDT in freshwater environments partitions primarily into sediment. Biodegradation of DDT in sediment may be significant. Bioconcentration factors between 51,000 and 100,000 have been reported for fathead minnows exposed to DDT (NLM, 1996). The Federal Water Quality Criteria for the protection of aquatic life for acute and chronic exposures to DDT and its metabolites in freshwater systems are 1.1 and 0.001 μg/L, respectively EPA, 1999a).

7.4.2.24 Dieldrin

Dieldrin is a chlorinated insecticide that is no longer registered for general use (NLM, 1996). Dieldrin is extremely persistent in the environment; however, photodegradation and evaporation of the compound may occur (Howard, 1991). In aquatic systems, dieldrin will preferentially adsorb to sediments and potentially bioconcentrate. Biodegradation in aquatic and terrestrial systems is not significant (Howard, 1991).

Plants. Dieldrin has been measured in a variety of crop plants with mean concentrations between 0.01 to 0.21 mg/kg reported (Carey et al., 1978). Concentrations of dieldrin in domestically grown leaf and stem vegetables are reported to average 0.000g mg/kg (Duggan et al., 1983). Information on the phytotoxicity of dieldrin could not be found in the literature.

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

Mammals. Dieldrin, like other chlorinated organic compounds, is stored in adipose tissue, liver, brain, and the muscle of mammals (International Agency for Research on Cancer, 1974). Fecal and urinary metabolites of dieldrin include trans-4,5-dihydroxy-4,5-dihydroaldrin and 4,5-seco-aldrin-4,5-dicarboxylic acid, respectively (The Chemical Society, 1975).

Information on the toxicity of dieldrin to wild animals is limited. Dieldrin is a central nervous system, liver, and kidney toxin (Sittig, 1985). Whitetail deer given dieldrin at 5 or 25 mg/kg for three years did not show signs of overt intoxication (Murphy and Korschgen, 1970). The growth of juvenile females was slower among the exposed deer than among the control animals (Murphy and Korschgen, 1970). The oral LD_{50} for mule deer exposed to dieldrin is between 75 and 150 mg/kg (Hudson et al., 1984). Oral LD_{50} values for laboratory mice, rabbits, and dogs are 38, 45, and 65 mg/kg/day, respectively (Registry of Toxic Effects of Chemical Substances [RTECS], 1996). Concentrations of dieldrin between 9 and 11 mg/kg have been associated with convulsions in exposed rats (Harr et al., 1970). Dieldrin has been shown to be both fetotoxic and treatogenic (RTECS, 1996). Dieldrin has also been shown to adversely affect fertility in male and female raccoons (NRC, 1977).

Based on a study involving rats fed dieldrin in their diets for 3 generations, a NOAEL value of 0.02 mg/kg/day has been derived (Treon and Cleveland, 1955). The critical endpoint for this study was reproduction.

Birds. Dieldrin has been detected in the eggs of birds. Concentrations of dieldrin measured in great blue herons from the Great Lakes ranged from 0.14 to 14 mg/kg (Ohlendorf et al., 1981). A nationwide survey of starlings conducted from 1970 to 1976 reported an average dieldrin concentration in starlings of 0.117 mg/kg (White, 1979).

Among the birds tested, California quail (*Callipepla californica*) are the most sensitive to dieldrin with an acute oral LD_{50} of 8.78 mg/kg/day (Hudson et al., 1984). The oral LD_{50} for mallards exposed to dieldrin is 381 mg/kg/day (Hudson et al., 1984). Based on a study by Mendenhall et al. (1983) using barn owls (*Tyto alba*) fed dieldrin in their diets for two years, a NOAEL value of 0.077 mg/kg/day has been derived. The critical endpoint for this study was reproduction. Signs of dieldrin poisoning in birds include tail feathers spread and pointed upward or downward, closed eyelids, fluffed feathers, immobility, hyperexcitability, ataxia, dyspnea, myasthemia, and terminal wing-beat convulsions (Hudson et al., 1984).

Aquatic Life. Concentrations of dieldrin in domestic fish collected from 1970 to 1976 averaged 0.045 mg/kg (Duggan et al., 1983). Bioconcentration of dieldrin in aquatic species is reported to range from moderate to significant (100 to 10,000) (Callahan et al., 1979). Bioconcentration factors in fish range from 3 to 6,000 (Sanborn and Yu, 1973). According to Reynolds and Prasad (1982), fish are highly susceptible to dieldrin poisoning. Ninety-six hour LC_{50} values for freshwater fish generally range between 1.2 and 19 μg/L (NLM, 1996). LC_{50} values for *Chironomus tentans* larvae and frog and tadpoles (*Pseudacris triseriata*) are 0.9 and 100 μg/L, respectively (EPA, 1980d).

The National Recommended Water Quality Criteria (EPA, 1999a) for the protection of freshwater organisms for dieldrin are 0.24 and 0.056 μg/L for the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC), respectively.

7.4.2.25 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).

Plants. Dinitrotoluenes are expected to be readily taken up by plants, based on structural analogies with 1,3-dinitrobenzene and p-nitrotoluene (McFarlane et al., 1987).

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

Mammals. DNT is absorbed through the gastrointestinal tract, respiratory tract, and skin in most species (EPA, 2003e). 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_{50} 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, 1999a). The criterion to protect freshwater aquatic life as derived using procedures other that 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).

7.4.2.26 Endosulfan I and Endosulfan Sulfate

Endosulfan is a chlorinated hydrocarbon insecticide and acaricide of the cyclodiene subgroup which acts as a poison to a wide variety of insects and mites on contact. Although it may also be used as a wood preservative, it is used primarily on a wide variety of food crops, including tea, coffee, fruits and vegetables, as well as on rice, cereals, maize, sorghum, and other grains. It is compatible with many other pesticides and may be found in formulations with dimethoate, malathion, methomyl, monocrotophos, pirimicarb, triazophos, fenoprop, parathion, petroleum oils, and oxine-copper. Technical endosulfan is made up of a mixture of two molecular forms of endosulfan, the alpha- and beta-isomers.

Endosulfan is moderately persistent in the soil environment, with a reported average half-life of 50 days (EXTOXNET, 2003). The two isomers have different degradation times in soil. The half-life for the alpha-isomer is 35 days, and is 150 days for the beta-isomer under neutral conditions. Endosulfan does not easily dissolve in water and has a very low solubility. It has a moderate capacity to adhere or adsorb to soils. Transport of this pesticide is most likely to occur if endosulfan is adsorbed to soil particles in surface runoff. It is not likely to be very mobile or to pose a threat to groundwater.

Plants. In plants, endosulfan is rapidly broken down to the corresponding sulfate (endosulfan sulfate). On most fruits and vegetables, 50 percent of endosulfan is lost within 3 to 7 days. Plant tissue residues are usually the result of surface deposition (ATSDR, 2000). Very limited data indicate that endosulfan and its metabolites are translocated in plants treated by immersion of the above-ground portion of the plants (Beard and Ware, 1969). Under laboratory controlled and semi-controlled greenhouse conditions, endosulfan (stereoisomers I and II) and its metabolites penetrated plant tissue and were translocated from the leaves to the roots of both bean and sugar beet plants (Beard and Ware, 1969).

Terrestrial Invertebrates. No information was found regarding the toxicity of endosulfan or endosulfan sulfate to terrestrial invertebrates.

Mammals. Endosulfan is rapidly degraded into mainly water-soluble compounds and eliminated in mammals with very little absorption into the gastrointestinal tract. In rabbits, the beta-isomer is cleared from blood plasma more quickly than the alpha-isomer, with reported blood half-lives of approximately 6 hours and 10 days, respectively (EXTOXNET, 2003).

Endosulfan is highly toxic via the oral route, with reported oral LD_{50} values ranging from 18 to 160 mg/kg in rats, 7.36 mg/kg in mice, and 77 mg/kg in dogs. It is also highly toxic via the dermal route, with reported dermal LD_{50} values for rats ranging from 78 to 369 mg/kg (EXTOXNET, 2003). Endosulfan may be only slightly toxic via inhalation, with a reported inhalation LC_{50} of 21 mg/L for one hour, and 8.0 mg/L for four hours. Animal data indicate that toxicity may also be influenced by species and by level of protein in the diet; rats that have been deprived of protein are nearly twice as susceptible to the toxic effects of endosulfan.

Stimulation of the central nervous system is the major characteristic of endosulfan poisoning. Symptoms noted in acutely exposed humans include incoordination, imbalance, difficulty breathing, gagging, vomiting, diarrhea, agitation, convulsions, and loss of consciousness.

Reversible blindness has been documented for cows that grazed in a field sprayed with endosulfan.

In rats, oral doses of 10 mg/kg/day caused high rates of mortality within 15 days, but doses of 5 mg/kg/day caused liver enlargement and some other effects over the same period. Data from animal studies reveal the organs most likely to be affected include kidneys, liver, blood, and the parathyroid gland (EXTOXNET, 2003).

Birds. Endosulfan is highly to moderately toxic to bird species, with reported LD_{50} values for mallards ranging from 31 to 243 mg/kg, and for pheasants ranging from 80 to greater than 320 mg/kg. The reported 5-day dietary LC_{50} is 2,906 ppm in Japanese quail. Male mallards from 3 to 4 months old exhibited wings crossed high over their backs, tremors, falling, and other symptoms as soon as 10 minutes after an acute, oral dose (EXTOXNET, 2003).

Aquatic Life. Endosulfan does not bioaccumulate to high concentrations in terrestrial or aquatic ecosystems. In aquatic ecosystems, residue levels in fish generally peak within 7 days to 2 weeks of continuous exposure to endosulfan. Maximum bioconcentration factors (BCF) are usually less than 3,000, and residues are eliminated within 2 weeks of transfer to clean water (NRCC, 1975b). Maximum BCFs of 600 and 22.5 in mussel tissue were reported by Ernst (1977) and Roberts (1972). Tissue concentrations fell rapidly upon transfer of the organisms to fresh water; for example, Ernst (1977) reported a depuration half-life of 34 hours.

Endosulfan is highly toxic to four fish species and both of the aquatic invertebrates studied. In fish species, the reported 96-hour LC_{50} values were 1.5 μg/L for rainbow trout, 1.4 μg/L for fathead minnow, 1.5 μg/L for channel catfish, and 1.2 μg/L for bluegill sunfish. The reported 96-hour LC_{50} values for scuds (*Gammarus lacustris*) and stoneflies (*Pteronarcys sp.*) were 5.8 μg/L and 3.3 μg/L, respectively. In the mussel (*Mytilus edulis*), endosulfan accumulated up to 600 times the ambient water concentration (EXTOXNET, 2003).

The National Recommended Water Quality Criteria (EPA, 1999a) for the protection of freshwater organisms for endosulfan are 0.22 and 0.056 μg/L for the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC), respectively.

7.4.2.27 Endrin

Endrin was first used as an insecticide, rodenticide, and avicide beginning in 1951 to control cutworms, voles, grasshoppers, borers, and other pests on cotton, sugarcane, tobacco, apple

orchards, and grain (EPA, 1979). Endrin tends to persist in the environment mainly in forms sorbed to sediments and soil particles. Endrin is extremely persistent when released to soil. A conservative estimate of its half-life in sandy loam soils is approximately 14 years (Nash and Woolson, 1967).

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

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

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

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

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

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

Aquatic Life. When released to water, endrin strongly adsorbs to sediment and bioconcentrates significantly in aquatic organisms (ATSDR, 1996c). 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, 1996c). The estimated half-life of endrin in water is more than 4 years (ATSDR, 1996c). 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 et al., 1993).

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

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

7.4.2.28 Heptachlor

Heptachlor is an organochlorine cyclodiene insecticide, first isolated from technical grade chlordane in 1946. During the 1960s and 1970s, it was used primarily by farmers to kill termites, ants, and soil insects in seed grains and on crops, as well as exterminators and home owners to kill termites. Heptachlor acts as a nonsystemic stomach and contact insecticide. An important metabolite of heptachlor is heptachlor epoxide, which is an oxidation product formed from heptachlor by many plant and animal species (EXTOXNET, 2003).

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

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

Mammals. Heptachlor is a highly to moderately toxic compound via the oral route of exposure, with reported oral LD_{50} values of 100 to 220 kg/kg in rats, 30 to 68 mg/kg in mice, 116 mg/kg in guinea pigs, and 100 mg/kg in hamsters. It is moderately toxic via the dermal pathway as well,

with reported dermal LD_{50} values of 119 to 320 mg/kg in rats, and greater than 2,000 mg/kg in rabbits (EXTOXNET, 2003). Heptachlor, like many organochlorines may interfere with nerve transmission, and may also cause an increase in activity of the enzymes involved in the breakdown of foreign chemicals. Heptachlor is readily taken up through the skin, lungs, and gastrointestinal tract. Once absorbed, it is systemically distributed and preferentially moves into body fat. In mammals, heptachlor is readily converted to its most persistent and toxic metabolite, heptachlor epoxide, in the liver. Heptachlor epoxide is stored mainly in fatty tissue, but also in liver, kidney, and muscle tissue. The acute toxicity of heptachlor epoxide, the main and most persistent of heptachlor's metabolites, may be greater. Effects due to heptachlor exposure may include hyperexcitation of the central nervous system, liver damage, lethargy, incoordination, tremors, convulsions, stomach cramps or pain, and coma. Results of animal tests show that chronic exposure to heptachlor or heptachlor epoxide adversely affects the liver, kidney, and red blood cells (EXTOXNET, 2003).

A study conducted with mink fed heptachlor in their diet for 181 days was used to derive a NOAEL value of 0.1 mg/kg/day (Crum et al., 1993). Fertility, litter size, and kit survival were reduced in the lowest exposure group (6.25 ppm).

Birds. Heptachlor is moderately to highly toxic to bird species; the reported acute oral LD_{50} in mallard ducks is 2,080 mg/kg. The reported 5-day dietary LC_{50} for Japanese quail is 99 ppm. Other reported 8-day dietary LC_{50} values for heptachlor are 450 to 700 ppm in bobwhite quail, and 250 to 275 ppm in pheasants (EXTOXNET, 2003). It is also reported to decrease the survivability of chicken eggs. Heptachlor and its more toxic metabolite, heptachlor epoxide, have been found in the fat of fish and birds. They have also been found in the liver, brain, muscle, and eggs of birds (EXTOXNET, 2003).

Aquatic Life. Both heptachlor and heptachlor epoxide are highly toxic to most fish species tested. The reported 96-hour LC_{50} values are: 5.3 to 13 μg/L in bluegill sunfish; 7.4 to 20 μg/L in rainbow trout; 6.2 μg/L in northern pike; 23 μg/L in fathead minnow; and 10 μg/L in largemouth bass. Heptachlor is also highly toxic to freshwater aquatic invertebrates. Both heptachlor and heptachlor epoxide have been shown to bioconcentrate in aquatic organisms such as fish, mollusks, insects, plankton, and algae. It has been found in several fish, mollusks, and other aquatic species at concentrations of 200 to 37,000 times the concentration of heptachlor in the surrounding water (EXTOXNET, 2003).

The Tier II secondary acute and secondary chronic values for heptachlor are 0.125 and 0.0069 µg/L, respectively (Suter and Tsao, 1996). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of heptachlor 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 lowest fish EC_{20} for heptachlor is 0.86 µg/L. The National Recommended Water Quality Criteria (EPA, 1999a) for the protection of freshwater organisms for heptachlor are 0.52 and 0.0038 µg/L for the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC), respectively.

7.4.2.29 MCPP

MCPP (Mecoprop) is a selective, hormone-type phenoxy herbicide. It is applied postemergence 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 others. The duration of MCPP's residual activity in soil is about two months. Adsorption of MCPP increases with an increase in organic matter in the soil. Un-aged MCPP and its salt forms are very mobile in a variety of soils. Because of this high mobility, it may potentially leach into groundwater. However, in general, phenoxy herbicides such as MCPP are not sufficiently persistent to reach groundwater (EXTOXNET, 2003).

Plants. 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 control. No other data regarding the toxicity of MCPP to plants was found in the literature (EXTOXNET, 2003).

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

Mammals. MCPP has a low acute toxicity to test animals. The LD_{50} is 930 – 1,210 mg/kg for rats and 650 mg/kg for mice. The LD_{50} 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. MCPP may be mutagenic at very high doses (EXTOXNET, 2003).

Birds. MCPP is practically nontoxic to birds. The LC_{50} is greater than 5,620 ppm for mallard ducks and 5,000 ppm for bobwhite quail. The LD_{50} (oral) is 740 mg/kg for Japanese quail and 700 mg/kg for bobwhite quail (EXTOXNET, 2003).

Aquatic Life. MCPP is virtually nontoxic to fish. Available data indicate a low potential of MCPP to bioaccumulate in fish. The 96-hour LC50 is 124 ppm for rainbow trout and greater than 100 ppm for bluegill sunfish (EXTOXNET, 2003).

7.4.2.30 Methoxychlor

Methoxychlor is an organochlorine insecticide effective against a wide range of pests encountered in agriculture, households, and ornamental plantings. Methoxychlor is one of a few organochlorine pesticides that have seen an increase in use since the ban on DDT in 1972. It is quite similar in structure to DDT, but has relatively low toxicity and relatively short persistence in biological systems. It may be found in formulations with malathion, parathion, piperonyl butoxide, and pyrethrins (EXTOXNET, 2003).

Plants. No information was located in the literature regarding the toxicity of methoxychlor to plants.

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

Mammals. Methoxychlor is practically nontoxic via the oral route, with reported oral LD_{50} values of 5,000 to 6,000 mg/kg in rats, 1,850 mg/kg in mice, and 2,000 mg/kg in hamsters. It is reportedly practically nontoxic dermally, with a reported dermal LD_{50} in rabbits of greater than 2,000 mg/kg. Symptoms of acute exposure include central nervous system depression, progressive weakness, and diarrhea. Data from animal studies indicate that target organs for methoxychlor include the kidneys, liver, mammary glands, and uterus (EXTOXNET, 2003).

Existing data suggest that methoxychlor is not carcinogenic, mutagenic, teratogenic, and is unlikely to cause reproductive effects in mammals. Rats exposed to methoxychlor in their diets for 11 months were observed for adverse effects on reproduction. Fertility and litter size were significantly reduced among rats fed diets containing 100 or 200 ppm methoxychlor (Gray et al., 1988). These data were used to derive a NOAEL value of 4 mg/kg/day.

Available evidence suggests that methoxychlor does not accumulate to any significant degree in fat or other tissues of mammals. At high dietary doses in rats, storage was minimal over the 18-week exposure period and was not detectable within two weeks after the study (EXTOXNET, 2003).

Birds. Methoxychlor is slightly toxic to bird species, with reported acute LD_{50} values of greater than 2,000 mg/kg for the mallard duck (*Anas platyrhynchos*), sharp-tailed grouse (*Tympanuchus phasianellus*), and California quail (*Callipepla californica*). The reported 5-day dietary LC_{50} for Japanese quail (*Coturnix japonica*) is greater than 5,000 ppm. Reported 8-day LC_{50} values are greater than 5,000 ppm in bobwhite quail (*Colinus virginianus*) and ring-necked pheasants (*Phasianus colchicus*). Dietary levels of as high as about 145 mg/kg/day had no effects on reproductive function of male and female chickens over 8 to 16 weeks of exposure (EXTOXNET, 2003).

Aquatic Life. Methoxychlor is very toxic to fish and aquatic invertebrates. Reported 96-hour LC_{50} values for technical grade methoxychlor (~ 90 percent pure) are less than 20 µg/L for cutthroat trout, atlantic salmon, brook trout, lake trout, northern pike, and largemouth bass. Reported LC_{50} values are between 20 and 65 µg/L for rainbow trout, goldfish, fathead minnow, channel catfish, bluegill, and yellow perch. Aquatic invertebrates with 96- or 48-hour LC_{50} values of less than 0.1 mg/L include Daphnia, scuds, sideswimmers, and stoneflies (EXTOXNET, 2003).

Predicted bioconcentration factors (BCF) were the highest in the mussel (12,000) and the snail (8,570). These predicted BCFs indicate that methoxychlor can accumulate in aquatic organisms that do not rapidly metabolize the compound. Practically no metabolism was seen in Daphnia or mayflies. Fish reportedly break down methoxychlor fairly rapidly and thus tend not to accumulate it appreciably. No biomagnification of residues was observed in largemouth bass fingerlings fed contaminated Daphnia, and no evidence of metabolism was seen in rainbow trout (EXTOXNET, 2003). The Tier II secondary chronic value for methoxychlor is 0.019 µg/L (Suter and Tsao, 1996).

7.4.2.31 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 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 above-ground plant parts than in below-ground 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 μg/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 μg/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).

Terrestrial Invertebrates. Earthworms (*Eisenia fetida*) exposed to benzo(a)pyrene (up to 500 mg/kg) in soil for 28 days did not exhibit DNA strand breaks (Honeycutt and Roberts, 1995), indicating the lack of genotoxicity at the exposure concentrations. PAH compounds were determined to be relatively non-toxic to the earthworm *Eisenia fetida* (Roberts and Dorough, 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 known as mixed-function oxidases in the liver. A few laboratory studies on rodents have revealed 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). Data from a study of mice fed benzo(a)pyrene in their diets were used to derive a NOAEL value of 1.0 mg/kg/day (MacKenzie and Angevine, 1981). The critical endpoint in this study was reproduction.

Sims and Overcash (1983) have reported 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 μg/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 μg 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 the production of reproductive and teratogenic effects in eggs of the sand sole (*Psettichthys melanostictus*) exposed to 0.1 μg benzo(a)pyrene/L for five days 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.

7.4.2.32 Polychlorinated Biphenyls

Polychlorinated biphenyls (PCB) are mixtures of different congeners of chlorophenyl. PCBs are a group of highly fat-soluble, semivolatile compounds that readily bioaccumulate and biomagnify in ecological receptors, especially upper trophic level carnivores and aquatic food webs. In general, PCBs adsorb strongly to soil and sediment and are soluble in fatty tissues. Volatilization and biodegradation of the lower chlorinated congeners also occur. The toxicological properties of individual PCB congeners are influenced primarily by: 1) lipophilicity, which is correlated with log K_{ow}, and 2) steric factors resulting from different patterns of chlorine substitution on the biphenyl molecule. In general, PCB isomers with high K_{ow} values and high numbers of substituted chlorines in adjacent positions constitute the greatest environmental concern. Biological responses to individual isomers or mixtures vary widely, even among closely related taxonomic species.

Plants. Total PCB concentrations in terrestrial plants are usually less than 0.3 mg/kg (dry weight) (Buckley, 1983). Soil-to-plant transfer factors for PCBs are generally low, ranging from 0 to 0.5 (Pal et al., 1980). Uptake by plants is dependent on the degree of chlorination of the PCB compound, with uptake greater for the lipophilic, highly chlorinated forms (Pal et al., 1980).

Although plant growth reductions resulting from PCB addition to soil have been measured, no mechanism of toxicity has been suggested. Because cumulative water use seems to be more sensitive to PCBs than plant growth, it has been suggest that effects on plants may be indirect, following an effect on transpiration (Strek and Weber, 1982). *In vitro* cultures of plant cells are capable of metabolizing and detoxifying PCBs (Fletcher et al., 1987). A benchmark value of 40 mg/kg has been suggested by Efroymson et al. (1997a) as a concentration protective of plant communities.

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

Mammals. PCB concentrations in small mammals collected from relatively uncontaminated sites usually range from 0 to 1.3 mg/kg (wet weight) (Talmage and Walton, 1991). PCBs are readily absorbed through the GI tract, respiratory system, and skin (Eisler, 1986b). Simple PCBs

are metabolized in the body and excreted in urine and bile (Eisler, 1986b). Exposure of vertebrates to PCB congeners can induce the production of mixed-function oxidase (MFO) enzymes that metabolize foreign compounds (Parkinson and Safe, 1981).

Among mammals, mink (*Mustela vison*) are especially sensitive to PCBs (Eisler, 1986b). A concentration of 0.64 mg/kg in the diet can induce reproductive failure in mink (Fleming et al., 1983). Mink diets containing 6.7 and 8.6 mg/kg (wet weight) Aroclor-1242 and -1254, respectively, resulted in the death of 50 percent of the animals within a nine-month period (Ringer, 1983). Based on toxicity data from old field mice (*Permyscus poliontus*) and mink, fed Aroclor-1254 in their diets, NOAEL values of 0.068 and 0.14, respectively, have been derived (McCoy et al., 1995; and Aulerich and Ringer, 1977). The critical endpoint for both of these studies was reproduction.

Certain PCB congeners have been shown to induce mutagenic, carcinogenic, and teratogenic effects in mammals (Eisler, 1986b). Symptoms of PCB toxicity in mink fed diets supplemented with PCBs included anorexia, bloody stools, reproductive failure, increased infant mortality, and the presence of fatty livers, hemorrhagic gastric ulcers, and kidney degeneration (Aulerich and Ringer, 1977).

Birds. The toxicity of PCBs to birds is influenced by the species of bird and its associated food habits, health, and body fat content, gender and age (Eisler, 1986b). Dietary LD_{50} values for various species of birds exposed to PCB aroclor mixtures were highest for Aroclor-1221 and generally decreased with increasing percent chlorination (Heath et al., 1972). Based on toxicity data from screech owl (*Otus asio*) and ring-necked pheasants (*Phasianus colchicus*) fed Aroclor-1242 and Aroclor-1254, respectively, NOAEL values were derived. The NOAEL for Aroclor-1242 was 0.41 mg/kg/day (McLane and Hughs, 1980) and the NOAEL for Aroclor-1254 was 0.18 mg/kg/day (Dahlgren et al., 1972). The critical endpoint for both of these studies was reproduction. Exposure to PCBs can result in adverse affects on reproduction, growth, metabolism, and behavior (Eisler, 1986b). Eggshell thinning may also occur as a result of exposure of the female birds to PCBs (Eisler, 1986b).

In 1979, 83 percent of European starlings (*Sturnus vulgaris*) collected nationwide contained measurable amounts of PCBs (Cain and Bunck, 1983). Maximum PCB concentrations in eggs of American bald eagles (*Haliaetus leucocephalus*) collected nationwide during 1979 ranged from 5.6 to 81 mg/kg (wet weight) (Wiemeyer et al., 1984).

Aquatic Life. PCBs are bioconcentrated, bioaccumulated, and biomagnified in freshwater environments. Bioconcentration factors for freshwater invertebrates exposed to Aroclor-1254 have been reported to range from 60 to 27,000 (Eisler, 1986b). National surveys of PCB concentrations in freshwater fish collected from 1978 to 1979 revealed total PCB concentrations in whole fish to average 1.4 mg/kg (wet weight) (Schmitt et al., 1983). Mean concentrations of Aroclor-1248, Aroclor-1254, and Aroclor-1260 in whole fish were reported to be 0.1, 0.2, and 0.3, respectively. Diet appears to be the major route of PCB uptake in freshwater fish (Eisler, 1986b). Aquatic invertebrates have been shown to contribute to the cycling of PCBs in freshwater environments by their uptake of sediment-associated PCBs and their subsequent migration to the water column or surface, where they can be ingested by other organisms (Eisler, 1986b).

Because of the strong hydrophobicity of PCBs, these compounds localize in sediments and accumulate in the lipids of freshwater biota (Hooper et al., 1990). The high affinity of PCBs for lipids and the relative resistance of highly chlorinated congeners to metabolism are responsible for the biomagnification of these compounds in aquatic environments (Sanders and Chandler, 1972). Dechlorination of PCBs by bacteria in anaerobic sediments has been reported to reduce the toxicity of these compounds (Brown et al., 1987).

The toxicity of PCBs to aquatic biota increases with percent chlorination of the PCB mixture (Birge et al., 1978). Laboratory toxicity tests have shown PCB toxicity to increase with exposure time. Also, younger developmental stages are more sensitive to PCBs than older stages (Eisler, 1986b). The National Recommended Water Quality Criterion for PCBs (criterion continuous concentration) is 0.014 µg/L (EPA, 1999a). The Tier II values for PCBs, which have been derived using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995), are presented below:

PCB Congener	Secondary Acute V Value (µg/L)	Secondary Chronic Value (µg/L)	Lowest Test EC_{20} for Fish (µg/L)	Lowest Test EC_{20} for Daphnids (µg/L)
Aroclor-1221	5.0	0.28	80	--
Aroclor-1232	1.0	0/58	148	--
Aroclor-1242	1.2	0.053	<2.9	--
Aroclor-1248	1.4	0.081	0.4	2.5
Aroclor-1254	0.6	0.033	0.52	1.2
Aroclor-1260	1,700	94	2.1	--

The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of a specific PCB 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). A similar value can be determined for daphnids, which reflects the highest tested concentration of a PCB causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species (Suter and Tsao, 1996). The data presented above indicate that Aroclor-1260 is the least toxic of the major PCB congeners.

7.4.2.33 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, 1996; 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).

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

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

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_{50} values for these experiments were 2.3 g/kg for rats and 4.2 to 6.2 g/kg for mice (DIALOG, 1996).

Mice fed bis(2-ethylhexyl)phthalate in their diets for 105 days were studied for effects on reproduction. While significant reproductive effects were observed among mice on diets containing 0.1 and 0.3 percent bis(2-ethylhexyl)phthalate, no adverse effects were observed among the 0.01 percent dose group. These data were used to derive a NOAEL value of 18.3 mg/kg/day (Lamb et al., 1987).

Birds. Ringed doves were fed bis(2-ethylhexyl)phthalate in their diets for 4 weeks during a critical lifestage and studied for reproductive effects (Peakall, 1974). No significant reproductive effects were observed in the maximally exposed doves (10 ppm). These data were used to derive a NOAEL value of 1.1 mg/kg/day based on reproductive effects.

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_{50}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.

7.4.2.34 Trichloroethene

Trichloroethene, or trichloroethylene, is not known to occur as a natural product (Howard, 1990). This chlorinated organic compound is primarily used as a degreaser of metals (Howard, 1990). Photooxidation is the primary mode by which trichloroethene is removed from the atmosphere (NLM, 1996). The compound is fairly stable in soil; however, it can leach to groundwater. The primary removal process in aquatic systems is evaporation. Biodegradation is expected to be significant only under anaerobic conditions (NLM, 1996).

Plants. Data on measured concentrations of trichloroethene in plants are limited to concentrations reported in grain-based foods, which ranged from 0.77 to 2.7 μg/kg (Heikes and Hopper, 1986). Phytotoxicity data for trichloroethene could not be found in the literature.

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

Mammals. Concentrations of trichloroethene in wildlife are not reported in the literature. Trichloroethene is a hepatotoxin and central nervous system toxin (NLM, 1996). The oral LD_{50} value for mice exposed to trichloroethene is 2.402 g/kg (RTECS, 1996). The inhalation LC_{50} value for mice exposed to trichloroethene is 8,450 ppm for a four-hour exposure period (RTECS, 1996). The LD_{50} for dermal exposure of rabbits to trichloroethene is greater than 20 g/kg (RTECS, 1996). Fetotoxic and teratogenic effects have been reported in the offspring of pregnant rats exposed to trichloroethene (RTECS, 1996). Symptoms of chronic poisoning in dogs from inhalation of trichloroethene include lethargy, anorexia, nausea, vomiting, and weight loss (ACGIH, 1971). There is a limited amount of evidence that suggests trichloroethene is carcinogenic in mammals (RTECS, 1996).

Mice exposed to trichloroethene through oral gavage for 6 weeks at dosage levels that ranged from 100 to 3,200 mg/kg/day were used to derive a chronic NOAEL value of 0.7 mg/kg/day (Buben and O'Flaherty, 1985). The critical endpoint in this study was hepatotoxicity.

Aquatic Life. Very little data exist on concentrations of trichloroethene in aquatic biota. Bioconcentration factors of between 17 and 39 have been reported for bluegill sunfish and rainbow trout exposed to trichloroethene (Burrows et al., 1989). LC_{50} values for fathead minnows (*Pimephales promelas*) and bluegill sunfish (*Lepomis macrochirus*) exposed to trichloroethene for 96 hours have been reported as 20 and 44.7 mg/L (Borthwick, 1977).

The Tier II Secondary Acute and Secondary Chronic Values derived using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995) are 440 and 47 μg/L, respectively. The lowest chronic toxicity values for fish and daphnids are reported as 11,100 and 7,257 μg/L, respectively (Suter and Tsao, 1996). The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration 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 value for trichloroethene has been determined to be 5,758 μg/L (Suter and Tsao, 1996).

7.4.3 Potential Receptors

Potential ecological receptors at the BBGR Ranges fall into two general categories: terrestrial and aquatic. Within these two general categories there are several major feeding guilds that could be expected to occur at the BBGR Ranges: herbivores, invertivores, omnivores, carnivores, and piscivores. All of these feeding guilds have the potential to be directly exposed to various combinations of surface soil at the BBGR Ranges and surface water and sediment in the tributaries of South Branch of Cane Creek and Ingram Creek in the vicinity of the BBGR Ranges via various activities (e.g., feeding, drinking, grooming, bathing, etc.). These feeding guilds may also be exposed to site-related chemicals via food web interactions.

Dermal absorption of PAHs and chlorinated herbicides and pesticides from soil is a potential pathway for all feeding guilds at the BBGR 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 volatile organic constituents have been detected in surface soil at the BBGR Ranges, they have only been detected sporadically and at relatively low concentrations. Therefore, inhalation of VOCs is not a significant exposure pathway at the BBGR Ranges. Inhalation of constituents sorbed to soil particles and inhaled as dust is a potential pathway for all of the feeding guilds at the BBGR Ranges.

Terrestrial species may also be exposed to constituents in surface water in the South Branch of Cane Creek and Ingram Creek tributaries through ingestion of and dermal contact with surface water in these ephemeral drainage features. The ephemeral nature of most of these drainage features in the vicinity of the BBGR Ranges limits the potential for exposure to surface water. Additionally, only Ingram Creek tributaries exhibit elevated concentrations of several constituents in sediment. The South Branch of Cane Creek tributaries do not exhibit elevated constituent concentrations in either surface water or sediment.

Aquatic and semi-aquatic (i.e., amphibians) 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. However, as discussed previously, the ephemeral nature of most of the drainage features at the BBGR Ranges limits the exposure potential for aquatic organisms. 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.

Although these inorganic compounds and PAHs in sediment may accumulate in lower trophic level organisms at the BBGR Ranges, biomagnification through the aquatic food web is not expected to be significant. Pesticides and herbicides in the sediment may accumulate and magnify in the food web.

7.4.3.1 Herbivorous Feeding Guild

The major route of exposure for herbivores is through ingestion of plants that may have accumulated contaminants from the soil, surface water, or sediment. The vegetation at the formerly maintained areas at the BBGR Ranges is mainly grasses and sedges, which are remnants of the maintained grass that was present when the BBGR Ranges were operational. Since terrestrial herbivores by definition are grazers and browsers, they could be exposed to chemicals that have accumulated in the vegetative tissues of the plants at the site. Terrestrial herbivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while grazing, grooming, or other activities.

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

Aquatic herbivores, such as muskrat (*Ondatra zibethica*) and mallard (*Anas platyrhynchos*) could be exposed to site-related constituents in surface water and sediment; however, most of the ephemeral drainage features at the BBGR Ranges do not provide suitable habitat to support aquatic herbivores. The exceptions to this are the creek that flows through the southern portion of Range 23 and South Branch of Cane Creek that flows adjacent to Range 18, both of which are perennial in nature. These creeks have the potential to support smaller aquatic herbivorous species.

7.4.3.2 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 (i.e., earthworms and other invertebrates) lives on or below the soil surface.

Typical invertivorous species that could be expected to occur at the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include American woodcock (*Philohela minor*), carolina wren (*Thryothorus ludovicianus*), shorttail shrew (*Blarina brevicauda* and *Blarina carolinensis*), and eastern mole (*Scalopus aquaticus*). Aquatic invertivores could include the wood duck (*Aix sponsa*) and blacknose dace (*Rhinichthys atratulus*). Most of the ephemeral drainage features at the BBGR Ranges do not provide suitable habitat to support aquatic invertivores. The exceptions to this are the creek that flows through the southern portion of Range 23 and the South Branch of Cane Creek that flows adjacent to Range 18, both of which are perennial in nature. These creeks have the potential to support smaller aquatic herbivorous species.

7.4.3.3 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 the ephemeral drainage features at the BBGR Ranges.

Typical omnivorous species expected to occur at the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include red fox (*Vulpes vulpes*), white-footed mouse (*Peromyscus leucopus*), and American robin (*Turdus migratorius*). Aquatic omnivores, such as raccoon (*Procyon lotor*) and creek chub (*Semotilus atromaculatus*) could be exposed to COPECs in surface water and sediment in the drainage features at the BBGR Ranges; however, the ephemeral nature of most of the drainage features precludes their presence due to the lack of suitable habitat. The exceptions to this are the creek that flows through the southern portion of Range 23 and South Branch of Cane Creek that flows adjacent to Range 18, both of which are perennial in nature. These creeks have the potential to support smaller aquatic herbivorous species.

7.4.3.4 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 contaminants in their tissues. Carnivores are quite often top predators in a local food web and are often subject to exposure to contaminants that have bioaccumulated in lower trophic-level organisms or

biomagnified through the food web. Food web exposures for carnivores are based on the consumption of prey animals that have accumulated COPECs from various means. Smaller, herbivores, omnivores, invertivores, and other carnivores may consume soil, 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 volatile organic compounds are not accumulated in animal tissues to any great extent (Shugart et al., 1991 and USAEHA, 1994). Therefore, food web exposures to these chemicals are expected to be minimal. PAHs have the potential to accumulate in lower trophic level organisms but not in higher trophic level organisms because they have mechanisms for metabolizing and excreting this class of compounds. Chlorinated herbicides and pesticides have the potential to bioaccumulate and biomagnify through the food chain; therefore, there is the potential for significant exposure to these classes of chemicals by carnivores. Carnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities.

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

Because most of the drainage features at the BBGR Ranges are relatively narrow and shallow and ephemeral in nature, they do not have the capability to support large aquatic carnivores on a full-time basis. Carnivorous fish such as largemouth bass (*Micropterus salmoides*) and spotted gar (*Lepisosteus oculatus*) would not be expected to occur in the drainage features at the BBGR Ranges due to the habitat restrictions. Carnivorous mammals such as the mink (*Mustela vison*), may feed along the creek that flows through the southern portion of Range 23 during certain periods of the year when significant water is present in the creek or in the South Branch of Cane Creek adjacent to Range 18, but most likely would not live adjacent to most of the drainage features at the BBGR Ranges because of the inability of these small drainage features to support large individuals of fish or other aquatic species. The South Branch of Cane Creek adjacent to Range 18 may be able to support aquatic carnivores since this creek in perennial in nature.

7.4.3.5 Piscivorous Feeding Guild

Piscivores are specialists that feed almost exclusively on fish. Therefore, they may be exposed to site-related chemicals that have accumulated in small fish that may inhabit the small drainage features at the BBGR Ranges during periods of significant precipitation. They may also be exposed to surface water and sediment in these drainage features through ingestion of drinking

water and during feeding. Because most of the drainage features at the BBGR Ranges are ephemeral in nature, they would only provide a drinking water supply during periods of significant precipitation. The small creek in the southern portion of Range 23 and the South Branch of Cane Creek adjacent to Range 18 may provide perennial drinking water sources. Although piscivorous species might visit the drainage features at the BBGR Ranges during certain periods of the year when the creek flow is significant, they would not be expected to live near the BBGR Ranges due to the fact that these drainage features are not large enough to support larger fish species. The exception is the South Branch of Cane Creek adjacent to Range 18. The South Branch of Cane Creek may 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 SVOCs or VOCs at the BBGR Ranges through the food web. Chlorinated herbicides and pesticides have the potential to bioaccumulate and biomagnify through the food chain; therefore, there is the potential for significant exposure to these classes of chemicals by piscivores, if they are present at the BBGR Ranges.

Typical piscivorous species that could occur near the BBGR Ranges and are commonly used as sentinel species in ecological risk assessment include great blue heron (*Ardea herodias*) and belted kingfisher (*Ceryle alcyon*). Larger piscivorous fish species (e.g., smallmouth bass, spotted gar) and piscivorous mammals (e.g., mink) are not expected to occur in the vicinity of the BBGR Ranges due to the habitat limitations of the small drainage features in this area and their inability to support larger fish and other aquatic species. The exception is South Branch of Cane Creek at Range 18. South Branch of Cane Creek in this area may provide adequate habitat to support larger fish species, and thus piscivorous species.

7.4.3.6 Threatened and Endangered Species

Four species listed as threatened or endangered by the USFWS have been recorded at FTMC. These threatened and endangered species are as follows:

- Gray bat (*Myotis grisescens*)
- Blue shiner (*Cyprinella caerules*)
- Mohr's Barbara buttons (*Marshallia mohrii*)
- Tennessee yellow-eyed grass (*Xyris tennesseensis*).

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

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

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

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

Creek golf course. The small, ephemeral streams in the vicinity of the BBGR Ranges do not provide the habitat favored by gray bats and their presence in the vicinity of the BBGR Ranges is unlikely. However, gray bat surveys have not been conducted on the small streams in the vicinity of the BBGR Ranges.

7.4.4 Complete Exposure Pathways

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

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

If any of these four components are absent, then a pathway is generally considered incomplete. Potentially complete exposure pathways are depicted in the SCM as Figure 7-1.

Ecological receptors may be exposed to constituents in soils via direct and/or secondary exposure pathways. Direct exposure pathways include soil ingestion, dermal absorption, and inhalation of 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 contaminants 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.

Additionally, birds (e.g., wild turkeys, quail) could be exposed to lead bullet fragments at the BBGR Ranges through ingestion. Birds that use these ranges for foraging could ingest lead bullet fragments as grit and retain them in their crops, if the bullet fragments are of the preferred size, texture, and shape.

Ecological receptors could be exposed to constituents in surface water via direct contact or through consumption of water. However, due to the fact that no constituents have been detected in surface water at elevated concentrations, surface water exposures are not considered significant.

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 contaminants from being carried by wind erosion. Acetone was detected in sediment samples albeit at very low concentrations. Therefore, inhalation of constituents originating from the sediment is not a significant exposure pathway. Exposure via dermal contact may occur, especially for benthic organisms and wading birds or other animals that may use the drainage features as feeding areas. Some aquatic organisms consume sediment and ingest organic material from the sediment. Inadvertent ingestion of sediments may also occur as the result of feeding on benthic organisms and plants.

While constituents in soils may leach into groundwater, environmental receptors will not come into direct contact with constituents in groundwater since there is no direct exposure route. The only potential exposure pathways for ecological receptors to groundwater would be via surface water exposure routes. As described in previous sections of this report, groundwater discharge to surface water in the small ephemeral drainage features is a potentially viable transport mechanism for dissolved constituents; 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 (i.e., exposure to constituents in surface water as a result of surface runoff) and is only expected to occur during periods of significant precipitation when the potentiometric surface of the groundwater intersects the level of the small drainage features at the BBGR Ranges. Additionally, no constituents were detected in surface water at elevated concentrations; therefore, groundwater intrusion into surface water does not appear to be significant at the BBGR Ranges.

Secondary exposure pathways involve constituents that are transferred through different trophic levels of the food chain and may be bioaccumulated and/or bioconcentrated. This may include constituents bioaccumulated from soil into plant tissues or into terrestrial species ingesting soils. These plants or animals may, in turn, be consumed by animals at higher trophic levels. Sediment-borne COPECs may bioaccumulate into aquatic organisms, aquatic plants, or animals which frequent surface waters and then be passed through the food chain to impact organisms at higher trophic levels.

In general, the constituents detected in surface soil at the BBGR Ranges may bioaccumulate in lower trophic level organisms (i.e., 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 (4,4'-DDE, 4,4'-DDT, MCPP, aldrin, beta-BHC, gamma-BHC, dieldrin, endosulfan sulfate, endrin, methoxychlor, and endrin ketone) were detected in surface soil that have a propensity to bioconcentrate through the food chain. Therefore, these constituents may be available to higher trophic level organisms through food chain interactions.

Much like surface soil, the majority of the constituents detected in sediment may bioaccumulate in lower trophic level organisms (i.e., 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. However, several chlorinated herbicides and pesticides (4,4'-DDT, MCPP, alpha-BHC, gamma-chlordane, and heptachlor) were detected in sediment that have a propensity to bioconcentrate through the food chain. Therefore, these constituents may be available to higher trophic level organisms through food chain interactions.

Summaries of the potentially complete exposure pathways for the terrestrial and aquatic ecosystems at the BBGR Ranges are presented in Tables 7-1 and 7-2, respectively.

7.5 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; and
- Calculate screening-level hazard quotients.

These steps are summarized below.

7.5.1 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 (i.e., protection of sediment benthic communities from adverse changes in structure or function).

The assessment endpoints for this SLERA were identified for each environmental medium and 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.

7.5.2 Ecological Screening Values

The ESVs 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 4 and are presented in the *Final Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000b). 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 BBGR Ranges (soil, surface water, sediment, and groundwater), a hierarchy has been developed which presents an orderly method for selection of ESVs. The hierarchy for selecting ESVs for soil is as follows:

- EPA Region 4 constituent-specific ESVs
- EPA Region 4 ecological screening values for general class of constituents
- EPA Region 5 ecological data quality levels (EDQL)
- EPA Region 3 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 4 constituent-specific ESVs
- NOAA (1999) Screening Quick Reference Tables chronic freshwater ambient water quality criteria
- EPA Region 5 EDQLs
- Office of Solid Waste and Emergency Response (OSWER) Ecotox Threshold values
- EPA Region 3 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 4 constituent-specific ESVs
- NOAA (1999) Screening Quick Reference Tables chronic freshwater ambient water quality criteria
- EPA Region 5 EDQLs
- OSWER ecotox threshold values
- EPA Region 3 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.

The surface water ESVs were used to assess constituents in groundwater as described in Section 7.3 of this report.

7.5.3 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 therefore, constituents will not be inadvertently eliminated from further assessment.

The statistical summaries (including the exposure point concentrations) for surface soil at the BBGR Ranges are presented in Tables 7-3 through 7-9, surface water in Tables 7-10 and 7-11, sediment in Tables 7-12 and 7-13, and groundwater in Table 7-14.

7.5.4 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; and
- Comparison to naturally occurring background concentrations.

Constituents that were detected in environmental media at the BBGR Ranges were evaluated against the ESVs by calculating a screening-level hazard quotient (HQ_{screen}) for each constituent in each environmental medium. A HQ_{screen} was calculated by dividing the maximum detected constituent concentration in each environmental medium by its corresponding ESV as follows:

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

where:

HQ_{screen}	=	screening-level hazard quotient;
$MDCC$	=	maximum detected constituent concentration; and
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. A 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, a HQ_{screen} value greater than one indicated that the MDCC was greater than the ESV and that the chemical might pose adverse ecological hazards to one or more receptors.

In order to better understand the potential risks posed by chemical constituents at the BBGR Ranges, a mean hazard quotient was also calculated 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 BBGR Ranges are presented in Tables 7-3 through 7-14.

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.

The comparison of detected constituent concentrations with naturally occurring constituent concentrations was conducted via a three-tier process outlined in a technical memorandum dated June 24, 2003 (Shaw, 2003b). The first tier of the background comparison process was a comparison of the maximum detected constituent concentration to the background threshold value (BTV). A study of the natural geochemical composition associated with FTMC (SAIC, 1998) determined the mean concentrations of 24 metals in surface soil, surface water, sediment, and groundwater samples collected from presumably unimpacted areas. Per agreement with EPA Region 4, the background threshold value (BTV) for each metal was calculated as two times the mean background concentration for that metal. The BTV for each metal was used to represent the upper boundary of the range of natural background concentrations expected at FTMC, and was used as the basis for evaluating metal concentrations measured in site samples. Site sample metal concentrations less than or equal to the corresponding BTV represent the natural geochemical composition of media at FTMC, and not contamination associated with site activity. Site sample metal concentrations greater than the corresponding BTV require further background assessment.

If maximum constituent concentrations were greater than the BTV, then the second tier of the background comparison was employed. Tier two of the background comparison consists of statistical comparisons of the site data to background data using the Slippage Test and the Wilcoxon Rank Sum (WRS) Test. If the site data failed either the Slippage Test or the WRS Test, then the site data were subjected to a geochemical evaluation to determine whether concentrations of inorganic compounds are naturally occurring or are elevated due to contamination. The three-tier background comparison process is described in detail in Appendix I of this report.

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 background using the three-tier background screening process. If constituent concentrations were determined to be less than their naturally occurring background concentrations, then a risk management decision could result in eliminating these constituents from further assessment.

7.6 Identification of Constituents of Potential Ecological Concern

A constituent was identified as a COPEC if the following conditions were met:

- The maximum detected constituent concentration exceeded the ESV
- The maximum detected constituent concentration was 10 times the BTV if the constituent was identified as a macronutrient, and
- Constituent concentrations were determined to be greater than naturally occurring concentrations based on the three-tier background screening procedure.

If a constituent in a given environmental medium did not meet these conditions, then it was not considered a COPEC at the BBGR 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 been initially identified for surface soil, surface water, sediment, and groundwater at the BBGR Ranges are presented in Tables 7-3 through 7-14.

In order to focus future ecological risk assessment efforts on the constituents that are the most prevalent at the BBGR Ranges and have the greatest potential to pose ecological risk, additional lines of evidence were assessed. Some of the additional lines of evidence used in this SLERA included the frequency of detection, the magnitude of the HQ_{screen} value, comparison to alternative ESVs, and the environmental fate of the constituents. 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 BBGR Ranges. These additional lines of evidence are discussed in the following sections. A summary of the COPECs in surface soil, surface water, sediment, and groundwater is presented in Table 7-15.

7.6.1 COPECs in Surface Soil

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

7.6.1.1 Surface Soil at Range 18

Antimony, copper, and lead were frequently detected at elevated concentrations in surface soil at Range 18. The highest concentrations of copper and lead were found in soil samples collected from the hillside in the southern portion of the range, which acted as an impact zone for Range 18.

Antimony. All of the detected concentrations of antimony (7 out of 16 samples) in surface soil at Range 18 exceeded the ESV and BTV. Additionally, all of the detected antimony concentrations in soil were greater than the background range for antimony. Because antimony is a known component of munitions and was routinely detected at elevated concentrations, it was identified as a COPEC in surface soil at Range 18.

Arsenic. Arsenic was detected in three out of 16 surface soil samples at concentrations that exceeded the BTV. All of the soil samples from Range 18 exhibited arsenic concentrations that were within the range of background. Arsenic passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected arsenic in surface soil from Range 18 was determined to be naturally occurring through geochemical evaluation except for one sample. The calculated HQ_{screen} value was 2.4, indicating the maximum detected concentration of arsenic in surface soil only slightly exceeded the ESV. Based on the relative infrequency of elevated arsenic detections, the low magnitude of the HQ_{screen} value, and the magnitude of the detected concentrations relative to background, arsenic was not considered a COPEC in surface soil at Range 18.

Beryllium. Beryllium was detected in five out of 16 samples at concentrations that exceeded the ESV. Six samples also exceeded the range of background. One surface soil sample had an anomalously high concentration of beryllium, which may indicate a component of contamination. All of the elevated concentrations of beryllium were from samples located in the hillside in the southern portion of Range 18 that acted as the impact area for this range. For these reasons, beryllium has been identified as a COPEC in surface soil at Range 18.

Copper. Copper was detected in nine out of 16 samples at concentrations that exceeded the ESV. Eleven samples exhibited copper concentrations that were greater than the background range. The highest concentrations of copper in surface soil were detected in samples collected from the hillside in the southern portion of the range that acted as the impact area. For these reasons, copper was identified as a COPEC in surface soil at Range 18.

Lead. Lead was detected in 33 out of 55 surface soil samples at concentrations that exceeded the ESV. Twenty-nine samples exhibited lead concentrations that were greater than the background range. The highest detected concentrations of lead were found in samples from the hillside located in the southern portion of Range 18 that acted as the impact area for this range. Lead is a known component of munitions, and the historical use of the site as a firing range indicates that the lead in soil is related to Army activities. For these reasons, lead was identified as a COPEC in surface soil at Range 18.

Nickel. Nickel was detected in two out of 16 surface soil samples at concentrations that exceeded the ESV. There was no discernable pattern to the elevated nickel concentrations in surface soil. Based on geochemical evaluation, only one surface soil sample at Range 18 exhibited an anomalously high concentration of nickel. The calculated HQ_{screen} value was 2.2, indicating the maximum detected concentration of nickel only slightly exceeded the ESV. Based on the infrequency of detection and the low magnitude of the HQ_{screen} value, nickel was not considered a COPEC in surface soil at Range 18.

Zinc. Zinc was detected in eight out of 16 surface soil samples at concentrations that exceeded the ESV. Three samples exhibited zinc concentrations that were greater than the background range. Two surface soil samples had anomalously high concentrations of zinc, which may indicate a component of contamination. The highest detected concentrations of zinc were found in samples from the hillside located in the southern portion of Range 18 that acted as the impact area for this range. Zinc is a known component of munitions, and the historical use of the site as

a firing range indicates that the zinc in soil is related to Army activities. For these reasons, zinc was identified as a COPEC in surface soil at Range 18.

Organic Compounds. The herbicide MCPP was detected in two out of 11 surface soil samples at concentrations that exceeded the ESV. 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) and nontoxic to fish (LC_{50} =124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (EXTOXNET, 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, 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, MCPP was not considered a COPEC in surface soil at Range 18.

The pesticides 4,4'-DDT, beta-BHC, and endrin ketone were all detected in one sample at concentrations that exceeded the ESV. The calculated HQ_{screen} values for these pesticides were 1.8, 1.5, and 1.9, respectively, indicating the maximum detected concentrations of these pesticides only slightly exceed their ESVs. Based on the infrequency of detection and the low magnitude of the HQ_{screen} values, these pesticides were not considered COPECs in surface soil at Range 18.

Several SVOCs were detected in surface soil at concentrations that exceeded their respective ESVs. These SVOCs included 2,4-dinitrotoluene, anthracene, benzo(a)pyrene, fluoranthene, and pyrene. 2,4-Dinitrotoluene was detected in one out of 18 samples, anthracene was detected in one out of 11 samples, benzo(a)pyrene was detected in one out of 11 samples, fluoranthene was detected in two out of 11 samples, and pyrene was detected in two out of 11 samples at elevated concentrations. The calculated HQ_{screen} values for these SVOCs were 1.2, 2.1, 4.4, 4.3, and 4.0, respectively. These SVOCs were detected in the same two samples, neither of which is associated with areas that would be indicative of munitions-related activities. The PAH background screening values for soil adjacent to asphalt (IT, 2000b) are 0.935 mg/kg for anthracene, 1.42 mg/kg for benzo(a)pyrene, 2.031 mg/kg for fluoranthene, and 1.626 mg/kg for pyrene. All of the detected concentrations for these four PAHs were less than these screening values. Based on the infrequency of detection, the low magnitude of the HQ_{screen} values, and the

detected concentrations relative to the PAH screening values, none of these SVOCs were considered COPECs in soil at Range 18.

7.6.1.2 Surface Soil at Range 20

Copper and lead were frequently detected at elevated concentrations in surface soil at Range 20. The highest concentrations of copper and lead were found in soil samples collected from the hillside opposite the firing line which acted as the impact zone for Range 20.

Antimony. Antimony was detected in two out of four surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 2.9. Although the HQ_{screen} value did not indicate a severe exceedance of the ESV, antimony is a known component of munitions and the highest concentrations of antimony were found in the impact area for this range. Because antimony is a known component of munitions and the highest concentrations were detected in areas that would suggest it was the result of Army activities, antimony was considered a COPEC in surface soil at Range 20.

Copper. Copper was detected in two out of four surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 6.2. The highest concentrations of copper were detected in samples from the hillside that acted as the impact area for this range. Based on these reasons and the fact that copper is a known component of munitions, copper was identified as a COPEC in surface soil at Range 20.

Lead. Lead was detected in 14 out of 15 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited lead concentrations that were greater than the background range. The highest concentrations of lead were detected in samples from the hillside that acted as the impact area for this range. Based on these reasons and the fact that lead is a known component of munitions, lead was identified as a COPEC in surface soil at Range 20.

7.6.1.3 Surface Soil at Range 23

Copper and lead were detected frequently in surface soil samples from Range 23 at elevated concentrations. The highest concentrations of these constituents were found in the target mounds located at various distances down-range of the firing lines.

Antimony. Antimony was detected in three out of 22 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 2.4, indicating the maximum detected concentration of antimony only slightly exceeded the ESV. However, based on the fact that

antimony is a known component of munitions, antimony was considered a COPEC in surface soil at Range 23.

Copper. Copper was detected in 11 out of 22 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited copper concentrations that were greater than the range of background copper. The highest concentrations of copper in surface soil were detected in the vicinity of the target mounds along each firing line. Based on the frequency of detection, the pattern of detection, and the fact that copper is a known component of munitions, copper was identified as a COPEC in surface soil at Range 23.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and BTV. The highest concentrations of lead were also detected in the vicinity of the target mounds along each of the firing lines. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and the fact that lead is a known component of munitions, lead was considered a COPEC in surface soil at Range 23.

Organic Compounds. Aroclor 1260 was detected in one out of 13 surface soil samples. The calculated HQ_{screen} value was 1.1, indicating the maximum detected concentration of Aroclor-1260 only slightly exceeded the ESV. Based on the infrequency of detection and the low magnitude of the HQ_{screen} value, Aroclor-1260 was not considered a COPEC in surface soil at Range 23. Several pesticides exceeded their respective ESVs in one or more surface soil samples. These pesticides included the following: 4,4'-DDE, 4,4'-DDT, aldrin, beta-BHC, dieldrin, endrin, gamma-BHC, and methoxychlor. 4,4'-DDE and 4,4'-DDT were detected in one out of 13 samples at elevated concentrations, with HQ_{screen} values of 1.3 and 1.2, respectively. Aldrin was detected in two out of 13 samples at elevated concentrations, with a HQ_{screen} value of 5.2. Beta-BHC and dieldrin were detected in three out of 13 samples at elevated concentrations, with HQ_{screen} values of 6.6 and 7.8, respectively. Endrin was detected in one out of 13 samples at an elevated concentration, with a HQ_{screen} value of 3.6. Gamma-BHC was detected in two out of 13 samples at elevated concentrations, with a HQ_{screen} value of 420. Methoxychlor was detected in one out of 13 samples at an elevated concentration, with a HQ_{screen} value of 1.1. There was no apparent pattern to the elevated concentrations of pesticides. Based on the relative infrequency of detection, the lack of an apparent pattern to the detections, and the overall relatively low HQ_{screen} values, none of the detected pesticides were included as COPECs in surface soil at Range 23.

The herbicide MCPP was detected in three out of 13 surface soil samples at elevated concentrations. 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) and nontoxic to fish (LC_{50} =124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (EXTOXNET, 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, 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, MCPP was not considered a COPEC in surface soil at Range 23.

Trichloroethene was detected in one sample out of 13 at an elevated concentration. 2,4-DNT and 2,6-DNT were detected at elevated concentrations in one and two samples, respectively, out of 22 samples. The calculated HQ_{screen} values for these three constituents were 1.8, 7.8, and 12.5, respectively. Based on the infrequency of detection and the low magnitude of the HQ_{screen} values, these constituents were not considered COPECs in soil at Range 23.

7.6.1.4 Surface Soil at Range 25

Antimony, copper and lead were detected frequently in surface soil samples from Range 25 at elevated concentrations. The highest concentrations of these constituents were found in the hillside that acted as the impact zone for this range.

Antimony. Antimony was detected in nine out of 62 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 5.9. There was no apparent pattern to the elevated antimony concentrations. However, based on the fact that antimony is a known component of munitions, antimony was considered a COPEC in surface soil at Range 25.

Copper. Copper was detected in 13 out of 62 surface soil samples at concentrations that exceeded the ESV. Twenty samples exhibited copper concentrations that were greater than the range of background copper. The highest concentrations of copper in surface soil were detected in the hillside that acted as the impact zone for this range. Based on the frequency of detection, the pattern of detection, and the fact that copper is a known component of munitions, copper was identified as a COPEC in surface soil at Range 25.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. The highest concentrations of lead were also detected in the hillside that acted as the impact zone for this range. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and the fact that lead is a known component of munitions, lead was considered a COPEC in surface soil at Range 25.

Zinc. Zinc was detected in 19 out of 62 surface soil samples from Range 25 at elevated concentrations. One surface soil sample exhibited an elevated concentration of zinc compared to naturally occurring levels. All of the other zinc concentrations are likely naturally occurring. The calculated HQ_{screen} value was 4.4. Based on the infrequency of detection above naturally occurring concentrations, zinc was not considered a COPEC in surface soil at Range 25.

Organic Compounds. 4,4'-DDT was detected two out of 12 samples at elevated concentrations relative to the ESV. The calculated HQ_{screen} value was 2.3. Based on the relative infrequency of detection at elevated concentrations and the low magnitude of the HQ_{screen} value, 4,4'-DDT was not considered a COPEC in soil at Range 25.

7.6.1.5 Surface Soil at Range 25-East

No constituents were detected in surface soil samples from Range 25-East at elevated concentrations; therefore, no COPECs were identified at this range.

7.6.1.6 Surface Soil at Range 26

Antimony, copper and lead were detected frequently in surface soil samples from Range 26 at elevated concentrations. The highest concentrations of copper and lead were found in the hillside that acted as the impact zone for this range.

Antimony. Antimony was detected in four out of 17 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 4.4. There was no apparent pattern to the elevated antimony detections. However, due to the fact that antimony is a known component of munitions it was included as a COPEC in surface soil at Range 26.

Copper. Copper was detected in 4 out of 19 surface soil samples at concentrations that exceeded the ESV. Nine samples exhibited copper concentrations that were greater than the range of background copper. The highest concentrations of copper in surface soil were detected in the hillside that acted as the impact zone for this range. Based on the frequency of detection,

the pattern of detection, and the fact that copper is a known component of munitions, copper was identified as a COPEC in surface soil at Range 26.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. The highest concentrations of lead were also detected in the hillside that acted as the impact zone for this range. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and the fact that lead is a known component of munitions, lead was considered a COPEC in surface soil at Range 26.

Organic Compounds. Several PAH compounds were detected at elevated concentrations in surface soil at Range 26. These PAH compounds include: anthracene, benzo(a)pyrene, fluoranthene, pentachlorophenol, phenanthrene, and pyrene. Anthracene was detected in one sample out of 11 at an elevated concentration. The calculated HQ_{screen} value for anthracene was 5.2. Benzo(a)pyrene was detected in two out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for benzo(a)pyrene was 25. Fluoranthene was detected in four out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for fluoranthene was 58. Pentachlorophenol was detected in one out of 11 samples at an elevated concentration. The calculated HQ_{screen} value for pentachlorophenol was 40.5. Phenanthrene was detected in one out of 11 samples at an elevated concentration. The calculated HQ_{screen} value for phenanthrene was 2.4. Pyrene was detected in three out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for phenanthrene was 50. There was no apparent pattern to the elevated concentrations of PAHs.

Benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene were positively detected in a single soil sample (HR-84Q-GP06) out of 11 samples collected. All of the remaining detections of these and other PAH compounds were estimated values. The following alternative ESVs for these PAH compounds (based on exposures to masked shrew) are presented in EPA (2003f) Region 5 Ecological Screening Levels: anthracene = 1,480 mg/kg, benzo(a)pyrene = 1.52 mg/kg, fluoranthene = 122 mg/kg, pentachlorophenol = 0.119 mg/kg, phenanthrene = 45.7 mg/kg, and pyrene = 78.5 mg/kg. All of the positively detected and estimated concentrations of PAH compounds in surface soil at Range 26 are less than these alternative ESVs except one detection of benzo(a)pyrene, which slightly exceeds the alternative ESV (alternative HQ_{screen} = 1.6).

Due to the fact that the detected concentrations of PAH compounds are all less than the alternative ESVs, except for benzo(a)pyrene, and the HQ_{screen} value for benzo(a)pyrene utilizing

the alternative ESV is relatively low (1.6), PAHs were not considered COPECs in surface soil at Range 26.

The pesticides beta-BHC, dieldrin, endosulfan sulfate, and the herbicide MCPP were detected in 1 of 11 surface soil samples at elevated concentrations. The calculated HQ_{screen} values for these compounds were 5.3, 1.02, 1.2, 13. There was no discernable pattern to the elevated concentrations of these pesticides. Due to the relative infrequency of detection, the low magnitude of the HQ_{screen} values, and the lack of discernable pattern relative to Army activities, these pesticides were not considered COPECs in surface soil at Range 26

7.6.1.7 Surface Soil at the Ranges South of Range 25

Antimony, copper and lead were detected frequently in surface soil samples from the Ranges South of Range 25 at elevated concentrations.

Antimony. Antimony was detected at elevated concentrations in nine out of 17 surface soil samples. The calculated HQ_{screen} value was 2.1. Because antimony is a known component of munitions and it was detected relatively frequently at elevated concentrations, antimony was identified as a COPEC in surface soil at the Ranges South of Range 25.

Copper. Copper was detected in 9 out of 16 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited copper concentrations that were greater than the range of background copper. Based on the frequency of detection at elevated concentrations and the fact that copper is a known component of munitions, copper was identified as a COPEC in surface soil at the Ranges South of Range 25.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. Based on the frequency of detection at elevated concentrations and the fact that lead is a known component of munitions, lead was considered a COPEC in surface soil at the Ranges South of Range 25.

7.6.2 COPECs in Surface Water

No constituents were detected in surface water samples from either the South Branch of Cane Creek tributaries or the Ingram Creek tributaries at elevated concentrations relative to ESVs or naturally occurring levels (Tables 7-10 and 7-11). Therefore, no COPECs were identified in surface water at the BBGR Ranges.

7.6.3 COPECs in Sediment

No constituents were detected at elevated concentrations in the sediment sample (HR-74Q-SW/SD01) collected from South Branch of Cane Creek near Range 18. To supplement this single sample, data for 17 additional sediment samples collected from South Branch of Cane Creek in association with other investigations at FTMC were evaluated. The Range 18 sediment sample location and the additional sediment sample locations are shown on Figure 7-2. Two sediment samples were collected at locations upstream of Range 18, and 15 samples were collected at locations downstream of Range 18. The upstream sample locations are approximately 2,200 feet and 3,500 feet from Range 18 sediment location HR-74Q-SD01. The closest downstream sediment sample location is approximately 250 feet from sediment location HR-74Q-SD01. The other downstream sediment sample locations are dispersed along South Branch of Cane Creek to a distance of 13,200 feet downstream. The two upstream sediment samples exhibited lead concentrations of 8.0 and 13.5 mg/kg, and the 15 downstream sediment samples had lead concentrations ranging from 9.0 to 25.5 mg/kg. None of these additional sediment samples from South Branch of Cane Creek exhibited lead concentrations above the ESV (30.2 mg/kg) for lead in sediment. Additionally, geochemical evaluation of all the detected concentrations of lead in sediment samples from the South Branch of Cane Creek indicates that lead in these sediment samples is naturally occurring. Therefore, no COPECs were identified in South Branch of Cane Creek tributary sediments.

Copper and lead were detected in one sample from the Ingram Creek tributaries at anomalously high concentrations relative to naturally occurring levels. These results indicate that these samples may contain a component of contamination. Copper and lead are also known components of munitions. For these reasons, copper and lead were identified as COPECs in sediment within the Ingram Creek tributaries.

The herbicide MCPP and the pesticides 4,4'-DDT, alpha-BHC, gamma-chlordane, and heptachlor were detected in sediment samples from Ingram Creek tributaries. 4,4'-DDT was detected in one out of 9 sediment samples. The calculated HQ_{screen} value was 9.7. MCPP was detected in two out of 9 sediment samples from Ingram Creek tributaries. No HQ_{screen} value could be calculated because MCPP does not have an ESV associated with it. Alpha-BHC was detected in one out of 9 sediment samples at an elevated concentration. The calculated HQ_{screen} value for alpha-BHC was 1.4. Gamma-chlordane was detected in two out of 9 sediment samples at elevated concentrations. The calculated HQ_{screen} value for gamma-chlordane was 52.4. Heptachlor was detected in two out of 9 sediment samples at elevated concentrations. The calculated HQ_{screen} value for heptachlor was 2.0.

Studies of the fate and transport of MCPP have shown that the duration of MCPP's residual activity in soil or sediment is about two months. Because this range has not been active for a number of years, it is expected that the MCPP detected in sediment 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) and nontoxic to fish (LC_{50} = 124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (EXTOXNET, 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, 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, MCPP was not considered a COPEC in sediment in the Ingram Creek tributaries.

The NOAA (1999) Screening Quick Reference Tables identify a sediment screening criterion for chlordane in freshwater sediments of 4.5 μg/kg. This value is the threshold effects level (lowest level at which adverse effects may be exhibited in test organisms). If this value is considered an alternative ESV for gamma-chlordane, then only one sediment sample from the Ingram Creek tributaries (HR-76Q-SD03) exhibited an elevated concentration of gamma-chlordane. This sediment sample was collected from the small ephemeral drainage feature southwest of Range 20. Although gamma-chlordane was only detected in two sediment samples from Ingram Creek tributaries, the one sediment sample with the maximum detected concentration may contain a component of gamma-chlordane contamination. For this reason, gamma-chlordane was considered a sediment COPEC in the Ingram Creek tributaries.

The other pesticides detected in sediment samples from the Ingram Creek tributaries (4,4'-DDT, alpha-BHC, and heptachlor) were not considered COPECs in sediment due to their relative infrequency of detection, limited areal extent, and low magnitude of the calculated HQ_{screen} values for several of the pesticides.

Acetone was detected in one out of nine sediment samples at an elevated concentration above its ESV. The calculated HQ_{screen} value for acetone in sediment was 1.2. Based on the infrequency of detection at elevated concentrations and the low magnitude of the HQ_{screen} value, acetone was not identified as a COPEC in sediment in the Ingram Creek tributaries.

7.6.4 COPECs in Groundwater

The pesticides 4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, gamma-BHC, and heptachlor were detected in one out of five groundwater samples at elevated concentrations. Alpha-chlordane was detected in two out of five groundwater samples at elevated concentrations. The 4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, and heptachlor detections were all from the same monitoring well located in the center of Range 25. Gamma-BHC was detected in one monitoring well at Range 18 and alpha-chlordane was detected in one monitoring well at Range 15 and one well at Range 23. The HQ_{screen} values for 4,4'-DDD, 4,4'-DDT, alpha-chlordane, endosulfan I, endrin, gamma-BHC, heptachlor in groundwater were calculated to be 13.9, 77.0, 8.8, 1.4, 23.5, 1.6, and 31.6, respectively.

It is important to note that none of the pesticides detected in groundwater were detected in surface water samples from the BBGR Ranges. As stated previously in this report, ecological receptors do not have a direct exposure pathway to groundwater. Ecological receptors can only be exposed to constituents in groundwater if it is expressed at the ground surface as seeps or is discharged to lakes or streams via springs. Exposure of ecological receptors to groundwater could then occur via surface water pathways. Because the pesticides in groundwater were detected infrequently and were not detected in surface water samples, it is expected that potential ecological exposure to these compounds is insignificant. Therefore, these compounds were not considered COPECs in groundwater at the BBGR Ranges.

7.7 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 BBGR 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 BBGR 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.

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 that precludes 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 contaminants to the soil. The removal of bullets prior to analysis may produce an underestimation of future exposure point concentrations for bullet-related contaminants 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.

Additionally, there is no consideration given to the bioavailabilty 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 (i.e., 1 to 10 percent for some inorganics in soil) than the 100 percent that was assumed in this assessment. 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 under- or overestimated 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 BBGR 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 the South Branch of Cane Creek and Ingram Creek tributaries 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. Similarly, the surface water and soil ESVs do not take into account site-specific conditions at the BBGR 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.

One significant source of uncertainty that may impart a non-conservative bias on the SLERA results is the exclusion of metals determined to be present at concentrations comparable to naturally occurring background concentrations from consideration as COPECs. The chemicals excluded from selection as COPECs based solely on their comparison to background concentrations are discussed below.

As noted above, the exclusion of chemicals from the list of COPECs based on comparison to naturally occurring levels is performed via a three-tiered protocol (Shaw, 2003b). Tier 1 – comparison of the maximum detected constituent concentration to the BTV – is generally considered to be sufficiently conservative so that the uncertainty associated with chemicals eliminated in this tier of the protocol is minimal. Therefore, only chemicals excluded as COPECs via tiers 2 or 3 of the background screening protocol are discussed herein.

Range 18 Surface Soil. Aluminum, barium, cobalt, iron, mercury, and selenium were eliminated from consideration as COPECs in surface soil at Range 18 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were Al = 576, Ba = 1.05, Co = 4.3, Fe = 212, Hg = 1.38, and Se = 1.9.

Per EPA (2003g) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2003g). Since the pH of soils at Range 18 is greater than 5.5, the exclusion of aluminum from the list of COPECs introduced minimal uncertainty into this SLERA.

Barium, cobalt, and mercury were detected in one out of 16 surface soil samples at a concentration that exceeded their respective ESVs and selenium was detected in three out of 16 samples at elevated concentrations. Due to the relative infrequency of detection at elevated concentrations and the low magnitude of the calculated HQ_{screen} values, elimination of these constituents from the list of COPECs in soil at Range 18 introduced a minimal level of uncertainty to the results of this SLERA.

Iron was detected in three out of 16 surface soil samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

Range 20 Surface Soil. Barium and zinc were eliminated from consideration as COPECs in surface soil at Range 20 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} value for barium was 1.05 and for zinc was 1.11. One out of 4 samples exhibited elevated concentrations of barium or zinc. Due to the relative infrequency of detection at elevated concentrations and the low magnitude of the calculated HQ_{screen} values, elimination of these constituents from the list of COPECs in soil at Range 20 introduced a minimal level of uncertainty to the results of this SLERA.

Range 23 Surface Soil. Aluminum, arsenic, barium, beryllium, cobalt, iron, manganese, selenium, and zinc were eliminated from consideration as COPECs in surface soil at Range 23 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were the following: Al = 468, As = 1.75, Ba = 3.45, Be = 1.45, Co = 1.59, Fe = 214, Mn = 45.8, Se = 1.33, and Zn = 1.2.

Per EPA (2003g) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2003g). Since the pH of soils at Range 23 is greater than 5.5, the exclusion of aluminum from the list of COPECs introduced minimal uncertainty into this SLERA.

Selenium was detected in one out of 22 soil samples at a concentration that exceeded the ESV, arsenic, cobalt, and zinc were detected in two out of 22 soil samples at concentrations that exceeded their respective ESVs, and beryllium was detected in three out of 22 samples at concentrations that exceeded the ESV. Based on their infrequency of detection at elevated concentrations and the relatively low magnitude of the HQ_{screen} values for these constituents, elimination of arsenic, beryllium, cobalt, selenium, and zinc from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of the SLERA.

Barium was detected in six out of 22 surface soil samples at concentrations that exceeded the ESV. EPA Region 5 has suggested an alternative ESV of 1,040 mg/kg for barium based on exposure to a masked shrew. If this alternative ESV is considered, then none of the detected barium concentrations are in excess of this alternative ESV. Because all of the detected values of barium are less than the alternative ESV, the exclusion of barium from consideration as a COPEC in soil at Range 23 is expected to introduce a minimal level of uncertainty to the results of the SLERA.

Iron was detected in one out of 22 surface soil samples at a concentration greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

The exclusion of manganese from the list of COPECs in soil remains a source of uncertainty in this SLERA.

Range 25 Surface Soil. Aluminum, arsenic, barium, beryllium, chromium, cobalt, iron, manganese, selenium, silver, and vanadium were eliminated from consideration as COPECs in surface soil at Range 25 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were the following: Al = 686, As = 3.69, Ba = 2.77, Be = 2.15, Cr = 252, Co = 2.71, Fe = 360, Mn = 48, Se = 2.8, Ag = 2.36, and V = 65.5.

Per EPA (2003g) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is

identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2003g). Since the pH of soils at Range 25 is greater than 5.5, the exclusion of aluminum from the list of COPECs introduced minimal uncertainty into this SLERA.

Alternative ESVs for barium include 330 mg/kg based on soil invertebrate toxicity and 1,000 mg/kg based on mammalian herbivore (vole) toxicity (EPA, 2003f). If 330 mg/kg is used as the alternative ESV, then 2 soil samples had concentrations that exceeded this alternative ESV. If 1,000 mg/kg is used as an alternative ESV, then none of the soil samples from Range 25 had barium concentrations that exceeded the alternative ESV. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of barium from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Alternative ESVs for beryllium include 36 mg/kg based on mammalian insectivore (shrew) toxicity and 40 mg/kg based on soil invertebrate toxicity (EPA, 2003f). All of the detected concentrations of beryllium were less than thee alternative ESVs; therefore, elimination of beryllium from consideration as a COPEC in surface soil at Range 25 introduces a minimal level of uncertainty to the SLERA.

The level of uncertainty introduced to this SLERA by eliminating chromium from consideration as a COPEC is unclear. The ESV for chromium is based on toxicity to hexavalent chromium, the more toxic form of chromium. It is highly likely that only a fraction of the chromium detected in soil at Range 25 is in the hexavalent form, if any. Consideration of an alternative ESV of 32 mg/kg (based on earthworm toxicity, Efroymson et al., 1997b) would result in fewer exceedences and lower HQ_{screen} values; however, a certain level of uncertainty remains.

Alternative ESVs for cobalt include 13 mg/g based on plant toxicity, 190 mg/kg based on avian insectivore (woodcock) toxicity, and 240 mg/kg based on mammalian insectivore (shrew) toxicity (EPA, 2003f). A number of soil samples exhibited cobalt concentrations that exceeded the alternative ESV based on plant toxicity, but none of the detected concentrations of cobalt exceeded the other alternative ESVs based on wildlife toxicity. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of cobalt from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Iron was detected in a number of surface soil samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron

was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

Alternative ESVs for selenium in soil range from 1.0 mg/kg for the protection of terrestrial plants (Efroymson et al., 1997a) to 70 mg/kg for the protection of terrestrial invertebrates (Efroymson et al., 1997b). Although several soil samples exhibited selenium concentrations that exceed the plant-based ESV, none of the soil concentrations of selenium exceeded the ESV based on protection of terrestrial invertebrates. Based on comparisons to these alternative ESVs, the elimination of selenium from the list of soil COPECs is expected to introduce a minimal level of uncertainty to the results of the SLERA.

An alternative ESV for silver in soil is 4.04 mg/kg, based on exposure to masked shrew (EPA, 2003f). Only one out of 62 surface soil samples exhibited a silver concentration that exceeded this alternative ESV. Based on the relatively low magnitude of the HQ_{screen} value and the fact that only one sample had silver a concentration that exceeded the alternative ESV, the elimination of silver from the list of COPECs likely introduced a minimal level of uncertainty to the results of the SLERA.

Elimination of arsenic, manganese, and vanadium from the list of soil COPECs remains a source of uncertainty in this SLERA.

Range 25 East Surface Soil. Barium, beryllium, and cobalt were eliminated from consideration as COPECs in surface soil at Range 25 East solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were the following: Ba = 1.85, Be = 1.05, and Co = 1.34.

Alternative ESVs for barium include 330 mg/kg based on soil invertebrate toxicity and 1,000 mg/kg based on mammalian herbivore (vole) toxicity (EPA, 2003f). None of the soil samples from Range 25 East had barium concentrations that exceeded the alternative ESVs. Based on the low HQ_{screen} value and the fact that none of the detected concentrations of barium exceeded the alternative ESVs, elimination of barium from the list of soil COPECs at Range 25 East introduced a minimal level of uncertainty to the results of this SLERA.

Alternative ESVs for beryllium include 36 mg/kg based on mammalian insectivore (shrew) toxicity and 40 mg/kg based on soil invertebrate toxicity (EPA, 2003f). All of the detected

concentrations of beryllium were less than these alternative ESVs; therefore, elimination of beryllium from consideration as a COPEC in surface soil at Range 25 East introduces a minimal level of uncertainty to the SLERA.

Alternative ESVs for cobalt include 13 mg/g based on plant toxicity, 190 mg/kg based on avian insectivore (woodcock) toxicity, and 240 mg/kg based on mammalian insectivore (shrew) toxicity (EPA, 2003f). One soil sample exhibited a cobalt concentration that exceeded the alternative ESV based on plant toxicity, but none of the detected concentrations of cobalt exceeded the other alternative ESVs based on wildlife toxicity. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of cobalt from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Range 26 Surface Soil. Aluminum, barium, beryllium, cobalt, iron, manganese, selenium, and zinc were eliminated from consideration as COPECs in surface soil at Range 26 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were the following: Al = 346, Ba = 5.94, Be = 1.14, Co = 1.32, Fe = 229, Mn = 39.4, Se = 1.93, Zn = 1.01.

Per EPA (2003g) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2003g). Since the pH of soils at Range 26 is greater than 5.5, the exclusion of aluminum from the list of COPECs introduced minimal uncertainty into this SLERA.

Alternative ESVs for barium include 330 mg/kg based on soil invertebrate toxicity and 1,000 mg/kg based on mammalian herbivore (vole) toxicity (EPA, 2003f). If 330 mg/kg is used as the alternative ESV, then one soil sample had a concentration that exceeded this alternative ESV. If 1,000 mg/kg is used as an alternative ESV, then none of the soil samples from Range 26 had barium concentrations that exceeded the alternative ESV. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of barium from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Alternative ESVs for beryllium include 36 mg/kg based on mammalian insectivore (shrew) toxicity and 40 mg/kg based on soil invertebrate toxicity (EPA, 2003f). All of the detected concentrations of beryllium were less than these alternative ESVs; therefore, elimination of

beryllium from consideration as a COPEC in surface soil at Range 26 introduces a minimal level of uncertainty to the SLERA.

Alternative ESVs for cobalt include 13 mg/g based on plant toxicity, 190 mg/kg based on avian insectivore (woodcock) toxicity, and 240 mg/kg based on mammalian insectivore (shrew) toxicity (EPA, 2003f). One soil sample exhibited a cobalt concentration that exceeded the alternative ESV based on plant toxicity, but none of the detected concentrations of cobalt exceeded the other alternative ESVs based on wildlife toxicity. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of cobalt from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Iron was detected in a number of surface soil samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

Alternative ESVs for selenium in soil range from 1.0 mg/kg for the protection of terrestrial plants (Efroymson et al., 1997a) to 70 mg/kg for the protection of terrestrial invertebrates (Efroymson et al., 1997b). One soil sample exhibited a selenium concentration that exceeded the plant-based ESV. None of the soil concentrations of selenium exceeded the ESV based on protection of terrestrial invertebrates. Based on comparisons to these alternative ESVs, the elimination of selenium from the list of soil COPECs is expected to introduce a minimal level of uncertainty to the results of the SLERA.

The maximum detected concentration of zinc in surface soil at Range 26 was 50.3 mg/kg resulting in a HQ_{screen} value of 1.01. All of the other detected values of zinc were less than the ESV. Based on the low HQ_{screen} value and the infrequency of detection at elevated concentrations, the uncertainty introduced to the results of the SLERA by eliminating zinc from the list of soil COPECs is minimal.

Elimination of manganese from the list of soil COPECs remains a source of uncertainty in this SLERA.

Ranges South of Range 25 Surface Soil. Arsenic, barium, beryllium, chromium, cobalt, iron, manganese, thallium, and zinc were eliminated from consideration as COPECs in surface soil at the Ranges South of Range 25 solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for these constituents were the following: As = 1.92, Ba = 8.73, Be = 1.67, Cr = 116, Co = 7.0, Fe = 242, Mn = 61.2, Tl = 3.99, and Zn = 1.07.

Arsenic was detected in 2 out of 16 samples at concentrations that exceeded the ESV. Due to the low frequency of detection at elevated concentrations and the relatively low HQ_{screen} value, elimination of arsenic from the list of soil COPECs is expected to introduce a minimal level of uncertainty to the results of the SLERA.

Alternative ESVs for barium include 330 mg/kg based on soil invertebrate toxicity and 1,000 mg/kg based on mammalian herbivore (vole) toxicity (EPA, 2003f). Only one soil sample from the Ranges South of Range 25 had a barium concentration that exceeded the alternative ESVs. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of barium from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Alternative ESVs for beryllium include 36 mg/kg based on mammalian insectivore (shrew) toxicity and 40 mg/kg based on soil invertebrate toxicity (EPA, 2003f). All of the detected concentrations of beryllium were less than these alternative ESVs; therefore, elimination of beryllium from consideration as a COPEC in surface soil at the Ranges South of Range 25 introduces a minimal level of uncertainty to the SLERA.

The level of uncertainty introduced to this SLERA by eliminating chromium from consideration as a COPEC is unclear. The ESV for chromium is based on toxicity to hexavalent chromium, the more toxic form of chromium. It is highly likely that only a fraction of the chromium detected in soil at the Ranges South of Range 25 is in the hexavalent form, if any. Consideration of an alternative ESV of 32 mg/kg (based on earthworm toxicity, Efroymson et al., 1997b) would result in fewer exceedences and lower HQ_{screen} values; however, a certain level of uncertainty remains.

Alternative ESVs for cobalt include 13 mg/kg based on plant toxicity, 190 mg/kg based on avian insectivore (woodcock) toxicity, and 240 mg/kg based on mammalian insectivore (shrew) toxicity (EPA, 2003f). Two soil samples exhibited cobalt concentrations that exceeded the

alternative ESV based on plant toxicity, but none of the detected concentrations of cobalt exceeded the other alternative ESVs based on wildlife toxicity. Based on the low frequency of detection at concentrations exceeding the alternative ESVs, exclusion of cobalt from the list of COPECs is expected to introduce a minimal level of uncertainty to the results of this SLERA.

Iron was detected in a number of surface soil samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

The maximum detected concentration of zinc in surface soil at the Ranges South of Range 25 was 53.5 mg/kg resulting in a HQ_{screen} value of 1.07. All of the other detected values of zinc were less than the ESV. Based on the low HQ_{screen} value and the infrequency of detection at elevated concentrations, the uncertainty introduced to the results of the SLERA by eliminating zinc from the list of soil COPECs is minimal.

Thallium was detected in two out of 16 soil samples from the Ranges South of Range 25 at concentrations that exceeded the ESV. Based on the infrequency of detection at elevated concentrations and the relatively low HQ_{screen} value, the uncertainty introduced to the results of the SLERA through elimination of thallium as a soil COPEC is expected to be minimal.

Elimination of manganese from the list of soil COPECs remains a source of uncertainty in this SLERA.

Surface Water. Barium was the only constituent in surface water that was eliminated from consideration as a COPEC solely on statistical and geochemical comparisons to naturally occurring levels. An alternative ESV for barium in surface water is 0.22 mg/L (EPA, 2003f). All of the detected concentrations of barium in surface water at the BBGR Ranges are less than this alternative ESV. Therefore, elimination of barium from consideration as a surface water COPEC imparts minimal uncertainty on the results of the SLERA.

South Branch Tributaries Sediment. Copper and lead were the only constituents detected in sediment in the South Branch of Cane Creek tributaries that were eliminated from

consideration as COPECs solely on statistical and geochemical comparisons to naturally occurring levels.

An alternative ESV for copper is 28 mg/kg based on the apparent effects threshold (AET) value reported in NOAA's (1999) Screening Quick Reference Table. If this alternative ESV is considered, none of the detected concentrations of copper in South Branch of Cane Creek sediments exceed the alternative ESV. Therefore, elimination of copper from consideration as a sediment COPEC introduces a minimal level of uncertainty to the results of the SLERA.

Elimination of lead from the list of sediment COPECs remains a source of uncertainty in this SLERA.

Ingram Creek Tributaries Sediment. Aluminum, barium, beryllium, chromium, iron, manganese, selenium, silver, and thallium were eliminated from consideration as COPECs in sediment from the Ingram Creek tributaries at the BBGR Ranges solely on statistical and geochemical comparisons to naturally occurring levels in sediment. HQ_{screen} values could not be calculated for aluminum, barium, beryllium, iron, manganese, selenium, or thallium because there are no sediment ESVs for these constituents. An unknown level of uncertainty remains in the results of the SLERA due to the elimination of these constituents from consideration as sediment COPECs. The calculated HQ_{screen} values for chromium and silver were 2.6 and 2.35, respectively. Only one out of nine sediment samples exhibited chromium or silver concentrations that exceeded the ESV. Due to the relatively low magnitude of the HQ_{screen} values and the low frequency of detection at elevated concentrations, elimination of chromium and silver from consideration as sediment COPECs is expected to impart a minimal level of uncertainty to the results of the SLERA.

Groundwater. Barium and manganese were the only constituents eliminated from consideration as COPECs in groundwater solely on statistical and geochemical comparisons to naturally occurring levels. An alternative ESV for barium in surface water is 0.22 mg/L (EPA, 2003f). All but one of the detected concentrations of barium in groundwater at the BBGR Ranges are less than this alternative ESV. Although the entire comparison of groundwater constituent concentrations to surface water ESVs is fraught with uncertainties, elimination of barium from consideration as a groundwater COPEC imparts minimal uncertainty on the results of the SLERA. The elimination of manganese from the list of groundwater COPECs remains a source of uncertainty in this SLERA.

7.8 Summary and Conclusions

The potential for ecological risks at the BBGR Ranges was determined through a SLERA. This ecological screening process consisted of a characterization of the ecological setting at the BBGR Ranges, development of a 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 BBGR Ranges, a description of the ecological receptors, a description of the complete exposure pathways, calculation of screening-level hazard quotients, and a description of the uncertainties within the process.

7.8.1 SLERA Summary

The following sections describe the findings of the SLERA process at each of the BBGR Ranges and the South Branch of Cane Creek and Ingram Creek tributaries in the vicinity of the BBGR Ranges.

7.8.1.1 Range 18 Surface Soil

The following constituents in surface soil at Range 18 exhibited maximum concentrations that exceeded their respective ESVs and naturally occurring levels: antimony, arsenic, beryllium, copper, lead, nickel, zinc, anthracene, benzo(a)pyrene, fluoranthene, pyrene, 4,4'-DDT, MCPP, beta-BHC, endrin ketone, and 2,4-dinitrotoluene. These 16 constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.25 (2,4-DNT) to 466 (lead).

Antimony, beryllium, copper, lead, and zinc were frequently detected at concentrations that exceeded their respective ESVs and naturally occurring levels; therefore, these five metals were identified as COPECs in surface soil at Range 18. All of the remaining constituents were infrequently detected at elevated concentrations, the HQ_{screen} values were relatively low and the pattern of elevated concentrations was not consistent with the use of this site as a firing range. Therefore, the remaining constituents detected in surface soil were not considered COPECs in surface soil at Range 18.

7.8.1.2 Range 20 Surface Soil

Antimony, copper, and lead were frequently detected in surface soil at Range 20 at elevated concentrations relative to their respective ESVs and naturally occurring levels. Additionally, the elevated concentrations of these three metals were detected in areas that would be expected to be maximally impacted by firing range activities (e.g., hillsides that act as impact areas) and are known components of munitions. Therefore, these three metals were identified as COPECs in

surface soil at Range 20. No other constituents were detected in surface soil at Range 20 at elevated concentrations.

7.8.1.3 Range 23 Surface Soil

The following constituents in surface soil at Range 23 exhibited maximum concentrations that exceeded their respective ESVs and naturally occurring levels: antimony, copper, lead, Aroclor-1260, 4,4'-DDE, 4,4'-DDT, MCPP, aldrin, beta-BHC, gamma-BHC, dieldrin, endrin, methoxychlor, 2,4-DNT, 2,6-DNT, and trichloroethene. These 16 constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.1 (Aroclor-1260 and methoxychlor) to 420 (gamma-BHC).

Antimony, copper, and lead were frequently detected in surface soil at Range 23 at elevated concentrations relative to their respective ESVs and naturally occurring levels. Additionally, the elevated concentrations of these three metals were detected in areas that would be expected to be maximally impacted by firing range activities (e.g., target mounds) and are known components of munitions. Therefore, these three metals were identified as COPECs in surface soil at Range 23. All of the remaining constituents were infrequently detected at elevated concentrations, the HQ_{screen} values were relatively low and the pattern of elevated concentrations was not consistent with the use of this site as a firing range. Therefore, the remaining constituents detected in surface soil were not considered COPECs in surface soil at Range 23.

7.8.1.4 Range 25 Surface Soil

Antimony, copper, lead, zinc, and 4,4'-DDT were detected in surface soil samples from Range 25 at concentrations that exceeded their respective ESVs and naturally occurring levels. These five constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 2.3 (4,4'-DDT) to 452 (lead).

Antimony, copper, and lead were frequently detected in surface soil samples from Range 25 at elevated concentrations relative to their respective ESVs and naturally occurring levels. Additionally, copper and lead were detected in patterns consistent with the use of this site as a firing range (e.g., highest concentrations in target impact zones) and are known components of munitions. Therefore, antimony, copper, and lead were identified as COPECs in surface soil at Range 25. Only one sample out of 62 samples collected contained zinc at a concentration that was higher than naturally occurring levels and only two samples out of 12 contained 4,4'-DDT at elevated concentrations. Based on the infrequency of detection and the relatively low magnitude

of the calculated HQ_{screen} values, zinc and 4,4'-DDT were not considered COPECs in surface soil at Range 25.

7.8.1.5 Range 25-East Surface Soil

No constituents were detected in surface soil samples from Range 25-East at elevated concentrations with respect to ESVs or naturally occurring levels; therefore, no COPECs were identified in surface soil at this site.

7.8.1.6 Range 26 Surface Soil

The following constituents in surface soil at Range 26 exhibited maximum concentrations that exceeded their respective ESVs and naturally occurring levels: antimony, copper, lead, anthracene, benzo(a)pyrene, fluoranthene, pentachlorophenol, phenanthrene, pyrene, MCPP, beta-BHC, and endosulfan sulfate. These twelve constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.02 (dieldrin) to 202 (lead).

Antimony, copper, and lead were frequently detected in surface soil samples from Range 26 at elevated concentrations relative to their respective ESVs and naturally occurring levels. The elevated concentrations of copper and lead were detected in areas that would be expected to be maximally impacted by firing range activities (e.g., target mounds and impact areas). Based on the frequency of detection at elevated concentrations, the pattern of the elevated detections, and the fact that they are known components of munitions, antimony, copper, and lead were identified as COPECs in surface soil at Range 26. All of the remaining constituents were infrequently detected at elevated concentrations, the HQ_{screen} values were relatively low and the pattern of elevated concentrations was not consistent with the use of this site as a firing range. Therefore, the remaining constituents detected in surface soil were not considered COPECs in surface soil at Range 26.

7.8.1.7 Ranges South of Range 25 Surface Soil

Antimony, copper, and lead were frequently detected in surface soil samples from the Ranges South of Range 25 at elevated concentrations relative to their respective ESVs and naturally occurring levels. There was no apparent pattern to the elevated detections of these three metals. Due to the fact that these three constituents are known components of munitions and were detected at elevated concentrations, they were identified as COPECs in surface soil at the Ranges South of Range 25.

7.8.1.8 Surface Water

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

7.8.1.9 Sediment

No constituents were detected in sediment from the South Branch of Cane Creek tributaries at elevated concentrations relative to ESVs and naturally occurring levels. Therefore, no COPECs were identified in sediment associated with the South Branch of Cane Creek tributaries in the vicinity of the BBGR Ranges.

Copper, lead, 4,4'-DDT, MCPP, alpha-BHC, gamma-chlordane, heptachlor, and acetone were detected in sediment from the Ingram Creek tributaries at elevated concentrations relative to ESVs and naturally occurring levels. These eight constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.2 (acetone) to 52.4 (gamma-chlordane). Copper and lead were frequently detected at elevated concentrations relative to ESVs and naturally occurring levels and are known components of munitions; therefore, they were identified as COPECs in sediment in the Ingram Creek tributaries. Gamma-chlordane was also identified as a COPEC in the Ingram Creek tributary sediment.

All of the remaining constituents detected in sediment from the Ingram Creek tributaries were infrequently detected at elevated concentrations, and the HQ_{screen} values were relatively low. Therefore, the remaining constituents detected in sediment were not considered COPECs in sediment in the Ingram Creek tributaries.

7.8.1.10 Groundwater

Several constituents in groundwater were detected at concentrations that were elevated with respect to ESVs and naturally occurring levels. These constituents include the following: 4,4'-DDD, 4,4'-DDT, gamma-BHC, endosulfan I. endrin, alpha-chlordane, and heptachlor. These seven constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.4 (endosulfan I) to 77 (4,4'-DDT).

The 4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, and heptachlor detections were all from the same monitoring well located in the center of Range 25. Gamma-BHC was detected in one

monitoring well at Range 18 and alpha-chlordane was detected in one monitoring well at Range 15 and one well at Range 23.

It is important to note that none of the pesticides detected in groundwater were detected in surface water samples from the BBGR Ranges. As stated previously in this chapter, ecological receptors do not have a direct exposure pathway to groundwater. Ecological receptors can only be exposed to constituents in groundwater if groundwater is expressed at the ground surface as seeps or is discharged to lakes or streams via springs. Exposure of ecological receptors to groundwater could then occur via surface water pathways. Because the pesticides in groundwater were detected infrequently and were not detected in surface water samples, it is expected that potential ecological exposure to these compounds is insignificant. Therefore, these compounds were not considered COPECs in groundwater at the BBGR Ranges.

7.8.2 Conclusions of SLERA

The SLERA for the BBGR Ranges determined that several inorganic constituents (i.e., antimony, copper, and lead) were routinely detected in surface soil at the BBGR Ranges at concentrations that exceeded their respective ESVs and naturally occurring levels. Beryllium and zinc were also detected in surface soils at Range 18 at elevated concentrations. Additionally, copper, lead, and gamma-chlordane were detected in a number of sediment samples from Ingram Creek tributaries at concentrations that exceeded their respective ESVs and naturally occurring levels. No constituents were identified as COPECs in surface water or groundwater at the BBGR Ranges. Although several constituents in surface water and groundwater exceeded their respective ESVs, additional lines of evidence indicated that none of the detected constituents in surface water or groundwater should be considered COPECs.

Thus, the potential exists that these constituents in surface soil and sediment may pose adverse ecological risks to the flora and/or fauna at the BBGR Ranges. Although several PAH compounds, chlorinated herbicides and pesticides, and VOCs were also detected in environmental media at the BBGR Ranges at concentrations that exceeded their respective ESVs and naturally occurring levels, 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 BBGR Ranges.

Because several constituents were detected in environmental media at the BBGR Ranges at concentrations that exceeded their respective ESVs and naturally occurring levels, and because conservative assessment techniques were used in the SLERA process, a more thorough

assessment (e.g., baseline ecological risk assessment) is warranted to reduce uncertainties inherent in the SLERA process and to determine the potential for ecological risk at the BBGR Ranges.

8.0 Summary, Conclusions, and Recommendations

This chapter summarizes the results of the RI conducted at the BBGR Ranges and presents the major conclusions of the investigation. In addition, recommendations are made regarding future actions at these sites.

8.1 Summary of RI Field Activities

RI field activities included environmental sampling, monitoring well installation, and XRF soil screening. Environmental sampling consisted of the collection and analysis of 339 surface and depositional soil samples, 144 subsurface soil samples, 19 groundwater samples, 12 surface water samples, and 12 sediment samples. The samples were analyzed for a variety of chemical parameters including metals, VOCs, SVOCs, pesticides, herbicides, PCBs, and explosive compounds, although not all samples were analyzed for all parameters. To facilitate groundwater sample collection and to provide site-specific geological information, 21 monitoring wells, including 4 residuum wells and 17 bedrock wells, were installed at the BBGR Ranges. In addition, XRF screening was performed to further delineate the extent of metals contamination in surface soil and to evaluate potential contamination within the Range 25 safety fan.

8.2 Nature and Extent of Contamination

To evaluate the nature and extent of contamination at the BBGR Ranges, the analytical data were compared to human health SSSLs, ESVs, and background screening values for FTMC. The SSSLs and ESVs were developed by Shaw for human health and ecological risk evaluations as part of investigations performed under the BRAC Environmental Restoration Program at FTMC. Site metals data were further evaluated using statistical and geochemical methods to determine if the metals detected in site media are site related.

8.2.1 Surface and Depositional Soil

A total of 339 surface and depositional soil samples were collected for chemical analysis. Metals, VOCs, SVOCs, pesticides, herbicides, two explosive compounds, and one PCB were detected in the samples.

Metals. The concentrations of 11 metals exceeded SSSLs and background: aluminum, antimony, arsenic, barium, chromium, copper, iron, lead, manganese, thallium, and vanadium. Eighteen metals were detected at concentrations exceeding ESVs: aluminum, antimony, arsenic, barium, beryllium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium,

silver, thallium, vanadium, and zinc. However, lead was frequently detected in surface soil at levels above screening levels and is the major constituent of the small-arms ammunition known to have been used at the BBGR Ranges. Lead was detected in all of the surface and depositional soil samples for which it was analyzed. Ninety-three lead results exceeded the residential SSSL (400 mg/kg) and background (40 mg/kg), and 191 results exceeded the ESV (50 mg/kg) and background. The lead concentrations were plotted as an isopleth map (Figure 4-1) that showed contamination at the BBGR Ranges is generally present in target areas and in hillside impact zones. The exceptions were Range 18 and the Ranges South of Range 25, both of which are relatively flat areas and exhibited widespread lead contamination over a large portion of the area of investigation. Other metals detected at concentrations exceeding risk-based screening values and known to be constituents of ammunition included antimony, copper, and zinc, although these metals were almost exclusively co-located with the elevated lead results. In addition to lead, the statistical and geochemical evaluation concluded that antimony and copper concentrations were anomalously high in numerous surface soil samples. Arsenic, beryllium, and nickel results in one sample each, and zinc results in three samples, were also determined to be anomalously high.

Organic Compounds. One explosive compound (2,4-dinitrotoluene) and five PAH compounds were detected at levels exceeding SSSLs in a limited number of the surface soil samples. VOC, pesticide, herbicide, and PCB concentrations in surface soil were all below SSSLs.

Several pesticides, six SVOCs (primarily PAHs), two explosive compounds (2,4-dinitrotoluene and 2,6-dinitrotoluene), one VOC (trichloroethene), one herbicide (MCPP), and one PCB (Aroclor 1260) were detected sporadically in surface soils at levels exceeding ESVs. These compounds were infrequently detected at levels above ESVs and showed no discernable pattern of contamination.

8.2.2 Subsurface Soil

One hundred and forty-four subsurface soil samples were collected for chemical analysis. Metals, VOCs, SVOCs, pesticides, one herbicide, and one explosive compound were detected in the samples. PCBs were not detected in the subsurface soil samples analyzed for these compounds.

Metals. The concentrations of nine metals exceeded SSSLs and background: aluminum, antimony, arsenic, chromium, iron, lead, manganese, thallium, and vanadium. However, lead was the major metal contaminant in subsurface soil. Lead was detected in all 128 subsurface soil

samples analyzed for lead and six results, ranging in concentration from 593 to 2,630 mg/kg, exceeded the residential SSSL (400 mg/kg). Subsurface soil lead concentrations were plotted as an isopleth map (Figure 4-3) that showed a very limited areal extent of subsurface soil contamination within the Range 20 impact zone and at some Range 23 target mounds. Antimony was also detected at concentrations exceeding risk-based screening values and is also known to be a constituent of ammunition. The statistical and geochemical evaluation concluded that concentrations of antimony (in 7 samples), copper (14 samples), and lead (14 samples) were anomalously high in the subsurface soil samples. Beryllium, mercury, and zinc results were also determined to be anomalously high in one sample each.

Organic Compounds. All detected levels of VOCs, SVOCs, pesticides, herbicides, and explosive compounds in subsurface soil were below SSSLs.

8.2.3 Groundwater

Nineteen groundwater samples were collected for chemical analysis. Metals, pesticides, explosives, and one VOC were detected in the samples. SVOCs, herbicides, and PCBs were not detected in the groundwater samples analyzed for these compounds.

Metals. The concentrations of three metals exceeded SSSLs and background: barium, manganese, and thallium. However, the statistical and geochemical evaluation concluded that these metals were present at naturally occurring levels.

Organic Compounds. The concentrations of three pesticides (beta-BHC, gamma-BHC, and heptachlor) and three explosive compounds (2,4-dinitrotoluene, 2,6-dinitrotoluene, and 4-amino-2,6-dinitrotoluene) exceeded their respective SSSLs in one well each. The explosives were detected sporadically in groundwater and showed no discernable pattern of contamination. Based on the historical use of the BBGR Ranges as weapons training ranges, it is likely that the pesticides are isolated, low-level detections.

8.2.4 Surface Water

Twelve surface water samples were collected for chemical analysis. Metals and one VOC were detected in the samples. SVOCs, pesticides, herbicides, PCBs, and explosives were not detected in the surface water samples analyzed for these compounds.

Metals. Only one metal (arsenic) exceeded its SSSL and background. However, the analytical result was "B"-flagged during data validation, indicating that arsenic was a laboratory-related

contaminant. The concentrations of two metals (barium and beryllium) exceeded ESVs and background. The beryllium result was "B" flagged indicating that it was a laboratory-related contaminant.

The statistical and geochemical evaluation concluded that all of the metals detected in surface water were present at naturally occurring levels.

Organic Compounds. Only one VOC (methylene chloride) was detected in the surface water samples. All results were below the SSSL and ESV.

8.2.5 Sediment

Twelve surface water samples were collected for chemical analysis. Metals, VOCs, pesticides, and one herbicide were detected in the samples. SVOCs, explosives, and PCBs were not detected in the samples.

Metals. Only one metal (lead, 533 mg/kg) exceeded its SSSL (400 mg/kg) and background (37.8 mg/kg). The elevated lead concentration was detected in one sediment sample collected at Range 23. The concentrations of four metals (chromium, copper, lead, and silver) exceeded ESVs and background. Copper and lead concentrations exceeded ESVs and background in one sample each from Ranges 18 and 20, and in two samples from Range 23. Chromium and silver concentrations exceeded ESVs and background in one sample from Range 25.

The statistical and geochemical evaluation concluded that the metals detected in sediment were present at naturally occurring levels except for copper and lead in one sample each from Range 23 (Figure 4-5).

Organic Compounds. VOC, pesticide, and herbicide concentrations in sediment were all below SSSLs. The concentrations of four pesticides exceeded ESVs: 4,4'-DDT (in one Range 20 sample), alpha-BHC (one Range 20 sample), gamma chlordane (one sample each from Range 20 and Range 25 East), and heptachlor (two Range 23 samples). In addition, one VOC (acetone) exceeded its ESV in one sediment sample from Range 23.

8.3 Contaminant Fate and Transport

Formation of leachate and transport in soil waters is a viable transport mechanism for organic compounds at the BBGR Ranges. However, concentrations of most compounds were below

default EPA screening levels (i.e., DAF_{20} SSLs), and groundwater concentrations are expected to remain below their regulatory limits.

Lead, arsenic, antimony, and copper are the only metals at the BBGR Ranges that exceed the DAF_{20} SSLs; however, these metals appear to be nearly immobile in the subsurface environment. Concentrations of antimony and lead in sediment indicate that erosion of surface soils is an active transport mechanism at the BBGR Ranges, and continued transport can be expected unless some action is taken.

A few PAHs, explosive compounds, and pesticides were detected at concentrations above DAF_{20} SSLs for surface soils; however, the impact of these compounds is largely limited to the surface soils. The pesticides are nearly immobile in the soil system, and the explosive compounds may be removed via microbial activity; thus, impact to the groundwater is not expected. Concentrations of the pesticide alpha-BHC in sediment indicate that erosion of surface soils may be transporting contaminated soils.

8.4 Streamlined Human Health Risk Assessment

An SRA was performed to evaluate risks to human health from exposure to environmental media at the BBGR Ranges. The SRA consisted of CSEM development, data evaluation, exposure and toxicity assessments, risk characterization, and uncertainty analysis. Four plausible receptors were evaluated in the SRA: recreational site user, groundskeeper, construction worker, and resident. The analytical data were evaluated, metals recognized as essential nutrients and those present at background concentrations were removed from evaluation, and COPCs – those chemicals that may contribute significantly to risk – were identified. Metals associated with bullets and shot (predominantly lead, but also antimony and copper) were the primary COPCs. Some nitroaromatic compounds, possibly from gunpowder, and some organochlorine pesticides, probably used for insect control, were sporadically detected and selected as COPCs.

Cancer risk and noncancer hazard estimates were calculated for each COPC and compared to threshold values to identify COCs (i.e., COPCs present at levels that raise concern for adverse human health effects). The following table summarizes risks by range and receptor. At ranges where unacceptable human health threats have been identified, the COC(s) and environmental media are listed.

Site	Receptor Scenario			
	Recreational Site User	Groundskeeper	Construction Worker	Resident
Range 18	Acceptable	Lead in surface soil	Lead in surface soil	Antimony, copper, and lead in surface soil
Range 20	Acceptable	Lead in surface and subsurface soil	Lead in surface and subsurface soil	Lead in surface and subsurface soil
Range 23	Acceptable	Acceptable	Lead in surface and subsurface soil	Lead in surface and subsurface soil
Range 25	Acceptable	Acceptable	Lead in surface soil	Lead in surface soil
Range 26	Acceptable	Acceptable	Lead in surface soil	Lead in surface soil
Ranges 25 East and 28	Acceptable	Acceptable	Acceptable	Acceptable
Ranges South of Range 25	Acceptable	Lead in surface soil	Lead in surface soil	Lead in surface soil

"Acceptable" indicates calculated risks are within acceptable limits.

As shown in the table above, all of the ranges are suitable for recreational site use. In the other receptor scenarios evaluated, lead in soil is the primary risk driver. No COCs were identified for Range 25 East and Range 28. Therefore, these ranges are suitable for release without restriction or further investigation.

An important source of uncertainty is the presence of bullets and lead fragments on the ground surface at all of the remaining BBGR Ranges. Environmental degradation of bullets could result in increased lead concentrations in soil, thereby increasing the hazard associated with human exposure to the BBGR Ranges.

8.5 Screening-Level Ecological Risk Assessment

A SLERA was performed to evaluate potential risks to ecological receptors from exposure to environmental media at the BBGR Ranges. This ecological screening process consisted of a characterization of the ecological setting at the BBGR 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 BBGR Ranges, a description of the ecological receptors, a description of the complete exposure pathways, calculation of screening-level hazard quotients, and an uncertainty evaluation.

Antimony, copper, and lead in surface soil were the predominant COPECs identified in the SLERA. These three metals were identified as COPECs at all of the BBGR Ranges except Range 25 East and Range 28. Beryllium and zinc were also identified as COPECs in surface soil at Range 18. Additionally, copper, lead, and gamma-chlordane were determined to be COPECs

in Ingram Creek tributary sediments. No constituents were identified as COPECs in surface water or groundwater associated with the BBGR Ranges. The SLERA concluded that a more thorough assessment is warranted to reduce uncertainties and to better determine ecological risks.

8.6 Recommendations

Based on the results of the RI, no further investigation is warranted to define the nature and extent of contamination at the BBGR Ranges. Shaw recommends conducting a BERA to complete the ecological risk assessment process. Following the BERA, a feasibility study should be performed to evaluate and select a remedial alternative for the protection of human health and the environment.