REMEDIAL INVESTIGATION REPORT
ANNAPOLIS LEAD MINE SUPERFUND SITE
CERCLIS ID #: (MO0000958611)
OPERABLE UNIT-3 (SOIL IN THE TOWN OF ANNAPOLIS, MISSOURI)
IRON COUNTY, MISSOURI

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Appendices

I. Introduction

The Remedial Investigation (RI) of the Annapolis Lead Mine (ALM) Operable Unit-3 (OU-3) has been completed by the United States Environmental Protection Agency (EPA). OU-3 is defined as soil in the town of Annapolis, Missouri. Field activities for OU-3 have been completed by the United States Environmental Protection Agency (EPA) in conjunction with the Missouri Department of Natural Resources (MDNR).

II. Purpose

The purpose of this report is to define the nature and extent of mine waste contamination, if any, in the town of Annapolis. This report is to serve as the basis for determining the optimum remedy for the ALM site OU-3 under the National Contingency Plan (NCP).

III. Site Background and Description

The ALM site is located approximately .75 miles north of Big Creek. The mine operated between the years of 1919 and 1940. Mining activities at the ALM site included the excavation of ore bodies, the crushing and concentrating of ore and storage of the concentrated metals prior to offsite shipment for smelting. The crushing and concentrating wastes (tailings) were disposed of on the surface of the property within a ravine that is a tributary of Sutton Branch Creek. The resulting pile of tailings has been stabilized under an engineered cap and occupies approximately ten acres of the site. Tailings residue is present in the substrate of Sutton Branch Creek for approximately .75 mile downstream of the site, where it merges with Big Creek. It has been estimated that 1,173,000 tons of tailings were deposited in the tailings pile area during the period of mining operations. The tailing piles and certain eroded deposits within OU-1 were the subject of a removal action which installed the cap prior to this RI. The OU-1 removal action resulted in the consolidation and covering of the tailings piles as well as the return of some of the outwash material to the pile prior to installation of the cap.

The former mine and impacted area are located approximately one mile east northeast of Annapolis, Missouri. Runoff from the former mine operation entered Sutton Branch Creek which flows downstream into Big Creek. The area affected by the mining wastes is considered rural/residential. OU-1 is defined as the Sutton Branch Creek floodplain from the probable point of entry (PPE) to the confluence with Big Creek, as well as the historic tailings pile and mine area and is approximately 200 acres in size. OU-2 is defined as Big Creek from the mouth of Sutton Branch Creek downstream to the confluence with the St. Francois River, which is a total of approximately 20 miles of stream. OU-3 is defined as the soil in the town of Annapolis. OU-3 is the focus area of this report. A site map can be found in Figure 1, Appendix B.

A. Site (Source) History

- 1919 - 1940 Operation period of the mine.
- 1982 St. Joseph Lead Company sold the surface rights of the property to private individuals but retained the mineral rights.
- 1987 Doe Run acquired the mineral rights through a buy out or ownership transfer
- 1992 MDNR collected water and sediment samples along Sutton Branch Creek, the receiving stream of runoff from the site

- 1993 MDNR referred the site to EPA as a potential Hazardous waste site
- 1996 EPA completed a Screening Site Inspection (SSI) at the site
- 1997 EPA conducted emergency response activities at the site. Site residents were relocated off site. These activities were in response to elevated blood lead levels found to exist in two children living with their family in a dwelling established in the foundations of the former grinding/milling plant
- 1999 EPA completed an Expanded Site Inspection and Removal Assessment (ESI/RA) The ESI/RA focused on documenting, for the purpose of listing the site on the National Priorities List (NPL) and/or initiating a removal action, the extent of metals contamination across the site and in the receiving stream
- 2004 EPA listed the site on the NPL. EPA initiated and completed a removal action on the northern portion of the site. The waste piles were consolidated and covered.
- 2005 EPA completed a RI/FS, Proposed Plan, and ROD for OU-1.

B. Previous Investigations of the Source Area

- **1992 Preliminary Assessment**

In September 1992, the MDNR collected water and sediment samples downstream of the ravine that drains the tailings pile. Analysis of the samples demonstrated that elevated levels of lead, arsenic, cadmium, zinc, nickel and copper exist in the sediments of the receiving stream, Sutton Branch Creek. The state conducted no source area sampling of sediment, soil, surface water, or ground water. The site was subsequently forwarded to the EPA as a potential hazardous waste site.

- **1996 Screening Site Inspection**

In June 1996, EPA completed a Screening Site Inspection (SSI) at the ALM site. The SSI focused primarily on evaluating the site in accordance with the national Hazardous Ranking System (HRS). XRF analysis of soil samples taken during the SSI revealed lead concentrations in the tailings pile as high as 2,570 parts per million (ppm) and lead concentrations around the on-site residence as high as 27,500 ppm. Around the former mine operations areas lead was found in soil as high as 28,300 ppm. Eight soil samples (plus one duplicate) were collected for laboratory analysis to confirm the XRF readings and to provide data for the soil exposure pathway. Three sediment and surface water samples also were collected from Sutton Branch Creek. Soil samples from the site and sediment samples collected from Sutton Branch Creek, contained elevated levels of six metals, which are listed in Table 1. Surface water samples from Sutton Branch Creek displayed elevated levels of lead, with concentrations up to 11.6 micrograms per liter (μg/L). Arsenic, cadmium, and thallium were also found to exist at elevated levels at the site. The SSI recommended that an Expanded Site Inspection be performed due to an observed release of hazardous materials to the surface water and soil at the site.

- **U.S. Fish and Wildlife Service (USFWS) studies**

Two USFWS Service studies conducted on aquatic life in Big Creek have shown evidence of heavy metal contamination in fish species. Fish collected downstream from the ALM site in Big Creek were evaluated by the USFWS in two studies; one completed in 1993, the other in 1997. Cumulatively, these studies revealed that lead and cadmium concentrations are elevated in fish found in Big Creek.

Both studies involved the enzyme o-aminolevulinic acid dehydratase (ALA-D), which catalyzes formation of a hemoglobin precursor, porphobilinogen (PBG), from aminolevulinic acid. ALA-D is highly sensitive to lead and relatively easy to measure. The inhibition of ALA-D activity is used as a biomarker for lead exposure in humans, waterfowl, and, more recently, in fish. The objective of the first study, conducted in 1989 and 1990, was to verify and calibrate the biomarker of lead exposure for use in a statewide assessment of metals pollution from lead and zinc mining, and to determine whether metals other than lead and zinc affect ALA-D activity. Big Creek was chosen as a sampling site because it is near the ALM site. The studies indicated that lead concentrations in fish blood at sampling locations downstream of the confluence of Big Creek and Sutton Branch Creek were elevated significantly higher than at upstream locations along Big Creek. Cadmium concentrations were greatest downstream of Annapolis and Sutton Branch Creek. Study authors cited the ALM site as a probable source and suggested continued monitoring. These studies were conducted prior to the Time-Critical Removal Action in 2003, which significantly reduced the loading of mine waste to Big Creek. The Glover Smelter is also a potential contributor to the creek.

- **1997 Emergency Response**

In March 1997, EPA collected additional dust and wipe samples from the then existing on-site residence. Soil and ground water well samples were also taken at this time. An XRF was used to screen surface soils at the site, and the ten sample locations subsequently were selected from those screened points to provide a wide range of concentrations for a site-specific XRF calibration model that might be required. The samples were collected in response to detection of high lead levels in the blood of the two children residing in the on-site residence. The wipe samples were analyzed for the eight Resource Conservation and Recovery Act metals, and the soil and ground water samples were analyzed for 24 metals by the EPA Region 7 laboratory in Kansas City, Kansas. Results from these samples along with the results from blood-lead samples taken from the children were used in making a determination that individuals living on site were being adversely impacted. In May 1997, the EPA performed a removal action which resulted in the Iron County Division of Family Services relocating the children and their immediate family from the site.

Table 1 (Appendix A) shows analytical results for ground water well samples collected during the emergency response.

- ***1999 Expanded Site Inspection and Removal Assessment***

An Expanded Site Inspection and Removal Assessment (ESI/RA) of the northern segment of the ALM site was completed by the EPA in February 1999. The ESI/RA focused on documenting the extent of metals contamination across the site and in the stream receiving site runoff. Nineteen ground water, 11 surface water, 19 surface and subsurface soil, and 13 sediment samples were collected during the sampling activities, including background and quality control (QC) samples. The samples were analyzed for total metals. Water samples also were analyzed for dissolved metals.

Over 100 in situ readings were collected with an XRF during the ESI/RA. Soil profiling samples also were collected with a Geoprobe® in waste source areas—including portions of the chat and tailings pile—to determine approximate depths of mining wastes across the site. The chat and tailings pile was found to contain mining waste to a depth of 21 feet.

During the ESI/RA, concentrations of on-site lead were found to be as high as 20,000 ppm. Lead was found in the sediment of Sutton Branch Creek at levels as high as 2,900 ppm. The surface water of Sutton Branch Creek exhibited lead at concentrations of 17.4 ppm. The ESI/RA estimated the amount of lead-contaminated tailings, chat, and soil (above 500 ppm) at 51,677 cubic yards. Much of this volume was located in the tailings pile which was estimated to contain approximately 39,000 cubic yards of mining waste.

Approximately ten percent of the screening locations were sampled for laboratory confirmation analysis (see Table 2). Analytical results indicated lead as high as 7,000 mg/kg in sample number -309, and 7 of the 12 samples collected were above the EPA removal action level (RAL) of 400 mg/kg. Arsenic also was found at levels exceeding three times above background concentrations and above the residential RAL screening level in four of the six confirmation samples. Cadmium and zinc were detected at levels exceeding three times above background but not exceeding health-based screening levels. Laboratory confirmation results for XRF lead screening from the November 1999 ESI/RA are listed Table 2.

The soil sample with the highest lead concentration (7,000 mg/kg) was collected 300 feet north of the chat and tailings pile from the mill slime pond. This area consistently produced the highest XRF screening values (six surface screening values ranged from 5,700 to 9,290 mg/kg). Areas of lead contamination above 500 mg/kg also were detected around the then existing on-site residence and other structures associated with the former mining operations. XRF screening results in other locations of the former mining area ranged from 105 mg/kg to 3,362 mg/kg for lead in soil. Although several other metals were detected during the sampling event, only arsenic was found above a health-based benchmark (cancer risk of 0.43 mg/kg); however, background concentrations also were found above the same benchmark.

Laboratory analyses substantiated visual observations of mining waste in Sutton Branch Creek. Elevated levels of arsenic, cadmium, lead, and zinc were reported in the ESI/RA in surface water samples collected from Sutton Branch Creek. Heavy metals were also found above designated background concentrations and ecological threshold values in sediment samples collected along the surface water pathway. Lead was found as high as 2,600 mg/kg in sediment samples collected from the chat and tailings pile outfall, and as high as 1,700 mg/kg at the confluence of Sutton Branch Creek and Big Creek (designated wetland area), .75 mile downstream of the site. Other contaminants— including arsenic, cadmium, and zinc—were found in sediment samples collected along Sutton Branch Creek, at levels above background and ecological-based screening levels.

Elevated concentrations of contaminants, possibly attributable to the site, were found in surface water collected from the furthest downstream sampling location in Big Creek—approximately 1,300 feet downstream of the confluence with Sutton Branch Creek. In addition, total and dissolved lead at levels above background screening levels and Ambient Water Quality Criteria (AWQC) standards were found in surface-water samples collected from Sutton Branch Creek. Cadmium was identified above background levels and the AWQC standard in one surface water sample from Big Creek, collected 100 feet downstream of the confluence with Sutton Branch Creek. Table 3 lists analytical results for surface water samples.

Data collected during the ESI/RA indicated that the ALM site has had an impact on the environment, primarily through the surface water pathway. Tailings from the site were migrating to Sutton Branch Creek. Evidence of elevated levels of lead and cadmium in Big Creek fish were found, and the threat to human health through the consumption of contaminated fish was considered high. Further, elevated metals have been found at a known wetland area (the confluence of Sutton Branch Creek and Big Creek). This contamination may be affecting the ecological system of this sensitive environment and other wetland systems further downstream of the confluence.

None of the domestic wells sampled within a 1-mile radius had contaminant concentrations exceeding maximum contamination levels (MCL). However, arsenic was reported in at least one private well at a concentration exceeding the EPA Reference Dose (RfD) or EPA Cancer Risk (CR) level. An on-site irrigation well was found to be contaminated with total lead and cadmium; this shallow ground water contamination is most likely attributable to the source(s) on site. However, poor construction of the well (the lack of a surface seal) may have resulted in elevated concentrations that are not necessarily representative of the local ground water. Lead and cadmium were identified in several wells on and adjacent to the site during the EPA SI in November 1997.

Tetra Tech START (contractors working for EPA) and EPA have estimated the volume of lead-contaminated soils that may require excavation and/or stabilization. Quantity calculations were derived from integrating visual inspection information, screening and analytical data, and mapping techniques. Based on this information and historical documentation, four lead-contaminated source areas were delineated for removal assessment purposes: the heavily eroded chat and tailings waste pile, the outwash area of the chat and tailings waste pile, the former mining operations area, and the mill slime pond. An estimated 51,677 cubic yards of lead-contaminated tailings, chat, and soil (above 500 mg/kg) were calculated for these four areas.

- **2003 Time Critical Removal Action**

An extensive removal has been performed by EPA in the northern portion of the ALM site (OU-1) and although not officially complete, the majority of the removal response activity has been completed. Contouring and erosion repair along with some sediment control actions and final seeding to establish a vegetative cover for the cap is all that remains at the time of this writing. The original waste piles, including some of the scattered mine wastes which were returned to the vicinity of the waste piles, have been contoured and covered with a clay cap. The purpose of the cap is to prevent further erosion of the waste pile and thus eliminate future spread of contamination from the waste pile from reaching Sutton Branch Creek and ultimately Big Creek.

- **2005 Record of Decision for OU-1**

The Record of Decision for OU-1 (2005) includes the following activities:

- Addition of phosphate to floodplain soils (away from the outer edge of riparian zone) during the dry season to improve the density of vegetation and to reduce the bioavailability of lead to terrestrial receptors.

- Mining wastes in heavily forested, thickly vegetated areas, such as the riparian buffer, will not be subject to excavation, consolidation, or capping.

- Excavation of sediments from Sutton Branch Creek in pockets, or depositional areas. The amount of excavation will be determined during the Remedial Design (RD) phase.

- Placement of excavated sediments in the existing repository area and cap with a soil cover.

- Stabilization of the Sutton Branch Creek channel with large rock and/or other material to prevent wash-outs and stream channel meandering. The extent of stabilization will be determined during the RD phase.

- Implementation of Institutional Controls

- Performance of annual monitoring to determine remedial effectiveness. The monitoring frequency will be evaluated to determine whether it should be more frequent or can be extended to periods beyond annual monitoring.

- Regular water quality monitoring (including phosphorus) will be carried out by MDNR at established monitoring stations, pursuant to the Clean Water Act (CWA).

- MDNR will manage post-removal maintenance of the protective cover consistent with all federal and state laws.

- The Remedial Action for OU-1 should occur in early summer 2007 and will be completed by the end of the summer of 2007.

- **2006-2007 OU-2 Remedial Investigation**

The investigation of the OU-2 area included over bank and stream sediment sampling, along with qualitative habitat evaluations at each sampling site. A total of 49 sediment samples were taken from Big Creek in areas where stream sediment deposition was evident. The results were below levels of concern for all receptors.

C. Review of Historic Aerial Photographs

The review of historic photographs can be found in the Administrative Record for the Annapolis Lead Mine Site, specifically in the Remedial Investigation/Feasibility Study (RI/FS) for OU-1.

D. Study Area Investigation

The investigation of the town of Annapolis (OU-3) followed the Superfund Lead-Contaminated Residential Sites Handbook (OSWER 9285.7-50, 2003). Eighty-five properties were sampled including church yards, school yards, and residential yards. The sampling technique was dependent upon the size and shape of the yard. Figure 1 shows the area sampled during this RI. Examples of the recommended soil sampling techniques are shown in Figures 2, 3, and 4, Appendix B. Soil was collected at each of the residential yards and analyzed for lead concentration (milligram/kilogram). Quart size (32 ounces) heavy duty freezer bags were used to contain the soil samples. The soil was mixed thoroughly to promote homogeneity. After mixing, the samples were analyzed by XRF spectrometry. Field personnel transported confirmation samples (10%) to the laboratory. Confirmation samples were analyzed for Target Analyte List (TAL) metals.

E. Site Characterization

i. Surface Features

Lead contamination of the surface soil was the primary focus of this investigation. Most of the properties sampled were private residences, as well as church yards and the school yard. Specific surface features were not noted. All areas were sampled following the guidance in the Superfund Lead-Contaminated Residential Sites Handbook (OSWER 9285.7-50, 2003).

ii. Contaminant Source Investigations

Contaminant source investigations have shown one possible source of lead contamination in Annapolis. The historical mining area (OU-1) is the primary possible source of contamination.

iii. Meteorological Investigations

a. Wind

The following information was developed from the Glover area approximately 13 miles upstream of Annapolis. This data should be similar for the Annapolis site. There are two primary wind patterns in the valley in the Glover area. Winds from the south predominate during daylight hours between roughly 8:00 A.M. through 9:00 P.M. dependent on season, with speed varying considerably up to eight miles per hour or greater. Winds from the north-northeast predominate during evening hours caused by valley drainage flow. These winds are generally light and variable, up to 11 miles per hour or greater.

During synoptic conditions associated with northwestern weather fronts, winds can achieve significant speeds, up to 16 miles per hour from the northwest. During storm events winds achieve considerably higher velocities.

b. Precipitation --The area averages 44 inches per year, mostly as rain.

c. Temperature

EPA reviewed monthly climate data for Iron County, Missouri, compiled by the High Plains Regional Climate Center. The nearest weather station is in Arcadia, Missouri, which is about 15 miles north of the site. The period of record was from June 1, 1918, to July 31, 2000. The data indicate that the climate of Iron County is characterized by relatively hot summers and moderately cold winters. Rainfall is well distributed throughout the year and constitutes most of the annual precipitation of 44 inches. The prevailing wind direction is southerly (U.S. Department of Agriculture [USDA] 1991). The average monthly and annual climate data are in Table 4, Appendix A.

F. Soils

i. Geology

No significant investigations of a geologic nature was required or undertaken for this report. The following information was taken from the St. Francois River Watershed Inventory and Assessment (MDC, 2001). The headwater area is dominated by the Ozark uplift (St. Francois Mountains) which has exposed outcrops of Precambrian igneous rock (granite, rhyolite, felsite) on as much as 50 percent of the surface on some slopes. The hard igneous rock has no overburden, and shut-ins, cascades, and waterfalls produce ancient rigid boundaries that control the course, gradient, and floodplain features of the first 80 miles of the river channel. Downstream, igneous rock is replaced by hard Cambrian dolomites and sandstone. Eventually, cherty Ordovician dolomite becomes the primary underlayment adjacent to the Wappapello Lake basin.

The absence of a deep cherty residuum in the igneous Ozark uplift and the formation of erosion resistant upland soils results in little gravel accumulation in the alluvial floodplain soils. Channel substrates contain a significant proportion of stable cobble, stone, and boulders, and stream bank soils are more cohesive than in most Ozark streams because of lower densities of gravel. The Big Creek watershed is not strongly influenced by the St. Francois Mountains uplift. It is similar to the adjacent Black River basin, with its deep, cherty limestone residuum. The result is an abundance of gravel in Big Creek (MDC, 2001).

ii. Soil Types

Site-specific soil investigations were required for this report. This sole reason for this investigation was to measure the concentration of heavy metals in the soil of the town of Annapolis. However, site-specific soil types were not required for this report. General soil types and conditions are described in the following paragraphs.

Soils formed in the hard, igneous rock of the upland ridge tops lack an overburden of chert or loess and are typically described as extremely bouldery, cobbly, or stony with outcrops sometimes occupying 50 percent of the surface area. Fertility is low, reactions are acidic, runoff is rapid, and water capacity is low, which produces extremely droughty conditions most suitable for woodland and limited grass production. Soil series most frequently associated with the uplands are Irondale, Syenite, Delassus, and Clarksville.

The finer silt-loam soils formed on the slopes also contain a large proportion of stones and boulders, and a chert overburden appears on some foot slopes. A fragipan is usually present which can restrict root depth to less than three feet. Soil fertility is low, reactions are acidic, runoff is rapid, but water capacity is high and droughty conditions are limited to hot, dry summer periods. Some of the soils on the slopes can be tilled, but erosion hazards and low crop yields tend to limit agriculture activities to hay and pasture production. Soil series most frequently associated with the slopes are Auxvasse, Killarney, Courtois, Fourche, and Wilber.

The sand-silt-clay loams formed in floodplains are highly fertile, but fertility tends to decrease to moderate in a downstream direction. Soils range from neutral to only slightly acidic, runoff is moderate, and water capacity is high. Most of the floodplain soils can be tilled without a serious erosion threat, but hay and pasture products can often produce better yields than row crops. Soil series most frequently associated with the floodplains are Wakeland, Haymond, and Pope (MDC, 2001).

iii. Hydrogeology

The hydrogeology of the Big Creek watershed is not greatly influenced by the St. Francois Mountains uplift, but, instead, is more closely related to the deep, cherty limestone residuum of the upper Black River basin. The unconsolidated alluvium throughout the watershed provides subsurface storage and allows rapid groundwater movement that sustains and stabilizes base flows (MDC, 2001).

iv. Demography

The population for Annapolis was 310 residents as of the year 2000. There are approximately 136 occupied houses in town (city-data.com, 2007).

v. Ground Water Investigations

No groundwater investigations were conducted. It was determined that investigation of groundwater would not be necessary based on the results from the investigations of groundwater in the mining area prior to the Time Critical Removal Action. Prior investigations showed concentrations that were limited and of little concern in the source area (OU-1).

vi. Ecology

Region 7 staff conducted a Baseline Ecological Risk Assessment (BERA) of OU-1. An addendum to the BERA was done to identify the impacts of the removal on the conclusions found in the Ecological Risk Assessment. The BERA and the Addendum to the BERA can be found in Appendix D.

vii. Surface Water and/or Sediment Investigations

No surface water investigations were conducted for OU-3.

G. Physical Characteristics of the Study Area

The physical setting of the study area was the town Annapolis (Figure 1, Appendix A).

i. Soil in the town of Annapolis included the following:

1. Residential Soil
2. School Soil
3. Church Soil
4. Other soils that were deemed attractive to young children.

ii. Results of Field Activity

The results of field activity showed that soil contamination in the town of Annapolis, based on XRF analyses, was found at two residences (see Appendix A, Table 5). One driveway had a mean lead concentration of 1,180 ppm and one Sampling Unit at a separate property had a mean lead concentration of 429 ppm. EPA will address the driveway as a Time Critical Removal Action. The elevated Sampling Unit was the only elevated Soil Sampling Unit in the town. EPA divided this property into four Sampling Units (SUs). Each SU was composed of five point composite samples. The initial screening of this yard resulted in an elevated lead level of 609 parts per million in SU #1. Based on the results of the other three SUs in the yard, the results of other properties in town, and the advice of EPA's Human Health Risk Assessor, EPA re-sampled the elevated sampling unit (SU #1) on this property, using a more thorough technique. The resample of SU #1 was composed of a 15 point composite sample (Figure 3, Appendix B). The results of this sampling indicated that SU #1 contained a mean of 429 parts per million lead, which is just above the screening level of 400 ppm. The mean concentration of the entire property was 277 ppm, which was below the screening level for lead in residential surface soils of 400 ppm.

IV. Nature and Extent of Contamination

A. Site Characterization Results

The results show that there is minimal soil contamination in the town of Annapolis.

i. Potential Routes of Migration

Potential routes of contaminant migration could be from the Glover smelter or the source area (OU-1). Sampling results show that neither of these potential sources has significantly impacted the soil in the town of Annapolis.

ii. Contaminant Persistence

This was not evaluated since the Contaminants of Concern (COCs) were not elevated in the sampling areas. However, the primary COC, lead, is very persistent in nature.

iii. Contaminant Migration

Contaminant migration was the primary reason for this investigation. EPA and MDNR agreed to evaluate the soils in the town of Annapolis due to the possibility of contaminated mine waste being transported from the historical pile to the town.

B. Baseline Risk Assessment

i. Human Health

No Human Health Risk Assessment (HHRA) was conducted for Operable Unit-3 because no significant contamination was found in the initial investigation. However, the HHRA for OU-1 used standard USEPA guidance along with both default and site-specific information to assess potential health risks for people living, working, or recreating in the area. A copy of the pertinent sections of the HHRA is included in Appendix C. The HHRA focused on evaluating potential exposure to lead and other mine-related materials under existing conditions. Risk and hazards for three potential receptor groups were evaluated in the HHRA including current and future residents, current and future recreationists, and future construction workers. Future residents and construction workers were used to assess residual risks in areas of the mine operations area that are above the floodplain. Current and future recreationists were used to assess potential risks and hazards associated with existing contamination in the floodplain of Sutton Branch Creek.

Quantitative risk and hazard estimates were developed for residents, construction workers and recreational users.

Residential Lead Exposures: The Integrated Exposure Uptake Biokinetic Model (IEUBK) was used to assess lead exposures for young children. Lead exposures for future residential children were assessed for exposure to soil in the former mine site. Hot spots in this area were evaluated separately. To illustrate the range for possible impacts to blood lead levels both default and alternative values for key parameters in the model were assessed in the uncertainties section. Most soils in the former mining operations area of the site that were sampled during post-removal activities have lead concentrations that are below levels of potential concern. A young child that lives or plays in these areas and be exposed to have greater than a 5 percent chance of having their blood lead concentrations exceed the health protection goals of 10 micrograms per deciliter (μg/dL); when this criterion is met, lead exposures are unlikely to represent a significant hazard. This conclusion would apply to most of the residences outside of the floodplain. Lead concentrations in 84 out of the 85 residences screened were below the screening value of 400 (parts per million) ppm.

For identified hotspots that were sampled in the former mining operations area, average lead concentrations could be high enough to represent a hazard to young children. In hotspot areas, lead exposures are predicted to be very high and lead concentrations in soil and dust could theoretically cause a young child exposed to average soil and dust concentrations in these areas to have a high probability of having a blood lead concentration exceeding the health protection goal of 10 μg/dL. Such exposures would only occur if residual lead contamination that exists below the 18 inch engineered soil cover were to be brought to and left on the surface after residential development at the site. Currently, no exposure pathways exist for residual lead beneath the clean soil cover.

ii. Ecological

No Baseline Ecological Risk Assessment (BERA) was conducted for Operable Unit-3 because no significant contamination was found in the initial investigation; however, the source area (OU-1) and Big Creek (OU-2) were included in the BERA for the ALM OU-1 Site. Additionally, the results were compared to the Consensus-Based Sediment Quality Guidelines (MacDonald and others, 2000). The Consensus-Based Sediment Quality Guidelines (SQGs) are chemical benchmarks developed from sediment toxicity tests on a variety of organisms. SQGs help determine whether contaminants are present in concentrations that could cause or contribute adverse effects on resident biota. The Probable Effects Concentration (PEC) SQG is the minimum level of which a specific contaminant would begin to have negative impacts on organisms. Lead and zinc are considered contaminants of concern for Ecological health. The PEC for Lead in Big Creek is 121 parts per million, while the PEC for Zinc in Big Creek is 459 parts per million. The lead PEC was exceeded in sample number UP-7. No other samples exceeded the PEC for lead or zinc. The BERA for OU-1 is included in Appendix D.

C. Source and Extent of Contamination

The majority of the contamination at OU-1 results from the presence of mine tailings that were the result of mine operations. Erosional forces operating on the tailings piles have distributed the mine waste throughout the Sutton Branch Creek basin. XRF surveys have revealed that heavy metal contamination was present throughout nearly the entire Sutton Branch Creek basin above the confluence of Sutton Branch Creek and Hampton Creek and below the confluence of the tailings pile drainage and Sutton Branch Creek. However, the concentrations decline below Hampton Creek. EPA is in the process of remediating Sutton Branch Creek. No significant contamination from mine waste was found in Big Creek (OU-2) or the town of Annapolis (OU-3). The properties screened as part of the OU-3 investigation along with the results are shown in Table 5, Appendix A.

D. Current Contaminant Distribution

Based on the sampling data, and after the removal action, there will be no contaminants in OU-3 resulting from mine waste.

E. Future of Site Contamination

Since no contamination will remain in OU-3, there will be no future of site contamination.

V. Conclusions

Based on the results of field investigations, the following conclusions are appropriate concerning risks and hazards associated with mine waste in OU-3:

- 83 out of 85 properties screened were below the screening level for lead in residential surface soils of 400 ppm. The soil lead screening level is the concentration of lead, if found in samples of residential surface soils, which would trigger further investigation.

- Metal contamination above levels of concern in Annapolis was found in one driveway and in one Soil Sampling Unit. The driveway will be addressed as a time critical removal action during the summer of 2007. The Soil Sampling Unit will not be addressed. EPA has determined that a soil cleanup action is not necessary at this time. The primary factors contributing to this decision include:

    - The lead soil concentration found in the southwest area of the property was only slightly above EPA's screening level;

    - The area with the slightly elevated concentration was small and not currently a play area or likely to become a play area in the future;

- There was no pattern to the contamination in the community that would connect the property to the mine waste that is the subject of EPA's actions at the Annapolis Lead Site; and

- The mean concentration of the lead across the property is well below the screening level.

- Lead was the only COC that was assessed for OU-3; however, Target Analyte List Metals (TALs) were measured in the laboratory confirmation samples and the concentrations of the TALs were below levels of concern.

- Lead exposure in the town of Annapolis is below levels of concern for all potential receptors.

VI. *Data Limitations and Recommendations for Future Work*

The majority of the data at the site was limited to XRF sampling for lead along with ten percent confirmation from the EPA Regional Laboratory. The XRF sample results and Laboratory Confirmation sample results were acceptable, with a 20.5 percent Relative Percent Difference (RPD) (see Appendix A, Table 6). Other COCs were not identified in samples that were not sent to the laboratory for confirmation (approximately 90 percent of the total samples). The experience of EPA Region 7 personnel with mining contaminants is sufficient to permit confidence that the ratio of contaminants are such that a fairly consistent correlation between the lead component and the other components of the mine waste exists.

Sufficient investigations have been made of this site to provide an adequate site characterization. No additional future investigations are required to determine appropriate actions for this site.

VII. Recommended Remedial Action objectives

Data have shown that there is no unacceptable risk from lead associated with mine waste in the town of Annapolis. The contaminated driveway will be addressed as a Removal Action. Since no risk has been found, there are no corresponding Remedial Action Objectives (RAOs).

VIII. Literature Cited

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APPENDIX A

TABLES

TABLE 1

SELECTED METAL RESULTS FROM EMERGENCY RESPONSE GROUND WATER SAMPLING, 1997 ANNAPOLIS LEAD MINE SITE, ANNAPOLIS, MISSOURI

Sample Location	Arsenic	Cadmium	Copper	Lead	Nickel	Thallium	Zinc
Ruble-Alcorn well	ND	2.36	ND	1.10	18.6	**44.1**	16.4
Irrigation well	ND	1.62	11.4	**51.8**	8.54	ND	2,470
Clark well	ND	ND	6.29	ND	ND	ND	72
MCL	50	6	1300	15	100	2	NL
MCLG	NL	6	1300	0	100	.5	NL

Notes: All results are in micrograms per liter (μg/L). Shaded cells indicate values exceeding the federal maximum contaminant level.

MCL Maximum Contaminant Level
MCLG Maximum Contaminant Level Goal
ND Below method detection limit
NL Not listed

TABLE 2

ESI/RA XRF CONFIRMATION RESULTS FOR LEAD, 1997
ANNAPOLIS LEAD MINE SITE, ANNAPOLIS, MISSOURI

Sample Location	**Sample Number**	**Laboratory Pb Concentration**	**Mean XRF Pb Concentration**
Former mining area: (6-12 inches)	-311	24	135
Former mining area, near future Ruble well: (0-6 inches)	-303	68	103
Former mining area: (0-6 inches)	-304	83	236
Outwash area: (6-12 inches)	-312	130	296
Former mining area near Clark irrigation well: (0-6 inches)	-307	210	358
Former mining area: (0-6 inches)	-305	410	530
Outwash area near chat/tailings pile: (0-6 inches)	-318	**1,000**	**1,127**
Outwash area near chat/tailings pile: (0-6 inches)	-317	**1,100**	**1,991**
Outwash area: (6-12 inches)	-310	**1,700**	**1,372**
Former mining area, Clark residence drive: (6-12 inches)	-301	**3,200**	**4,970**
Former mining area, west of Clark residence drive: (0-6 inches)	-302	**3,300**	**2,547**
Mill slime pond: (0-12 inches)	-309	**7,000**	**6,450**
Residential PRG for lead	400		
Industrial PRG for lead	750		

Notes: All results are in milligrams per kilogram (mg/kg). Shaded cells indicate values exceeding the appropriate Region 9 Preliminary Remediation Goal for lead in soil. The actual removal action level (RAL) for the site will be established by the EPA.

Pb Lead
PRG Preliminary remediation goal
XRF X-ray fluorescence spectrometer

TABLE 3

ESI/RA SURFACE WATER SAMPLE RESULTS, NOVEMBER 1997
ANNAPOLIS LEAD MINE SITE, ANNAPOLIS, MISSOURI

Sample Location	Arsenic	Cadmium	Copper	Lead	Zinc
Big Creek					
Big Creek (100 feet downstream of Sutton Branch confluence)	16U/7U	**2.48/0.9U**	1.5U/1.5U	1U/1U	4.32/5.01
Big Creek and Sutton Branch Creek (confluence)	16U/7U	1U/0.9U	1.5U/1.5U	**3.94/5.60**	14.1/5.36
Big Creek, 250 feet upstream of confluence with Sutton Branch Creek	16U/7U	1U/0.9U	1.5U/1.5U	1U/1U	4.07U/4.07U
Sutton Branch Creek					
Sutton Branch Creek (3,500 feet downstream of PPE)	16U/7U	1U/0.9U	1.5U/1.5U	12.5/13.0	5.37/4.07U
Sutton Branch Creek (2,000 feet downstream of PPE)	16U/7U	1U/0.9U	1.5U/1.5U	**17.4/8.62**	5.53/4.07U
Mine tailings outfall (PPE)	16U/7U	1U/0.9U	1.5U/1.5U	2.14/1U	4.07U/4.07U
Sutton Branch Creek (500 feet upstream of PPE)	16U/7U	1U/0.9U	1.59/1.5U	**10.6**/1U	4.07U/4.07U
Sutton Branch (0.75 miles upstream)	16U/7U	1U/0.9U	1.59/1.5U	1.52/1U	6.15/4.07U
Health-based Benchmark					
AWQC	190	1	11	2.5	100

Notes: All concentrations are in micrograms per liter ($\mu g/L$). The first value for each analyte is the total concentration and the following value is the dissolved concentration. Shaded analytes indicate concentrations exceeding the AWQC. Bold type indicates a value three times above background concentrations.

AWQC Ambient water quality criteria
PPE Probable point of entry
U Below method detection limit

TABLE 4
AVERAGE MONTHLY CLIMATE DATA FOR IRON COUNTY, MISSOURI

	Average Max. Temperature (°F)	Average Min. Temperature (°F)	Average Total Precipitation (in.)	Average Total Snow Fall (in.)
January	43.2	21.2	2.56	3.2
February	48.2	24.5	2.41	3.0
March	57.9	32.3	3.94	2.3
April	69.5	42.9	4.53	0.2
May	77.0	51.3	4.79	0.0
June	84.6	60.2	4.19	0.0
July	89.1	64.0	3.58	0.0
August	88.0	62.6	3.68	0.0
September	80.9	54.9	3.72	0.0
October	71.2	43.5	3.40	0.1
November	57.0	33.2	4.19	0.8
December	46.4	25.4	3.14	1.8
Annual	67.7	43.0	44.14	11.4

Notes:
Source: High Plains Regional Climate Center. 2005. "Arcadia. MO (230224), Period of Record Monthly Climate Summary." Accessed June 13, 2005. On-Line Address: http://www.hprcc.unl.edu/cgi-bin/cli_perl_lib/cliMAIN.pl?mo0224.

Key for Table
°F = Degrees Fahrenheit in. = Inches Max. = Maximum Min. = Minimum

Table 5. Property Results

Yard	Sampling Units ≥ 400	Overall Yard Concentration (mean)	Contaminated Driveway
1	0	114	No
2	0	117	No
3	0	126	No
4	0	144	No
5	0	111	No
6	0	113	No
7	0	108	No
8	0	94	No
9	0	112	No
10	0	203	No
11	0	124	No
12	0	99	No
13	0	122	No
14	0	104	No
15	0	146	No
16	0	264	No
17	0	123	No
18	**1=429**	277	No
19	0	131	No
20	0	123	No
21	0	141	No
22	0	136	No
23	0	143	No
24	0	146	No
25	0	200	No
26	0	129	No
27	0	218	No
28	0	107	No
29	0	152	No
30	0	104	No
31	0	159	No
32	0	247	No
33	0	165	No
34	0	107	No
35	0	90	No
36	0	107	No
37	0	99	No
38	0	106	No
39	0	125	No
40	0	109	No
41	0	106	No
42	0	222	No
43	0	138	No
44	0	118	No
45	0	144	No
46	0	120	No
47	0	262	No
48	0	229	No

Yard	Sampling Units ≥ 400	Overall Yard Concentration (mean)	Contaminated Driveway
49	0	83	No
50	0	114	No
51	0	125	No
52	0	115	No
53	0	87	No
54	0	117	No
55	0	80	No
56	0	114	No
57	0	126	No
58	0	96	No
59	0	194	No
60	0	106	No
61	0	80	No
62	0	121	No
63	0	152	No
64	0	201	No
65	0	93	No
66	0	150	No
67	0	115	No
68	0	Below Detection Limit	No
69	0	149	No
70	0	182	No
71	0	Below Detection Limit	No
72	0	74	No
73	0	80	No
74	0	84	No
75	0	91	No
76	0	107	No
77	0	145	No
78	0	233	No
79	0	122	No
80	0	179	No
81	0	106	No
82	0	135	No
83	0	82	No
84	0	219	No
85	0	94	Elevated driveway of 1,180 ppm*

*will be addressed as a Removal Action

Table 6. Laboratory Confirmation Samples

Sample #	XRF Concentration	Lab Confirmation	Relative Percent Difference
LC-1	177	80	54.8%
LC-2	ND	58	
LC-3	395	228	42.3%
LC-4	130	177	26.6%
LC-5	***609**	555	8.9%
LC-6	120	113	5.8%
LC-7	128	118	7.8%
LC-8	166	133	19.9%
LC-9	71	88	19.3%
LC-10	ND	89	
LC-11	125	120	4%
LC-12	107	58	45.8%
LC-13	132	128	3.3%
LC-14	379	**°584**	35.1%
LC-15	262	343	23.6%
LC-16	94	113	16.8%
LC-17	ND	56	
LC-18	162	214	24.3%
LC-19	144	150	4%
LC-20	139	148	6.1%
LC-21	122	90	26.2%
LC-22	129	153	15.7%
LC-23	ND	40	
LC-24	ND	17	
LC-25	ND	38	
			20.5 % is the mean RPD. The XRF performed within 79.5% mean accuracy when compared to the Laboratory Results, which EPA considers an acceptable level.

* This Sampling Unit was re-sampled. The second round of sampling resulted in a concentration of 429 ppm.

° This laboratory confirmation is above 400 ppm, the screening level for lead in soil. However, the confirmation samples are used to support the XRF, which is the decision making tool. In this case the XRF resulted in a concentration of 379 ppm, which is below the screening level of 400 ppm.

APPENDIX B

FIGURES

Annapolis, MO

Figure 1. Town of Annapolis

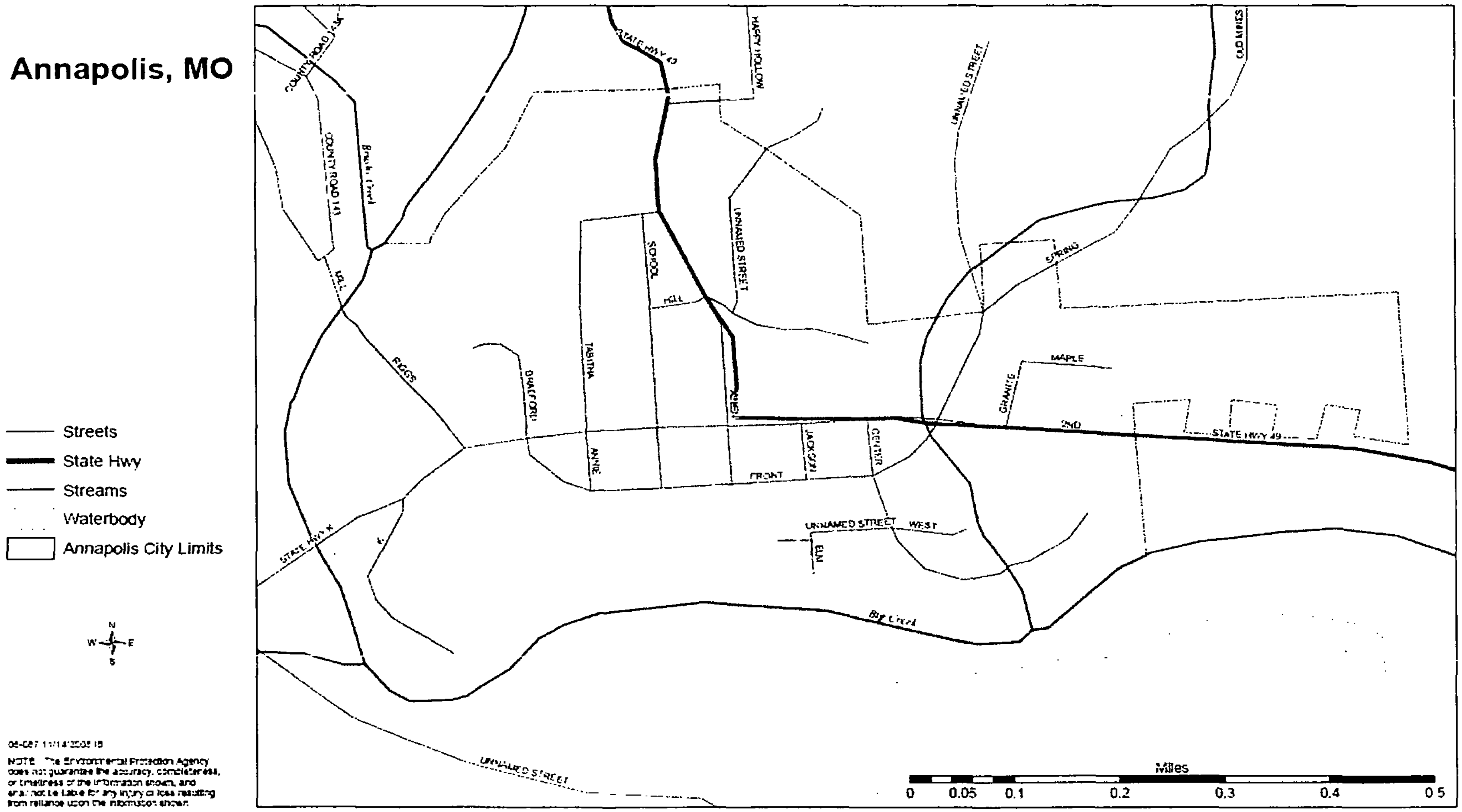


Figure 1. Town of Annapolis

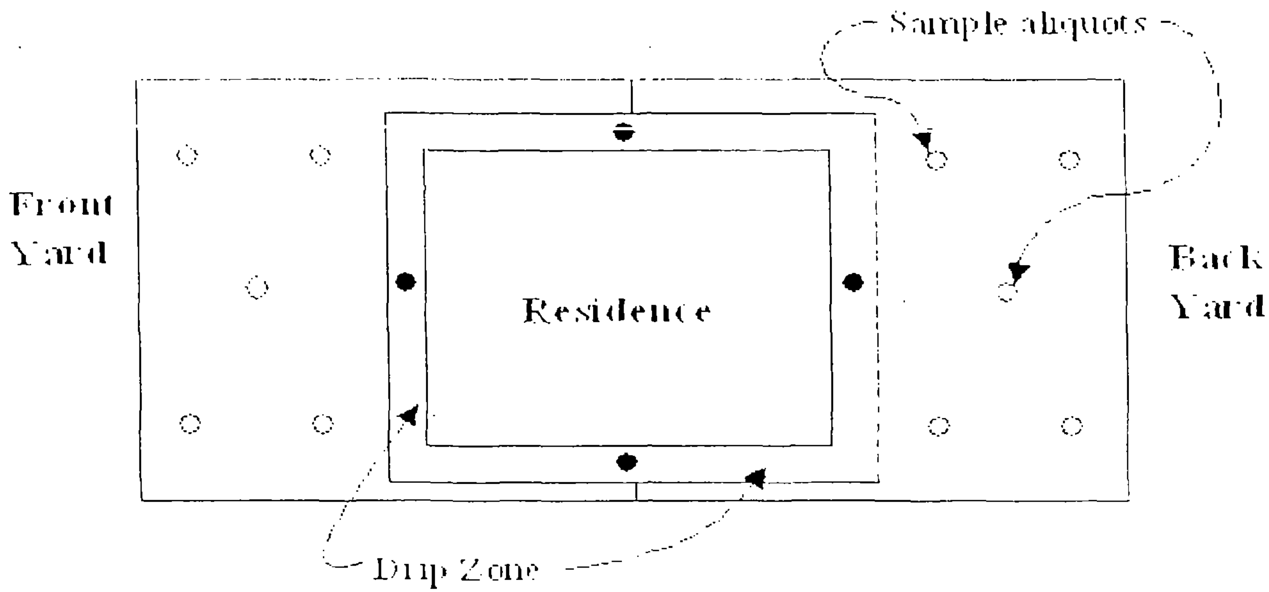


Figure 2. Recommended minimum soil sampling in yards less than or equal to 5,000 square feet with small side yard.

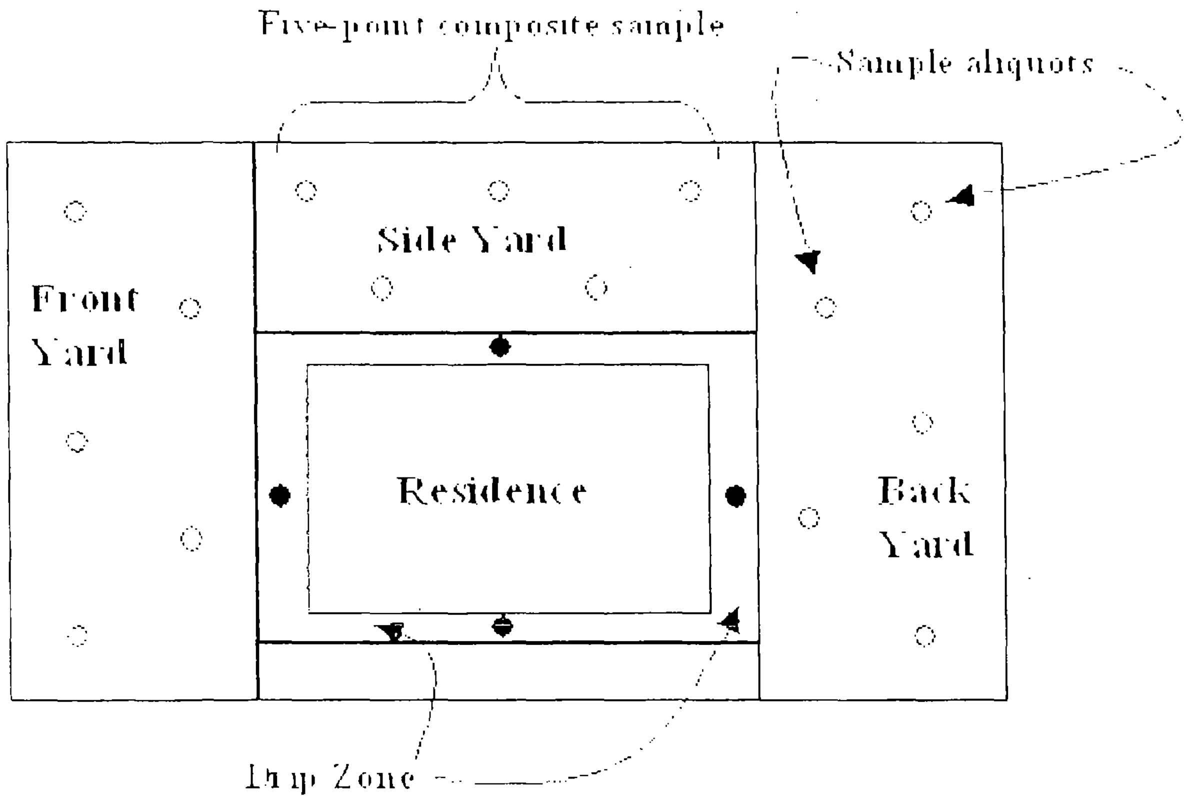


Figure 3. Recommended minimum soil sampling in yards less than or equal to 5,000 square feet with substantial side yard.

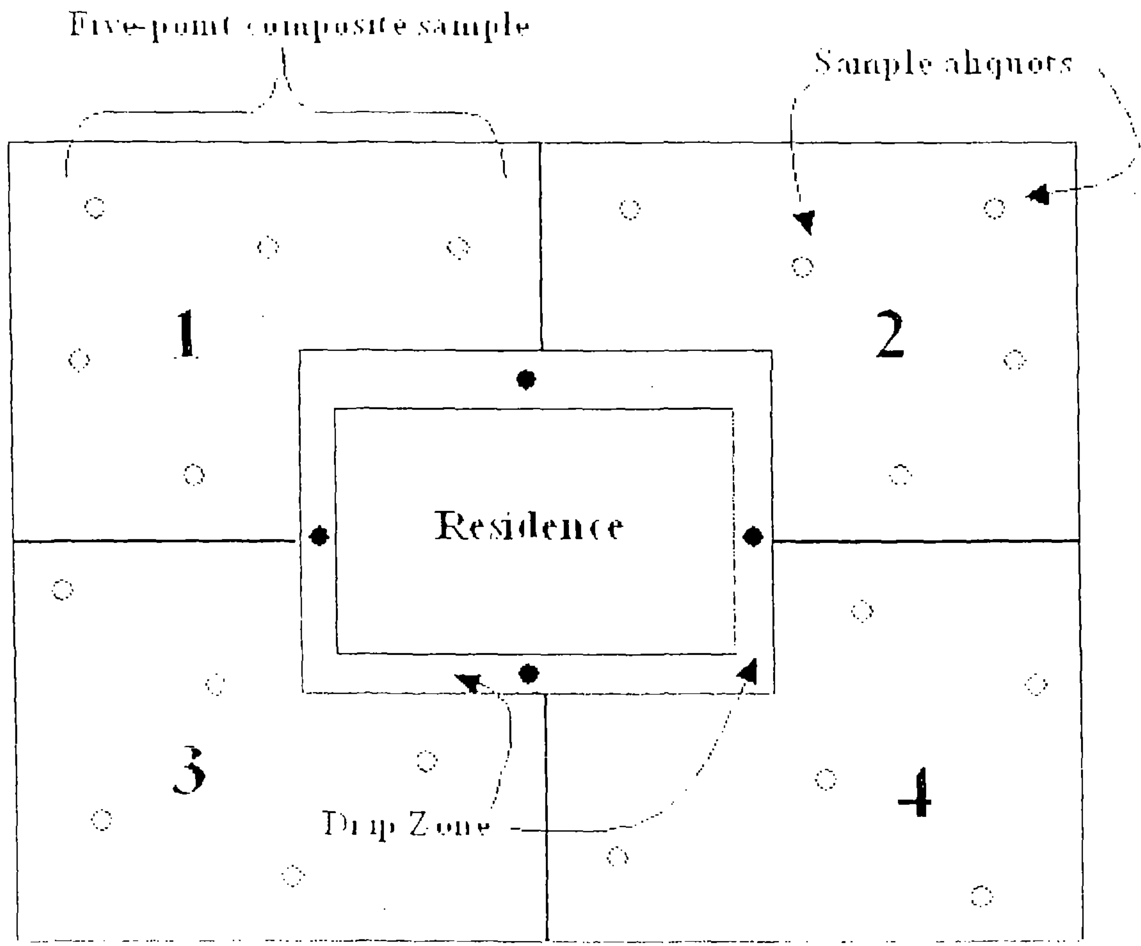


Figure 4. Recommended minimum soil sampling in yards greater than 5,000 Square feet.

APPENDIX C

HUMAN HEALTH RISK ASSESSMENT FOR OU-1

Sections 4-8

Section 4: Exposure Assessment
Section 5: Toxicity Assessment
Section 6: Risk Characterization
Section 7: Uncertainties
Section 8: Summary and Conclusions

Section 4
Exposure Assessment

Populations that may be exposed to chemicals at a site and pathways by which these populations may come into contact with site chemicals are identified in the exposure assessment. In identifying potential pathways of exposure, both current and possible future land use of the site and surrounding area is considered in this HHRA. The following sections present the exposure assessment, including methods and assumptions used to quantify potential exposures at the site. The exposure assessment is conducted in accordance with the following documents and others cited in the text:

- *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A). Interim Final.* EPA/5401/1-891002. December 1989.
- *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part D – Standardized Planning, Reporting and Review of Superfund Risk Assessments).* EPA/540-R-97-033. January 1998.
- *Exposure Factors Handbook.* Office of Research and Development. National Center for Environmental Assessment. EPA/600/P-95/002Fa. August 1997.
- *Risk Assessment Guidance for Superfund, Volume 1: Human Health Evaluation Manual. Supplemental Guidance, Standard Default Exposure Factors.* March 1991.
- *Guidance Manual for the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children.* EPA PB93-963510, OSWER 9285.7-15-1.
- *Recommendations of the Technical Review Group for Lead for an Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil.* EPA-540-R-03-001 January 2003. http://www.epa.gov/superfund/programs/lead/products/adultpb.pdf

Exposure assessment defines, in qualitative or quantitative fashion, the ways that people living, working or recreating in the study area might be exposed to heavy metals, particularly lead, released as a result of historic mining operations.

For the Annapolis Lead Mine, the basic approach to the assessment is twofold. First, for the portion of the site north of Highway 49, the assessment addresses residual risks associated with lead contamination left in place following a recently completed removal action. Second, for floodplain areas, particularly those areas south of Highway 49, the assessment addresses potential risks from existing contamination eroded from the mine site and deposited during flood events.

Currently, the Annapolis Lead Site source area and adjacent Sutton Branch Creek floodplain are mainly undeveloped land. There is one home adjacent to and north of Highway 49 and next to the Sutton Branch Creek. The yard of this home has been sampled, and no concentrations of lead above 400 mg/kg were detected. Further, one occupied residence exists adjacent to the southern floodplain of Sutton Branch Creek. Future development of the site is not restricted by specific zoning regulations, so future residential or commercial development is theoretically possible in areas outside of the floodplain. Future use of most of the target area will most likely be for recreational activities. Commercial or industrial applications seem unlikely and are not quantitatively evaluated. Since hypothetical future residential land use is evaluated and would be more restrictive, evaluation of potential commercial or industrial land use does not seem necessary to achieve the goals of this assessment.

4.1 Exposure Assessment Process

Exposure is defined as human contact with a chemical or physical agent (USEPA 1989). Exposure assessment is the estimation of magnitude, frequency, duration, and pathway(s) of exposure to a chemical. Assessment of exposure consists of three steps:

- Characterization of Exposure Setting
- Identification of Exposure Pathways
- Quantification of Exposure

The first step involves identifying the environmental setting of a site (e.g., climate) and the current and potential future human populations on and near the site. Human populations are described with regard to characteristics that could affect exposure to site-related chemicals, including location relative to the site, activities, and the presence of sensitive subgroups (e.g., pre-school children).

Step two of the exposure assessment identifies pathways by which human populations might be exposed to site-related chemicals. Chemical sources, release and transport mechanisms, and inter-media transfer are evaluated. Exposure pathways are identified based on the location and activities of potentially exposed populations and on the types of potentially contaminated media.

The final step, exposure quantification, has two components: estimation of exposure point concentrations and calculation of chemical intake. Exposure point concentrations are chemical concentrations at the point of human contact. Site-specific chemical data from previous investigations for media of concern are used to estimate exposure point concentrations. Exposure point concentrations and equations for estimating these concentrations are presented in the following sections.

4.2 Exposure Setting

The following section details physical settings and human use factors that may influence risk to human health at the ALS.

4.2.1 Physical Setting

The physical setting details physical characteristics of the environment that may influence exposure and risk to the health of human receptors.

4.2.1.1 Climate

Climatological data are included because climate may influence human activity patterns. For example, the daily temperature may affect both the frequency and duration of participation in outdoor activities, types of clothing worn, and the types of activities. USEPA investigated climate based on climatology data from nearby Arcadia, Missouri, which is 15 miles north of the site (Tetra Tech 2005). The area has moderately cold winters and relatively hot summers with January being the coldest month (average maximum temperature of 43.2 °C) and July being the warmest month (average maximum temperature of 89.1 °C). Precipitation averages 44 inches per year and is distributed throughout the year. The majority of the precipitation is rain, but snow does fall annually but remains for only short periods of time. The prevailing wind is in the southerly direction.

4.2.1.2 Hydrology and Hydrogeology

Sutton Branch Creek is a tributary of Big Creek, which are both in hydrogeologic unit 8020202 of the Upper St. Francis Basin. Sutton Branch Creek is considered a small, losing stream which is joined by Hampton Creek, also a losing stream, just before the confluence at Big Creek. Big Creek is rated a class I to II tributary of the St. Francis River, and it is considered to have navigable waterways from Highway K, upstream of the ALS, to Sam A. Baker State Park and its confluence with the St. Francis near Lodi, Missouri and U.S. Highway 67. The St. Francis River originates in St. Francois County and travels through the Ozarks to its outlet at the Mississippi River in Lee County, Arkansas. Descriptions of paddling trips can be found in several guidebooks of the area, suggesting that recreational use of Big Creek is likely and ongoing. There are no known drinking water draws from Sutton Branch Creek or Big Creek by residents, but there is potential use of surface water by downstream residents for irrigation water for gardens and yards. The MoDNR also draws some surface water from Big Creek for use at the Sam A. Baker State Park (Sverdrup 1995a). Big Creek itself is outside the geographical scope of this risk assessment, and information on this creek is provided here only to illustrate the connections among the site and surrounding resources.

An in-depth characterization of Sutton Branch Creek hydrology and hydrogeology took place during January 2005, and the findings are summarized in the RI report (Tetra Tech 2005). The Sutton Branch Creek floodplain is characterized as a wide, flat depositional environment, covered by dense herbaceous vegetation, where water tends to spread and decrease in velocity. The in-stream portions of Sutton Branch Creek are characterized as having a gravel streambed with connectivity to the floodplain at various locations. Some stream modifications have been noted due to natural aggrading and degrading and construction activities by the county. Loss of bank material by erosion has been estimated to be 62.22 tons per year.

Residents are known to use the local groundwater for drinking and irrigation. The locations of wells and groundwater details are discussed in previous reports (E & E 1997; 1999). A total of 245 households (E & E 1999) are within a 4-mile radius of the ALS. Approximately 14 people rely on wells within 0.25 miles of the ALS, and there are artesian, shallow, and drilled wells within the ALS boundaries. Groundwater depth at a monitoring station in Bixby, Missouri is approximately 235-237 feet bgs (MoDNR 2005), and the average well depth of the area is 228 feet with depths ranging from 80 to 525 feet (E & E 1999). At the site, one irrigation well is completed in surface deposits at 10 feet of depth and two drinking wells are completed in bedrock at depths of approximately 220 and 130 feet. Two artesian wells are located within 0.5 miles of the ALS. One artesian well, located 800 feet from the mine tailings pile, is used by approximately 50 people, despite warnings by the health department that the water is not drinkable. The wells of residents near the ALS are situated in shallow alluvium or bedrock and have variable yield due to the lack of lateral continuity in the sedimentary rocks isolated by igneous rocks. Movement of groundwater is via vertical jointing.

4.2.1.3 Geology

Geology is summarized from the QAPP report (E & E 1997). The ALS is in the St. Francois Mountains Physiographic Province of Missouri on westward sloping topography with drainage into Sutton Branch Creek. The area is underlain by Precambrian highland mass with on lapping Paleozoic carbonates and silicates. Lead deposits of the region are in the Cambrian Bonne Terra formation, which is mostly dolomite but may have pure limestone areas. Ore obtained from this formation, and specifically at the ALS, has a whitish appearance due to the presence of limestone. Stratigraphy of completed groundwater wells are associated with unconsolidated valley alluvium (20 to 25 feet thick) and underlying Cambrian sandstone and dolomite.

4.2.2 Biological Setting

The area surrounding the Annapolis Lead Site and the Sutton Branch Creek floodplain is dominated by pastureland and upland wooded areas. The area around the confluence of the Sutton Branch and Big Creeks is designated as palustrine, deciduous broad-leafed forested, temporarily flooded wetland. Beaver (*Castor canadensis*), white-tailed deer (*Odocoileus hemionus*), coyote (*Canis latrans*), red fox (*Vulpes fulva*), striped skunk (*Mephitis mephitis*), rabbit (*Lepus spp.*), waterfowl, squirrel, various bird species, reptiles, and amphibians were identified at the ALS in 2003 and 2004 (USEPA 2005a). Potential species of concern also present in Big Creek are the southern brook lamprey (*Icthyomyzon gagei*), the Big Creek crayfish (*Oronectes peruncus*), and the silver-jaw minnow (*Notropis buccatus*).

Sutton Branch Creek is small and intermittently dry in reaches adjacent to and downstream of the mine operations area. The waterway does not support any fishery along this course. An exception may be the stream reach just upstream of the confluence with Big Creek. At this point, the Sutton Branch Creek has been joined by the Hampton Branch, which is significantly larger, and flows are perennial. In this reach, fish from Big Creek could move up a short distance into the tributary. These fish would likely best be characterized as part of the Big Creek community. Since the Hampton Branch drains a watershed that is not contaminated with mine wastes, any fish that live in its waters would not be affected by releases from the mine operations area.

4.3 Receptor Populations

Receptor populations were selected based on current and potential future land use, activities of the receptor populations, and a complete exposure pathway to contaminated media.

4.3.1 Location of the Current Population

The ALS is entirely within Iron County, Missouri, which covers 551 square miles of southeastern Missouri and has a density of 19.4 people per square mile (US Census 2000b). Based on the most recent census, Iron County has a population of 10,376 persons, a decrease of 0.3% from the 1990 census (US Census Bureau 2000c). The current population within a 4-mile radius of the area under investigation is approximately 1,300 persons, 180 within a 1-mile radius. Other nearby populations include approximately 100 people working at the ISP, Inc. manufacturing facility (Sverdrup 1995a), the South Iron School District with 489 registered students (MDESE 2005), and an unknown number of recreational users. Annapolis, located west of the site, has 310 residents (US Census 2000a). The downstream village of Vulcan has 157 residents and the village of Des Arc has 187 residents. Approximately 15-miles downstream of the site, along Big Creek and the St. Francis River, is Sam A. Baker State Park.

4.3.2 Current and Future Land Use

The ALS is owned by four different landowners. There is an abandoned single-family residence in a former mine building at the ALS source area. One residence, the Mayberry property, is located north of Highway 49, adjacent to Sutton Branch Creek. There is at least one occupied residential dwelling in the southern segment of the ALS area, just adjacent to and above the Sutton Branch Creek floodplain. No gardens are known to exist within the boundaries of the target area, but future use for gardening and the consumption of homegrown vegetables is theoretically possible.

Part of the floodplain area may be harvested for hay. However, the area is not obviously cultivated, is not associated with a farm residence, and no evidence of grazing was observed on a recent site visit by CDM and USEPA (July 8, 2005). Hay harvested from the area would have to be transported to a livestock feeding area and would very likely be mixed with hay and other feed from other sources. Livestock are probably not raised exclusively on hay from the floodplain. Since lead does not biomagnify in the food chain, the amount of lead that might be taken up and retained by livestock fed intermittently with contaminated hay should not be great. In addition, cattle are expected to be exposed to the greatest concentrations of lead through incidental ingestion of soil while feeding (Neuman and Dollhopf 1992). Since cattle are not feeding directly at the site but are instead fed hay cultivated from the site, lead levels in the tissue of cattle would be expected to be greatly reduced. Any lead contained in hay eaten by grazing ruminants will be partitioned to the liver and kidney rather than muscle (Sedki et al. 2003). Liver and kidney are not likely to be the primary tissues of consumed by receptor populations, as less than 2 percent of total beef consumption (based on all sex, age and demographic subgroups) is attributed to edible organ meats, specifically liver and kidney (USEPA 1997b). Furthermore, people consuming meat from livestock raised locally are likely to obtain only a portion of their meat from animals fed contaminated hay. Resulting secondary exposures to lead in the relatively small area of the floodplain that produces hay are likely to be small.

The ALS was also evaluated for industrial/commercial land uses. The property is currently under private ownerships and zoning in the area is non-restrictive. The location of the site and the lack of significant growth in the area suggests that industrial use is unlikely. There is the potential for additional residences to be constructed, either as an additional outbuilding of one of the adjacent residences or through property subdivision and future residential development. Mine tailings have been removed from the site for use by county road crews, the school for the playground, and for concrete (MoDNR 1993). Exposure via mine tailings after removal and transportation from the site is beyond the scope of this assessment due to the lack of information about concentration of contaminants in the removed material, limited knowledge of receptors exposed, and no information on the current distribution of the material.

Recreational activities may be conducted at the ALS by local residents. Off-road vehicle traffic was noted during a site visit on an unimproved road from Highway 49 to Big Creek. Big Creek, a MoDNR Outstanding Resource Water, is immediately adjacent to the Sutton Branch Creek floodplain and is popular for recreational activities including canoeing, kayaking and fishing. This resource is likely to attract people to the area for recreation. Big Creek is likely to be much more attractive than the Sutton Branch Creek floodplain or the mine operations area for people seeking recreation. However, some people, in particular children and residents in the area, might occasionally make use of the mine operations area and/or the floodplain. Fishing is most likely not possible in the portion of Sutton Branch Creek directly west of the source area, however, some bait collection and wading or other water play is, at least, possible. Sutton Branch Creek is difficult to access in most places because of dense riparian vegetation, and recreational use of this creek is likely to be very limited.

A small amount of fish habitat may exist in the lowest portion of the Sutton Branch Creek, between the confluence with Hampton Branch and the confluence with Big Creek. Fish in this reach, however, are likely to move in and out of Big Creek and/or the uncontaminated Hampton Branch. Anglers that may take fish from this area would best be assessed when examining potential exposures for Big Creek downstream of source areas. Such an evaluation is outside the scope of this risk assessment, and is not further addressed.

Possibly, a limited number of crayfish could also live in the lower reaches of Sutton Branch Creek. Populations of crayfish in the Creek are likely to be small, based on direct observation of the creek during the site visit in July, 2005. Harvesting any significant number of crayfish from the Creek would be difficult, and consumption of contaminated crayfish is not expected to be a significant pathway.

Overall, Sutton Branch Creek provides limited habitat for fish or crayfish. Any significant take of either type of organism from the Creek is highly unlikely. Some animals may make use of habitat near the confluence of Sutton Branch Creek with Big Creek. Assessment of potential exposures from consumption of such organisms is best addressed as part of an analysis of the aquatic environment in Big Creek. Such an evaluation is outside the scope of this assessment.

4.3.3 Sensitive Subpopulations

Subpopulations at the ALS and in the vicinity were identified to characterize groups that could be a greater risk than other people in similar exposure situations. Greater risks for some populations could be attributed to such factors as increased sensitivity, multiple exposure pathways, or a relative increased exposure potential based on the exposure period or contact with contaminated media. Subpopulations of concern depend upon site-specific characteristics and may include infants and young children, pregnant women, the elderly, individuals with respiratory problems, or individuals engaging in a specific activity (e.g. fishing).

Demographics of Iron Country in the 2000 U.S. Census describes the population as 5.9% under the age of 5, 25.0% under the age of 18, and 17.1% over the age of 65 (US Census 2000c). Median age is 39.7 years. The population of Iron County is 51.3% female. Average number of people per household is 2.46 persons, and average number of people per family is 2.94 persons.

Children were identified as a potential sensitive subpopulation because of the presence of children in the nearby school, the potential for children to live or recreate on-site, the demography of the county, and their potential for greater sensitivity or exposure to heavy metals. In fact, the USEPA conducted an emergency response action at the mine operations area of the ALS to remove two children with elevated blood lead levels (USEPA 2004b). Childhood development has been shown to be affected by contamination of heavy metals, especially lead. Children with increased levels of lead in the blood may have damage to the brain, anemia, muscle weakness, stomachache, or other health effects. Lead can also pass from a mother to the fetus and may lead to premature birth, decreased birth weight, and learning deficiencies. Besides greater sensitivity to certain chemicals, children may have a greater exposure than adults. Behaviors which may increase exposure in children include playing in the creek, digging and playing in soil, and frequent hand-to-mouth contact.

Residents or recreational users who consume fish caught in contaminated areas or consume homegrown vegetables, cultivated in contaminated soils or irrigated with contaminated water, may also be sensitive subpopulations due to increased exposure via the diet. Some heavy metals may accumulate to some extent in fish and could, in theory, be a significant source of exposure for anglers that take significant numbers of fish from contaminated areas of creeks and rivers. Frequent consumption of these fish, especially those in close contact with sediment such as catfish, may increase exposure levels to certain contaminants. The consumption of vegetables grown in contaminated media, or irrigated with contaminated surface water, may also increase exposure to some metals, especially if residual soil is present on root vegetables during consumption.

4.3.4 Selection of Receptor Populations

Three different receptor populations were selected based on proximity to sources, sensitivity, and activities or use of land both on-site and in near proximity. The receptors selected are detailed in the following sections.

The receptors selected for the evaluation of risk to human health include:

- Current/Future Residents-Adult and Child (0-6 years) Scenario;
- Current and Future Recreational Users-Older Child (7 through16 years) Scenario; and
- Future Construction Workers.

4.3.4.1 Future Residents

Both child and adult future residents were chosen as receptor populations for the human health risk assessment. Currently, two residences are located on the site, but neither is located in areas where lead concentrations exceed the screening level of 400 mg/kg; future residential use, specifically in the source area, is theoretically possible. Future residential construction in the Sutton Branch Creek floodplain is considered unlikely, however, and residual contamination in floodplain areas are not used in estimates of exposure for current and hypothetical future residents. The resident receptor population has the greatest exposure period of all potential receptors due to their likely presence at the site on a daily basis over an extended period of time. Ingestion of soil and interior dust are considered to be primary pathways of exposure for the current and future resident. However, potential exposure to contaminated groundwater used for domestic purposes is also evaluated.

For the evaluation of residential exposures, data are available only for residual lead concentrations for the mine operations area. This area is the only one where residential development is at all likely. Evaluation of health impacts due to exposure to lead in residential settings is accomplished through the use of the USEPA's Integrated Exposure Uptake Biokinetic (IEUBK) model (USEPA 1994b) for young children. Young children are more susceptible to the toxic effects of lead, and generally receive the highest exposures to lead in soil and dust. Thus, protection of young children will also protect adult residents in the same environment. Thus, hypothetical future residential exposures are evaluated solely through evaluation of lead exposure for young children.

Where exposure to very young children is not expected (e.g. recreational exposure settings or construction workers), the adult lead model is used to estimate potential hazards due to potential lead exposure, as described below.

Other factors could contribute to potential exposure in residential populations. Residents may engage in recreational activities, and therefore, be exposed to additional contaminated media. These individuals may live in areas impacted by mining wastes and may recreate near their homes in contaminated areas, which may lead to exposure through both residential and recreational activities. Residents may also consume fish (e.g., crayfish) and may consume produce from gardens in areas contaminated by mine tailings or watered with surface water or groundwater from contaminated areas. These additional exposure pathways could increase health risks in residents.

Risks based on recreational use of the ALS by residents are evaluated based on use by a "resident" or local recreational user. That is, recreational exposure parameters are chosen to reflect relatively frequent recreational use that may occur for residents with immediate access to contaminated areas.

4.3.4.2 Current and Future Recreational User

Currently and in the future, some recreational use of both the mine operations area and the floodplain area are theoretically possible. Neither of these areas is attractive for recreation, especially given the immediate access to Big Creek. However, residents that live in the areas, particularly children, might visit these areas infrequently. Recreational users of the site are assumed to be local residents (though not residents that might live on the site in the future) that, because of proximity, do visit the site.

Access to the ALS is both possible and probable, at least occasionally, for some recreational users. Health risk to recreational users was investigated due to the accessibility of the site, status of Big Creek as an Outstanding Water Resource, which should attract people to the area, and the potential for exposure through multiple exposure pathways. There are currently no site restrictions in place for use of the source area or the Sutton Branch Creek floodplain, except fencing around much of the removal area in the northern segment. Although the ALS is located on private property, signs of recreational use were evident in limited areas based observations during a recent site visit. There does not appear to be much recreational value of Sutton Branch Creek or the floodplain area. Sutton Branch Creek is choked with vegetation and is dry during portions of the warmer months when use would be most prevalent. The only likely recreational use for Sutton Branch Creek is for children infrequently exploring the area. The floodplain area south of Highway 49 may be used by adjacent residents as an extension of their yard, although no current signs of such activity are obvious. Due to the terrain and distance to nearby residences, very young children are not likely to play in this area or other portions of the site; however, older children may frequent these areas. Older children ranging in age, from 7 to 16 years are quantitatively evaluated as recreational receptors in the HHRA.

For much of the Sutton Branch Creek floodplain, data are available for potentially mine-related constituents other than lead. Thus, potential risks and hazards for recreational visitors to the site are evaluated for exposures to lead, using the Adult Lead Methodology as well as to other chemicals of potential concern that exist in surficial floodplain soils.

4.3.4.3 Future Construction Worker

A large percentage of the ALS is undeveloped and is not currently under restrictions for land use. Thus, the potential for future development must be considered. Current or future property owners could sub-divide their land or build residences on their existing properties. The population of Missouri is projected to grow at a rate of 14.9% over the next 30 years, which is less than the national average of 29.2% (US Census 2000b). Iron County had a decrease in population of 0.3% from 1990 to 2000 (US Census 2000c). The minimal growth of the Missouri population in general and the loss of Iron County residences specifically suggests that large-scale development of the ALS is unlikely for the foreseeable future.

Still, some potential for isolated construction activities exists, especially in the northern segment. County road crews have been active in the past during improvement projects in the Sutton Branch Creek channel and removing mining material for incorporation into concrete mix. If additional development or road construction were to occur at the site, construction workers could be exposed to contaminants at the site. Worker exposures would be less than those for hypothetical future residents because of shorter exposure times, frequencies, and durations, as compared to residents in the area. However, construction workers involved in manual activities may have intensive contact with contaminants in soils, including subsurface soils that contain residual contamination. Construction activities are likely to penetrate the 18-inch clean fill barrier to contamination, especially during excavation activities (e.g. for foundations).

Since construction is anticipated only in the mine operations area outside of the floodplain, data on residual contamination is only available for lead. Thus, construction workers are only evaluated for potential future exposure to lead. This issue is further discussed in Section 7, Uncertainties.

4.4 Site Conceptual Exposure Model (SCEM)

The primary source of contamination at the ALS consists of crushed and concentrated mine waste from the mining of galena ore during historical mining activities. The majority of the waste was deposited in a 10-acre, natural ravine at the southern end of the mining operations area (USEPA 2005a). Over time, the mine pile eroded and the mine tailings traveled with topographic features to the Sutton Branch channel. The creek transported material downstream to its confluence with Big Creek. Contaminated media spread across the floodplain of Sutton Branch, being deposited as water velocities slowed.

Recently, contaminated soil with lead concentrations exceeding 400 mg/kg were excavated from the former mining operations area north of Highway 49 and consolidated into an on-site repository at the site of the former waste pile. The pile was then capped and seeded (USEPA 2004c). Thus, few areas of the site have surface concentrations of lead above 400 mg/kg (mainly within the Sutton Branch Creek floodplain), and current exposure potential for the former mine area is low. Any future exposures would occur only if residual contamination was brought to the surface following future site development. Thus, only future exposures are evaluated for most areas of the former mine site.

Releases from primary (mine waste pile) and secondary sources (soil and air) also resulted in contamination of surface water and sediment, and conceivably may have resulted in contamination of groundwater and biota. Evidence for these releases includes elevated levels of contaminants measured in some media, observed mine wastes on stream banks and in floodplain area, and noticeable erosion of wastes from the source pile.

In contrast, samples collected from nearby domestic wells indicate lead concentrations below levels of concern, suggesting that currently used shallow groundwater has not been affected. Concentrations in biota from Sutton Branch Creek have apparently not been collected and no contamination of biota that can be traced directly to the mine site are available. Fish studies in Big Creek are difficult to interpret since upstream sources (e.g. the Glover smelter) exist and could be significant sources of metals in biota.

These sources and releases, along with the above discussion of possible receptors, form the foundation for the development of a SCEM. This model (Figure 4-1) illustrates potential pathways for exposure of humans to contaminated media. As shown in the SCEM, environmental media potentially impacted by the release and transport of contaminants may include:

- Soil
- Indoor dust (Tracked from outdoor soil)
- Outdoor air (Windblown Particulates)
- Plants/homegrown produce
- Fish
- Surface water
- Sediment
- Groundwater

All of the above potential exposure media are further evaluated to identify those that may be important for risk management of the site. Complete and significant exposure pathways are further discussed in the following sections.

4.5 Exposure Pathways

An exposure pathway generally consists of the following elements:

- A chemical source and mechanism of release
- An environmental transport medium for the released chemical
- A point of potential human exposure with the contaminated medium
- A route of exposure (inhalation, ingestion, dermal absorption) into the receptor

For a given site, not all exposure pathways may be "complete." That is, one or more of the above components may be missing. Further, exposures for some pathways may be too small to be significant for the HHRA. Therefore, an analysis of exposure pathways is included to identify complete and significant exposure pathways that may be important for risk management decisions.

Sources of contamination, mechanisms of contaminant release from sources, and subsequent transport of contaminants through the environment are examined in this section to identify potentially contaminated media at the site. Potential exposure pathways for human receptors are discussed in subsequent sections.

4.5.1 Exposure Pathways of Concern

As discussed above, an exposure pathway generally consists of a chemical source, mechanism for release and transport, a point of exposure to the contaminated medium, and a route of exposure into the receptor. The absence of any one of these elements would result in an incomplete exposure pathway. Furthermore, if one of these steps is very inefficient, exposure potential may be negligible, even though the pathway is theoretically complete. Potential exposure pathways are therefore identified in the SCEM and evaluated to determine whether they are complete and significant. The SCEM (Figure 4-1) identifies complete pathways that may represent significant potential for exposure and are therefore the focus of the HHRA. Current and future residents, constructions workers, or current and future recreational users of the site could be exposed to site-related contaminants, especially arsenic and lead, via several pathways, as illustrated in the SCEM (Figure 4-1).

4.5.1.1 Ingestion

Contaminated media may pose risk to receptors through ingestion of contaminated media, whether such ingestion is incidental or intentional. Ingestion of contaminated material may be in minor quantities, but depending on bioavailability, may lead to relatively great exposure.

Purposeful Ingestion

Ingestion of secondary and tertiary sources of contamination may pose some risk to residents and recreational users. Purposeful ingestion of contaminated media by construction workers is highly unlikely, and so, this pathway is considered incomplete.

Groundwater from 5 wells sampled at the ALS contained concentrations of heavy metals. Two of the wells are used for drinking, and an Artesian well located at the northern end of the site may be used by as many as 50 residents for drinking, despite warnings from the state. The use of groundwater for drinking by residents is evaluated quantitatively. Recreational users and construction workers do not have access to groundwater for drinking purposes, so this pathway is considered incomplete.

The consumption of fish is another potential exposure route for recreational users. Fish (e.g., crayfish) will be exposed to both contaminated surface water and sediment, and fish may accumulate some metals in their tissues. Surface water and sediment of Sutton Branch have had measurable concentrations of heavy metals in the past, although these concentrations are expected to be diminishing since the removal action was completed. Currently, mine wastes (chat) are mostly not visible in the Sutton Branch Creek, a significant change from pre-removal conditions. Furthermore, this creek is too small to support a fishery, and no complete exposure pathway exists for anglers on this creek. As discussed previously, some fish habitat may exist on the creek between confluences with the Hampton Branch and Big Creek. Fish in this reach are likely to move between Sutton Branch Creek and Big Creek and can be most reasonably assessed when addressing potential exposures for anglers that frequent the creek. Such an evaluation is outside the scope of this assessment.

Another purposeful route of exposure for residents may occur from the ingestion of homegrown produce. Vegetables may accumulate contaminants if they are grown in contaminated soil. Plants may accumulate some metals. For instance, plants can take up arsenic from soil and will deposit it in the leaves, so consumption of leafy vegetables may increase exposure to arsenic (ATSDR 1989). Lead is mostly stored in the roots of plants instead of in the shoots or seeds; therefore, consumption of root vegetables may increase exposure to lead (ATSDR 1999).

Uptake and accumulation of metals in vegetables, and subsequent consumption, can lead to increased exposure in residents with gardens. The garden scenario is incomplete based on current site conditions. The only residence with potential use of contaminated areas for a garden is located upland of the floodplain and south of Highway 49. Areas in the mine operations area could be potentially used for a garden if future residential development takes place. However, based on data from other sites, the uptake of arsenic, lead and other metals for soils at mine sites is likely to be insignificant. Thus, this pathway is not included in the quantitative analysis. The pathway is discussed in more detail in Section 7, Uncertainties.

Incidental Ingestion

Incidental ingestion of surface soil is evaluated for all potential receptors at the ALS. If redevelopment were to occur in the northern segment of the site, subsurface contamination may be brought to the surface and current and future residents could be exposed to contaminants while working or playing in their yards. Incidental ingestion of soils may occur via hand to mouth activities. This pathway may be significant, especially for younger children who tend to ingest larger quantities of soil during play. Also, construction workers involved in earthwork (i.e. excavating, grading, landscaping, etc.) in the northern segment of the site may be exposed to contaminants during construction activities and could potentially ingest subsurface contamination via hand-to-mouth activities. Recreational users of the southern segment may also incidentally ingest contaminants while playing in the area.

Incidental ingestion of interior dust is evaluated using the IEUBK model for future residential children assuming a soil-to-indoor dust transfer factor of 0.7, the default in the model.

Incidental ingestion of surface water and sediment during wading or swimming by recreational users of Sutton Branch Creek is a potentially complete exposure pathway. An actual quantitative amount of material ingested may be difficult to quantify but is likely greatest for children who may ingest small amounts of water and/or sediment during wading, bait collection, or other play activities in the waters of Sutton Branch Creek.

4.5.1.2 Dermal Contact

Direct contact with wastes at the mine operations area has been limited by the recent removal action, but tailings waste has migrated to floodplain areas and to sediments in Sutton Branch Creek. Receptors may be exposed through dermal contact with these media currently. In the future, dermal exposure might be possible if residual contamination beneath the 18-inch clean cover in the mine operations area is brought to the surface during excavation.

Dermal exposure pathways are not expected to contribute significantly to overall exposure because most metals are inefficiently absorbed through the skin. However, some measurements exist for absorption of arsenic in soil through the skin and these data can be used to estimate dermal exposure to this COPC. Thus, dermal absorption is quantitatively estimated for arsenic in soils and sediments.

For other soil COPCs, lead, iron, and manganese, no dermal absorption estimates are made. The IEUBK model recognizes the insignificance of this pathway by not including dermal absorption as a route of exposure for lead. In similar fashion, significant absorption of iron and manganese from soil, sediment or indoor dust seems highly unlikely and also is not quantified.

Dermal exposure is theoretically possible for all receptors evaluated in this assessment. However, assessment of dermal exposure to arsenic in soils or dust for hypothetical future residents or construction workers is not possible because of lack of post-removal data for constituents other than lead. Recreational users may come into contact with soil during play or other activities near the bank of the creek and/or elsewhere in the flood plain. These receptors may also come into dermal contact with in-stream sediments. Dermal exposure is evaluated only for current and future recreational visitors to the floodplain.

Dermal contact with contaminated groundwater or surface water is also theoretically possible for the site. However, as indicated above, little groundwater contamination attributable to the site has been detected, and obvious mine waste contamination is no longer present in surface sediments in Sutton Branch Creek. The latter observation suggests that any source of metals to surface water has been reduced significantly since the completion of the removal action. These observations, in turn, suggest that dermal contact with COPCs in groundwater and surface water should be small or negligible. However, in keeping with the evaluation of potential exposure to arsenic via dermal contact, this pathway is evaluated for this single COPC. Only hypothetical future residents are anticipated to use groundwater for domestic purposes. Thus, these are the only receptors evaluated for dermal exposure to arsenic in groundwater.

4.5 1.3 Inhalation

Finally, receptors at the ALS may have an increased exposure to certain contaminants via inhalation of dust and particulates. Any existing surface contamination is mostly covered by vegetation, and wind speeds needed to carry particulates at the site are not likely to be reached. In the future, some materials in the subsurface at the site may be brought to the surface and could represent a source of metals to ambient air. However, such exposures are unlikely to represent significant exposure. For example, a calculation for arsenic suggests that in residential settings, risks associated with inhalation of arsenic may be 2 orders of magnitude less than risks associated with ingestion of contaminated soil. Arsenic is a good test case, because the slope factor for arsenic via inhalation is an order of magnitude higher than that for ingestion. Thus, risks due to inhalation of arsenic should be relatively high compared to those for ingestion. That inhalation risks are still much lower than those for ingestion suggest that the inhalation pathway will be insignificant for all COPCs.

The inhalation pathway is not quantified for any receptors for the ALS.

4.5.2 Receptor-Specific Exposure Assumptions for Evaluation of COPCs Other Than Lead

Exposure assumptions were identified based on characteristics of specific receptor groups reasonably assumed to be affected by mine wastes. Exposure assumptions are presented for estimates of RME. Chemical intake estimates for RME use upper range values for some, but not all, exposure assumptions so that their combination results in a reasonable upper range estimate of exposure for that pathway. Exposure parameters used to evaluate RME are summarized in Tables 4-1 through 4-4. Three receptors exist for the ALS: current/future resident, current/future recreational user, and future construction worker. The assumptions specific to these pathways are further characterized for child and adult receptors, where appropriate. Exposure parameters specific to each receptor are evaluated below.

Often possible risks and hazards for a site are also estimated using parameters consistent with central tendency exposure (CTE). Such estimates were not included in this risks assessment because potential lead exposure was assumed to be the "driver' for site-related health hazards. Thus, the emphasis in this assessment is on estimation of lead exposure using the IEUBK model and Adult Lead Methodology, for which the concepts of RME and CTE do not apply.

Note that the exposure assumptions identified in this section do not apply to the evaluation of lead exposure. Lead is assessed using the IEUBK and Adult Lead Methodology and is separately discussed in Section 4.8, Methods for Evaluating Exposure to Lead.

4.5.2.1 Current/Future Resident

The current and future resident exposure is evaluated for both an adult resident and a young child. Exposure parameters are discussed below. Note again that no soil data are available for constituents other than lead for areas assessed for residential exposure. Thus, lead is the only COPC evaluated for exposure to soil and indoor dust for residents. Soil data used to evaluate lead exposures for residents in the mine area were collected after the removal action, immediately below the 18 inch cap of clean soil. Evaluation of lead exposures is discussed separately in Section 4.7 below. Finally, note also that the only on-site residents live in areas of the site where lead concentrations in soil are less than the screening level of 400 mg/kg. Thus, although a current residential scenario exists, exposure to these residents is expected to be minimal.

Lack of data to characterize post-remediation conditions at the mine operations area for COPCs other than lead is a potentially significant data gap. The impact of this data gap is further discussed in Section 7, Uncertainties.

Exposures to contaminants in groundwater are evaluated for both an adult resident and a young child. Residents are assumed to use groundwater as a drinking water source and for other domestic purposes such as bathing. Exposure parameters are discussed below.

Exposure Frequency
The exposure frequency is the number of days per year an individual participates in a particular activity. An exposure frequency of 350 days/year is used to evaluate residential exposures for children and adults (EPA 1991). This value assumes that a person spends all but 15 days of vacation each year at home.

Exposure Duration
The duration of exposure is the number of years over which exposure may occur. For residential RME exposure durations, exposure durations for ingestion of soil and dust of 24 and 6 years are used for adult and child residents, respectively (EPA 1989). Exposure to noncarcinogens is based on exposure assumptions for adults and children separately (EPA 1991). All other pathways are based on 30 years for adults.

Body Weight
For adult residents, the value selected for body weight is 70 kg. This value is the representative mean body weight for people between the ages of 18 and 75 (EPA 1991). For child residents (ages 0 to 6 years), a value of 15 kg is used for the body weight parameter (EPA 1991).

Averaging Time
Averaging time is the period in days over which intake is averaged. For noncarcinogenic chemicals, intakes are averaged over the exposure duration (exposure duration [years] * 365 days/year). For carcinogens, intake calculations average the total cumulative dose over a lifetime (70 years * 365 days/year).

Consistent with typical EPA practice, a lifespan of 70 years is used in this HHRA. Averaging times differ for carcinogens and noncarcinogens because the effects of carcinogenic chemicals are assumed to have no threshold. Therefore, any exposure to a carcinogen carries a finite risk of cancer during the individual's lifetime. Within reason, this means that a single large exposure to a carcinogen is expected to carry the same risk as the same dose divided into many small exposures. Therefore, carcinogen intakes are expressed in terms of lifetime exposures, regardless of the actual exposure duration (EPA 1989). For noncarcinogenic chemicals, hazards are anticipated to be proportional to average daily exposure, and intakes are therefore averaged over the exposure duration multiplied by 365 days. The averaging time for a child resident is 6 years or 2,190 days.

Ingestion Rate
Ingestion rates used are EPA recommended default values (EPA 1991). Ingestion rate of groundwater used for drinking water is 2 L per day for adults and 1 L per day for children.

Skin Surface Area
For dermal contact with groundwater, the total body surface area for adults and children is assumed to be exposed while bathing. Since surface area is a dependent variable the 50th percentile value is used in order to correlate with average body weights. The exposed skin surface area for the adult resident is 18,000 cm^2, the average of the 50th percentile for males and females greater than 18 years of age (EPA 2004f). The skin surface exposure area for the child is 6,600 cm^2, the average of the 50th percentile for males and females between the ages of 1 year old and 6 years old (EPA 2004f).

Dermal Permeability Coefficient
Dermal permeability coefficients are chemical-specific and were obtained from EPA (EPA 2004f).

Dermal Contact Event Frequency
The dermal contact event frequency is assumed to be one event per day (EPA 2004f) for both adult and child residents.

Dermal Contact Event Duration
The EPA (EPA 2004f) recommended RME event duration for dermal contact during bathing is used. The event duration is assumed to be 0.58 hours per day and 1 hour per day for the adult and child resident, respectively.

4.5.2.2 Current and Future Recreational User

A current and future recreational user is evaluated based on an older child scenario (7 to 16 years). The evaluation of this receptor is considered to be protective of all users because children are the most sensitive receptor for non-carcinogenic effects, and they are the most likely receptor with the most frequent exposure through recreational use of the site. Potential exposures to COPCs in surface water, sediment and surface soil are evaluated for the recreational receptor. Surface soil data collected from the floodplain area are used to evaluate recreational exposures associated with soil. Maximum detected COPC concentrations in surface water and sediment collected from Sutton Branch Creek in the floodplain area are used to evaluate exposures to these media. Evaluation of recreational exposures is uncertain because data on actual recreational use are seldom available. Uncertainties in the quantitative evaluation of this receptor are discussed in some detail in Section 7, Uncertainties.

Exposure Frequency
Exposure frequencies of 2 days per week over the warmest 6 months of the year (52 days total) are used to evaluate exposures in the older child recreational users. This assumption is expected to reflect maximal exposure frequency for a local recreational user living near contaminated land and using the floodplain almost as an extension of their yard. These same assumptions are used for exposure to soils in the floodplain and for exposure to sediments and surface water in the Sutton Branch. Younger children (0 to 6 years) are not evaluated because they are less likely to spend time in floodplain areas because of the need for supervision by adults during recreational activities in these areas.

Exposure Duration
Recreational visitors are assumed to live in the area; therefore, exposure for noncarcinogens and carcinogens is assumed to continue for the entire period from ages 7 through 16 (10 years) based on professional judgment..

Exposure Time
Exposure time for a wading scenario was assumed to be 2 hours/day (USEPA 1997a).

Body Weight
The body weight (BW) was set to 43 kg, which is the average of the mean body weights of boys and girls from age 7 through age 16 (USEPA 1997a).

Averaging Time
A lifetime expectancy of 70 years (USEPA 1989) was used for all receptor groups as the averaging time for exposure to carcinogenic contaminants. For noncarcinogenic chemicals, intakes are averaged over the exposure duration multiplied by 365 days. Therefore, the averaging time is 3,650 days for a child recreational user.

Ingestion Rate
Recreational users will likely have an ingestion rate similar to that of adult residents. The daily incidental ingestion rate for sediment is therefore assumed to be 100 milligrams per day (mg/day), which is 100 percent of the daily soil ingestion rate presented for an older child (USEPA 1997a). In the absence of guidance on this exposure assumption, the above rate was selected as a conservative measure. This value may overestimate sediment ingestion rates; moist sediments might adhere more strongly to skin than drier soil, but creek water would tend to wash the sediments off before the soiled skin reaches the mouth or food. All exposure is assumed to occur at the site during the event; thus, the fraction ingested (FI) was conservatively assumed to be 100 percent.

Recreational users are assumed to ingest 50 ml/hour of surface water during wading (USEPA 1989). This value is actually appropriate for swimming, which is not possible in Sutton Branch Creek. However, no values for activities such as wading appear to exist. This assumption is likely to overestimate possible exposures via surface water ingestion.

Skin Surface Area
A child recreational user is assumed to wear a short-sleeved shirt and shorts (no shoes); therefore, the exposed skin surface area is limited to hands, forearms, lower legs, and feet. The skin surface exposure area for the child is 4,000 cm2, the average of the 50th percentile for males and females between the ages of 7 and 16 years and the percentage of total body surface area by body part (30 percent for hands, feet, forearms and lower legs) for adults (USEPA 1997a). These values assume that relative surface areas for body parts remains constant over the age range of 7 to 16 years.

Soil-to-Skin Adherence Factor
A dermal adherence factor of 0.2 mg/cm^2 was assumed for recreational users for exposure to floodplain soils. An adherence factor of 1 mg/cm^2 was assumed for exposure to sediments in Sutton Branch Creek (USEPA 2004f).

Dermal Absorption Factors
Chemical-specific dermal absorption fraction for arsenic is 0.03 (USEPA 2004f). No other COPCs are quantitatively evaluated for dermal exposure.

Dermal Contact Event Frequency
Dermal contact event frequency is assumed to be one event per day (USEPA 2004f) for recreational users.

4.5.2.3 Construction Worker

Construction workers are only assessed for potential exposure to residual lead in the mine operations area. Thus, no exposure parameters are identified for assessing exposure to other COPCs. Post-remediation soil data was used to evaluate lead exposures for construction workers in the mine area.

4.5.3 Bioavailability of Metals in Soil and Dust

No site-specific bioavailability studies have been conducted for the site. A relative bioavailability of 100 percent is therefore assumed for all COPCs, except lead. This assumption in essence indicates that COPCs are absorbed into the body in similar amounts as the chemical form of the COPC used to define toxicity in human epidemiological or animal laboratory studies.

The oral bioavailability for lead was assumed to be the default in the IEUBK model, 30 percent absolute absorption from the GI tract.

4.6 Exposure Point Concentrations

Exposure point concentrations (EPCs) represent chemical concentrations in environmental media that a person could potentially contact. In a typical baseline risk assessment following USEPA guidance, a conservative estimate of the average concentration that a person might contact is used as the exposure point concentration (e.g., the 95% Upper Confidence Limit (UCL) on the arithmetic mean chemical concentration).

For this assessment, potential exposures for residents, recreationists, and construction workers were performed on the following basis:

- soils beneath the 18" clean soil cover in the mine operations areas of the ALS separately, excluding hotspots (lead only);
- hotspots in soil beneath the 18" soil cover in the mine area of the ALS (see Figure 3-3);
- surface soils in the Sutton Branch Creek floodplain;
- Sutton Branch Creek surface water and sediment in the mine operations and floodplain areas separately; and,
- groundwater in the mine operations area only.

Sample locations are presented on Figures 3-1 through 3-5. Exposure point concentrations are estimates of average concentrations of lead in the mine operations area and lead and other COPCs within the floodplain in the ALS.

The UCL of the arithmetic mean was used as exposure point concentrations for surface and subsurface soil, except as described below. These estimates were then used to assess exposure for residents, recreationists, and construction workers in both the mine operations and floodplain areas of the ALS. The UCL provides a conservative estimate of the mean concentration, such that randomly drawn subsets of site data will have means that are equal to or less than the UCL, some pre-determined percentage of the time. UCLs were calculated according to methods outlined in *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (USEPA 2002). For nondetects, COPCs are assumed to be present at one-half of the laboratory reporting limit, if this value was below the maximum detected value. Computation of an UCL of the population mean depends upon the data distribution. Typically, environmental data are positively skewed, and a default lognormal distribution is often used to model such distributions. EPA's ProUCL (USEPA 2004d) program, Version 3.0, was used to test normality or lognormality of the data distribution and to compute conservative and stable UCLs of population means. ProUCL computes the UCLs of the population means both using parametric (distribution sensitive) and nonparametric (distribution insensitive) procedures. ProUCL calculations and UCLs for surface and subsurface roadway soil are presented in Appendix D.

UCLs were not calculated for surface water, sediment, and groundwater because of the low sample numbers. Instead, the maximum detected concentration of each COPC was used as the EPC for surface water, sediment, and groundwater.

An exception to the use of UCL for exposure point concentrations is lead. Inputs to the IEUBK model are intended to be simple averages. Thus, EPCs for lead were estimated as the simple average of soil data for the mine operations area and the floodplain. Use of the arithmetic mean for the two hotspots identified in the mine operations area that could theoretically support residential development in the future is subject to some uncertainty because of relatively high variability in soil lead concentrations in these areas. This issue is further discussed in Section 7, Uncertainties.

Variability in lead concentrations is also relatively high for the floodplain area south of Highway 49. In this case, however, receptors are expected to access the site randomly; that is, no areas that might be particularly attractive for recreational use are apparent when walking the site. Thus, recreational visitors are anticipated to contact soils throughout the floodplain. The average concentration of lead in this area is 912 mg/kg. This value is not substantially less than UCLs calculated using several bootstrap procedures, which fell in the range of 1,400 to 1,500 mg/kg. Use of these higher values would not affect the conclusions of the risk assessment.

Table 4-5 and Appendix D present the ALS EPCs for surface water, sediment, and groundwater.

Table 4-5: Annapolis Mine Site EPCs

Media	Chemical	Location	EPC Used	EPC
Surface Water[1]	As	Sutton Branch Floodplain Area	Maximum	0.0125 mg/L
	Mn	Sutton Branch Floodplain Area	Maximum	0.0115 mg/L
	Pb	Sutton Branch Floodplain Area	Maximum	0.025 mg/L
	Tl	Sutton Branch Floodplain Area	Maximum	0.101 mg/L
Sediment [1]	As	Sutton Branch Floodplain Area	Maximum	25.60 mg/kg
	Pb	Sutton Branch Floodplain Area	Maximum	1070 mg/kg
Groundwater[2]	As	Mining Area	Maximum	0.00083 mg/L
	Fe	Mining Area	Maximum	41.5 mg/L
	Pb	Mining Area	Maximum	0.0038 mg/L
	Tl	Mining Area	Maximum	0.0023 mg/L
Soil	Pb	Mining Area w/o hotspots	Mean	159.4 mg/kg
	Pb	Mining Area, Clark hotspot	Mean	6959.7 mg/kg
	Pb	Mining Area, Mayberry hotspot	Mean	2639.7 mg/kg
	As	Floodplain Area	95% Approximate Gamma UCL	34.45580 mg/kg
	Mn	Floodplain Area	95% student's-T UCL	1497.025 mg/kg
	Pb	Floodplain Area	Mean	912.4 mg/kg

[1] Maximum detected concentrations of analytes in surface water and sediment from Sutton Branch Creek within the floodplain area are used as EPCs.
[2] The shallow irrigation well, CC104-001 was excluded from the dataset used to estimate groundwater EPCs because this well is screened in an aquifer (10 feet below ground surface) not normally used for drinking water in the area. This shallow irrigation well was included in the dataset used to select COPCs (Table 3-3) and many of the maximum detected values presented in Table 3-3 are from the water sample collected from this well.

4.7 Exposure Calculations for COPCs Other Than Lead

Chronic daily intakes (CDI) are calculated for arsenic, manganese, and thallium using the exposure assumptions described above. CDIs are estimated for each selected exposure pathway. The equations used to calculate CDIs for each exposure pathway are shown below.

4.7.1 Ingestion and Dermal Contact with Soils or Sediments

The following equation is used to estimate CDIs for ingestion and dermal exposure of soil and sediment exposure:

Ingestion of Contaminated Soils

Pathway Intake Equation:

Chronic Daily Intake (CDI) (mg/kg-day) = Cs x CF_s x IRS x EF x ED x 1/BW x 1/AT

Where:

Cs= Chemical Concentration
CF_s = Soil Conversion Factor
IRS = Soil Ingestion Rate
EF = Exposure Frequency
ED = Exposure Duration
BW = Body Weight
AT_c = Averaging Time-Cancer
AT_n = Averaging Time-Non-Cancer

Dermal Contact with Contaminated Soils and Sediments

Pathway Intake Equation:

Dermally Absorbed Dose (DAD) (mg/kg-day) = DA event x EV x ED x EF x SA x 1/BW x 1/AT

Where:

DA_{event} = C_s x CF_s x AF x ABS_d

C_s = Chemical Concentration

CF_s = Soil Conversion Factor

SA = Skin Surface Area

ABS_d = Dermal Absorption Fraction

AF - Adherence Fator

ET = Exposure Time

EF = Exposure Frequency

ED = Exposure Duration

EV = Event Duration

BW_c = Body Weight

AT_c = Averaging Time-Cancer

AT_n = Averaging Time-Non-Cancer

4.7.2 Ingestion of and Dermal Contact with Groundwater

The following equation is used to estimate CDIs for ingestion of and dermal contact with groundwater pathway:

Ingestion of Contaminated Groundwater

Pathway Intake Equation:

Chronic Daily Intake (CDI) (mg/kg-day) = C_w x EF x [(IRW_a x ED_a x 1/BWa) + (IRW_c x ED_c x 1/BW_c)] x 1/AT

Where:

C_w = Chemical Concentration

IRW_a = Water Ingestion Rate-Adult

IRW_c = Water Ingestion Rate-Child

EF = Exposure Frequency

ED_a = Exposure Duration-Adult

ED_c = Exposure Duration-Child

BW_a = Body Weight -Adult

BW_c = Body Weight -Child

AT_c = Averaging Time-Cancer

AT_n = Averaging Time-Non-Cancer

Dermal Contact with Contaminated Groundwater

Pathway Intake Equation:

Dermally Absorbed Dose (DAD) (mg/kg-day) = $EF \times CF_w \times EV \times [(DA_a \times SA_a \times ED_a \times 1/BW_a) + (DA_c \times SA_c \times ED_c / BW_c)] \times 1/AT$

Where:

$DA_a = K_p \times C_w \times EVS_a$
$DAc = K_p \times C_w \times EVS_c$
C_w = Chemical Concentration in Water
CF_w = Water Conversion Factor
SA_a = Skin Surface Area-Adult
SA_c = Skin Surface Area-Child
K_p = Permeability Constant
ET_a = Exposure Time-Adult
ET_c = Exposure Time-Child
EF = Exposure Frequency
ED_a = Exposure Duration-Adult
ED_c = Exposure Duration-Child
EVS_a = Event Duration-Adult showering
EVS_c = Event Duration-Child bathing
BW_a = Body Weight-Adult
BW_c = Body Weight-Child
AT_c = Averaging Time-Cancer
AT_n = Averaging Time-Non-Cancer

Specific values used for these daily intake calculations can be found in Appendix E, Table 4.

4.8 Methods for Evaluating Exposure to Lead

Exposures to lead are not evaluated using the same methods as those described above for other site COPCs. Methods used to evaluate such exposures are described in the following sections for young children and for adults.

4.8.1 Use of the IEUBK Model for Young Children

Blood lead level calculations for young children used *Windows* Version 1.0, Build 261 of the IEUBK model. Except as described below, default parameters in this model were used in the analysis.

Concentration of Lead in Drinking Water

Site-specific measurements of lead in groundwater from drinking wells suggest concentrations of 3.8 μg/L or lower, which are not notably different from the default value of 4 μg/L; thus, the default value for this parameter was retained.

Dietary Intake of Lead

Updated dietary lead intake values are available from the Technical Review Workgroup (TRW) and were used in all modeling. An important source of lead exposure for the IEUBK model is lead consumed with food. Current data from the USDA Total Diet Study (FDA 2001) and the National Health and Nutrition Examination Survey III (NHANES III) (CDC 1997) indicate that dietary lead exposure has decreased since the current default estimates for dietary lead were developed for the IEUBK model. USEPA's TRW for the IEUBK model have provided updated dietary lead intake estimates for use in the model, and have indicated that use of these new dietary estimates may influence risk management decisions at sites where lead is a key contaminant.

Updated dietary intake estimates are provided by USEPA's TRW for lead (http://www.epa.gov/superfund/programs/lead/ ieubkfaq.htm#fda). The recommended updated dietary intake estimates are used in evaluating potential lead exposures in young children. By age group, the updated dietary intake values are:

Age Range (years)	Dietary Lead Intake (µg/d)
0-1	3.16
1-2	2.6
2-3	2.87
3-4	2.74
4-5	2.61
5-6	2.74
6-7	2.99

µg/d = micrograms per day

Alternative estimates for other inputs to the IEUBK model might also be considered in evaluating potential lead exposure. These inputs are not universally accepted and are discussed under uncertainties rather than being included in the quantitative analysis here. Alternative estimates could be considered for soil ingestion rates and soil-to-dust transfer.

Other IEUBK Model Input Parameters

All other input parameters to the IEUBK model were retained as model defaults. These parameters include geometric standard deviation (GSD), 1.6; maternal blood lead concentration, 2.5 microgram per deciliter (µg/dL); concentration of lead in air, 0.1 micrograms per cubic meter (µg/m^3); and other sources of lead exposure, 0 µg/d. A complete list of input parameters for the IEUBK model runs is provided along with the output from these runs in Appendix F.

4.8.2 Use of the Adult Lead Methodology

USEPA's adult lead methodology (ALM) (USEPA 1996b) was used to assess intermittent or variable exposures to lead at the site by recreational users (older children) and construction workers. This model actually predicts lead exposure to the fetus of a pregnant women and is therefore not directly applicable to the older child (ages 7 to 16 years) that is evaluated for intermittent lead exposure. However, this model should be conservative for this age group and is the only methodology that can be applied to older children. Use of the model for this age group is likely to overestimate any potential health impacts to lead. This issue is further discussed in later sections.

The model recommended by USEPA (1996b) for use in evaluating lead exposures does not include inputs for either dermal or inhalation exposure to lead in soil. Implicitly, USEPA has determined that these exposure routes are typically insignificant compared to incidental soil ingestion. This conclusion is consistent with the IEUBK model for evaluating lead exposure in young children (USEPA 2002). This model does not consider dermal exposure to lead, and demonstrates that even inhalation exposure represents only a small fraction of total lead exposure in residential situations. Neither dermal nor inhalation exposure are considered in the quantitative estimates of possible impacts of lead exposure on blood lead levels.

For evaluation of adult exposures, the methodology consists of algorithms that concentrate on estimated fetal blood lead concentrations in pregnant women exposed to lead-contaminated soils. Thus, women of child-bearing age are the target receptor group for adult lead exposure. The adult lead model can thus be applied to recreational and adolescent receptors, provided that the appropriate model conditions are met (USEPA 2005e). Empirical data on biokinetic slope factors appear to be similar for young children and adults; however, there is uncertainty in applying a similar estimate for adolescents. Reported low baseline blood concentrations for children between the ages of 12 and 18 years of age (Brody et al., 1994) may be due to a growth spurt in which there is a shift of lead from blood to bone.

Exposure assumptions used in the ALM are discussed below and are summarized in Table 4-6.

Interpretation of Predictions from the Adult Lead Methodology
Interpretation of output from the Adult Lead Methodology is based on fetal blood lead level. EPAs health protection goal, that the probability of blood lead concentrations exceeding 10 µg/dL be 5 percent of less, is used to assess potential lead impacts for a developing fetus.

Background Blood Lead Concentration
The background adult blood lead concentration is the typical blood lead concentration in women of child bearing age in the absence of exposures to the site that is being assessed. Baseline blood lead concentrations (PbB) seem to vary by age, socioeconomic status and race/ethnicity. Lower PbB are often found among non-Hispanic white women, and higher levels among non-Hispanic black women. USEPA (2002) provides a range of values for each of these parameters, and some guidance for choosing values appropriate for a given site. Since site-specific data are unavailable, data from the NHANES III survey were used to determine appropriate values for the site. A PbB of 1.53 was used in this evaluation; this value is representative for all races in the Midwest Region (Table 3a, NHANES III, CDC 1997).

Biokinetic Slope Factor
The biokinetic slope factor relates the increase in adult blood lead concentration to average daily lead uptake (µg/dL blood lead increase per µg/day lead uptake). The default value of 0.4 µg/dL per µg/day provided by USEPA (1996b) is based on steady-state conditions. This value is used for all receptors.

Geometric Standard Deviation

In USEPA's adult lead methodology, the geometric standard deviation (GSD) is the estimated value of GSD among women of child-bearing age that have exposures to similar onsite lead concentrations but that have non-uniform response to site lead and non-uniform offsite lead exposures. GSD estimates seem most sensitive to how heterogeneous the population that may use the site is compared to the US population. USEPA provides a default GSD for four census regions and race/ethnicity (USEPA 2002). A GSD of 2.18 was used in this evaluation; this value is representative for all races in the Midwest Region (Table 3a, NHANES III, CDC 1997).

Averaging Time

An averaging time (AT) of 182 days is used to calculate PbB for construction workers and recreations users. A construction worker is assumed to work at the site only during the warmer six months of the year.

Absorption Fraction

This parameter is the absolute gastrointestinal absorption fraction (AFs) for ingested lead in soil and lead dust derived from soil. The default value of 0.12 (unitless) recommended by the TRW (USEPA 1996b) is used for the PRG calculation. The default value is based on the assumption that the absorption factor for soluble lead is 0 2 ($AF_{soluble}$) and that the relative bioavailability of lead in soil compared to soluble lead ($RBF_{soil/soluble}$) is 0.6:

$$AF_s = AF_{soluble}\ (0.2) * RBF_{soil/soluble}\ (0.6) = 0.12$$

Soil Ingestion Rate

USEPA (2005e) recommends a default value of 100 mg/day for construction workers engaged in short-term activities that may involve intimate contact with soils (e.g. excavation). USEPA does not recommend CTE values for soil ingestion rates in children older than 6 years. The soil ingestion rate of 50 mg/day is also used for the recreational user.

Exposure Frequency

Exposure frequency (EF) is the number of days per year that an individual may be exposed to site-related contaminants. Construction workers generally participate in only part of the construction or remedial activities, so that a few weeks of exposure are probably all that a single individual might be exposed (e.g. during excavation of a building foundation). Exposures for construction workers are generally short-term and the kinetics of lead exposure require several months before a new equilibrium of blood lead concentration is reached. For this analysis, an EF of 132 days/year is used for construction workers. This site-specific estimate corresponds to 22 days per month (5 days per week) for a 6 month period.

Significant uncertainty exists regarding the number of days a recreational receptor may visit the site. For this analysis, a range of values -- 27, 52 and 132 days (professional judgment) per year -- is used for the recreational user. This range provides an illustration of the sensitivity of the Adult Lead Methodology to exposure frequency. The top of the range, which corresponds to 5 days of exposure per week during the warmest 6 months of the year is likely to be an extreme value, given the lack of access and attractiveness of much of the site for recreational use. Thus, the upper end of the range should provide a ceiling on any lead exposures that might occur.

4.9 Summary

The preceding sections outline an approach to exposure assessment for the Annapolis Lead Site that includes the following:

- Calculation of exposures (and hence risks and hazards) given the baseline conditions within the Sutton Branch floodplain and after completion of soil remediation activities at the source area;
- Quantitative exposure evaluation for residents only in areas outside of the floodplain in the former mine operational area (lead only);
- Use of site-specific information for concentration of lead in tap water;
- Use of standard USEPA default exposure parameters for all non-site specific assumptions, and standard USEPA algorithms for estimation of potential risks and hazards due to exposure to arsenic, manganese, and thallium;
- Use of the USEPA's IEUBK model for estimation of lead exposure for young children and USEPA's Adult Lead Methodology for estimation of lead exposure for construction worker and recreational scenarios; and
- Development of a matrix of plausible lead exposure estimates for recreational visitors to the sites based on a range of exposure frequencies.

Results of the exposure assessment are combined with toxicity criteria identified in Section 5 and are presented in Section 6, Risk Characterization. Important uncertainties in exposure assessment are discussed in Section 7, Uncertainties.

Figure 4-1. Site Conceptual Exposure Model (SCEM) for the Annapolis Lead Mine

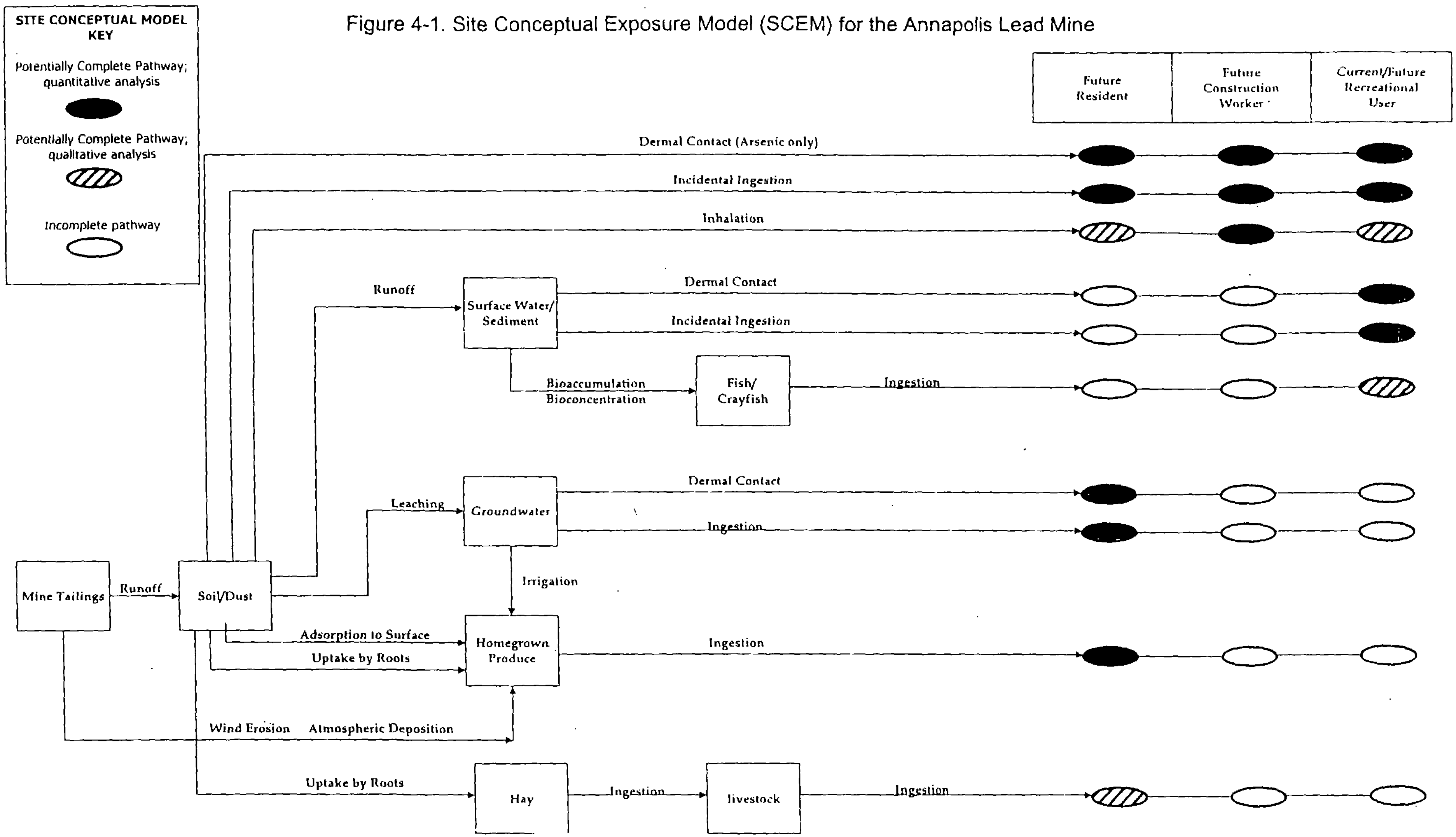


TABLE 4.1.RME

VALUES USED FOR DAILY INTAKE CALCULATIONS

REASONABLE MAXIMUM EXPOSURE

Scenario Timeframe: Future Exposure
Medium. Groundwater
Exposure Medium: Groundwater

Exposure Route	Receptor Population	Receptor Age	Exposure Point	Parameter Code	Parameter Definition	RME Value	Units	Rationale/ Reference	Intake Equation Model Name
Dermal	Resident	Adult	Dermal Contact while Showering	DAa	Dermally Absorbed Dose per Event-Adult	chemical-specific	mg/cm^2-event	USEPA 2004	
				CW	Chemical Concentration	chemical-specific	mg/L	Site-specific	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				CF	Water Conversion Factor	0.001	L/cm^3		((DAa x EDa x SAa x 1/BWa) +
				SAa	Skin Surface Area-Adult	18,000	cm^2	USEPA 2004	(DAc x EDc x SAc x 1/BWc)) x EF x EV x 1/AT
				Kp	Permeability Constant	chemical-specific	cm/hour	USEPA 2004	DAa = Kp x CW x CF x t-event$_a$
				EF	Exposure Frequency	350	days/year	USEPA 1991	DAc = Kp x CW x CF x t-event$_c$
				EV	Event Frequency	1	event/day	USEPA 2004	
				EDa	Exposure Duration - Adult	24	years	USEPA 1991	
				t-event$_a$	Event Duration-Adult Showering	0.58	hours/event	USEPA 2004	
				BWa	Body Weight - Adult	70	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	10950	days	USEPA 1991	
				DAc	Dermally Absorbed Dose per Event-Child	chemical-specific	mg/cm^2-event	USEPA 2004	
				SAc	Skin Surface Area-Child	6,600	cm^2	USEPA 2004	
				EDc	Exposure Duration - Child	6	years	USEPA 1991	
				t-event$_c$	Event Duration-Child bathing	1.00	hours/event	USEPA 2004	
				BWc	Body Weight - Child	15	kg	USEPA 1991	
		Child	Dermal Contact while Bathing	DA	Dermally Absorbed Dose per Event-Child	chemical-specific	mg/cm^2-event	USEPA 2004	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				Cw	Chemical Concentration	chemical-specific	mg/L	Site-specific	DA x EV x ED x EF x SA x 1/BW x 1/AT
				CF	Water Conversion Factor	0.001	L/cm^3		
				SA	Skin Surface Area-Child	6,600	cm^2	USEPA 2004	DA = Kp x CW x CF x t-event$_c$
				Kp	Permeability Constant	chemical-specific	cm/hour	USEPA 2004	
				EF	Exposure Frequency	350	days/year	USEPA 1991	
				EV	Event Frequency	1	event/day	USEPA 2004	
				ED	Exposure Duration-Child	6	years	USEPA 1991	
				t-event$_c$	Event Duration-Child bathing	1.00	hours/event	USEPA 2004	
				BW	Body Weight-Child	15	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	2190	days	USEPA 1991	
Ingestion	Resident	Adult	Drinking Water	CW	Chemical Concentration	chemical-specific	mg/L	Site-specific	
				IR-Wa	Water Ingestion Rate-Adult	2	L/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	350	days/year	USEPA 1991	CW x EF x ((IR-Wa x EDa x 1/BWa) +
				EDa	Exposure Duration-Adult	24	years	USEPA 1991	(IR Wc x EDc x 1/BWc)) x 1/AT
				BWa	Body Weight - Adult	70	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	10950	days	USEPA 1991	
				IR-Wc	Water Ingestion Rate-Child	1	L/day	USEPA 1991	
				EDc	Exposure Duration-Child	6	years	USEPA 1991	
				BWc	Body Weight - Child	15	kg	USEPA 1991	
		Child	Drinking Water	CW	Chemical Concentration	chemical-specific	mg/L	Site-specific	
				IR-W	Water Ingestion Rate-Child	1	L/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	350	days/year	USEPA 1991	CW x IR-W x EF x ED x 1/BW 1/AT
				ED	Exposure Duration-Child	6	years	USEPA 1991	
				BW	Body Weight - Child	15	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	2190	days	USEPA 1991	

USEPA 1989. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A) Interim Final. EPA/540/1-89/002. Office of Emergency and Remedial Response. U.S EPA. Washington DC.

USEPA 1991. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Supplemental Guidance Standard Default Exposure Factors. OSWER Directive 9285.6-03. Office of Emergency and Remedial Response. U.S. EPA. Washington, D.C

USEPA 1997 Exposure Factors Handbook. Volume 1, General Factors. Office of Research and Development. EPA/600/P-95/002Fa. August 1997.

USEPA. 2004d. Risk Assessment Guidance for Superfund, Volume I Human Health Evaluation Manual. (Part E, Supplemental Guidance for Dermal Risk Assessment). Final. Office of Emergency and Remedial Response. EPA/540/R/99/005 OSWER 9285.7-02EP. PB99-963312
a- Calculated from USEPA 1997 based on the a total body surface area to individual body part ratio for hands, feet, lower arms, and legs of adults. Proportion of body surface is assumed to be the same for children ages 7-1

TABLE 4.2 RME

VALUES USED FOR DAILY INTAKE CALCULATIONS

REASONABLE MAXIMUM EXPOSURE

Scenario Timeframe: Current/Future Exposure
Medium: Soils
Exposure Medium: Soils

Exposure Route	Receptor Population	Receptor Age	Exposure Point	Parameter Code	Parameter Definition	RME Value	Units	Rationale/ Reference	Intake Equation Model Name
Dermal	Resident	Adult	Dermal Contact with Soils/ Indoor Dust	DAa	Dermally Absorbed Dose per Event Adult	chemical-specific	mg/cm^2-event	USEPA 2004	
				CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				CF	Soil Conversion Factor	0.000001	kg/mg		((DAa x EDa x SAa x 1/BWa) +
				SAa	Skin Surface Area-Adult	5,700	cm^2	USEPA 2004	(DAc x EDc x SAc x 1/BWc)) x EF x EV x 1/AT
				ABSd	Dermal Absorption Factor	chemical-specific	percent	USEPA 2004	DAa = ABSd x CS x CF x AFa
				EF	Exposure Frequency	350	days/year	USEPA 1991	DAc = ABSd x CS x CF x AFc
				AFa	Adherence Factor-Adult	0.01	mg/cm^2	USEPA 2004	
				EDa	Exposure Duration - Adult	24	years	USEPA 1991	
				EV	Event Frequency	1.00	event/day	USEPA 2004	
				BWa	Body Weight - Adult	70	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	10950	days	USEPA 1991	
				DAc	Dermally Absorbed Dose per Event-Child	chemical-specific	mg/cm^2-event	USEPA 2004	
				SAc	Skin Surface Area-Child	2,800	cm^2	USEPA 2004	
				AFc	Adherence Factor-Child	0.2	mg/cm^2	USEPA 2004	
				EDc	Exposure Duration - Child	6	years	USEPA 1991	
				BWc	Body Weight - Child	15	kg	USEPA 1991	
		Child	Dermal Contact with Soils/ Indoor Dust	DA	Dermally Absorbed Dose per Event	chemical-specific	mg/cm^2-event	USEPA 2004	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	DA x EV x ED x EF x SA x 1/BW x 1/AT
				CF	Soil Conversion Factor	0.000001	kg/mg		
				SA	Skin Surface Area	2,800	cm^2	USEPA 2004	DA = ABSd x CS x CF x AF
				ABSd	Dermal Absorption Factor	chemical-specific	percent	USEPA 2004	
				EF	Exposure Frequency	350	days/year	USEPA 1991	
				AF	Adherence Factor	0.2	mg/cm^2	USEPA 2004	
				ED	Exposure Duration	6	years	USEPA 1991	
				EV	Event Frequency	1.00	event/day	USEPA 2004	
				BW	Body Weight	15	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	2190	days	USEPA 1991	
	Construction Worker	Adult	Dermal Contact with Soils/ Indoor Dust	DA	Dermally Absorbed Dose per Event	chemical-specific	mg/cm^2-event	USEPA 2004	
				CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				CF	Soil Conversion Factor	0.000001	kg/mg		DA x EV x ED x EF x SA x 1/BW x 1/AT
				SA	Skin Surface Area	3,300	cm^2	USEPA 2004	
				ABSd	Dermal Absorption Factor	chemical-specific	percent	USEPA 2004	DA = ABSd x CS x CF x AF
				EF	Exposure Frequency	132	days/year	Site-specific	
				AF	Adherence Factor	0.3	mg/cm^2	USEPA 2004	
				ED	Exposure Duration	1	years	Site-specific	
				EV	Event Frequency	1.00	event/day	Site-specific	
				BW	Body Weight	70	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	365	days	USEPA 1991	

TABLE 4 2.RME (continued)

VALUES USED FOR DAILY INTAKE CALCULATIONS

REASONABLE MAXIMUM EXPOSURE

Scenario Timeframe: Current/Future Exposure
Medium: Soils
Exposure Medium: Soils

Exposure Route	Receptor Population	Receptor Age	Exposure Point	Parameter Code	Parameter Definition	RME Value	Units	Rationale/ Reference	Intake Equation Model Name
Dermal (continued)	Recreational User	Child	Dermal Contact with Soils	DA	Dermally Absorbed Dose per Event	chemical-specific	mg/cm^2-event	USEPA 2004	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	DA x EV x ED x EF x SA x 1/BW x 1/AT
				CF	Soil Conversion Factor	0.000001	kg/mg		
				SA	Skin Surface Area	4,000	cm^2	USEPA 1997[a]	DA = ABSd x CS x CF x AF
				ABSd	Dermal Absorption Factor	chemical-specific	percent	USEPA 2004	
				EF	Exposure Frequency	52	days/year	*Site-specific*	
				AF	Adherence Factor	0.2	mg/cm^2	USEPA 2004	
				ED	Exposure Duration	10	years	Site-specific	
				EV	Event Frequency	1.00	event/day	*USEPA 1991*	
				BW	Body Weight	43	kg	USEPA 1997	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	
Ingestion	Resident	Adult	Hand-to-Mouth Contact with Surface Soil/Indoor Dust	CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	
				IR-Sa	Ingestion Rate-Adult	100	mg/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	350	days/year	USEPA 1991	CS x EF x (IR-Sa x EDa x 1/BWa) +
				EDa	Exposure Duration-Adult	24	years	USEPA 1991	(IR-Sc x EDc x 1/BWc) x 1/AT
				BWa	Body Weight-Adult	70	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	10950	days	USEPA 1991	
				IR-Sc	Ingestion Rate-Child	200	mg/day	USEPA 1991	
				EDc	Exposure Duration-Child	6	years	USEPA 1991	
				BWc	Body Weight-Child	15	kg	USEPA 1991	
		Child	Hand-to-Mouth Contact with *Surface Soil/Indoor Dust*	CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	
				IR-S	Ingestion Rate	200	mg/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	350	days/year	USEPA 1991	CS x IR-S x EF x ED x 1/BW x 1/AT
				ED	Exposure Duration	6	years	USEPA 1991	
				BW	Body Weight	15	kg	USEPA 1991	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non Cancer	2190	days	USEPA 1991	
	Construction Worker	*Adult*	Hand-to-Mouth Contact with Surface Soil/Indoor Dust	CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	
				IR-S	Ingestion Rate	330	mg/day	*USEPA 1997*	*Chronic Daily Intake (CDI) (mg/gk-day) =*
				EF	Exposure Frequency	132	days/year	Site-specific	CS x IR-S x EF x ED x 1/BW x 1/AT
				ED	Exposure Duration	1	years	Site-specific	
				BW	Body Weight	70	kg	USEPA 1991	
				ATC	Averaging Time Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	365	days	USEPA 1991	
	Recreational User	Child	Hand-to-Mouth Contact with Surface Soil	CS	Chemical Concentration	chemical-specific	mg/kg	Site specific	
				IR-S	Ingestion Rate	100	mg/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	52	days/year	Site-specific	CS x IR-S x EF x ED x 1/BW x 1/AT
				ED	Exposure Duration	10	years	Site-specific	
				BW	Body Weight	43	kg	*USEPA 1997*	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	

USEPA 1989. Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual (Part A) Interim Final EPA/540/1-89/002 Office of Emergency and Remedial Response U.S. EPA. Washington, DC.

USEPA 1991. Risk Assessment *Guidance for Superfund, Volume I* Human Health Evaluation Manual, Supplemental Guidance Standard Default Exposure Factors OSWER Directive 9285 6-03 Office of Emergency and Remedial Response U.S. EPA. Washington, D.C.

USEPA. 1997 Exposure Factors Handbook Volume 1 General Factors. Office of Research and Development. EPA/600/P-95/002Fa. August 1997.

USEPA 2004a Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual. (Part E, Supplemental Guidance for Dermal Risk Assessment). Final. Office of Emergency and Remedial Response. EPA/540/R/99/005. OSWER 9285.7-02EP PB99 963312.

a- Calculated from USEPA 1997 based on the a total body surface area to individual body part ratio for hands, feet, lower arms and legs of adults Proportion of body surface is assumed to be the same for children ages 7-16.

TABLE 4.3.RME

VALUES USED FOR DAILY INTAKE CALCULATIONS

REASONABLE MAXIMUM EXPOSURE

Scenario Timeframe: Current/Future Exposure
Medium: Sediment
Exposure Medium: Sediment

Exposure Route	Receptor Population	Receptor Age	Exposure Point	Parameter Code	Parameter Definition	RME Value	Units	Rationale/ Reference	Intake Equation/ Model Name
Dermal	Recreational User	Child	Dermal Contact with Sediment while Wading	DA	Dermally Absorbed Dose per Event	chemical-specific	mg/cm^2-event	USEPA 2004	
				CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	Dermally Absorbed Dose (DAD) (mg/kg-day) = DA x EV x ED x EF x SA x 1/BW x 1/AT
				CF	Sediment Conversion Factor	0.000001	kg/mg		
				SA	Skin Surface Area	4,000	cm^2	USEPA 1997*	
				ABSd	Dermal Absorption Factor	chemical-specific	percent	USEPA 2004	DA = ABSd x CS x CF x AF
				EF	Exposure Frequency	52	days/year	Site-specific	
				AF	Adherence Factor	1	mg/cm^2	USEPA 2004	
				ED	Exposure Duration	10	years	Site-specific	
				EV	Event Frequency	1.00	event/day	USEPA 2004	
				BW	Body Weight	43	kg	USEPA 1997	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	
Ingestion	Recreational User	Child	Incidental Ingestion of Sediment while Wading	CS	Chemical Concentration	chemical-specific	mg/kg	Site-specific	
				IR-S	Ingestion Rate	100	mg/day	USEPA 1991	Chronic Daily Intake (CDI) (mg/gk-day) = CS x IR-S x EF x ED x 1/BW x 1/AT
				EF	Exposure Frequency	52	days/year	Site-specific	
				ED	Exposure Duration	10	years	Site-specific	
				BW	Body Weight	43	kg	USEPA 1997	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	

USEPA. 1989. Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual (Part A) Interim Final EPA/5401/1-89/002. Office of Emergency and Remedial Response. U.S. EPA. Washington DC.

USEPA 1991 Risk Assessment Guidance for Superfund, Volume I Human Health Evaluation Manual, Supplemental Guidance Standard Default Exposure Factors. OSWER Directive 9285 6-03 Office of Emergency and Remedial Response. U.S. EPA Washington, D.C.

USEPA 1997. Exposure Factors Handbook Volume 1 General Factors. Office of Research and Development EPA/600/P-95/002Fa. August 1997.

USEPA. 2004d. Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment) Final Office of Emergency and Remedial Response. EPA/540/R/99/005. OSWER 9285.7-02EP PB99-963312

a- Calculated from USEPA 1997 based on the a total body surface area to Individual body part ratio for hands, feet, lower arms, and legs of adults. Proportion of body surface is assumed to be the same for children ages 7-16.

TABLE 4.4.RME

VALUES USED FOR DAILY INTAKE CALCULATIONS

REASONABLE MAXIMUM EXPOSURE

Scenario Timeframe: Future Exposure
Medium: Surface Water
Exposure Medium: Surface Water

Exposure Route	Receptor Population	Receptor Age	Exposure Point	Parameter Code	Parameter Definition	RME Value	Units	Rationale/ Reference	Intake Equation Model Name
Dermal	Recreational User	Child	Dermal Contact while Wading	DA	Dermally Absorbed Dose per Event-Child	chemical-specific	mg/cm^2-event	USEPA 2004	Dermally Absorbed Dose (DAD) (mg/kg-day) =
				Cw	*Chemical Concentration*	chemical-specific	mg/L	Site-specific	DA x EV x ED x EF x SA x 1/BW x 1/AT
				CF	Water Conversion Factor	0.001	L/cm^3	USEPA 1991	
				SA	Skin Surface Area-Child	4,000	cm^2	USEPA 1997[a]	DA = Kp x CW x CF x t-event
				Kp	Permeability Constant	chemical-specific	cm/hour	USEPA 2004	
				EF	Exposure Frequency	52	days/year	Site-specific	
				EV	*Event Frequency*	1	event/day	USPEA 2004	
				ED	Exposure Duration - Child	10	years	Site-specific	
				t-event	Event Duration-Child Wading	2.00	hours/event	Site specific	
				BW	Body Weight - Child	43	kg	USEPA 1997	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	
Ingestion	Recreational User	Child	Incidental Ingestion while Wading	CW	Chemical Concentration	chemical-specific	mg/L	Site-specific	
				IR-W	Water Ingestion Rate-Child	0.05	L/day	USEPA 1989	Chronic Daily Intake (CDI) (mg/gk-day) =
				EF	Exposure Frequency	52	*days/year*	*Site-specific*	CW x IR-W x EF x ED x 1/BW 1/AT
				ED	Exposure Duration-Child	10	years	Site-specific	
				BW	Body Weight - Child	43	kg	USEPA 1997	
				ATC	Averaging Time-Cancer	25550	days	USEPA 1991	
				ATN	Averaging Time-Non-Cancer	3650	days	USEPA 1991	

USEPA. 1989. Risk Assessment Guidance for Superfund, Volume I Human Health Evaluation Manual (Part A). Interim Final EPA/5401/1-89/002. Office of Emergency and Remedial Response. U.S. EPA. Washington, DC

USEPA. 1991 Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Supplemental Guidance Standard Default *Exposure Factors. OSWER Directive 9285.6-03 Office of Emergency* and Remedial Response. U S EPA. Washington, D.C.

USEPA 1997. Exposure Factors Handbook. Volume 1. General Factors. Office of Research and Development. EPA/600/P-95/002Fa. August 1997.

USEPA. 2004d Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual. *(Part E, Supplemental Guidance for Dermal* Risk Assessment) Final Office of Emergency and Remedial Response. EPA/540/R/99/005. OSWER 9285 7-02EP PB99-963312.

a- Calculated from USEPA 1997 based on the a total body surface area to individual body part ratio for hands, feet, lower arms and legs of adults. Proportion of body surface is assumed to be the same for children ages 7-18.

Table 4-6 Exposure Parameters Used in the Adult Lead Model

Exposure Parameter	Definition	Parameter Value	Reason for Variable Selection	Reference
PbB- Fetal	Target fetal blood lead – no more than 5% should exceed	10 µg/dL	Recommended by USEPA	USEPA 1996b
IR	Soil ingestion rate			
	Construction Worker	100 mg/day	Recommended by USEPA	USEPA 2005e
	Recreational User, Adolescent	50 mg/day	Recommended by USEPA	USEPA 1997a
R fetal/maternal	Ratio of fetal to maternal blood lead	0.9	Recommended by USEPA	USEPA 1996b
PbB adult, 0	Background adult blood lead concentration			
	Adult Receptors	1.7 –2.2 µg/dL	USEPA Range	USEPA 1996b
	Adult (Construction Worker)	1.53 µg/dL	NHANES III Survey data	USEPA 2002
	Recreational User, Adolescent	1.53 µg/dL	NHANES III Survey data	USEPA 2002
BKSF	Biokinetic slope factor	0.4 µg/dL/µg/day	Recommended by USEPA	USEPA 1996b
GSD	Geometric standard deviation	1.8-2.1	USEPA recommended range	USEPA 1996b
	GSD Used in Assessment	2.18	NHANES III Survey data	USEPA 2002
EF	Exposure Frequency			
	Construction Worker	132 days/year	Site-specific	Professional Judgment
	Recreational User	27, 52 and 132 days/year	Site-specific	Professional Judgment
AT	Averaging time			
	Construction Worker	182 days/year	Recommended by USEPA	USEPA 1996b
	Recreational User	182 days/year	Site-specific	Professional Judgment
AF	*Absorption Fraction*	*0.12*	*Recommended by USEPA*	USEPA 1996b

µg/dL = micrograms per deciliter
mg/day = milligrams per day
µg/day = micrograms per day

Section 5 Toxicity Assessment

The purpose of a toxicity assessment is to review and summarize available information on the potential for each COPC to cause adverse effects in exposed individuals. Adverse effects include both carcinogenic and noncarcinogenic health effects in humans as well as animals. COPCs for the ALS include arsenic, iron, lead, manganese, and thallium.

For most adverse effects caused by chemicals, a positive relationship exists between dose (intake of a chemical through a particular exposure pathway, such as ingestion) and response. Generally, as dose increases, type and severity of adverse response also increases. Furthermore, time of onset of toxic responses often shortens.

A key facet of any toxicity assessment is the use of dose-response information to describe a quantitative relationship between human exposure and potential for adverse health effects. Quantitative toxicity criteria are generally numerical expressions developed by USEPA of the relationship between chronic average daily dose (exposure) and toxic response (adverse health effects). As described below, separate toxicity criteria are developed for assessment of carcinogenic and noncarcinogenic health effects.

The USEPA has developed a hierarchy for reviewing human health toxicity values. This hierarchy has three tiers: (1) USEPA's Integrated Risk Information System (IRIS); (2) USEPA's Provisional Peer Reviewed Toxicity Values; and (3)other toxicity criteria (e.g. toxicity criteria developed by California EPA). For this document, toxicity values were obtained following USEPA's hierarchy, beginning with IRIS. Since dermal toxicity criteria are not available, oral toxicity criteria were used to evaluate risks and hazards from dermal exposure. Differences in absorption between oral and dermal exposure were corrected using absorption estimates obtained from USEPA RAGS Part E guidance (2004f). No toxicity criteria have been developed for lead. Instead, risks associated with lead exposure are evaluated for residential receptors using USEPA's IEUBK model (version 1.0, Build 261) and USEPA's adult lead methodology (ALM) (USEPA 1996b). Section 5.3 discusses lead modeling.

The following sections briefly outline how toxicity criteria for carcinogens and noncarcinogens are developed and expressed, and summarize toxicity values for COPCs. The general basis for the development of toxicity values for carcinogens and noncarcinogens is presented in Section 5.1 and 5.2, respectively. Sections 5.1 and 5.2 also present toxicity criteria for COPCs. Toxicity profiles for arsenic, iron, lead, manganese, and thallium are included in Appendix C.

5.1 Chemical Carcinogens

5.1.1 Evidence of Carcinogenicity

USEPA has developed a classification system for carcinogens to characterize overall weight of evidence of carcinogenicity based on the availability of human, animal, and other supportive data. Three major factors are considered:

- The quality of evidence from human studies;
- The quality of evidence from animal studies; and,
- Other supportive data that are assessed to determine whether the overall weight of evidence should be modified.

The USEPA classification system for the characterization of the overall weight of carcinogenicity has the following five categories:

- **Carcinogenic to Humans (formerly Group A - Human Carcinogen).** This category indicates that there is sufficient evidence from epidemiological studies to support a causal association between an agent and cancer. This descriptor may also be used if there is a lesser weight of epidemiological evidence strengthened by other evidence.
- **Likely to be Carcinogenic to Humans (formerly Group B - Probable Human Carcinogen).** This category generally indicates that there is at least limited evidence from epidemiological studies of potential carcinogenicity to humans. However, the weight of evidence does not reach that required for "Carcinogenic to Humans".
- **Suggestive Evidence of Carcinogenic Potential (formerly Group C - Possible Human Carcinogen).** This category indicates that the potential for carcinogenicity to humans has been raised, but the weight of evidence is not strong enough for a more definitive conclusion.
- **Inadequate Information to Assess Carcinogenic Potential (formerly Group D - Not Classified).** This category indicates that the weight of evidence for carcinogenicity is not adequate to use one of the other descriptors described above.
- **Not Likely to be Carcinogenic to Humans (formerly Group E - Evidence of Noncarcinogenicity to Humans).** This category indicates that the weight of evidence is strong enough to declare a chemical not likely to be carcinogenic to humans.

5.1.2 Cancer Slope Factors

The USEPA IRIS Work Group has used a variety of specialized models to estimate the upper bound risk of carcinogens for numerous compounds. Data from animal or epidemiological studies are used to determine slope factors, which are expressed as

$(mg/kg\text{-}day)^{-1}$. The cancer slope factor (CSF) describes the increase in an individual's risk of developing cancer over a 70-year lifetime per unit of exposure where the unit of exposure is expressed as milligrams per kilogram per day (mg/kg-day).

CSFs are calculated using methods intended to be protective of human health, and are based on the assumption that cancer risks decrease linearly with decreasing dose. The 95 percent upper confidence limit estimate for the slope is used in most cases to compensate for animal to human extrapolation and other uncertainties. The resulting CSFs are considered to be upper bound estimates, which are unlikely to underestimate carcinogenic potential in humans.

When the upper-range CSF is multiplied by the lifetime average daily dose of a potential carcinogen, the product is an estimate of the upper-bound lifetime individual cancer risk associated with exposure at that dose. The calculated risk is an estimate of the increased likelihood of cancer resulting from exposure to a chemical. For example, if the product of the CSF and the average daily dose is 1×10^{-6}, the predicted upper-bound cancer risk for the exposed population is one million, or 0.0001 percent. This risk is in addition to any "background" risk of cancer not related to the chemical exposure.

Calculation of carcinogenic risk relies on data derived from human epidemiological studies or chronic animal bioassays. The likelihood that a chemical is a human carcinogen is a function of the following factors:

- The number of tissues affected by the chemical;
- The number of animal species, strains, sexes, and number of experiments and doses showing a carcinogenic response;
- The occurrence of clear-cut dose-response relationships as well as a high level of statistical significance of the increased tumor incidence in treated, compared to control groups;
- A dose-related decrease in time-to-tumor occurrence or time-to-death with tumor; and
- A dose-related increase in the proportion of tumors that are malignant.

The USEPA prefers that data of sufficient quality from epidemiologic studies are used for estimating risks. However, animal studies can be drawn upon and are typically conducted using relatively high doses in order to observe adverse effects. Because humans are expected to be exposed at lower doses, data are adjusted by using a mathematical model. Data from animal studies are fitted to an appropriate model to extrapolate the dose-response to lower doses. The low-dose slope of the dose-response curve is subjected to various adjustments (e.g., calculation of 95 percent upper confidence limit), and inter-species scaling factors may be applied to derive slope factors for humans. Dose-response data derived from human epidemiological studies are fitted to dose-time-response curves on an individual basis. These models provide conservative but plausible estimates of upper limits on lifetime risk. Although the actual risk is unlikely to be higher than the estimated risk, it could be considerably lower, and may even be zero.

Table 5-1 presents oral CSFs for the ALS COPCs.

5.2 Systemic Toxicants

Oral reference doses (RfDs) and reference concentrations for inhalation (RfCs) are toxicity values developed by USEPA for chemicals exhibiting noncarcinogenic effects. RfDs and RfCs are usually derived from no-observable-adverse-effect levels (NOAELs) taken either from human studies, often involving workplace exposures, or from animal studies and are adjusted downward using uncertainty or modifying factors. Uncertainty factors are generally applied to adjust for the possibility that humans are more sensitive than experimental animals and that there may be sensitive subpopulations (e.g., children, pregnant women, individuals with hay fever or asthma). In addition, modifying factors are applied to address uncertainties related to the database. For example, a modifying factor of 2 to 10 may be applied in instances where the database on a particular chemical lacks information on possible reproductive or developmental toxicity.

RfDs and RfCs are intended as estimates of the daily exposure to a COPC that would not cause adverse effects even if exposure occurred continuously over a lifetime. These values are presented in units of mg/kg-day for comparison with estimated chronic daily intake into the body. Intakes that are less than the RfD or RfC are not likely to cause adverse health effects. Chronic daily intakes that are greater than the RfD or RfC indicate a possibility for adverse effects. The quantitative relationship between the estimated chronic daily intake (dose) and the RfD (or RfCs) is termed the hazard index (HI).

Oral RfDs and RfCs for the ALS COPCs are presented in Tables 5-2 and 5-3.

5.3 Lead Modeling

USEPA has not published conventional quantitative toxicity criteria for lead because available data suggest a very low or possible no threshold for adverse effects, even at exposure levels that might be considered background. Any significant increase above such background exposures could represent a cause for some concern. In lieu of evaluating risk using typical intake calculations and toxicity criteria, USEPA has developed a computer model (the Integrated Exposure Uptake Biokinetic [IEUBK] model) for prediction of blood-lead levels in children exposed to lead from a variety of sources, including soil, dust, air, diet, lead-based paint, and maternal blood. Estimated blood-lead levels are compared to target blood-lead concentrations to assess possible risks. The model can be used to assess risks to individual children or populations of children. For a single child, risk is calculated as the probability that the child's blood-lead level will exceed the level of concern (10 micrograms pre deciliter [μg/dL]).

USEPA has also developed an Adult Lead Methodology that assesses lead exposure to the fetus of a pregnant woman. This methodology is used to predict blood lead concentrations in adults and in fetuses for exposure scenarios that do not involve residential exposure of young children. Therefore, this model is not directly applicable to the older child (ages 7 to 16 years) that is evaluated for intermittent lead exposure. However, this model should be conservative for this age group and is the only methodology that can be applied to older children.

Both the IEUBK and ALM approaches are discussed in detail in Section 4, Exposure Assessment.

Table 5-1 Cancer Toxicity Values for COPCs

Chemical of Potential Concern	Carcinogen	Oral Cancer Slope Factor	Oral to Dermal Adjustment Factor[1]	Adjusted Dermal Cancer Slope Factor[2]	Inhalation Slope factor	Units	Weight of Evidence/ Cancer Guideline Description	Source	Date (MM/DD/YY) (Date Checked)[3]
Arsenic	C	1.5	NA	NA	15.1	(mg/kg /day)-1	A	IRIS	7/15/2005
Iron	NA	NA	NA	NA	NA	(mg/kg /day)-1	NA	NA	NA
Lead	C	[4]	NA	[4]	[4]	(mg/kg /day)-1	B2	IRIS	7/15/2005
Manganese	NC	NA	NA	NA	NA	(mg/kg /day)-1	D	IRIS	7/15/2005
Thallium (as thallium chloride)	NA	NA	NA	NA	NA	(mg/kg /day)-1	NA	NA	NA

1 Oral to Dermal Adjustment Factor from Exhibit 4-1, RAGS Part E, Supplemental Guidance for Dermal Risk Assessment Final. EPA/540/R/99/005. July 2004.
2 Adjusted Dermal Cancer Slope Factor (1/mg/kg/day) = Oral Cancer Slope Factor (1/mg/kg/day) / Oral to Dermal Adjustment Factor.
3 Toxicity values were obtained from USEPA online toxicity database, IRIS, July 2005.
4 Lead was evaluated using the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children, Version 1.0
IRIS EPA online toxicity database, http://www.epa.gov/IRIS
NA = not available/ not applicable
NC = noncarcinogen
USEPA Weight of Evidence:
A – Human Carcinogen
B1 – Probable human carcinogen – indicates that limited human data are available
B2 - Probable human carcinogen – indicates sufficient evidence in animals.
C – Possible human carcinogen
D – Not classified as human carcinogen

Table 5-2 Non-Cancer Oral Toxicity Values for COPCs

Chemical of Potential Concern	Chronic/ Subchronic	Oral RfD Value	Units	Oral to Dermal Adjustment Factor[1]	Adjusted Dermal RfD[2]	Units	Primary Target Organ	Combined Uncertainty/ Modifying Factors	Sources of RfD	Date of RfD (MM/DD/YY) (Date Checked)[3]
Arsenic	Chronic	3.00E-04	mg/kg-day	NA	NA	mg/kg-day	Hyperpigmentation, Keratosis, and Vascular System	3/1	IRIS	7/15/05
Iron[5]	Chronic	3.00E-01	mg/kg-day	NA	NA	NA	GI tract	NA	NCEA	7/22/05
Lead	NA	[4]	NA	NA	[4]	NA	Central Nervous System, Developmental	NA	NA	NA
Manganese[6]	Chronic	2.40E-02	mg/kg-day	4%	9.6E-04	mg/kg-day	Central Nervous System	NA	USEPA Region 9	8/1/05
Thallium (as thallium chloride)	Chronic	7.0E-05	mg/kg-day	NA	NA	NA	GI Tract and Central Nervous System	3000/1	IRIS	7/15/05

1 Oral to Dermal Adjustment Factor from Exhibit 4-1, RAGS Part E, Supplemental Guidance for Dermal Risk Assessment. Final. EPA/540/R/99/005. July 2004.
2 Adjusted Dermal Reference Dose (mg/kg-day) = Oral Reference Dose (mg/kg-day) x Oral to Dermal Adjustment Factor.
3 Toxicity values were obtained from USEPA online toxicity database, *IRIS*, July 2005.
4 Lead was evaluated using the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children, Version 1.0.
5 The oral RfD for iron is an outdated value that may overestimate potential hazards. This RfD is further discussed in Section 7, Uncertainties.
6 The oral RfD for manganese used in this HHRA is the oral RfD from the current Region 9 PRG table; this value is more conservative than the oral RfD on IRIS of 1.4E-01 mg/kg-day.

Note: There are no non-cancer inhalation toxicity criteria available for the above COPCs on *IRIS* USEPA online toxicity database, http://www.epa.gov/IRIS
NCEA = National Center for Environmental Assessment
NA = not available/ not applicable

Table 5-3 Non-Cancer Inhalation Toxicity Values for COPCs

Chemical of Potential Concern	Inhalation Reference Dose	Units	Primary Target Organ	Combined Uncertainty/ Modifying Factors	Sources of RfC	Date of RfC (MM/DD/YY) (Date Checked)
Arsenic	NA	NA	NA	NA	NA	NA
Iron	NA	NA	NA	NA	NA	NA
Lead	NA	NA	NA	NA	NA	NA
Manganese	5E-05	mg/m^3-day	Central Nervous System	1E+03	IRIS	7/15/05
Thallium (as Thallium chloride)	NA	NA	NA	NA	NA	NA

IRIS USEPA online toxicity database, http://www.epa.gov/IRIS
NA = not available/ not applicable

Section 6
Risk Characterization

In this section, exposure assessments (Section 4) are integrated with results of the toxicity assessment (Section 5) to produce quantitative expressions of carcinogenic risk and noncarcinogenic hazards. These quantitative risk and hazard estimates are presented along with a qualitative analysis of their meaning for people living, working or recreating in the study area.

Potential health hazards due to exposure to lead were evaluated independently because toxicity criteria, such as cancer slope factors and reference dose, are not available for this contaminant. Instead of standard risk and/or hazard calculations, the IEUBK model was used to estimate potential lead exposures for young children living in the study area and the Adult Lead Methodology is used to evaluate exposures for older children recreating in the area and adult workers. Quantitative results from the IEUBK model and Adult Lead Methodology and their interpretation for people living, working or recreating in the study area are presented separately.

6.1 Overview of Risk Characterization

Health hazards associated with exposure to lead are assessed using exposure models developed by USEPA. These models, the IEUBK model for young children and the Adult Lead Methodology for adolescents and construction workers, estimate the probability that a child exposed to given concentrations of lead in site media will have a blood lead concentration exceeding 10 µg/dL. When this probability falls below a health protection goal of 5 percent, lead exposures are typically considered to be acceptable.

Carcinogenic risks are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. The upper-bound excess lifetime cancer risk is estimated by multiplying the lifetime exposure (Section 4) by the cancer slope factor (Section 5). Excess lifetime cancer risks are generally expressed in scientific notation and are probabilities. An excess lifetime cancer risk of 1×10^{-6} (one in one million), for example, represents the incremental probability that an individual will develop cancer as a result of exposure to a carcinogenic chemical over a 70-year lifetime under specified exposure conditions.

The potential for noncarcinogenic effects is evaluated by comparing an exposure level over a specified time period with a reference dose derived for a similar exposure period. This ratio of exposure to toxicity is referred to as a hazard quotient (HQ). A hazard index (HI) is the sum of the HQs from individual chemicals of potential concern. Where an HI is equal to or less or less than one, potential exposures are at or below a "safe" level as defined by USEPA reference doses. Where HI's are greater than one, exposure may be sufficient to imply a hazard to human health. However, this conclusion is generally reached only where such an HI is based on exposure to chemicals that affect the same target organ or system. Chemicals are assumed to have additive toxicity only when they display similar toxicity profiles at low levels of exposure.

To gain perspective on estimates of risks and hazards, EPA uses targets that help to define when remediation or mitigation may be warranted. Typically, cancer risks that do not exceed 1 in ten thousand are considered acceptable, but decisions on the need for remediation are made on a case-by-case basis. Cancer risks below 1 in one million are typically considered *de minimus*. In addition, protection of young children for health effects of lead exposure is considered achieved if the odds of a typical or hypothetical child (or group of similarly exposed children) with blood lead levels of 10 µg/dL or greater is no more than 5 percent (USEPA 1994b). The results of risk calculations are compared to these target values to aid in determining whether additional response action is necessary at the site.

Cancer risk and noncancer hazard calculations for COPCs are discussed in the following sections. Potential health risks associated with lead are discussed separately in Section 6.2. Cancer risks for other COPCs are presented in Section 6.3. Estimated noncancer health hazards are presented for each of the receptors in Section 6.4. Separate estimates are presented for each of the exposure scenarios, including:

- Future Residents
- Construction Workers
- Recreational Visitors

For the Annapolis Lead Mine, the basic approach to the characterizing risks and hazards is twofold. First, for the portion of the site north of Highway 49, the assessment focuses on residual risks associated with lead contamination left in place following a recently completed removal action. Second, for floodplain areas, particularly those areas south of Highway 49, the assessment addresses potential risks from existing contamination eroded from the mine site and deposited during flood events.

6.2 Estimates of Lead Exposure

The main concern for the ALS is potential exposure to lead in mine wastes generated and released at the site. Although other COPCs were identified in this and previous reports, the "risk driver" for the site appears to be lead.

Potential health risks due to exposure to lead were assessed using USEPA's IEUBK model for lead exposure of young children, ages 0 to 84 months of age. USEPA's Adult Lead Methodology was used to assess non-residential exposures to lead. Results of these analyses are discussed in the following sections.

6.2.1 Residential Lead Exposures

The approach to evaluating the mine operations area was to assume that future residential development might occur and that such development would bring contaminated materials to the surface where future residents might be exposed. Although, unlikely, such a scenario is not specifically excluded.

Residential blood lead levels were calculated using site-specific (e.g. soil lead concentrations) and default exposure assumptions. The approach used site-specific information where available to evaluate key inputs to the USEPA's IEUBK Model for estimating lead exposure in young children. If site-specific information was sufficient, default inputs to the model were replaced with ones more applicable to the site. Otherwise, default parameters provided with the model were retained.

The focus of the IEUBK Model for lead in children is the prediction of blood lead concentrations in young children exposed to lead from several sources and by several routes. The model utilizes four interrelated modules (exposure, uptake, biokinetic, and probability distribution) to estimate blood lead levels in children exposed to lead contaminated media. The IEUBK Model can be used to predict the probability that a child exposed to given set of concentrations of lead in environmental media will have blood lead concentrations exceeding a health protection goal of concern (typically 10 µg/dL). For this assessment, estimates for blood lead concentrations were calculated for the former mining operations area and for identified hotspot areas, using the IEUBK model. The model was run using a combination of default and site-specific parameters. For this assessment the only non-default site-specific parameters available were media concentrations. IEUBK modeling results are based on updated dietary uptake values, a GSD of 1.6, and an assumed soil to dust transfer factor of 0.70 for residential children from birth to seven years (84 months) of age.

Recent studies have indicated that some model default parameters may overestimate exposures to lead. Additional realizations of the model were evaluated to illustrate the range of possible blood lead levels in children exposed in identical exposure conditions using non-default model parameters. These additional analyses are discussed in Section 7, Uncertainties. IEUBK model results for all model variations are presented in Appendix F.

Most soils in the mining operations area of the site that were sampled during post removal activities have lead concentrations that are below levels of potential concern. Young children that might live or play in these areas would not be expected to have greater than a 5 percent chance of having their blood lead concentrations exceed 10 µg/dL; when this criterion is met, lead exposures are unlikely to represent a significant hazard. In fact for many areas the probability that a child's blood lead concentrations would exceed 10 µg/dL is less than 1 percent. Since children receive more exposure than adults in the same setting, and are more sensitive to the harmful effects of lead, lead concentrations at these locations will not represent a significant hazard for adults either.

For the identified hotspots in the former mining operations area, average lead concentrations could be high enough to represent a hazard to young children, if residential development where to occur. Two such hotspots where identified, one near the former Clark residence (Clark hotspot) and one located north of the

Mayberry residence (Mayberry hotspot). [NOTE: The Mayberry hotspot lies outside of the yard of this residence and does not imply any source of lead on the current Mayberry property.] In hotspot areas, lead exposures are predicted to be very high and lead concentrations in soil and dust could theoretically cause the majority of children living in these areas to have blood lead concentrations exceeding 10 μg/dL. A child living at and playing in a yard characterized by average hotspot lead concentrations could have much greater than a 5 percent chance of having blood lead levels that exceed 10 μg/dL.

This result assumes that the current 18 inch clean soil cover is disturbed such that residual lead contamination beneath the cover is brought to and remains at the surface. A child living in the area could then be exposed directly to contaminated soils. The probability that a child's blood lead level will exceed the level of concern (10 μg/dL) is 99% and 91% at the former Clark residence hot spot and at the hotspot north of the Mayberry residence, respectively, assuming that a child is exposed to the average lead concentrations in these areas.

Locations with average lead concentrations that could represent a hazard for young children occur near the former Clark residence, the former Mayberry residence, near Sutton Branch Creek southwest of the former Clark residence, and at the toe of the tailings cap. No modeling estimates are provided for the latter two hotspots because the spot southwest of the former Clark residence is in the floodplain and residential development is not anticipated and because the hotspot at the toe of the cap is anticipated to fall under the requirement of the State of Missouri to maintain the integrity of the waste depository.

Results of the IEUBK modeling are presented in Table 6-1.

Table 6-1 Summary of IEUBK Model Runs
Annapolis Lead Mine Site

Future Residential Scenario		
Exposure Area	**Exposure Point Concentration (mean of dataset) (mg/kg)**	**Probability of a Child (Birth to 84 Months in Age) Expected to have a Blood Lead Concentration above 10 μg/dL**
Former Mining Operations Area, Surface Soil	159	0.2%
Hot Spot Areas		
Former Clark Residence	6960	99.4%
North of Mayberry Residence	2640	89%
Note: Recommended (TRW) New Dietary Intakes were Used		

6.2.2 Construction Worker Lead Exposures

Future development of the mine operations area would require some excavation that would penetrate the 18" cover placed on the site and construction workers could be exposed to residual lead contamination beneath this cover. No exposure for construction workers is anticipated for floodplain areas because no development is expected within areas subject to periodic flooding.

USEPA's Adult Lead Methodology was used to assess lead exposures for adult workers in the former mining operations area. For a majority of the site, lead concentrations are below levels of potential concern; however, in areas identified as hotspots, lead concentrations could be of concern for future construction workers. For areas not identified as hotspots, the average PbB is estimated to be 2.1 µg/dL for a construction worker exposed to average lead residual lead concentrations and the probability of fetal blood lead levels above 10 µg/dL is 2 percent for these individuals.

Estimated PbB levels for a construction worker exposed to average lead concentrations, and the probability of fetal blood lead levels above 10 µg/dL for these individuals are substantially higher for excavation activities in hotspot areas. For the Clark hotspot, these estimates are 26 µg/dL and 86 percent, respectively. Analogous estimates for the Mayberry hotspot are 11 µg/dL and 48 percent. These estimates suggest that exposures in hotspot areas could be unacceptable for future construction workers. Estimates of lead exposures for the construction worker based on USEPA's Adult Lead Methodology are summarized on Table 6-2 and calculation worksheets are presented in Appendix F.

Table 6-2 Summary of Estimated Lead Exposures for the Construction Worker Based on USEPA Adult Lead Methodology

Receptor	Construction Worker		
Exposure Area	Exposure Point Concentration (mean) mg/kg)	Probability that fetal PbB will be greater than 10 µg/dl	PbB of adult worker (µg/dL)
Former Mining Operations Area, Surface Soil	159	2%	2.1
Hot Spot Areas, Former Mining Operations Area			
Former Clark Residence	6960	86%	25.8
North of Mayberry Residence	2640	48%	10.7

PbB = blood lead level

6.2.3 Recreational User Lead Exposures

Recreational visitors to the site may contact existing surface contamination in much of the floodplain for Sutton Branch Creek. No exposure to subsurface contamination is anticipated for the mine operations area or areas within the floodplain where the removal occurred. In the former mine operations area, a cover of 18" has been placed over residual contamination. Part of the removal action is to maintain and repair this cover until it is fully vegetated and stabilized. Thus, residual contamination in these areas is not expected to be brought to the surface where they might represent a source of exposure. All estimates of lead exposure to soil are for existing surface contamination in the floodplain south of Highway 49.

USEPA's Adult Lead Methodology was used to assess lead exposures for adolescents recreating in the floodplain area. There is significant uncertainty associated with the recreational scenario; therefore, a range of exposures were evaluated. Lead exposures were estimated for a range of exposure frequencies, from one, two and five visits per week. The central estimate PbB for a recreational adolescent exposed to average lead concentrations in the floodplain ranged from 2 µg/dL to 3 µg/dL. Estimates for potential effects on the fetus are provided in the Table 6-3. The age range that is evaluated in this assessment is not typically associated with child-bearing; however, pregnancy is possible toward the upper end of the age range. Based on USEPA's Adult Lead Methodology the probability of fetal blood levels exceeding 10 µg/dL would range from 1 to 5 percent, again based on an individual exposed to average soil lead concentrations in the area. Percentages are at or below the USEPA health protection goal of no more than 5 percent probability of exceeding 10 µg/dL.

Note that the estimate of a 5 percent chance of exceeding the health protection goal is based on an exposure frequency of 5 events per week. This frequency is an extremely high estimate for an area with poor access and low attractiveness. The estimate shows that even extreme exposure parameters do not result in significantly elevated estimates of lead exposure for recreational visitors.

Separate estimates for exposure to sediments were not developed for potential recreational exposures, since the EPC for lead in sediment (330 mg/kg) is lower than that for floodplain soils. Exposure to sediments alone or in combination with floodplain soils would not be expected to cause impacts greater than those for soils alone. Also, exposure to lead in surface water was not quantified since this pathway would be negligible in comparison to the soil ingestion pathway. Estimates of lead exposures for the adolescent recreating in the floodplain area based on USEPA's Adult Lead Methodology are summarized on Table 6-3 and calculation worksheets are presented in Appendix F.

Table 6-3 Summary of Estimated Lead Exposures for the Recreational Adolescent Based on USEPA Adult Lead Methodology

Receptor	Recreational Adolescent		
Exposure Area	Exposure Point Concentration (mean) mg/kg)	Probability that fetal PbB will be greater than 10 µg/dl	PbB of recreational adolescent (µg/dL)
Flood Plain Area	912.4		
Exposure Frequency			
1 visit per week		1.1%	1.8
2 visits per week		1.8%	2.2
5 visits per week		5.0%	3.1

PbB = blood lead level

6.3 Cancer Risks

Cancer risks at the site are due to exposure to arsenic; potential health risks due to exposure to arsenic in soil were assessed using standard USEPA exposure equations and a combination of site-specific and USEPA default exposure assumptions. Cancer risk estimates were assessed for future residents in the former mining operations area (groundwater exposure only) and for recreational users visiting the floodplain area or playing in Sutton Branch Creek. Results from cancer risk calculations are discussed below.

6.3.1 Cancer Risk for Future Residents

Cancer risks for future residents were estimated using assumptions for an RME. Thus, all risk estimates are expected to fall in the upper range of those possible. In some cases, as discussed below and in Section 7, Uncertainties, the estimates can be interpreted as upper bounds. All potential cancer risks at the site are associated with exposure to arsenic in groundwater used for domestic purposes. Post-removal analytical data for arsenic in soil are not available; therefore, potential exposures associated with soil conditions are not evaluated.

The maximum detected concentration of arsenic in groundwater was used as the exposure point concentration; this exposure point concentration may overestimate potential for exposure. Cancer risk estimates for residential exposure to groundwater may represent the upper bound for the site. Cancer risk for ingestion of groundwater is 2×10^{-5}. Cancer risk associated with dermal contact with arsenic during bathing is 1×10^{-7}. Total cancer risk associated with groundwater exposures is 2×10^{-5}. Cancer risks for future residents based on RME therefore fall within USEPA's acceptable risk range. Cancer risk for the resident is presented in Table 6-4. Cancer risk calculations for residents are presented in Appendix E.

6.3.2 Cancer Risks for Future Construction Workers

Construction workers are only assessed for exposure to residual contamination in soils in the mine operations area. Since post-removal data for arsenic are not available for this medium, no cancer risk estimates were developed for these receptors.

6.3.3 Cancer Risks for Recreational Users

Cancer risks for recreational users were estimated for ingestion of and dermal contact with surface soils, sediment, and surface water. There is significant uncertainty associated with the recreational scenario; therefore, a range of exposures were evaluated. Cancer risk associated with incidental ingestion and dermal contact of surface soil is 5×10^{-6}, with incidental ingestion and dermal contact of sediment is 4×10^{-6}; and with incidental ingestion and dermal contact of surface water is 5×10^{-7}, based on 2 visits per week during the warmest months of the year. Total cancer risks for the recreationist are therefore 1×10^{-5}. This estimate falls in the middle of USEPA's risk management range. These risk estimates may overstate actual potential risks for the site. The greatest potential risks are associated with incidental ingestion of soils and sediments. For estimation of risks, maximum detected concentrations were used as exposure point concentrations for sediment, which may result in overestimate potential risk associated with this medium. Since people recreating at the site are unlikely to consistently visit only the most contaminated areas and since sediment contamination is likely to change substantially over time, actual exposure point concentrations, and, hence, cancer risks, may be less than those calculated.

Cancer risk for the recreational user is presented in Table 6-4 at the end of this section. Cancer risk calculations for the recreational users are presented in Appendix E.

6.4 Noncancer Hazards

Assessment of noncancer hazards followed the same basic approach used for assessment of cancer risks. As previously discussed, HQs for individual COPCs are added together to produce an HI. When such an HI exceeds one, HIs are then recalculated separately by adding individual HQs for COPCs that affect the same target organ or system.

6.4.1 Noncancer Hazards for Future Residents

No non-cancer hazards are estimated for exposure to COPCs in soils for future residents because of the lack of post-removal concentrations of metals other than lead. Thus, the only hazard estimates are those for exposure to groundwater used for domestic purposes.

Maximum detected concentrations were used to estimate hazards associated with the groundwater exposure pathway. The use of these concentrations most likely overestimates potential exposures associated with groundwater. The HI associated with ingestion of groundwater is 11 for a child resident and 6 for an adult. The individual HIs for iron are 9 and 2 for iron and thallium, respectively for the child resident. The individual HQs should be emphasized because iron and thallium affect different target organs, and additive effects may not be expected. Similarly, the individual HQs for the adult resident are 5 and 1 respectively. All of these estimates exceed the target HQ of 1.

6.4.2 Noncancer Hazards for Construction Workers

Non-cancer hazards are not estimated for construction workers because of lack of post-removal data for metals other than lead in soils in the mine operations area. The only exposures that are quantified are those associated with use of groundwater for domestic purposes.

6.4.3 Noncancer Hazards for Recreational Users

The noncancer health hazard index for recreational users associated with incidental ingestion and dermal contact of surface soil in the floodplain area is 0.1 , which is based on 2 visits per week to the site during the warmest 6 months of the year. The HI associated with the ingestion of and dermal contact with sediment are all less than one. The HI associated with incidental ingestion of dermal contact with surface water is less than one. The majority of the HI is associated with ingestion of thallium in surface water. As discussed previously, maximum detected COPC concentrations were used as exposure point concentrations for surface water and sediment, and likely these concentrations may overestimate any actual exposures that may take place in the study area. Noncancer hazards for the recreational users are presented in Table 6-4. Calculations for noncancer health hazards for recreational users are presented in Appendix E.

Table 6-4 Summary of Cancer Risks and Non Cancer Hazards for Receptors for the ALS

Exposure Area	Exposure Scenario	Receptor	Cancer Risk Estimate (1)	Non Cancer Hazard Index
Former Mine Operations Area	Domestic Use of Groundwater (Ingestion and Dermal Contact during bathing/showering)	Future Resident, Adult, Cancer Risk	2×10^{-5} (2)	6
		Future Resident, Child	Not Calculated	11
	Ingestion of and Dermal Contact with Soil	Future Resident	Exposures associated with soil in the former mining area are evaluated for lead only in the IEUBK model	
	Ingestion of and Dermal Contact with Soil	Future Construction Worker	Exposures associated with soil in the former mining area are evaluated for lead only using USEPA's Adult Lead Methodology	
Sutton Branch Floodplain	Incidental Ingestion of and Dermal Contact with Surface Soil	Recreational Visitor	5×10^{-6}	0.1
Sutton Branch Creek	Incidental Ingestion of and Dermal Contact with Surface Water	Recreational Visitor	5×10^{-7}	0.3
	Incidental Ingestion of and Dermal Contact with Sediment	Recreational Visitor	4×10^{-6}	0.06

(1) Cancer risk for resident includes exposure for 6 years as a child and 24 years as an adult.

(2) Cadmium in groundwater could contribute to cancer risks about equally with arsenic (Appendix B). This estimate is based on an oral slope factor from California EPA that is not widely accepted and is subject to much uncertainty. If hypothetical risks due to oral exposure to cadmium were added to those associated with arsenic, resulting risks would still be within EPA's risk management range.

Section 7
Uncertainties

Uncertainties can arise from several sources in a HHRA including data collection and interpretation, assumptions used to characterize exposures, and toxicity values. To compensate for uncertainty surrounding input variables, conservative assumptions are often made that tend to overestimate rather than underestimate risk. In cases where data are limited, assumptions may be based on professional judgment or subjective estimates that may under- or overestimate risks.

7.1 Types of Uncertainty

Three primary sources of uncertainty include:

- Scenario uncertainty
- Parameter uncertainty
- Model uncertainty

Scenario uncertainty results from missing or incomplete information needed to fully define exposure and dose. This uncertainty may include errors or gaps in site characterization, professional judgment, assumptions regarding exposed populations, and steady-state conditions. Sources of parameter uncertainty include measurement and sampling errors, inherent variability in environmental and exposure-related parameters, and the use of generic surrogate data or default assumptions when site-specific data are not available. Parameter uncertainty often leads to model uncertainty. One source of modeling uncertainty is relationship errors, such as errors in correlations among chemical properties or limitations in mathematical expressions used to define environmental processes. Errors due to the use of mathematical or conceptual models as simplified representations of reality are also sources of modeling uncertainty.

Often analysis of uncertainties is divided into "true uncertainty" and "variability." The former is uncertainty due to lack of knowledge of data. Variability is uncertainty due to irresolvable variation in physical, chemical, and biological process, human behavioral patterns, seasonal changes, and data for site characterization. An example of uncertainty in this HHRA involves selection of an exposure frequency for recreational site users. Little site-specific information is available and this parameter is based on professional judgment. An example of variability in this HHRA involves estimates of exposure concentrations. These estimates are upper range estimates of mean concentrations based on variability in data used in the calculations.

These three types of uncertainty have been identified in each of the four parts of this risk assessment: data evaluation, exposure assessment, toxicity assessment and risk characterization. Uncertainty within each of these components is discussed below.

7.2 Uncertainties Associated with Data Evaluation

Uncertainty is present in the data before it is even evaluated for risk assessment. Such uncertainty includes potential sampling bias, errors in laboratory extraction and analysis, and the protocol employed to assess contaminants identified as nondetected. Where COPCs are reported above detection limits, a higher level of confidence is placed on the analytical results. Sampling errors and biases and assumptions for use of nondetect data are almost always more important from uncertainty considerations.

The impact of errors in laboratory analysis can be assessed to some extent by examining results from independent chemical analyses. An analytical XRF instrument was used to measure soil concentrations of metals and metalloids during the recent removal action in the former mine operational area. The instrument was apparently well calibrated for measuring lead that was the focus of the removal action in the former mine operational area. Confirmation samples for some sampling locations were sent to an independent analysis using standard Contract Laboratory Program (CLP) methods. The correlation in findings of these two analyses is extremely good, implying that both methods produced consistent results. However, the slope of the regression line is about 0.8, indicating that the XRF values were consistently lower by about 20 percent than the concentrations reported from the off-site laboratory (Figure 7-1). Laboratory analyses are generally regarded as more accurate than those from field instruments. Thus, the data on which the risk assessment are based may be biased low, but by no more than 20 percent. Such a bias would not have any substantive impact on the results and conclusions of the risk assessment.

Soil data used to assess risks for the mine operational area were collected in the latter half of 2004. Data were collected using a grid system to systematically measure lead concentrations throughout the site. This systematic data collection for grids measuring only about 50 feet on a side provide a very complete site characterization, and exposure point concentrations based on these data are likely to accurately represent the potential for exposure.

Soil samples were not sieved, however, as suggested by USEPA's Technical Review Group for Lead (USEPA 2005c) to obtain the soil fraction most likely to adhere to skin and be subsequently ingested by young children. Some enrichment of lead in small soil particle sizes has been seen in past investigations, although this is not a universal finding. Thus, lead concentrations measured using unsieved soil could somewhat under or overestimate lead concentrations in the soil fraction that is expected to contribute most to exposure. This uncertainty cannot be resolved with currently available information. However, even significant enrichment in small particles would not materially change the basic conclusions of the risk assessment. In most of the mine operations area, for example, lead concentrations average much less than the screening level of 400 mg/kg. An enrichment "error" is unlikely to be sufficiently large to cause average concentrations to exceed 400 mg/kg. Further, average concentrations in hotspot areas appear to be much higher than those that might be acceptable for surface soils in residential developments. Enrichment would only further support the conclusion of potential excess lead exposure if residual contamination below the current 18 inch cover was brought to the surface and made available for direct contact exposure.

In the floodplain area south of Highway 49, data were collected in what appears to be a stratified random manner. Although the sampling was not as dense for this area as in the north, the manner in which contaminants were deposited suggests that contamination should be spread relatively evenly over the area. Thus, fewer samples would be necessary to characterize contaminant distribution than would be the case if contamination was spottier as is often the case when dealing with mining source areas. Available data are likely to represent actual exposure concentrations for the floodplain area accurately. For both the mine operational and the floodplain areas, uncertainties are likely to be associated mainly with measured variability.

Finally, data for soil constituents other than lead are not available to characterize post-removal conditions in the former mine operations area. Lack of these data and any associated quantitative exposure assessment suggests that risks and hazards to hypothetical future residents and construction workers are likely to be underestimated. Available data suggest, however, that such underestimation may not greatly affect the results and conclusions of the risk assessment. Two COPCs, besides lead, were identified for soils in the ALS, arsenic and manganese.

Arsenic was detected in most soil samples at concentrations above its Region 9 residential soil PRG. However, because of very high toxicity of arsenic, most background concentrations of arsenic exceed the PRG. Overall, the concentrations of arsenic observed at the site are not greatly elevated. For example, the exposure concentration for arsenic in the floodplain soils was less than 40 mg/kg. In many areas of the country, background arsenic concentrations can be in the range of a few up to 15 or 20 mg/kg. In contrast, the highest concentrations of lead at the site may be in the range of 20,000 mg/kg, while background concentrations may be less than 100 mg/kg. The amount of arsenic in the galena ore found at the ALS was apparently relatively low. Further, the highest concentrations seem to occur in areas where higher concentrations of lead are also found. Figure 7-2 shows this correlation between arsenic and lead in soils for the floodplain area. Probably, risk management decisions based on potential for lead exposure will also address any risks associated with residual arsenic. However, data are not available to directly support this conclusion.

The manganese concentration at one location slightly exceeded its residential soil PRG. Since almost all concentrations of manganese are below screening levels for soils, the level of contamination associated with past mining activities does not appear to be substantial. The moderate levels of manganese observed in soils suggest little potential for substantial exposure or hazard. Lack of data for manganese is not likely to have any significant impact on conclusions of the risk assessment.

7.3 Uncertainties Associated with Exposure Parameters

The combination of exposure assumptions and exposure point concentrations used in the assessment is expected to provide conservative estimates for exposure of individuals living near the ALS. However, uncertainties and their potential impacts on use of risk results for risk management should be understood. The exposure assessment relies on assumptions for a variety of exposure parameters. Assumptions used are variously based on:

- Site-specific information
- USEPA guidance
- Professional judgment

Choices made for adult exposure parameters are within the ranges suggested by USEPA and should be conservative for assessing adult exposure.

7.3.1 Soil Ingestion Rate

Soil ingestion rates are particularly important for the IEUBK model. Soil ingestion rates have been assessed and reassessed multiple times. Some recent information suggests that default soil ingestion rates may somewhat overestimate the average daily ingestion rates for young children. Uncertainty in this parameter was addressed by estimating possible lead exposure using both default and updated soil ingestion rates. These estimates bracket the range of plausible exposures based on available soil ingestion information.

Alternate soil and dust intake estimates provided by USEPA Region 8 (e-mail from Wendy O'Brien June 1, 2005 based on soil ingestion estimates from a study conducted in Anaconda, MT) are summarized below:

Age Range (years)	Soil and Dust Intake (g/d)
0-1	0.024
1-2	0.038
2-3	0.038
3-4	0.038
4-5	0.028
5-6	0.026
6-7	0.024

g/d = grams per day

Alternative estimates for possible lead exposure are provided in Table 7-1. This table also addresses uncertainties in soil-to-dust transfer coefficient, as discussed in Section 7.4.

Table 7-1 Summary of IEUBK Model Runs

Results for Child 0 to 84 months in age
Note: New Dietary Intakes were Used
Results presented are the probability in percent that a child exposed to soil at the EPC would have a blood lead concentration above 10 µg/dL

Exposure Area	Exposure Point Concentration (mean of dataset) mg/kg)	Soil to Dust Transfer Factor of 70%		Soil to Dust Transfer Factor of 24%	
		Alternate Soil and Dust Intake (1)	Default Soil and Dust Intake	Alternate Soil and Dust Intake (1)	Default Soil and Dust Intake
Former Mining Operations Area, Surface Soil	159.4	0.001%	0.2%	0.001%	0.04%
Hot Spot Areas					
Former Clark Residence	6959.7	79%	99.4%	61%	98%
North of Mayberry Residence	2639.7	25%	89%	10%	77%
(1) EPA 1996. Baseline Human Health Risk Assessment. Anaconda Smelter NPL Site. Anaconda, Montana.					

7.3.2 Exposure Frequency

Frequency and duration of exposure are also important determinants of exposure that are characterized by USEPA default values. No site-specific information on frequency of exposure or residence times is available for the ALS.

Exposure frequency for residents (groundwater ingestion) is estimated at the high end of possible frequencies, allowing only for a 2-week-per-year vacation. Most individuals may spend more time than this away from home and/or may spend limited time at home on most weekdays because of work commitments. These individuals may receive less exposure than that estimated in this assessment. However, a significant number of individuals (for example non-working parents) may spend significant amounts of time each day at their homes. The exposure frequency used in this assessment is not expected to be appropriate for most individuals in the potentially exposed population. However, the exposure frequency is reasonable for the most heavily exposed individuals and will be protective for the population as a whole.

Exposure frequency for construction workers and recreational visitors are very uncertain and are based completely on professional judgment. It is not possible to predict beforehand how long excavation activities would last during construction, and no data are available to estimate how frequently people might visit the site recreationally. Some quantitative sense of the range of lead exposure associated with recreational use of the floodplain was gained by comparing scenarios with a range of plausible exposure frequencies. Even an exposure frequency up to 5 days per week, an extremely high estimate given the lack of accessibility and attractiveness of the site for recreation still produced estimates of exposure that were below levels of potential concerns. It seems highly unlikely that recreational exposure could lead to unacceptable lead exposures under any foreseeable circumstances.

7.3.3 Exposure Duration

Exposure duration can also have a significant impact on exposure estimates. National norms suggest that the 90th percentile for time at one residence is about 30 years. If the population near the ALS is either more sedentary or more mobile than the nation as a whole, risks could be either under- or overestimated. In many cases, small rural communities have many residents that stay in the community for long periods of time, and some information suggests that this may be true for Annapolis.

Uncertainties in exposure duration are, however, unlikely to be of great significance in evaluation of residential exposure. For example, if a more reasonable upper range estimate for time at one residence was either 20 or 40 years, RME estimates would go down or up by only 33 percent.

Exposure duration for construction workers and recreational visitors are subject to significant uncertainties much like those for exposure frequencies. Exposure durations used in this assessment could either under- or overestimate potential risks and hazards for these receptors.

7.3.4 Evaluation of Inhalation Exposure

The risk assessment did not include estimates of risk and hazard due to exposure via inhalation of COPCs in the quantitative analysis. Since this pathway is at least potentially complete currently and/or in the future, this approach may lead to some underestimation of potential risks and hazards. However, such underestimation is probably negligible in terms of its impact on the conclusions of the risk assessment.

A screening level calculation using generic exposure parameters for inhalation exposure suggests that risks and hazards due to inhalation of arsenic will be a small fraction of those associated with incidental soil ingestion and dermal contact of arsenic (Table 7-2). The potential small contribution of the inhalation pathway would not change the reported risks or hazards, which are presented with only 1 or 2 significant figures. The screening calculation supports the decision to exclude the inhalation pathway from consideration in the quantitative assessment. Note also that the calculation used arsenic as an example COPC. The inhalation slope factor for arsenic is 10 times higher than the oral slope factor. Thus, the relative contribution of the inhalation pathway to overall cancer risk should be larger than that for many chemicals for which inhalation and oral toxicity criteria are similar. Even so, the inhalation pathway contributes only about 0.5 percent to total cancer risk from arsenic in the example calculation.

Table 7-2 Potential relative contribution of inhalation exposure to potential cancer risks for residents.

Exposure Parameters	Cancer Risk
Soil Ingestion Rate – 100 mg/d	Soil Ingestion 7.0×10^{-6}
Inhalation Rate – 20 m^3/day	
Particle Emission Factor - 1.32E-09 m^3/kg	Inhalation 3.6×10^{-8}
Exposure Frequency – 350 days/year	
Exposure Duration – 24 years	
Body Weight – 70 kg	

Averaging Time – 25550 days	
Slope Factor (oral) – 1.5 per mg/kg-d	
Slope Factor (inhalation) – 15 per mg/kg-d	

Calculations assume an arsenic concentration in soil of 10 mg/kg.

7.3.5 Evaluation of Dermal Exposure to Sediment

The risk assessment assumed that hands, forearms, lower legs and feet would be exposed to sediments in Sutton Branch Creek during recreational activities. This assumption may be a significant overestimate, since water play would likely wash off sediment from much exposed skin area fairly rapidly. Probably, the actual area of exposed skin would be much smaller for children wading in the creek. Risk and hazard estimates due to dermal exposure may be over 20 percent of total risks and hazards, and overestimation of exposed skin surface could result in some overestimation of total risk or hazard. A contribution of 20 percent is, however, relatively small and would be unlikely to alter the conclusions of the risk assessment.

Note: Uncertainties in the approach to dermal exposure applies only to arsenic in sediments. No other COPCs were evaluated for dermal exposure.

7.4 Uncertainties Associated with the Soil-to-Dust Transfer Factor (IEUBK Model)

Transfer of COPCs in soils to indoor dust is an important process for estimation of exposure. The default soil-to-dust transfer factor in the IEUBK model is 0.7. To estimate potential exposures for the ALS, the soil-to-dust transfer default factor of 0.7 was used. In recent studies of soil-to-dust transfer in Butte and Anaconda, Montana, measured dust concentrations of lead and arsenic, respectively, have been 24 and 43 percent of outdoor soil concentrations (USEPA 1994, 1996a). The estimate from Butte, Montana is based on data from homes in a community that was very dusty with large amounts of uncovered and unvegetated mine wastes when the data were collected. The estimate from Anaconda, Montana is based on data from a community where most of the arsenic released was in the form of very small particulate matter from a smelter. The ALS does not have large amounts of uncovered or unvegetated mine wastes and no smelting of ores was conducted in the town in the past. Thus, these two conditions that would seem to favor transfer of outdoor contamination indoors may not have been as important in determining soil-to-dust transfer for the ALS. Further, other estimates of soil-to-dust transfer are also much less than the default of 0.7. At another milling/smelting site in Utah, arsenic soil-to-dust transfer was estimated to be about 20 percent at Winchester Estates near the Midvale NPL site (CDM 2002). Even though fine particulates were released at this site during smelting operations, soil-to-dust transfer was still low.

Overall, available data from mining/milling/smelting sites appear to indicate that soil-to-dust transfer may be less than the default of 0.7. For this risk assessment, the range of soil-to-dust transfer from 0.24 to 0.7 was used to bracket the plausible range of transfer coefficients. Likely, even the estimates based on the lower value can be considered within the reasonable range for risk management decisions.

A comparison of lead exposure estimates for young children using default and alternative inputs to the IEUBK model are summarized in Table 7-1. Generally, the alternative inputs would have little impact on conclusions of the risk assessment. Concentrations of lead in hotspot areas are high and would represent unacceptable exposures regardless of choice of input parameters. Else where in the mine operations area, lead concentrations are below levels of concerns, and choice of input parameters would, again, not change this finding.

7.5 Uncertainties Associated with Uptake of COPCs into Garden Vegetables

Potential exposures and risks due to consumption of home-grown produce raised in contaminated soil were not quantitatively characterized. Potentially, this approach could lead to underestimation of possible risks and hazards. The decision not to evaluate exposure via garden vegetables is based on a study report of uptake of arsenic and lead into garden vegetables at the Kennecott mine site in Utah (EPA 1995). In this study, the uptake of arsenic and lead was low. Slopes of regression lines for vegetable concentrations versus soil concentrations ranged from a low of 0.000089 for lead uptake into zucchini to 0.0068 for lead uptake into beet greens. Regression slopes were less than those suggested in Baes et al (1984) who present generic uptake values for arsenic, lead and other metals. For some commonly grown vegetables (tomatoes, zucchini, leafy vegetables), results from Kennecott were 10 to 100 times lower than the Baes estimates. Results suggest that uptake into root crops may be greatest and that uptake into fruits (tomatoes and zucchini for example) is extremely limited. For example, arsenic could not be detected, using methods with detection limits in the 0.1 μg/mg range, in tomatoes.

Further, correlation coefficients for regressions in the study were generally low, suggesting a poor correlation between constituents in soil and those in vegetables. Soil concentrations may be overall poor predictors of concentrations in home grown vegetables. Vegetable data were not collected at the ALS site, making any attempt at quantification of the pathway very uncertain.

In addition, some fraction of arsenic taken up into vegetables is converted to less toxic organic forms (ATSDR 2000). The fraction of arsenic that is converted reduces exposure to the more toxic inorganic forms and therefore reduces potential risks at the site. Given that arsenic concentrations at the site are moderate (the EPC for As in flood plain soils is 34 mg/kg), that uptake is poor especially into some of the most popular types of vegetables, and that some fraction of arsenic taken up into plants will be detoxified, the potential for significant exposure and risk due to consumption of contaminated vegetables appear to be small.

Finally, gardens soils are typically amended, often on a yearly basis, with top soil, manure, etc., which would serve to dilute concentrations of arsenic and lead. These organic amendments might also reduce bioavailability of COPCs through binding to acid and sulfur groups. Further, continued harvest of crops from the same plot would reduce COPC concentrations over time. Overall, the garden vegetable consumption pathway would seem to be of minor concern for the ALS site.

7.6 Uncertainties Associated with Toxicity Assessment

7.6.1 Uncertainties in Cancer and Noncancer Toxicity Criteria

A potentially large source of uncertainty is inherent in the derivation of the USEPA toxicity criteria (i.e., RfDs and cancer slope factors). In many cases, data must be extrapolated from animals to sensitive humans by the application of uncertainty factors to an estimated NOAEL or LOAEL for noncancer effects. While designed to be protective, it is likely in many cases that uncertainty factors overestimate the magnitude of differences that may exist between human and animals, and among humans.

In some cases, however, toxicity criteria may be based on studies that did not detect the most sensitive adverse effects. For example, many past studies have not measured possible toxic effects on the immune system. Moreover, some chemicals may cause subtle effects not easily recognized in animal studies. The effects of lead on cognitive function and behavior at very low levels of exposure serve as examples.

In addition, derivation of cancer slope factors often involves linear extrapolation of effects at high doses to potential effects at lower doses commonly seen in environmental exposure settings. Currently, it is not known whether linear extrapolation is appropriate. Probably, the shape of the dose response curve for carcinogenesis varies with different chemicals and mechanisms of action. It is not possible at this time, however, to describe such differences in quantitative terms.

It is likely that the assumption of linearity is conservative and yields slope factors that are unlikely to lead to underestimation of risks. Yet, for specific chemicals, current methodology could cause slope factors, and, hence, risks to be underestimated.

Use of USEPA toxicity criteria could either over- or underestimate potential risks, but it is difficult to determine either the direction or magnitude of any errors. In general, however, it is likely that the criteria err on the side of protectiveness for most chemicals.

The RfD for iron is particularly uncertain. This criterion is an outdated provisional value from 1993 that does not reflect the latest research conducted by the Institute of Medicine (IOM), which published its revised iron dietary intake in 2001. The IOM report specified an upper tolerable intake for iron in children as 40 mg/per day for infants and children up to 13 years. This equates to a dose of 2.7 mg/kg-day for a 15 kg child and an even higher value for infants. Based on this IOM analysis, the risk assessment may be overestimating the HQ for iron by about 10-fold. A ten-fold reduction in hazard quotients for iron would mean that potentially significant impacts associated with ingestion of groundwater would be reduce to levels below the target

HQ of 1. Thus, newer information suggests that hazards identified in the risk assessment in Section 6, may be in error.

7.6.2 Bioavailability of Arsenic and Lead

Bioavailability of arsenic and lead are important issues for accurate assessment of possible risks and hazard associated with these common mine-site contaminants. Bioavailability of both arsenic and lead have been shown to be much lower than default estimates used in this risk assessment at several mine sites. For example, measured bioavailability of arsenic has ranged from less than 10 percent to perhaps 50 percent for several mine and smelter wastes tested in juvenile swine (Henningsen et al. 1999). However, extrapolation among mining sites is difficult, because results of bioavailability assays have varied over a wide range and are apparently affected significantly by such factors as ore type(s), soil geochemistry, and milling and smelting operations. Detailed information that would be necessary to estimate bioavailability of arsenic and lead in mine wastes at the ALS are not available, and no site-specific estimate of bioavailability of either contaminant can be made at this time. Risk and hazard estimates for arsenic and lead may be overestimated as a result of this data gap. Bioavailability of either arsenic or lead could be revisited if better support is needed in the future for risk management decisions.

7.7 Uncertainties Associated with Risk Characterization

Risk assessment guidance (USEPA 1989) stresses the importance of considering uncertainties in interpreting and applying results of any risk assessment. Assumptions are made using professional judgment and the scientific literature on site-specific risk assessments. In general, assumptions made throughout this risk assessment are conservative in that they would tend to overestimate exposure and resultant risk rather than underestimate it. In some instances, a range of plausible exposure estimates for lead were included in the risk assessment to better reflect the range of reasonable exposure estimates based on most recent information and recognized uncertainties.

A key uncertainty in the risk assessment is the assumption that parts of the site may be developed for residential exposure in the future. Although theoretically possible, the former mine operational area would not seem to be a particularly attractive site for building and it seems unlikely that such development would completely reverse the current cover and bring residual contamination, without dilution, to the surface. Part of the site, the repository of mine wastes, must be maintained in perpetuity by the State of Missouri. Other parts of the site could be developed, but such development would likely involve excavation of limited areas, and would not necessarily result in spreading of contaminated subsurface soils over all of the soils in the immediate yard of any residence. Thus, the assumption of residential exposure to currently covered residual contamination is likely a very conservative assumption for evaluating the removal area.

The risk characterization is also uncertain because of some basic assumptions made concerning the removal action recently completed for the site. Specifically, the assessment concluded that development on the repository in the former mine operational area would be excluded in perpetuity because of the requirement that the State of Missouri maintain the integrity of the cap on this repository. Furthermore, the assessment assumed that repair and maintenance of the 18" clean fill cover over the site would continue until the site was completely revegetated and the soil cover was stable.

While both of these assumptions concerning the removal action seem reasonable, one cannot be completely certain that the repository and soil cover will always remain uncompromised. For future residents and construction workers, this uncertainty applies only to the repository, since the assessment assumed that residential development could be possible on the rest of the mine operational area. However, no exposure was assumed for recreational visitors to any subsurface source of residual mine-related contamination. Since integrity of the cover material may be the least certain of the two assumptions regarding the removal action, uncertainties are greater for recreational visitors.

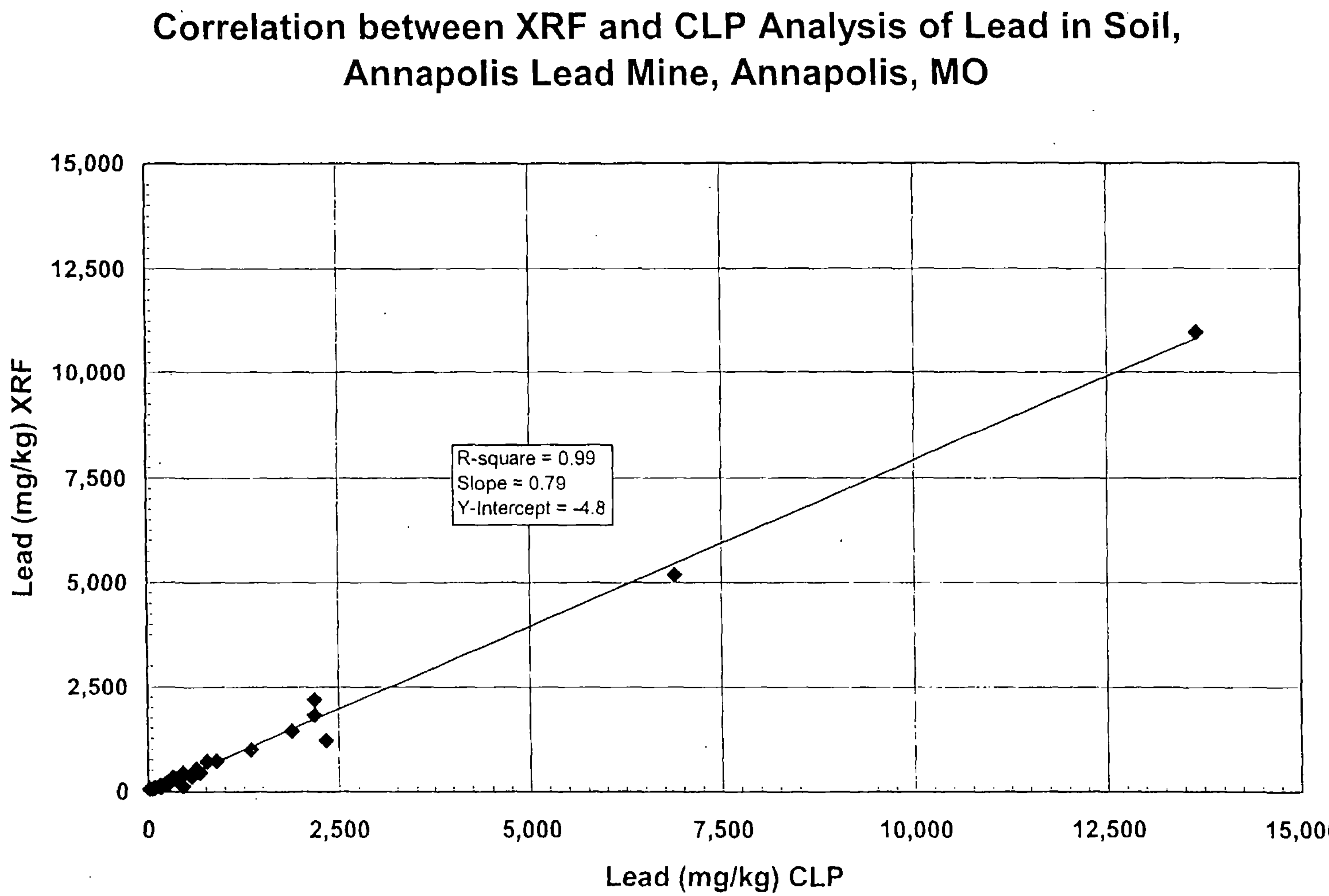


Figure 7-1: Correlation between XRF and CLP Analysis of Lead in Soil.

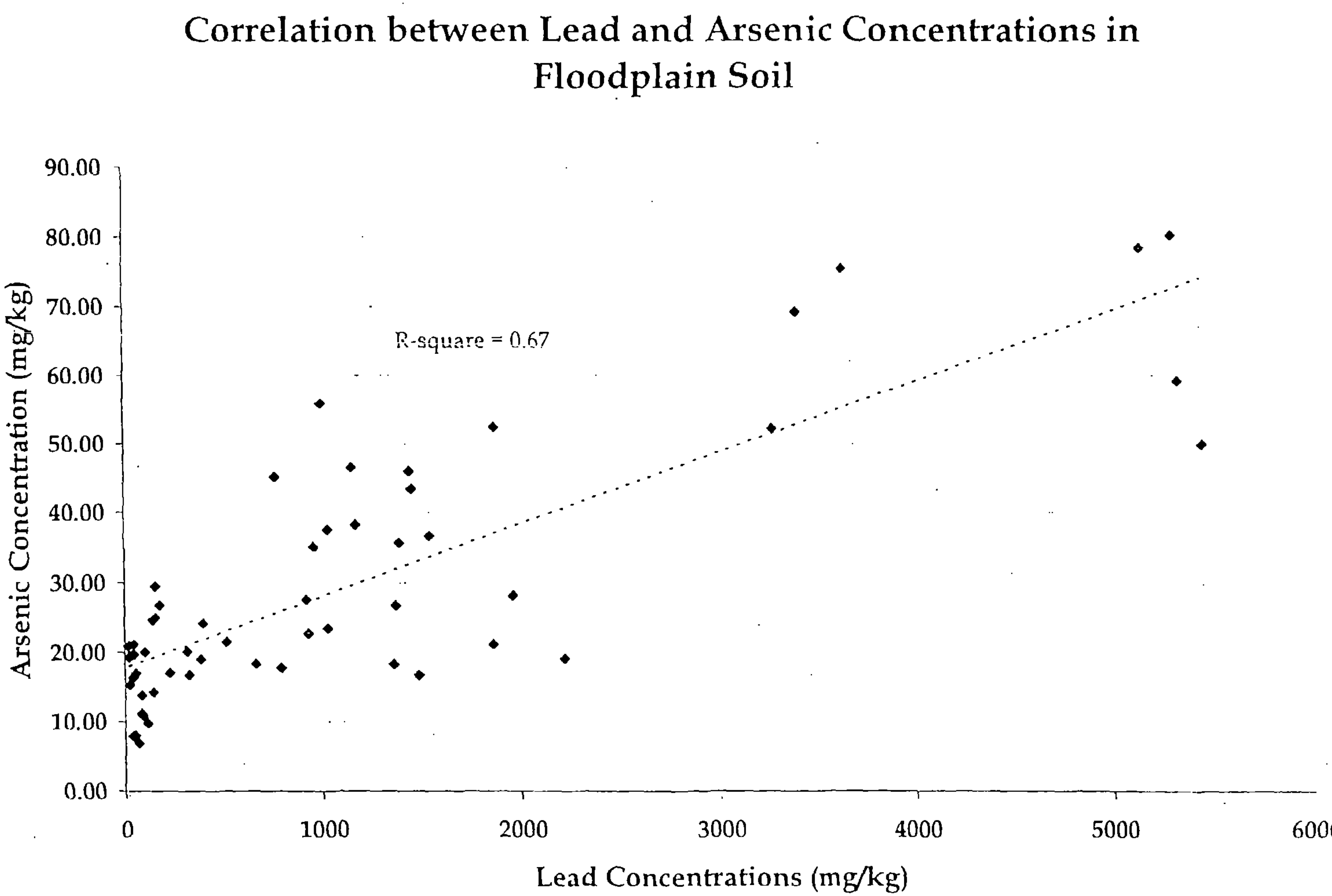


Figure 7-2: Correlation between Lead and Arsenic Concentrations in Floodplain Soil.

Section 8
Summary and Conclusions

The HHRA for the Annapolis Lead Mine used standard USEPA guidance along with both default and site-specific information to assess potential health risks for people living, working, or recreating in the area. The assessment focused on evaluating potential exposure to lead and other mine-related materials under existing conditions. For the former mine operations area, this focus requires estimation of potential risks and hazards following a recently completed removal action. In this area, the risk assessment provides information on residual risk and hazard posed by contamination left in subsurface soils after the removal. In the floodplain of the Sutton Branch, no removal activities have taken place. The risk assessment for this area provides a basic baseline risk assessment that can be used to help assess the need for remediation. Throughout the assessment, evaluation of mine operations area and the floodplain is kept separate to aid risk managers in understanding the different risk scenarios and focus employed.

The approach, results, and uncertainties of the risk assessment are summarized below, followed by a listing of conclusions supported by these results.

8.1 Summary of HHRA Approach

A HHRA was conducted for the site based on basic USEPA guidance (USEPA 1989), supplemented with more recent guidance and policy as appropriate. Site characterization data collected in recent field investigations and during the removal action in the former mine operations area were used in this HHRA to evaluate the possible exposure concentrations for residual contamination in the mine operations area and existing contamination in the Sutton Branch floodplain. Exposure concentrations help define risks and hazards due to exposure to metals and arsenic detected in site media (soil, surface water, sediment, and groundwater). Assumptions, methods, and results are summarized below.

Potentially Exposed Populations. Risks and hazards for three potential receptor groups were evaluated in the HHRA including current and future residents, current and future recreationists, and future construction workers. Future residents and construction workers were used to assess residual risks in areas of the mine operations area that are above the floodplain. Current and future recreationists were used to assess potential risks and hazards associated with existing contamination in the floodplain of the Sutton Branch.

Media of Concern and Exposure Pathways. Based on site data, media of concern are soil, indoor dust, air, vegetation, fish, groundwater, surface water, and sediment. Only a subset of these media was, however, assessed quantitatively in the risk assessment. Current and future residents and future construction workers were evaluated for direct contact with surface or subsurface soils and indoor dust (incidental ingestion and dermal contact (arsenic only)). Current and future residents were also evaluated for ingestion and dermal contact with potable water derived from shallow groundwater beneath the mine operations area. Recreationists were evaluated for incidental ingestion of surface soil, surface water and sediment and for dermal contact with soil, sediment, and surface water.

RME exposures were evaluated for the above receptors for all COPCs except lead. USEPA guidance generally defines RME as an exposure well above the average, but within the range of those possible. Estimates of central tendency exposures (CTE) were not included in the assessment. Risk and hazard estimates based on CTE were not thought to be essential because virtually all unacceptable exposures for the site were lead. This metal is evaluated using alternative methodology and the concepts of RME and CTE do not apply. Exposures to other COPCs, evaluated using RME estimates, were at or below levels of concern. CTE estimates would be less and therefore would not be highly informative.

Chemicals of Potential Concern. COPCs were selected for the ALS using comparisons of maximum concentrations of metals and arsenic detected in soil with residential PRGs developed by USEPA Region 9. Constituents with maximum concentrations above their PRGs were selected as COPCs. COPCs for surface soil included arsenic, lead, and manganese. COPCs selected for subsurface soil are the same as those selected for surface soil. COPCs selected for sediment include arsenic and lead. Surface water COPCs selected were arsenic, lead, manganese, and thallium. Groundwater COPCs selected were arsenic, iron, lead, and thallium. These COPCs are likely to represent all mining-related contaminants at the site that could be of concern for human health.

Evaluation of Exposure to Lead. Lead exposure is not assessed using standard risk assessment methods. Instead, exposures in residential settings are typically evaluated using USEPA's IEUBK model. Other exposure scenarios can be assessed using the Adult Lead Methodology, also developed by USEPA. For the ALS, the IEUBK model was used to assess risks to hypothetical future residents at the mine operations area, under the assumption that future residential development would transfer subsurface residual contamination to the surface. The Adult Lead Methodology was used to characterize potential lead exposures for construction workers involved in residential development in the mine operations area and to characterize potential lead exposures for recreational visitors to floodplain areas of the Sutton Branch.

8.2 Summary of HHRA Results

Quantitative risk and hazard estimates were developed for residents, construction workers and recreational users.

Residential Lead Exposures. The IEUBK model was used to assess lead exposures for young children. Lead exposures for future residential children were assessed for exposure to soil in the former mine site. Hot spots in this area were evaluated separately. To illustrate the range for possible impacts to blood lead levels both default and alternative values for key parameters in the model were assessed in the uncertainties section.

Most soils in the former mining operations area of the site that were sampled during post-removal activities have lead concentrations that are below levels of potential concern. A young child that might live or play in these areas and be exposed to average soil and dust concentrations would not be expected to have greater than a 5 percent chance of having their blood lead concentrations exceed the health protection goals of 10 µg/dL; when this criterion is met, lead exposures are unlikely to represent a significant hazard. This conclusion would apply to the current Mayberry residence and any residences outside of the floodplain south of Hwy 49. Lead concentrations in yards of these residences are below the screening value of 400 mg/kg.

For identified hotspots in the former mining operations area that were sampled, average lead concentrations could be high enough to represent a hazard to young children. In hotspot areas, lead exposures are predicted to be very high and lead concentrations in soil and dust could theoretically cause a young child exposed to average soil and dust concentrations in these areas to have a high probability of having a blood lead concentration exceeding the health protection goal of 10 µg/dL. Such exposures would only occur if residual lead contamination that exists below the 18 inch clean cover were to be brought to and left on the surface after residential development at the site. Currently, no exposure pathways exist for residual lead beneath the clean soil cover.

Nonresidential Lead Exposures. USEPA's Adult Lead Methodology (ALM) (USEPA 1996b) was used to assess intermittent or variable exposures to lead at the site by recreational users (older children) and construction workers. The current and future recreational user was evaluated based on an older child/adolescent scenario (7 to 16 years) for exposures to lead in soil while recreating in the Sutton Branch floodplain. Lead exposure for these individuals is not expected to cause more than a 5 percent chance of blood lead concentrations in a fetus exceeding 10 µg/dL for recreational visitors exposed to average floodplain soil or sediment concentrations.

Lead exposures for a construction worker were assessed for exposure to soil in the former mine operations area. Hot spots were evaluated separately. The predicted blood lead level for a construction worker exposed to average concentrations of lead in soil outside of hotspot areas was 2.1 µg/dL; the probability that fetal blood lead concentrations would exceed the health protection goal of 10 µg/dL was 2 percent for such an individual. This finding suggests that hazards associated with lead exposure are not expected for construction workers in most areas of the former mine site; however, only surface soil data were available to evaluate potential exposures after the removal action occurred. Construction workers would most likely be exposed to contamination in subsurface soil during construction activities and lack of subsurface data for some areas of the site where no removal took place may cause underestimation of potential exposure for future workers in the former mine area. Potential exposure was also evaluated for construction workers working in two hot spot areas; near the former Clark residence and north of the Mayberry residence. Exposures at these areas are above levels of concern for both the worker and the fetus. The central estimate worker blood lead levels could range from 10.7 µg/dL to 25.8 µg/dL for individuals exposed to average lead concentrations in soil. Probabilities that fetal blood lead concentrations would exceed USEPA's health protection goal of 10 µg/dL are 48 to 86 percent for a fetus of a construction worker exposed to average concentrations of lead in soils in these hotspots.

Total Carcinogenic Risks for Residents. Cancer risks at the site are due to exposure to arsenic; potential health risks due to exposure to arsenic were assessed using standard USEPA exposure equations and a combination of site-specific and USEPA default exposure assumptions. Post removal analytical data for arsenic in soil are not available; therefore, potential exposures associated with soil are not evaluated. Cancer risk for the groundwater exposure pathway was 2×10^{-5}; this estimate is based on ingestion of arsenic in groundwater and dermal contact with arsenic during bathing or showering. The groundwater dataset was small and maximum concentrations were used to estimate cancer risks. Cancer risks for future residents based on RME fall within USEPA's acceptable risk range. The lack of soil data for COPCs other than lead may underestimate risk for the future residential scenario.

Total Carcinogenic Risks for Workers. Cancer risks could not be estimated for the construction workers because data for soil constituents other than lead are not available to characterize post-removal conditions for the mine operations area. This data gap could result in some underestimation of risk for the site.

Total Carcinogenic Risks for Recreational Users. Cancer risk for recreational visitors to the floodplain (1×10^{-5}) falls within the USEPA's risk management range of 1×10^{-6} to 1×10^{-4}.

Noncancer Hazards for Residents. Noncancer hazards were estimated using standard USEPA exposure equations and a combination of site-specific and USEPA default exposure assumptions. Post removal analytical data for arsenic in soil are not available; therefore, potential exposures associated with soil are not evaluated. HI's for the groundwater exposure pathway were greater than one for both adults and children. These HI's, 6 and 11 for adults and children, respectively, are due mainly (80%) to potential exposure to iron. As discussed in Section 7, Uncertainties, the RfD for iron is outdated and subject to considerable uncertainty. Recent information suggests that the RfD could be too conservative by an order of magnitude. If the RfD that was used in the assessment was replaced by one ten times higher, HQs for iron would fall at or below the target of 1. Given available evidence, HQs for iron seem likely to fall into the acceptable range.

The HQ associated with ingestion of thallium in groundwater was also greater than one for the child (HI of 2). Maximum analytical results from three groundwater wells in the area were used to evaluate residential exposures to groundwater; thallium concentration was elevated in one of the residential drinking water wells sampled. Maximum concentrations used as exposure point concentrations may overestimate potential hazards.

Noncancer Hazards for Workers. Noncancer hazards could not be estimated for the construction workers because data for soil constituents other than lead are not available to characterize post-removal conditions for the mine operations area. This data gap could result in some underestimation of hazards for the site.

Noncancer Hazards for Recreational Users. Noncancer health hazards for all COPCs and pathways for recreational users were less than one, even when exposure frequency was assumed to be 5 days per week. This finding suggests that adverse noncancer health effects for recreational users at the site are not expected.

8.3 Conclusions

Based on the results of field investigations and the HHRA, the following conclusions are appropriate concerning human health risks and hazards associated with mine wastes in the ALS. Note that all risk and hazard estimates for COPCs other than lead are based on RME.

- Residual contamination in the mine operations area is generally below levels of concern for lead. However, hotspots exist in limited areas that could be associated with unacceptable exposures to lead. Unacceptable exposure could be realized for both future construction workers and future residents.

- Lead exposures for the mine operations area would be realized only if residual contamination beneath the 18" soil cover is excavated into or is brought to the surface in a residential yard. In the absence of development of the area, no complete exposure pathways for residual soil contamination would exist, provided that the cover remains intact.

- Lead is the only COPC that was assessed for the mine operations area; post-removal data was not available for other constituents. Some information suggests that higher concentrations of arsenic co-exist with elevated concentrations of lead. Thus, appropriate management of lead exposures is also expected to address risks due to exposure to arsenic. However, data are insufficient to demonstrate that this conclusion holds for all portions of the site. The other COPC, manganese, is present at concentrations above its screening criterion in only one sample in the floodplain. Manganese contamination does not appear to be sufficiently high to cause significant health impacts.

- Lead exposures for recreational visitors to the floodplain are not expected to reach unacceptable levels. These exposure estimates are based on frequent visits to an area that appears to be unattractive for recreation. The floodplain areas are heavily vegetated which may limit exposure to contaminated soils. The conclusion that risks and hazards for recreational visitors fall below levels of concern can be accepted with confidence.

- Lead exposures due to recreational contact with surface water and sediment in Sutton Branch Creek appear to be too low to cause unacceptable risk.

- Cancer risk due to exposure to arsenic in groundwater falls within USEPA's risk management range. However, hazards due to exposure to iron and thallium do fall above the target hazard of one and may imply some potential for unacceptable noncancer hazards. Groundwater risk and hazard estimates are based on maximum detected concentrations in a limited data set. Additional data would have to be collected to determine if these estimates are accurate and widespread in shallow groundwater. Further, the high HQ for iron is based on an outdated RfD and newer information suggests that the HQ could be overestimated by a factor of 10. If so, the major source of potential noncancer hazard would be eliminated.

- Cancer risks and noncancer hazards for recreational exposures in the floodplain and creek fall within the risk management range for cancer risk and below one for hazards. These results suggest that recreational exposure to COPCs other than lead may be in an acceptable range.

APPENDIX D

BASELINE ECOLOGICAL RISK ASSESSMENT AND ECOLOGICAL RISK ADDENDUM

ANNAPOLIS LEAD MINE

OU-1

1.0 INTRODUCTION

This document is a streamlined Ecological Risk Assessment (ERA) for the Operable Unit of the Annapolis Lead Mine National Priority List Site. A streamlined ERA differs from a baseline ERA in that the screening level steps 1 & 2 below are unnecessary. Historical data collections documenting the contaminants of concern at the Annapolis Lead Mine Site supplied enough information needed to move into a baseline risk assessment. This site is of potential ecological concern because of the piles of waste materials from past mining operations, which had previously occurred in the area during the early 1900s. The mine waste has been shown to contain high levels of lead, cadmium, nickel, arsenic and zinc. Of special concern are the mine waste piles eroding into Sutton Branch creek, a smaller tributary to the larger Big Creek.

1.1 PURPOSE OF THE ECOLOGICAL RISK ASSESSMENT

The purpose of an ERA at a Superfund Site is to describe the likelihood, nature and severity of adverse effects which environmental chemical contamination may be having on the ecosystems present at the site. Risk managers use this information, along with other relevant information, to make decisions on whether or not remedial activities are needed to protect the environment. If remediation is necessary, then a Feasibility Study (FS) is performed. The FS evaluates a range of alternative remedial actions that may meet risk management goals at the site.

Specifically, the purpose of an ERA at the Annapolis Lead Mine (ALM) Site is to:

- Meet the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the National Contingency Plan (NCP).
- Determine if there is future need for risk management decisions and a Feasibility Study (FS) for nonresidential areas of the ALM.

1.2 ECOLOGICAL RISK ASSESSMENT PROCEDURE

The United States Environmental Protection Agency (EPA) has standard guidance for the performance of ecological risk assessments at Superfund Sites (EPA 1997). The process consists of the following eight steps:

1. Screening level problem formulation and effects evaluation
2. Screening level exposure and risk evaluation
3. Baseline risk assessment problem formulation
4. Study design and data quality objectives
5. Field verification of sampling design
6. Site investigation
7. Risk characterization
8. Risk management

It is important to realize that the eight steps listed above are not intended to represent a linear sequence of mandatory tasks. Rather, some tasks may proceed in parallel and some tasks may be judged to be unnecessary at this site.

1.3 SCOPE OF THIS RISK ASSESSMENT

The scope of an ERA can vary widely from site to site, depending on the nature and extent of contamination at the site, and on the importance and value of potentially threatened ecological systems. As noted above, current EPA guidance recommends that the ecological risk assessment process begin with a screening level evaluation in order to determine whether there is a need for a full baseline ecological risk assessment and, if so, to define the proper scope of the site investigation and risk assessment (EPA 1997). At this site, there is a need for a detailed baseline ecological risk assessment. This conclusion is based on the following considerations:

- EPA analytical data documents metal contamination (arsenic, cadmium, lead, zinc) in sediments and surface water above background concentrations. The mine waste tailings pile is the main source.

- There is obvious and substantial contamination of Sutton Branch Creek and the adjacent floodplain with visible and buried tailings. Sutton Branch Creek is a small tributary to Big Creek but contains a diverse number of organisms including an endemic crayfish named the Big Creek crayfish (*Oronectes peruncus*). Big Creek is a perennial flowing waterbody and a Missouri Outstanding Resource Water that is ecologically and recreationally important.

- Tailings are known to contain a variety of different metals that are potentially toxic to a wide variety of different environmental receptors.

- At some locations, evidence of terrestrial phytotoxicity is readily detectable by simple visual inspection (i.e. nothing is growing on the mine waste).

- In June, 1992, there was a massive "fish-kill" event in Big Creek that occurred after the rupture of a dam (in Annapolis) constructed of rhyolite tailings (Schmitt 1997). This information suggests that fish may succumb to pulses of mine waste, indicating that flood events of eroding mine waste are potentially hazardous to the health of fish.

Based on these observations, it is concluded that a detailed scientific analysis of available data at the site is required in order to estimate the magnitude of the ecological risks to both the aquatic and the terrestrial environment from tailings and other mining-related contamination within the Annapolis Lead Mine Site.

2.0 SITE DESCRIPTION

2.1 SITE HISTORY

The Annapolis Lead Mine (ALM) Site is in the location of a former lead (Pb) mine which reportedly operated during the 1920s to the 1940s. The mining activities included the excavation of ore bodies, the crushing and concentrating of the ore and the storage of the concentrated metals prior to off site shipment for smelting. The wastes from crushing and concentrating were disposed of on the surface of the property within a small ravine. It is believed that an estimated 1,173,000 tons of tailings have been disposed of on 10 acres of the site. Through the years the waste has eroded off the 10-acre pile down into the adjacent floodplain.

2.2 PREVIOUS INVESTIGATIONS

Surface water sampling in Sutton Branch Creek indicated the presence of lead in excess of chronic and acute Ambient Water Quality Criteria (AQWC). A chronic lead concentration of 2.5 µg/L is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable fate (EPA 2002). An acute lead concentration of 65 µg/L, is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect (EPA 2002).

In 1992, Missouri Department of Natural Resources (MDNR) collected Sutton Branch surface water and sediment samples downstream of the PPE and lead was detected at (93 µg/L) and 4800 (mg/kg), respectively (Table 1).

A Screening Site Inspection (SSI) investigation was completed in June 1996 (Sverdrup). Results of Sverdrup's sediment sampling indicated the presence of arsenic, cadmium, cobalt, copper, lead, nickel, silver, thallium, and zinc. Lead was detected in Sutton Branch Creek sediments approximately 2300 feet upstream of the Probable Point of Entry (PPE) (13 mg/kg), and approximately 2000 feet downstream of the PPE (3970 mg/kg) (Table 1).

Sverdrup also screened soils around the former mine areas with a field portable X-ray fluorescence spectrometer (XRF) and had concentrations of lead as high as 28,300 mg/kg (Sverdrup 1996) (Table 1).

The EPA tasked Ecology and Environment (E & E), Superfund Technical Assessment and Response Team (START) to conduct an Expanded Site Inspection (ESI) and Removal Assessment (RA) at the ALM. Samples were taken in November 1997 and January 1998. START found lead in surface water samples taken from Sutton Branch Creek as high as 17.4 (µg/L) and lead sediment concentrations as high as 2,900 (mg/kg) (E & E 1999) (Table 1).

EPA's ecological risk assessor and on scene coordinator investigated the site in October 2003 and continued to find high lead levels in the sediment and water column of both Sutton Branch and Big Creek (Table 1a).

Table 1. Selected historical heavy metal results from site investigations of soil, sediment and water column sampling at Annapolis Lead Mine Site, Annapolis, Missouri (MDNR 1992, Sverdrup 1996, E & E 1999). Results in bold are at or above background /threshold /criteria values.

Sample Description	Arsenic	Cadmium	Copper	Lead	Nickel	Zinc
Soil Samples (XRF)	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**
Mine waste pile [b] Soil sample	**113**	**9.54**	**138**	**28300**	**56**	**676**
Soil near Clark residence [b] (Figure 3)	**53.9**	**10.9**	**266**	**27500**	**45.5**	**776**
Background concentrations [b]	0.95	1.77	16.8	300	14.6	93.3
Sediment Samples	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**	**(mg/kg)**
Sutton Branch Sediment sample Upstream of PPE [b]	1.86	1.04	5.5	13	4.32	15.3
Sutton Branch Sediment sample Downstream of PPE [b]	**68.3**	**3.41**	**62.6**	**3970**	**48.6**	**170**
Sutton Branch Sediment sample Downstream of PPE [C]	**88**	**2**	**62**	**2900**	**72**	**170**
Sutton Branch Sediment sample Downstream of PPE [a]	**140**	**3.5**	**102**	**4800**	**150**	**240**
Big Creek Sediment sample at Sutton Branch confluence [a]	**150**	**3.6**	**100**	**4400**	**180**	**250**
McDonald et al 2000 TEC Values	9.79	0.99	31.6	35.8	22.7	121
Water Column	(μg/L)	(μg/L)	(μg/L)	(μg/L)	(μg/L)	(μg/L)
Sutton Branch Creek Surface water samples at confluence to Big Creek [a]	<5	<2	<5	**47**	<50	14
Sutton Branch Creek Surface water samples downstream of PPE [a]	<5	<2	<5	**93**	<50	<11

Sutton Branch Creek Surface water samples [c] downstream of PPE	<16	<1	1.59	**17.4**	N/A	5.53
EPA's Ecotox AWQC* for surface water	190	0.25	11	2.5	160	100

[a] MDNR Samples [b] Sverdrup Samples [c] E & E (START) < = actual value of the sample is less than the reported value PPE-- Probable Point of Entry (where mine waste is visibly entering the creek) *AWQC—Ambient Water Quality Criteria, chronic, dependent upon water hardness (100 mg/L as $CaCO_3$) Average $CaCO_3$ hardness in Sutton Branch Creek is 72.2, average hardness in Big Creek is 87.7 (EPA 2004)

Table 1a. EPA preliminary heavy metal analysis results from site investigations of sediment and water column sampling Annapolis Lead Mine Site, Annapolis, Missouri October 28, 2003. Results in bold are at or above Ecotox values.

Sample Description Sediment	Arsenic (mg/kg)	Cadmium (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	pH
Sutton Branch upstream of PPE (≅ 1000 ft)	<5	<1	9.94	N/A	13.8	7.15
Sutton Branch downstream of PPE (≅ 1000 ft)	<5	<1	**2600**	N/A	**144**	7.80
Big Creek above Sutton Branch confluence (≅ 1000 feet)	6.81	**1.15**	13.9	N/A	18.3	7.80
Big Creek at confluence with Sutton Branch	<5	<1	**387**	N/A	43.4	7.34
McDonald et al. 2000	9.79	0.99	35.8	22.7	121	---
Water Column	(μg/L)	(μg/L)	(μg/L)	(μg/L)	(μg/L)	pH
Sutton Branch upstream of PPE	<25	<25	<25	N/A	<25	7.15
Sutton Branch downstream of PPE	<3	<3	**75.5**	N/A	<25	7.80
Big Creek above Sutton Branch confluence (≅ 1000 feet)	<25	<3	**51.3**	N/A	<50	7.80

Big Creek at confluence with Sutton Branch	<25	<3	<50	N/A	<25	7.34
EPA's Ecotox AWQC* for surface water	*190*	*0.25*	*2.5*	*160*	*100*	---

< = actual value of the sample is less than the reported value
*AWQC=Ambient Water Quality Criteria, chronic, water hardness as $CaCO_3$, 100mg/L
Average $CaCO_3$ hardness in Sutton Branch Creek is 72.2, average hardness in Big Creek is 87.7 (EPA 2004)

Two U.S. Fish and Wildlife Service studies conducted on aquatic life in Big Creek have shown evidence of heavy metal contamination in fish species. Both studies involved the enzyme δ-aminolevulinic acid dehydratase (ALA-D). ALA-D activity is highly sensitive to lead and is used as a biomarker for lead exposure in humans, waterfowl, and fish (Schmitt *et al.* 1993, 1997).

2.3 SITE LOCATION

The Annapolis Lead Mine (ALM) site is located in Iron County approximately one mile east of Annapolis in southeastern Missouri (Figure 1). The geographic coordinates are 37^0 21' 40" north latitude and 90^0 40' 30" west longitudes.

2.4 SITE DESCRIPTION

The ALM site property boundary is roughly rectangular in shape with the width portion located in Big Creek's floodplain south of the pile. The northeast and northwest areas of the site are bordered by wooded uplands. The entire area is approximately 50 acres of which approximately 10 acres is unvegetated mine waste.

The mine waste pile is composed of grey colored material that is mostly fine-grained (grain size ranging from 0.004-0.06 mm up to 2-4 mm) and slippery when wet. This material is highly erodable, resulting in steep-sided features and an outwash area spreading westward toward Sutton Branch Creek, and then Sutton Branch carries the material south into Big Creek (Figure 2).

Sutton Branch Creek is on the West Side of county road No. 138. Sutton Branch flows south approximately 3500 feet before becoming a losing stream (during summer months). Hampton Creek, also a losing stream, joins Sutton Branch Creek just before its confluence with Big Creek (Figure 2). Hampton Creek is also impacted by mine waste because it is located in Big Creek's floodplain; therefore it receives waste during flood events. START sampled Hampton Branch Creek, in November 1997, and the highest lead concentration in the water column was <1 μg/L and lead in sediment was 32 mg/kg. One arsenic level in Hampton Creek's sediment was 8.4 mg/kg which exceeds EPA's

Ecotox Threshold (ET) Effects Range Low (ERL) (EPA 1996)[1] value of 8.2 for all chemical forms of arsenic, but not the Threshold Effect Concentration (TEC) which is 159.79 mg/kg.

The pile is approximately 350 feet east of Sutton Branch Creek. The mine waste enters Sutton Branch Creek during storm events via a natural ravine and drainage ditch that is a tributary to Sutton Branch Creek. Mine waste dominates the substrate of Sutton Branch Creek after the probable point of entry (PPE) enters Sutton Branch. The waste has been eroding into Sutton Branch and Big Creek for 60-80 years. Consequently, Big Creek's floodplain has mine waste as deep as six feet in some areas. Also, during dry conditions the waste can be blown distances by wind and contaminate the surroundings including Sutton Branch Creek. Localized karst features such as springs and caves are present within four miles of the site.
Relatively hot summers and moderately cold winters characterize the climate in Iron County. Rainfall, which constitutes the majority of the annual precipitation, is well distributed throughout the year. Snow falls almost every winter, but snow cover usually lasts just a few days. Total annual precipitation is 44 inches. The prevailing wind is southerly.

Land Use

Approximately 276 permanent residents live within a 1.5 mile radius of the site. The total population within a 4-mile radius is estimated at 1338. The nearest school is 1.25 miles west of the site. To the north is an old city dump potentially containing more mine waste. The land surrounding the site is wooded to partially wooded and timber harvesting has been done in the past. There is limited agriculture production in and around the site (Figure 3).

Several old mine building ruins are present on the site (Figure 3). In addition, there are concrete mining buildings present that were used as dwellings for two families. There are no fences or gates associated with the property. It is known that residents have used the mine waste for several purposes including selling the waste to concrete companies, road crews and even as fill or surface covering around the playgrounds at the Annapolis school yard (Sverdrup 1996).

Northwest of the site is an artesian well that Annapolis residents have used for drinking water. The well has been tested and contains elevated levels of heavy metals and the residents have been informed that they should not drink the water from this well (Sverdrup 1996).

According to local residents, Sutton Branch Creek contains minnows and crayfish, which may be eaten by the local residents and the local wildlife. Big Creek, classified by

[1] The ET (ERL) is used when a Sediment Quality Criteria (SQC) or a Sediment Quality Benchmark (SQB) has not been calculated for a chemical. The ERL value represents the lower 10^{th}-percentile concentration associated with observation of biological effects (EPA 1996).

MDNR as an Outstanding State Resource Water[2], is used primarily for swimming and recreational fishing especially downstream of Annapolis. Commonly caught fish include small and large mouth bass, green and bluegill sunfish, crappie, walleye, and catfish.

The confluence of Sutton Branch Creek into Big Creek is a known palustrine, deciduous broad-leafed forested temporarily flooded wetland (USFWS 1992). There are several other wetland areas along Big Creek downstream of Sutton Branch confluence. The wetlands are sensitive ecological systems and the contamination may be impacting these environments, especially after flood events. State-listed rare species include the southern brook lamprey, the Big Creek crayfish, and the State-watched silverjaw minnow, which may occur in Big Creek.
Big Creek flows through Sam A. Baker State Park, which is located approximately 15-miles downstream from the site. Sam A. Baker State Park houses one of Missouri's largest wilderness preserves, the Mudlick Mountain Natural Area.

2.5 ENVIRONMENTAL SETTING

The ALM contains a number of important habitats and a wide variety of ecological receptors. These habitats and receptors are summarized below.

2.5.1 Aquatic Plant Communities

Periphyton (attached algae-such as on rocks), in all reaches, are important aquatic primary producers that may sustain both aquatic and terrestrial species.
Periphyton covers the rocks in the headwaters of Big Creek above the Glover smelting plant. Algae are non-existent throughout Sutton Branch and into Big Creek Aquatic emergent plants are also non-existent in Sutton Branch Creek. The water willow (*Justicia americana)*, usually abundant in Ozark streams, was not found in Sutton Branch or at the confluence with Big Creek.

2.5.2 Terrestrial Riparian Plant Communities

There were few trees on the banks of Sutton Branch Creek at the PPE where the mine waste was visible. Horsetail (*Equisetum arvense*) was the only abundant plant growing in mine tailings along Sutton Branch Creek's riparian zone. There were several species of trees, shrubs, and herbaceous plants both upstream and downstream of the PPE.

Table 2. The following table is a list of plants observed at the ALM in April 2004:

Common Name	**Scientific Name**
Trees and shrubs	
Cottonwood	*Populus deltoides*
Sycamore	*Platinus occidentalis*
Boxelders	*Acer negundo*
Red Cedar	*Juniperus virginiana*
Dogwood	*Cornus drummondii*

[2] Outstanding State Resource Water—For these waters no degradation of water quality is allowed. That means whatever heavy metal concentrations occur naturally in Big Creek (background levels) then that is the water quality standard for that waterbody.

Oak	*Quercus sp.*
Slippery Elm	*Ulmus rubra*
Ohio Buckeye	*Aesculus glabra*
Cherry	*Prunus sp.*
Redbud	*Cercis canadensis*
Willow	*Salix sp.*
Shortleaf Pine	*Pinus echinata*
Sumac	*Rhus copallina*
Herbaceous plants	
Mint	Labiatae
Violets	*Viola sp.*
Wake robin	*Trillium sp*
Pinks	Caryophyllaceae
Spring beauty	*Claytonia virginica*
Goldenrod	*Solidago sp.*
Horsetail	*Equisetum arvense*
Grasses	Graminae
Poison Ivy	*Toxicodendron radicans*
Virginia Creeper	*Parthenocissus quinquefolia*
Dogbane	Apocynaceae
Moss	Bryopsida
Meadow Rue	*Thalictrim minus*
May Apple	*Podophyllum peltatum*
Mullein	*Verbascum thapsus*
Strawberry	Rosaceae

2.5.3 Wetlands

According to the USFWS (2004), palustrine and riverine wetlands are located along Big Creek above and below the confluence of Sutton Branch Creek including downstream towards Sam A Baker State Park. (Figure 4).

2.5.4 Amphibians, Reptiles and Aquatic Fauna

Amphibians and reptiles can be found throughout Sutton Branch and Big Creek. Frogs, snakes, turtles, skinks and an abundance of fence lizards were observed at the ALM. There was a beaver dam and observed beaver activity in and along Sutton Branch Creek. Salamanders were discovered under logs in the woods. Waterfowl were not observed during our sampling event at the ALM and there were no observed stonerollers (*Campostoma anomalum*) which are a typical Ozark stream fish. Darters were observed in both waterbodies and the state watched Big Creek crayfish *(Orconectus peruncus)* was also found in Sutton Branch Creek.

A qualitative survey sample of aquatic macroinvertebrates was taken by EPA in October 2003. Aquatic macroinvertebrates were sampled in riffles using a 500 ③m D-net. Two samples were taken in Sutton Branch Creek (above and below the PPE) and two in Big Creek (above and below Sutton Branch confluence) (Figure 5). The following tables list the results of the qualitative aquatic macroinvertebrate survey:

Table 2a. Percent composition of selected invertebrate groups, ALM site, October 2003.

Station	Ephemeroptera	Plecoptera	Trichoptera	Coleoptera	Chironomidae
SBC ~1000 ft above PPE	35.8	6.0	13.4	13.4	3.0
SBC ~1000 ft below PPE	16.5	10.2	21.2	9.4	23.6
BC ~1000 ft above SBC Confluence	30.6	3.3	43.4	12.4	8.3
BC ~500 ft below SBC confluence	32.4	11.0	30.4	14.6	8.4

SBC-Sutton Branch Creek, PPE--Probable Point of Entry, BC—Big Creek

Table 2b. Qualitative aquatic macroinvertebrate survey taxonomic list, ALM, October, 2003. SBC=Sutton Branch Creek, BC=Big Creek, PPE= Probable Point of Entry.

Taxonomic List	SBC above PPE	SBC below PPE	BC above SBC confluence	BC below SBC confluence
Acari			1	
Amphipoda	3			
Coleoptera				
Larvae Elmidae	1	1		38
Larvae Elmidae *Stenelmis*			25	5
Adult Elmidae *Stenelmis*			3	2
Psephenidae *Psephenus*	8	11	1	
Psephenidae *Acneus*			1	
Decapoda				
Orconectes peruncus	4			
Diptera				
Chironomidae	2	30	20	26
Simuliidae *Simulium*	2	2		
Tipuliidae *Antocha*	2	15	4	4
Ephemeroptera				
Baetidae *Baetis*	7	6	10	13
Heptageniidae *Stenonema*	15	15	43	59
Isonychidae *Isonychia*			19	27

Siphloneuridae			2	1
Caenidae *Caenis*	1			
Ephemerellidae				
Serratella	1			
Gastropoda		1		2
Isopoda				
Assellidae *Lirceus*	1			1
Megaloptera				
Corydalidae *Corydalis*		2	1	3
C. Nigronia	6			
Odonata				
Coenagrionidae *Argia*		1		
Gomphidae		1		
Olighochaeta	1	2		
Plecoptera				
Leuctriade *Leuctra*		13		
Perlidae *Acroneuria*	4		8	34
Trichoptera				
Hydrospychidae	1	4	85	84
Glossosomatidae	3			
Helicopsyche				
Helicopsyche	1		7	3
Philopotamidae				
Chimarra	4	23	13	7
Total	**67**	**127**	**242**	**309**
Taxa richness	**19**	**15**	**16**	**16**
EPT index	**9**	**5**	**8**	**8**

2.5.5 Terrestrial Fauna

Table 2c. Several species were observed throughout the ALM in October 2003 and April 2004 and include the following:

Common Name	Scientific Name
Amphibian	
Frog	Anura
Salamanders	Caudata
Reptiles	
Garter snake	*Thamnophis sirtalis*
Copperhead	*Agkistrodon contortrix*
King Snake	*Lampropeltis calligaster calligaster*
Box turtle	*Terrapene carolina triunguis*
Fence lizard	*Sceloporus undulatus hyacinthinus*
Five-lined skink	*Eumeces fasciatus*
Birds	
Bird American crow	*Corvus brachyrhynchos*
American robin	*Turdus migratorius*

Bald eagle	*Haliaeetus leucocephalus*
Black capped chickadee	*Parus atricapillus*
Canada goose	*Branta canadensis*
Woodpecker	Picidae
Hummingbird	Trochilidae
Great blue heron	*Ardea herodias*
House sparrow	*Passer domesticus*
Killdeer	*Charadrius vociferus*
Mourning dove	*Zenaida macroura*
Osprey	*Pandion haliaetus*
Owl	*Tytonidae or Srigidae spp.*
Red-tailed hawk	*Buteo jamaicensis*
Red-winged blackbird	*Agelaius phoeniceus*
Song sparrow	*Melospiza melodia*
Turkey	*Meleagris gallopavo*
Swallows	Hirundinidae
Turkey vulture	*Cathartes aura*
Mammals	
Beaver	*Castor canadensis*
Coyote	*Canis latrans*
Squirrel	Sciuridae
Rabbit	*Lepus spp.*
Mice	Muridae
Mink	*Mustela vison*
Red fox	*Vulpes fulva*
Striped skunk	*Mephitis mephitis*
Whitetail deer	*Odocoileus hemionus*

2.5.6 Special Status Species

Table 2d. The following table lists rare, threatened and endangered species and species on the state of Missouri's watch list. These species are known to occur in the ALM area: (MDC, http://mdc.mo.gov/nathis/birds/birdatlas/maintext/0400123.htm).

Common Name	Scientific Name
State Rare	
Liverwort	*Mertzgeria furcata*
Cooper's hawk	*Accipiter striatus*
Great egret	*Ardea alba*
Alligator snapping turtle	*Macroclemys temminckii*
Black bear	*Ursus americanus*
Bald eagle (nesting rare)	*Haliaeetus leucocephalus*
Pied-billed grebe	*Podilymbus podiceps*
State Endangered Species	
Indiana bat	*Mytosis sodalis*

Snowy egret	*Egretta thula*
Northern harrier	*Circus cyaneus*
State Watch List	
Green tree frog	*Hyla cinerea*
Red shouldered hawk	*Buteo linaetus*
Cerulean warbler	*Dendroica cerulea*
Bewick's wren	*Thryomanes bewickii*
Big Creek crayfish	*Oronectes peruncus*
St Francois River crayfish	*Oronectes quadruncus*
Silver-jaw minnow	*Notropis buccatus*
River Otter	*Lutra canadensis*
Eastern Collard Lizard	*Crotophytus collaris*
Yellowwood	*Cladratis kentukea*
Butternut	*Juglans nigra*
Heartleaved plantain	*Plantago cordata*
Wood Stonecrop	*Sedum ternatum*
Shining Ladies Tresses	*Spiranthes lucida*

3.0 PROBLEM FORMULATION

3.1 NATURE AND EXTENT OF CONTAMINATION

The most obvious areas of mining-related impacts within the ALM site are extensive deposits of tailings eroding off the mine waste pile into Sutton Branch Creek (Figure 6). Some of these deposits are currently exposed, and some are buried beneath various depths of soil. Exposed tailings deposits support little or no vegetative cover. These barren or sparsely vegetated areas of exposed tailings are usually referred to as "chat piles". Figures 6 and 7 (Appendix A) show the visual appearance of the chat pile areas.

While the mine waste deposits in and along Sutton Branch Creek are the most obvious signs of mining-related impacts at the ALM, other areas are also impacted. Many areas within the current and historic flood plain have been contaminated due to past flood events which resulted in the distribution of dissolved or suspended contaminants into soils. Areas outside the current and historic flood plain have been impacted by a variety of transport pathways, including removal of mine waste by Iron County road crews, concrete companies and private individuals (Sverdrup 1996), and by the deposition of tailings re-distributed by wind.

3.2 MINING-RELATED CONTAMINANTS OF POTENTIAL CONCERN

The process of identifying contaminants of potential ecological concern for the ALM was based on historical information (Table 1), other Region 7 heavy metal CERCLA sites, ALM START field portable X-Ray fluorescence collected in 1999, 2003 and 2004, and EPA data collected at the ALM in 2003 and 2004. The chemicals selected for evaluation at these sites are summarized below. The following five mine waste-related chemicals were selected for quantitative evaluation in this assessment because they exceeded background concentrations, McDonald et al. 2000 TEC sediment values and AWQC for surface water:

- Arsenic
- Cadmium
- Lead
- Nickel
- Zinc

For the purposes of simplicity, these five chemicals will be referred to as "metals", even though it is recognized that arsenic is a metalloid and not a true metal.
Each of these chemicals of potential concern is capable of causing adverse effects in a wide variety of ecological receptors, including both aquatic receptors (fish, benthic organisms, algae) and terrestrial receptors (plants, birds, mammals). However, it is also important to recognize that all of these chemicals occur naturally in the environment, and that one of the chemicals (zinc) is required in small amounts for good health by nearly all living organisms. Arsenic, cadmium, lead, nickel and zinc were evaluated as chemicals of potential concern for both the aquatic and the terrestrial components of this risk assessment.

3.3 SUMMARY OF HEAVY METAL RISK TO THE ENVIRONMENT

3.3.1 Arsenic

Terrestrial Effects

Arsenic is found naturally in the environment, and low doses (<2μg/day) have actually been found to be beneficial to tadpoles, silkworm caterpillars and other organisms (Eisler 2000). However, soils contaminated with high levels of arsenic (>15 mg/kg) can cause lethal and sublethal effects on flora and fauna. Arsenic occurs in several forms including inorganic and organic states. Inorganic trivalent arsenic (As^{+3}) is more mobile, more soluble, more toxic, and therefore more of a problem than other forms of arsenic. It is frequently found as a component of sulfidic ores in arsenides of nickel, cobalt, copper and iron. Arsenic can be absorbed by ingestion, inhalation and permeation of the skin. Plants can uptake arsenic from the soil or through their leaves. Elevated soil levels of arsenic (>15 mg/kg) may cause phytotoxic effects in plants such as inhibition of photosynthesis. Earthworms (*Lumbricus terrestris*) held in soils containing 40 to 100 mg dry weight of pentavalent arsenic (As^{+5}) for 8-23 days had significantly reduced survival rates. Adverse effects were noted in mammals at single oral doses of 2.5 to 33 mg As/kg body weight Sensitive species of birds died following a single oral dose of 17.4 to 47.6 mg As/kg body weight (Eisler 2000).

Aquatic Effects

Adverse effects of arsenic on aquatic organisms have been reported at concentrations of 19 to 48 μg/L in water, 33 mg/kg in diets, and 1.3 to 5 mg/kg fresh weight in tissues. One of the most sensitive aquatic plants species was algae, which showed reduced growth in the range of 19-22 μg/L. Developing toad embryos were dead or malformed in 7 days at 48 μg/L. Toxic and other effects of arsenic to aquatic life depends on many biological and abiotic factors including temperature, pH, Eh (oxidation-reduction), organic content, phosphate concentration, suspended solids, and the presence of other substances and toxicants. As in the terrestrial environment trivalent arsenic (As^{+3}) is more toxic than pentavalent (As^{+5}) and sensitivity to arsenic is greatest in the early developmental stages for all organisms (Eisler 2000).

3.3.2 Cadmium

Terrestrial Effects

Cadmium is neither essential nor beneficial to biological organisms. In fact, cadmium is a known teratogen and carcinogen, a probable mutagen, and it has been implicated as the cause of many deleterious effects to fish and wildlife (Eisler, 2000). The availability of cadmium to living organisms from their immediate physical and chemical environments depends on factors such as adsorption and desorption rates of cadmium from terrigenous materials, pH, Eh, chemical speciation, and many other modifiers. The main routes of cadmium exposure are via inhalation and ingestion. Factors that are reported to affect dietary cadmium adsorption from the gastrointestinal tract are age, sex, chemical form, levels of protein, levels of calcium, and presence of other elements (Nriagu 1981).

Cadmium is taken up in plants and translocated with subsequent transfer into the terrestrial food web. Cadmium then biomagnifies in terrestrial food webs and tends to accumulate in the liver and kidneys of older apex organisms (Scheuhammer 1987). Freshwater and marine organisms accumulate cadmium from water containing cadmium concentrations not previously considered hazardous to many species of aquatic organisms (Currie et al. 1998).

The lethal effects of cadmium are thought to be caused primarily by free cadmium ions that are not bound to metallothioneins or other metal binding proteins. However, birds and mammals are comparatively resistant to the lethal effects of cadmium. For example, the lowest oral dose producing death in rats was 250 mg/kg-body weight (as cadmium flouroborate).

The sublethal effects of chronic cadmium exposure are growth retardation, anemia, renal effects and testicular damage (Eisler, 2000). Teratogenic effects on animals appear to be greater for cadmium than for other metals, including lead, mercury and arsenic (Ferm and Layton 1981). Until other data become available, wildlife dietary levels exceeding 100 μg Cd/Kg fresh weight on a sustained basis should be viewed with caution (Eisler 2000).

Aquatic Effects

Freshwater biota are the most sensitive receptor group. Concentrations of 0.8 to 9.9 μg/L in water are lethal to freshwater aquatic insects, crustaceans, and fish during exposures of 4-33 days (Eisler 2000). Cadmium inhibits Na+/K+-ATPase activity in tissues, which causes disruptions in osmoregulation (fluid and ion balance within cells). Cadmium is known to concentrate in organs such as the liver, heart, and kidney. Cadmium concentrations in these organs may cause anemia, enlarged heart, and other abnormalities (Eisler 2000). However, water hardness, especially Ca^{2+}, and alkalinity diminishes the biocidal properties of cadmium in freshwater.

3.3.3 Lead

Terrestrial Effects

Lead is neither essential nor beneficial to living organisms. All existing data show that its metabolic effects are adverse. Lead is toxic in most of its chemical forms and can be incorporated into the body via inhalation, ingestion, dermal absorption, and placental transfer to the fetus. Additionally, lead is a cumulative metabolic poison that affects behavior as well as vascular, nervous, renal and reproductive systems (Eisler 2000). Lead modifies the function and structure of kidney, bone, the central nervous system, and the hematopoietic system and produces adverse biochemical, histopathological, neuropsychological, fetotoxic, teratogenic and reproductive effects (Eisler 2000).

Lead does not biomagnify in terrestrial food webs. Older organisms tend to contain the greatest body burdens, and lead accumulations are greatest in bony tissues. Most metals, but especially lead, will accumulate in the roots of plants more than in shoots and seeds. Lead inhibits growth and photosynthesis in plants. Plants readily accumulate lead from soils with low pH or low organic content (Boggess 1977).

In birds, the toxic and sublethal effects of lead vary greatly by species, age, sex, and the form and dose of lead administered. Decreased ALAD activity is a useful indicator of lead exposure, and nestlings are the most sensitive life stage. Among sensitive species of birds, survival was reduced at dietary doses of 50 to 75 mg/kg body weight, and reproductive impairment has been noted at dietary doses of 50 mg/kg. In the Tri-State Mining District of Oklahoma, Missouri and Kansas, songbirds and waterfowl collected in the area had blood and tissue concentrations above levels that constitute lead poisoning (2.6 to 5.2 mg/kg dry weight), and ALAD activity was reduced by more than half in these birds (Beyer et al. 2004).

In mammals, there is general agreement on several points regarding the effects of lead. Significant differences occur between species in response to lead exposure; effects of lead are more pronounced with organolead than inorganic forms, younger developmental stages are the most sensitive; and the effects are exacerbated by elevated temperatures and dietary deficiencies in minerals, fats and proteins. Lead adversely effects the survival of sensitive species at varying concentrations: 5 to 108 mg/kg BW in rats (acute oral), 0.32 mg/kg BW in dogs (chronic oral), and 1.7 mg/kg BW in horses (chronic oral).

Aquatic Effects

Lead is toxic to all aquatic biota, with organolead compounds being more toxic than inorganic lead. Organolead compounds are man-made compounds in which a carbon atom of one or more organic molecules is bound to a lead atom. Of these, the Tetraalkyllead compounds, Tetraethyllead [TEL] and Tetramethyllead [TML] are the most common. Inorganic compounds include metallic lead, alloys, lead oxide, and lead sulphate (PbS, this form is found at the ALM). In aquatic systems, lead concentrations are usually highest in lower trophic levels such as algae and benthic (stream bottom) organisms. Sediments are not only lead sinks, but they are also a constant source of contamination. Fish that eat the benthos within contaminated sediments consume higher levels of heavy metals than piscivorous fish (i.e., bass). Lead can affect neural and hormonal systems that control activity and metabolic rates in fish. Lead can increase mucous production in gill tissue, decreasing oxygen diffusion across gill membranes. Lead inhibits the formation of heme (a component of hemoglobin) in all aquatic biota including amphibians (Eisler 2000).

3.3.4 Nickel

Terrestrial Effects

Nickel is an essential element and it is ubiquitous in the environment. However, human activities, such as mining and smelting contribute to nickel loadings in terrestrial and aquatic systems. The chemical and physical forms of nickel strongly influence bioavailability and toxicity (Eisler 2000). In mammals (including humans) nickel inhalable dust, nickel subsulfide, nickel oxide, and especially nickel carbonyl induce acute pneumonitis (inflammation of the lungs), central nervous system disorders, skin disorders such as dermatitis, and cancer of the lungs and nasal cavity (Eisler 2000). Mammals, such as the common shrew (*Sorex araneus*), have background Ni concentrations of 0.1 to 5 mg/kg in their kidneys. Exposed shrews had Ni concentrations up to 37 mg/kg in their kidney tissue which indicates that Ni may bioaccumulate in mammals (Eisler 2000).

Water solubility of nickel in soils and its bioavailability to plants are affected by soil pH, with decreases in pH below 6.5 mobilizing nickel. Nickel (Ni) is found in terrestrial plants at concentrations usually less than 10 mg/kg DW. Exposed plants may bioaccumulate Ni into their tissues and the hyperaccumulators (i.e. *Alyssum*) could contain up to 120,000 mg/kg DW (Eisler 2000). Earthworms in uncontaminated soils may contain as much as 38 mg Ni /kg DW. Birds located in unpolluted ecosystems have Ni concentrations in their organs from 0.1 to 2.0 mg/kg DW. In nickel contaminated areas, Ni concentrations were elevated in feathers (31 to 36 mg/kg DW), eggs, and internal tissues of birds when compared to conspecifics (same species) collected at reference sites (Eisler 2000). Waterfowl in contaminated areas are especially at risk because plants accumulate high levels of Ni in their tissues and the waterfowl consume those plants. Heavy metals, such as Ni, can bioaccumulate in the terrestrial food web. Bioaccumulation refers to an increase in the concentration of a chemical in specific organs or tissues at higher levels than would normally be expected. How the biologically accumulated chemical behaves within the tissues of organisms can depend upon the species, the age of the species, its health and gender and any other chemicals impacting the organism.

Aquatic Effects

Nickel concentrations have been found to be elevated in aquatic plants and animals located in contaminated areas. Elevated Ni concentrations were found in macrophytes, aquatic insects, tadpoles, zooplankton, fish tissues, and crayfish. There is also evidence of biomagnification through the food chain (Eisler 2000). Several species of amphibian populations, i.e. tree frog, (*Hyla sp)*, American toad, (*Bufo americanas)* and cricket frogs *(Acris sp.)*, have shown a decline in nickel rich waters (>19 μg Ni/L) (Eisler 2000). Nickel toxicity reduces photosynthesis, growth, and nitrogenase activity of algae. The metabolism of soil/sediment bacteria at nominal concentrations (from 11 to 113 μg Ni/L) causes adverse effects on mollusks, protozoans, yeasts, higher plants, insects and fish (Eisler 2000).

3.3.5 Zinc

Terrestrial Effects

Zinc is an essential trace element for all living organisms and is ubiquitous in the tissues of plants and animals. Zinc is a potent inducer of low molecular weight proteins (metallothioneins), which play an important role in zinc homeostasis and in protection against zinc poisoning in animals. Zinc is known to interact with numerous chemicals, including cadmium and lead, resulting in greatly modified patterns of accumulation, metabolism, and toxicity when compared to zinc alone. For example, zinc tends to diminish the toxic effects of cadmium and lead in terrestrial birds and mammals, probably due to metallothionein induction. However, the balance between excess zinc and insufficient zinc is important; either deficiency or excess can adversely effect growth, development and survival.

Zinc toxicosis has been documented to cause pancreatic alterations in a number of bird and mammal species. Excess zinc is also known to cause bone deformations and osteomalacia (a softening of bone tissue). In plants, sensitive species suffer mortality at zinc levels exceeding 100 mg/kg (oak and red maple seedlings). Adverse effects on earthworm survival have been documented at levels of 460 to 662 mg/kg zinc.

In birds, reduced survival has been documented in ducks on diets containing 2500 to 3000 mg/Kg-BW-day. Zinc poisoning in mallards is characterized by ataxia, paresis, and total loss of muscular control of the legs. Zinc concentrations in the livers and kidneys of waterfowl in the Tri-State Mining District were significantly increased above reference values. Recent work has demonstrated zinc poisoning in waterfowl from the area. Diagnosis was based on the finding of mild to severe degenerative abnormalities of the exocrine pancreas associated with hepatic and pancreatic zinc concentrations known to be toxic (Sileo et al., 2004).

Although zinc is relatively non-toxic to mammals, excessive intake of zinc produces a wide variety of effects, including neurological, hematological, hepatic, renal, cardiovascular, developmental, and genotoxic effects. Reproductive effects have been documented in rats fed 500 mg/kg zinc in the diet for three weeks (Saxena et al., 1989).

Aquatic Effects

Concentrations of zinc in tissues of aquatic organisms usually far exceed the required rate for normal metabolism. Zinc levels are further elevated in organisms near mining operations (Eisler 2000). Significant adverse effects of zinc on growth, survival, and reproduction occur in sensitive species of aquatic plants, protozoans, sponges, mollusks, Arthropods, fishes, and amphibians at water concentrations between 10 and 25 μg Zn/L. Zinc has been found to accumulate in juvenile fishes and unlike lead (Pb), zinc concentrations decrease in fish tissue with age (Eisler 2000). Freshwater aquatic plants are usually absent in waters containing >2000 μg Zn/L. Cadmium >10 μg/L significantly increases the toxicity of available zinc to freshwater plants. Elevated temperatures may also increase zincs toxicity to mollusks (Eisler 2000). An increase in salinity may further increase the potential for sorbed zinc in sediments to be released. Aquatic insects are impacted by zinc at levels >1330 µg/L with some species of mayflies being affected at 30-37 µg Zn/L (Eisler 2000). Amphibian embryos are more sensitive to zinc than older stages. Developmental abnormalities were evident in most species of amphibians at concentrations >1500 µg Zn/L (Eisler 2000). Toxic effects of zinc target the pancreas (cellular atrophy), bone (osteomalacia), and gill epithelium (hypoxia) in fish (Eisler 2000).

3.3.6 OTHER FACTORS OF POTENTIAL ECOLOGICAL CONCERN

In addition to the mine waste at the ALM, the Glover lead smelter is upstream of the town of Annapolis and has shown to contribute heavy metals to the waters of Big Creek. The EPA sampled pore water (water that collects in the interstitial spaces of sediment), surface water, and sediment in Big Creek above and below the Glover smelter. The results are as follows:

Table 3. Heavy metal pore water, surface water, and sediment analysis in Big Creek above and below the Glover Smelter, EPA, April 2004. See Figure 2 for sample site locations.

Pore water analysis (µg/L) *Higher than the National AWQC surface water value for lead which is 2.5. < = actual value of the sample is less than the reported value

COC	Above Glover Site 1	Below Glover & above Annapolis Site 2	Below Annapolis, ALM floodplain Site 11
Arsenic	<7	<7	<7
Cadmium	<1	<1	<1
Lead (Pb)	<10	14.0*	<10
Nickel	<6	25.5	16.3
Zinc	<4	23.9	<4
Surface water analysis (µg/L)			
Arsenic	N/A	<7	18.7
Cadmium	N/A	<1	<1

Lead (Pb)	N/A	<10	<10
Nickel	N/A	<6	7.64
Zinc	N/A	33.7	<4
Sediment analysis (mg/kg) Values in bold are above McDonald et al 2002 values (Table 1)			
Arsenic	N/A	**14.2**	**10.7**
Cadmium	N/A	**4.17**	**1.62**
Lead (Pb)	N/A	**115**	24.3
Nickel	N/A	8.25	4.82
Zinc	N/A	108	23.2

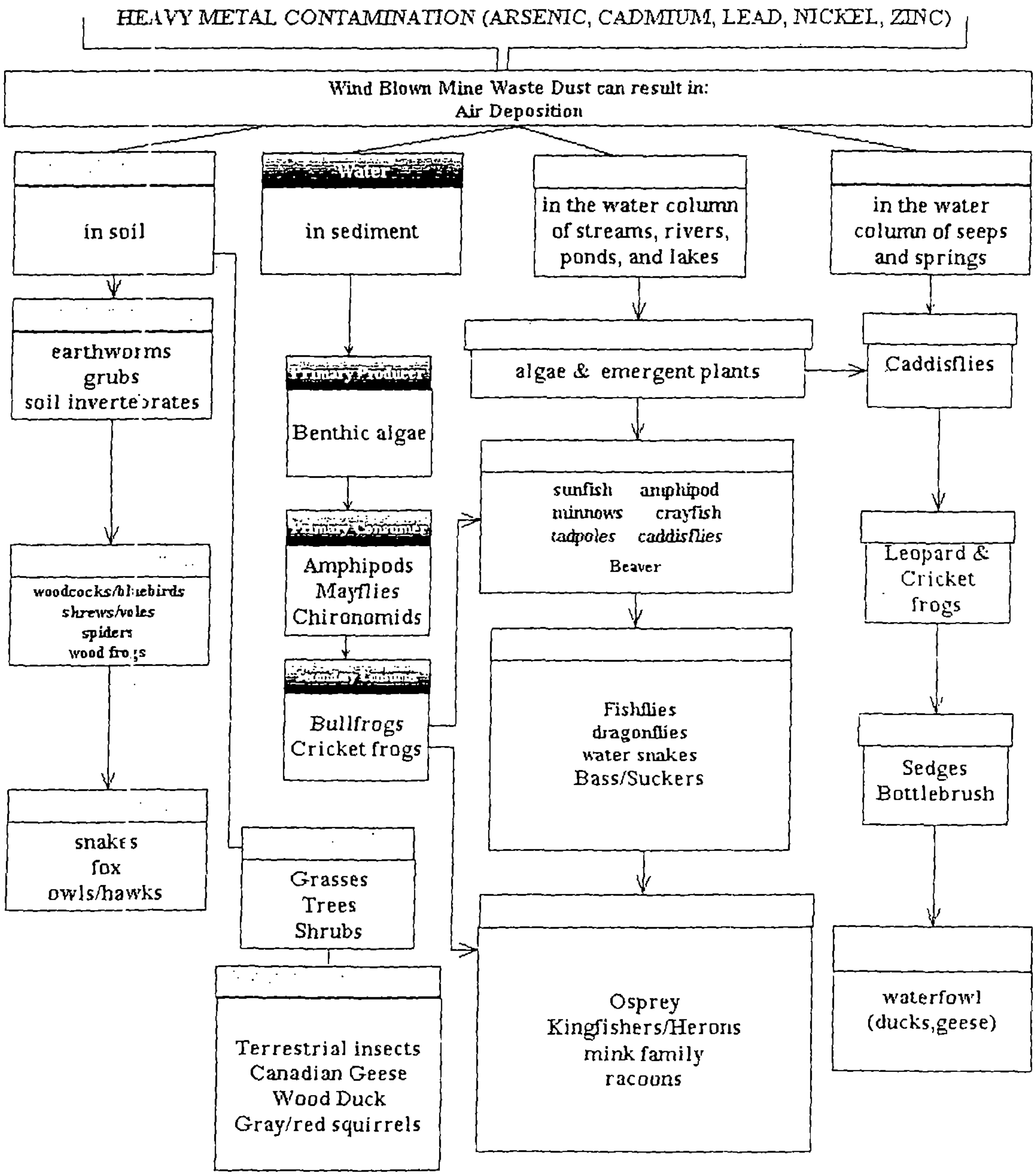
According to Table 3, there is additional metal contamination (arsenic, cadmium and lead) in Big Creek pore water and sediment, which may flow downstream toward the ALM. The additional contamination may be coming from the Glover Smelter. This information could be important for the ALM site when future state monitoring is performed in Big Creek.

3.4 CONCEPTUAL MODEL: ECOSYSTEMS AND RECEPTORS

Most metals and metalloids are capable of causing adverse effects on a wide variety of environmental receptors. At the ALM site, the potential for adverse effects exists for both the aquatic and terrestrial ecosystem. Based on the known pattern of mine waste deposits along Sutton Branch and Big Creek and in the adjacent flood plain, it is evident that the potential for adverse effects exists both for the aquatic ecosystem (fish, benthic invertebrates, amphibians, aquatic plants, etc.) and the terrestrial ecosystem (land animals, birds, insects, trees, grasses, shrubs, etc), both within and outside the riparian area.

The following conceptual site model (Table 3a) illustrates the complex pathways by which chemical contaminants may pass from one portion of the ecosystem to the other.

Table 3a. Annapolis Mine Site Conceptual Model to determine potential ecological receptors that may be exposed to heavy metal chemicals of concern.



3.5 RISK MANAGEMENT GOALS

Risk management goals define the ecological values to be protected and help ensure that the risk assessment process will supply the information needed to support the risk management decision process. Risk managers and risk assessors used information on the area ecology, regulatory endpoints, and publicly perceived environmental values to derive the management goals for this assessment. The ecological risk assessment subgroup responsible for guiding this assessment was directed by Catherine Wooster-Brown (EPA-ERA), Venessa Madden (EPA-ERA), Jason Gunter (EPA-Superfund Project Manager on detail as an ERA), Steven Kinser (Superfund Remedial Project Manager) Heath Smith (EPA On Scene Coordinator, OSC), Doug Ferguson (EPA-OSC), Robert Hinkson (MDNR), and Evan Kifer (MDNR). Based on the results of inputs from all of these parties, the overall management goal for this site was defined as follows:

3.6 ASSESSMENT ENDPOINTS

Assessment Endpoints are derived from general and specific management goals, and identify the specific environmental values which the risk manager has selected to be protective at the site. Specific Assessment Endpoints selected by the risk manager following a consideration of advice and input from a number of concerned parties at this site are listed below:

Assessment Endpoints for Terrestrial Receptors

- Survival, growth, diversity and abundance of the riparian vegetation community under chronic exposure to contaminants and other chemical and physical stressors in the 100 year flood plain habitats of Sutton Branch and Big Creek.

- Survival, growth, and reproduction of wildlife populations under chronic exposure to contaminants and other chemical and physical stressors in the 100 year flood plain habitats of Sutton Branch and Big Creek.

Assessment Endpoints for Aquatic Receptors

- Survival of fish, aquatic invertebrates, and algal populations under acute exposure to contaminants of concern and other chemical and physical stressors in Sutton Branch and Big Creek.

- Survival, growth and reproduction of fish, aquatic invertebrates, and algal populations under chronic exposure to contaminants of concern and other chemical and physical stressors in Sutton Branch and Big Creek.

3.7 RECEPTORS

In general, Assessment Endpoints can not be measured directly, so certain indicator species or groups of species are selected to represent each ecosystem identified as an assessment endpoint. At this site, the receptors selected for evaluation are listed below:

<u>Terrestrial Receptors</u>

- Short-tail Shrew
- Otter

- Woodcock
- Great Blue Heron
- Canadian Goose
- Soil invertebrates
- Horsetail

Aquatic Receptors
- Fish
- Aquatic macroinvertebrates
- Aquatic plants (algae, macrophytes)

These receptors were selected because they represent a broad range of ecological niches, and include a wide variety of direct and indirect (food-chain) exposure pathways which may occur at the site.

3.8 RISK HYPOTHESES

Risk hypotheses are statements which describe assumed relationships between chemical contamination and chemical-mediated effects on the ecosystem. Key hypotheses that were selected for investigation at this site are listed below:

- Metal concentrations in surface water reach levels that are sufficiently high, at least on some occasions, to cause either acute and/or chronic adverse effects in exposed populations of fish, benthic invertebrates, and/or aquatic plants.

- Metal concentrations in sediment are sufficiently high, at least in some locations, to cause adverse effects in exposed fish and/or benthic organisms.

- Benthic macroinvertebrate diversity, particularly Ephemeroptera, are diminished in Sutton Branch Creek (SBC) below the probable point of entry (PPE) when compared to macroinvertebrate diversity above the PPE (Table 2a).

- Metal concentrations in soil are sufficiently high, at least in some locations, that growth and survival of terrestrial plants is inhibited.

- Metal concentrations in soil and the terrestrial food web (i.e. earthworms) are sufficiently high, at least in some areas, which may cause adverse effects in some terrestrial receptors, either directly or via the food chain.

A key property of useful risk hypotheses is that they can be tested. There are a number of different ways to test each of the hypotheses above to determine whether the data support or contradict the hypothesis. The following section outlines the approach used at this site to test the hypotheses above.

3.9 MEASUREMENT ENDPOINTS AND HYPOTHESIS-TESTING APPROACH

Measurement Endpoints are attributes or characteristics of the environment or of selected environmental receptors that can be measured quantitatively and which can be related to the Assessment and Management Endpoints established for the site. The following studies provide two main types of measurement endpoints:

Concentration Values In Environmental Media
This includes not only the main abiotic (non-living) media of potential concern (surface water, sediment, soil, tailings), but also a number of biotic (living) media (fish, benthic organisms, aquatic and terrestrial plants, small mammals, birds) that may be ingested as a food item by one or more of the indicator species at the site.

Site-Specific Observations on Receptor Demographics
This includes data on what species are present, in what abundance, and in what condition. Such population and community based data are often a fairly direct means for evaluating the extent of site-related impacts on ecological receptors. A few studies have been conducted at the ALM site. The process of testing the hypotheses regarding the impact of metals on each major component of the ecosystem (benthics, fish, plants, terrestrial animals) was performed using the general method discussed below.

STEP 1: PREDICTIVE EVALUATION

One approach for evaluating ecological risks from environmental contaminants is to predict the potential for adverse effects by comparing estimated levels of exposure of various environmental receptors to appropriate Toxicity Reference Values (TRVs). These TRVs may either be expressed in units of concentration of a chemical (C) in an environmental medium (e.g., mg/kg in soil, μg/L in water), or in units of dose or intake (mg/kg-day) of the chemical by the environmental receptor. TRVs are derived from review of published toxicity studies, and identify concentration values or dose values that correspond to a No-Observed-Adverse-Effect Level (NOAEL) and/or a Lowest-Observed-Adverse-Effect-Level (LOAEL). Each TRV is both chemical-specific and species-specific (EPA 1999). The comparison takes the form of a ratio, referred to as the Hazard Quotient (HQ), as follows:

$$HQ = \frac{C\ (mg\ kg)}{TRV\ (mg\ kg)}$$

$$HQ = \frac{Dose\ (mg\ kg\text{-}day)}{TRV\ (mg\ kg\text{-}\ day)}$$

If the value of the HQ is less than or equal to one (1), it is believed that no unacceptable impacts will occur in the exposed population of receptors. If the value of the HQ exceeds 1, and then an unacceptable impact may occur, with the predicted likelihood and/or severity of the impacts increasing as the value of HQ increases.

In many cases, the effect of a chemical exposure depends on the length of time the exposure occurs (exposure duration). For this reason, it is often appropriate to identify separate TRVs for acute and chronic exposures. Hazards from acute exposures are usually estimated from peak concentration values, while hazards from chronic exposure are usually based on longer-term average concentrations.

In some cases, two alternative TRV values are available for a particular metal for a particular receptor, one based on the NOAEL, and one based on the LOAEL. These two alternative TRV values reflect the range of uncertainty which exists in the actual threshold between the presence and absence of an adverse effect. If the HQ based on the NOAEL does not exceed a value of one, it is concluded that the chemical does not pose a hazard. If the HQ based on the LOAEL exceeds a value of one, it is expected that the chemical could pose a significant hazard. If the HQ based on the LOAEL is less than one but the HQ based on the NOAEL is greater than one, the chemical is probably close to a level that could cause adverse effects, but whether or not significant effects would actually occur cannot be judged with certainty (EPA 1999). These concepts are summarized below:

LOAEL-Based HQ NOAEL-Based HQ Interpretation

LOAEL based HQ	NOAEL based HQ	Interpretation
< 1	≤ 1	No Hazard
≤ 1	> 1	Possible Hazard
> 1	> 1	Likely Hazard

For TRVs based on dose (i.e., those expressed in units of mg/kg-day), it is important to understand that there may be a wide range of doses experienced by different members of a population. For the ALM site, co-located data was used in HQ calculations. For example, when calculating the HQ for vermivores (earthworm eating organisms), soil data was collected in the same area where the earthworms were collected. The co-located soil data was used instead of the highest (acute) or the average (chronic) heavy metal soil values at the ALM. It is assumed that co-located soil and earthworm data depicts more realistic HQs for this particular site.

If an environmental medium is contaminated with more than one chemical, or if a receptor is exposed to more than one contaminated environmental medium, a screening-level estimate of total hazard may be derived by summing the chemical-specific and/or medium-specific HQ values. The result is termed the Hazard Index (HI) (EPA 1999).

$$HI = \sum (HQ)$$

Summing HQ values in this way assumes that the toxicological mechanisms of action of all of the different chemicals are similar, and that the adverse effects caused by each chemical are additive. In cases where the effects occur by different mechanisms, it is likely the screening-level HI will tend to overestimate actual risk.

The chief advantages of this HQ-based approach are:
1) The only site-specific data required to support the calculations are measured environmental concentration levels, and;

2) The resulting HQ and HI values provide a direct quantitative index of the relative

severity of any anticipated adverse effects. However, there are also limitations to this approach that stem from uncertainties in both the numerator (the estimate of dose or exposure) and the denominator (the TRV) used to calculate the HQ. For example, TRV values are based on toxicity tests performed under laboratory conditions, which may or may not account for factors which can influence (either increasing or decreasing) toxicity in the field (e.g., reduced bioavailability, interaction with other chemicals, combined stress from other sources). Also, some TRVs are based on limited and sometimes internally inconsistent toxicity data and TRVs may not be available for all receptors of concern at a site. Therefore, some TRVs may be relatively uncertain, especially when extrapolation of findings across different species is required. In addition, estimation of actual exposure levels is often difficult (especially for terrestrial receptors) due to lack of site-specific data on intake rates, home ranges, etc. Because of these potential limitations, the HQ/HI approach is best considered to be a screening-level means of evaluation. That is, if an HQ or HI is above a level of concern, this is an indication that effects may be occurring, but further studies (e.g., direct observation of exposed receptors) are sometimes needed in order to confirm if this is really the case.

STEP 2: DIRECT OBSERVATIONS OF ECOSYSTEM STATUS

A second approach for evaluating impacts of environmental contamination on ecological receptors is to make direct observations on the receptors in the field, seeking to determine whether any receptor population has unusually low numbers of individuals, or whether the diversity (number of different species) of a particular category of receptors (i.e. plants, benthic organisms, birds) is lower than expected.

The chief advantage of this approach is that direct observation of community status does not require making the numerous assumptions and estimates needed in the HQ approach. However, there are also a number of important limitations to this approach. The most important of these is that both the abundance and diversity of an ecological population depend on many site-specific factors (habitat suitability, availability of food, predator pressure, etc.), and it is often difficult to know what the expected (un-impacted) abundance and diversity of an ecological population should be in a particular area. This problem is generally approached by seeking an appropriate "reference area" (either the site itself before the impact occurred, or some similar site that has not been impacted), and comparing the observed abundance and diversity in the reference area to that for the site. However, it is sometimes quite difficult to locate reference areas that are truly a good match for all of the important habitat variables at the site, so comparisons based on this approach do not always establish firm cause-and-effect conclusions regarding the impact of environmental contamination on a receptor population.

This problem is further complicated by the natural variability in population parameters over time. That is, measurements of diversity and abundance at any one point in time may not be representative of long-term average values. Thus, comparisons between a site area and a reference area are of greatest value when based on cumulative observations over long periods of time.

Data on population and community structure are especially valuable for hypothesis testing. In general, the basic hypothesis to be tested is that there is a direct correlation between the concentration of metals in the environment and the level of effect observed in exposed populations. One way to test this hypothesis is to plot a measure of population or community status (e.g., number of macroinvertebrates or fish) as a function of one or more measures of environmental contamination (e.g., the concentration of metals in sediments or water). If metals are causing an effect on the population-based measurement endpoint, then it is expected that there will be an observable trend in the measurement endpoint as a function of the environmental endpoint. If a trend is observed and if the trend is statistically significant, this may be taken as good evidence for a cause and effect relationship. However, the converse is not necessarily true. That is, absence of a statistically significant trend is not necessarily proof that metals are having no effect. This is because the measurement endpoint may depend not only on metals but on numerous other variables (e.g., water temperature, flow rate, prey abundance, habitat quality), and the effect of metals may be partly or entirely obscured by variability in the effects of these other stressors. This problem is further complicated by the difficulty usually encountered in obtaining accurate measurements of population status and/or environmental contamination levels. That is, if one or both values are not known precisely, this "measurement error" can prevent the detection of cause and effect relationships, which do exist. Thus, if no significant relationship is detected between a community endpoint and the level of environmental contamination, the correct interpretation is that the effect of the contamination (if any) is not sufficiently large to be detected in the face of other independent variables and/or measurement error.

STEP 3: WEIGHT OF EVIDENCE EVALUATION

As discussed above, each of the methods available for evaluating potential impacts of environmental pollution on ecological receptors has advantages but also has limitations. For this reason, conclusions based on only one method of evaluation may be misleading. Therefore, the best approach for deriving reliable conclusions is to combine the findings of all methods for which data are available, taking the relative strengths and weaknesses of each method into account. If the methods all yield similar conclusions, confidence in the conclusion is greatly increased. If different methods yield different conclusions, then a careful review must be performed to identify the likely basis of the discrepancy, and to decide which method is more likely to yield the correct conclusion.

For example, consider the case where an impact on a receptor species is detected by direct observation and the HQ and/or HI values for one or more of chemicals of concern are greater than one. In this situation, the two independent lines of evidence tend to support each other, and it is reasonable to conclude that the chemicals are causing, or at least contributing, to the observed impact. Similarly, if the HQ and HI values are less than one and no effect can be observed, then both lines of evidence support each other and indicate that hazard is not significant. However, in the case where an impact on a receptor species is observed but HQ and/or HI values are less than 1, then the two lines of evidence are not in agreement, and it is reasonable to question whether the effect is attributable to the chemicals and to consider whether other factors may be responsible. Likewise, in the case where HQ and/or HI values are greater than one but no effect can be observed, then it is reasonable to suspect that the predicted hazards may be higher than actual, either because exposure or dose has been over-estimated or because the TRV is too conservative.

4.0 NEAR, MID, AND FAR PILE SEPARATE EXPOSURE ASSESSMENT

Exposure was not based on a site-wide average, instead risks were separately assessed for Near, Mid, and Far pile sites. The following are short location descriptions of the Near, Mid, and Far site areas. Please see Figure 2, in Appendix A, to view a map of these areas.

Near Pile—The Near Pile area is located on the east side of County Road 138 in a confined area that was directly in and around the mine waste piles. This area did not extend to Sutton Branch Creek or Highway 49. See Figure 2, Appendix A.

Mid-Pile—The Mid-Pile area is located across County Road 138 and extends into and to the west of Sutton Branch Creek. The Mid-Pile area also crosses Highway 49 and extends down to Big Creek ending approximately 1000 ft above and 500 ft below Sutton Branch confluence (Figure 2).

Far Pile—The Far Pile area is located by the bridge off of Highway 49 (Figure 2).

4.1 PORE WATER, WATER COLUMN, AND SEDIMENT DATA

The following tables list the analytical results of the aquatic samples taken from Sutton Branch and Big Creek. Pore water is water from the interstitial spaces taken just below the sediment surface in the middle of the stream with a pore water extractor (EPA 2004). Bank pore water uses the same pore water extractor, but the sample is taken from the edge of the stream where there is no surface water above the sediment. Please see Figure 2 to locate site numbers.

Table 4.a. EPA heavy metal results for pore water, bank pore water, water column and sediment samples taken from Sutton Branch Creek, Annapolis Mine Site, April 2004. U=The analyte was not detected at or below the reporting limit.

Sutton Branch Creek							
Background levels site 3	7U	1U	2U	29U	10U	6U	4U
Background levels site 4	7U	1U	2U	29U	10U	6U	4U
Pore water-ug/L	**Arsenic**	**Cadmium**	**Copper**	**Iron**	**Lead**	**Nickel**	**Zinc**
Site 5	U	U	U	473	274	45.8	75.8
Site 6	U	U	U	U	10.7	7.57	U
Site 7	U	U	U	U	U	U	U
Site 8	U	U	U	U	U	U	U
Site 9	U	U	U	U	14.4	15.4	U
Mean	**U**	**U**	**U**	**473.0**	**99.7**	**22.9**	**75.8**
Bank Pore Water--ug/L							
Site 5	U	U	U	735	31.3	40.6	27.5
Site 6	18.9	U	U	U	35.7	11	5.23
Site 7	U	U	U	U	11	14.2	U
Site 8	U	U	U	U	U	U	U
Site 9	12.4	U	U	U	12.8	12	U
Mean	**15.7**	**U**	**U**	**735.0**	**22.7**	**19.5**	**16.4**
Water column--ug/L							
Site 5	U	U	U	U	U	U	U
Site 6	U	U	9.12	U	U	U	U
Site 7	19.9	U	U	U	U	U	U
Site 8	7.95	U	U	U	U	U	U
Site 9	*	U	U	U	U	7.86	U
Mean	**13.9**	**U**	**9.1**	**U**	**U**	**7.9**	**U**
Sediment—mg/kg							
Site 5	U	U	N/A	N/A	U	U	U
Site 6	5.04	U	N/A	N/A	365	U	41.5
Site 7	10.6	U	N/A	N/A	721	4.7	55.9
Site 8	8.79	U	N/A	N/A	164	5.16	37.5
Site 9	7.69	U	N/A	N/A	962	3.19	49.8
Mean	**8.1**	**U**	N/A	N/A	**553**	**4.4**	**46.2**

Table 4b. EPA heavy metal results for pore water, bank pore water, water column and sediment samples taken from Big Creek, Annapolis Mine Site, April 2004. U = The analyte was not detected at or below the reporting limit

Big Creek

Background levels site 1	7U	1U	2U	29U	10U	6U	4U
Pore water-ug/L	**Arsenic**	**Cadmium**	**Copper**	**Iron**	**Lead**	**Nickel**	**Zinc**
Site 2	U	U	U	9180	14	25.5	23.9
Site 10	U	U	U	U	U	7.96	U
Site 11	U	U	U	U	U	16.3	U
Mean	**U**	**U**	**U**	**9180.0**	**14.0**	**16.6**	**23.9**
Bank Pore Water-ug/L							
Site 2	U	U	U	U	U	6 93	70.3
Site 10	10.8	U	U	U	U	6.76	U
Site 11	U	U	U	U	U	12.5	U
Mean	**10.8**	**U**	**U**	**U**	**U**	**8.7**	**70.3**
Water column-ug/L							
Site 2	U	U	U	U	U	U	U
Site 10	12.4	U	U	U	U	7.7	U
Site 11	18.7	U	U	U	U	7.64	U
Mean	**15.6**	**U**	**U**	**U**	**U**	**U**	**U**
Sediment-mg/kg							
site 2	U	U	N/A	N/A	U	U	U
site 10	7.13	U	N/A	N/A	40.3	3.61	13.3
site 11	10.7	1.62	N/A	N/A	24.3	4.8	23.2
Mean	**8.9**	**U**	N/A	N/A	**32.3**	**4.2**	**18.3**

Table 4c lists the results for heavy metals found in Near, Mid and Far soils at the ALM. Near and Far Pile soil was collected in the field by the EPA and analyzed by EPA's Science and Technology Center. The EPA Superfund Program analyzed ALM soil in Near and Mid areas for lead only with X-Ray Fluorescence (XRF) in 2003 and 2004 (Table A1 , Appendix A).

Table 4.c. Metals found in Near, Mid, and Far Pile soils at the ALM, EPA, 2004. Values in bold were used in the Hazard Quotient (HQ) equations, Table 9a. See Figure 2 in Appendix A for site locations.

Near Pile

	Metal in Soil mg/kg						
Site	*Ar*	*Cd*	*Pb*	*Pb (XRF)*	*Ni*	*Zn*	*pH*
Near	73.8	1.92	654	**	85.2	226	7.7
Near	35.7	1.26	2090		32	106	
Near	79.6	5.5	9400		43.9	643	
Near	45.4	3.52	6800		27.6	524	
MEAN	58.6	3.05	4736	1411	47.2	374.7	
95 % CI		(1.21, 4.89)	(748, 8723)	(1198,1625)			(129, 621)
95% UCL		4.89	8723	1625			
CLV*			**654**				

*Co-Located Value (Soil collected where earthworms were collected Table 9a)

** See Table A1 in Appendix A for Pb (XRF) data

95% CI = Confidence Interval, 95% UCL = Upper Confidence Limit

Mid-Pile-There is only XRF Pb data available (Table A1, Appendix A). Values are based on 199 15.24 x 15.25 meters (50ft^2) cells (mg/kg). There is not a CLV for Mid-Pile.

	Metal in Soil mg/kg
	Pb (XRF)
MEAN	756
95 % CI	(639,840)
95% UCL	840

Far Site- A Mean, 95% CI, and 95% UCL for Far Pile soil is not available

			Metal in Soil mg/kg			
Site	Ar	Cd	Pb	Ni	Zn	pH
Far	15.5	7.37	149	10.3	71	7.4

For the purposes of assessing exposures of ecological receptors, all soil samples were collected in the depth interval from zero to two feet. This depth interval was selected because most plant species have roots that exist within this zone, and burrowing mammals are also likely to be exposed in this zone.

Inspection of Table 4.c reveals the following main observations:

- There is variability in the concentration of each metal of concern in different soil samples.
- The CLV is the lowest lead concentration in the Near pile soils and therefore concentrations above 654 mg/kg could be toxic to earthworms.
- Between soil categories, there is a clear pattern of decreasing concentration of all of the metals from Near to Far Sites.
- The soil pH in areas is greater than 7, but less than 8.

4.3 FISH TISSUE DATA

Two Big Creek fish surveys were performed in 1989 (Schmitt et al. 1993) and (Schmitt et al. 1997) and found elevated concentrations of lead in fish tissue (4.57 mg/kg wet weight-whole carcass) and high concentrations of cadmium (1.2 mg/kg wet weight-whole carcass). Schmitt (et al. 1984) also recorded greatly elevated lead levels in whole fish (9 to 18 mg/kg) fresh weight from the Big River (0.3 mg/kg fresh weight in edible tissue is considered hazardous to human health) located in the lead belt slightly northeast of Annapolis. For comparison, the highest lead concentration recorded to date in the National Biocontamination Monitoring Program is 6.7 mg/kg fresh weight in whole tilapia from Honolulu in 1979 (Lowe et al. 1985). Schmitt et al. (1993, 1997) fish tissue data from the Big Creek studies were utilized to evaluate potential impacts on fish.

5.0 HAZARD TO FISH

5.1 EXPOSURE PATHWAYS

Fish species known to inhabit Big Creek include small and large mouth bass, green and bluegill sunfish, crappie, walleye, and catfish.
Fish in the creek may be exposed to chemicals of concern by three pathways:

1. Direct contact (gill respiration) with chemicals dissolved or suspended in surface water

2. Ingestion of food items (benthic organisms, plant material, other fish, etc.) that have incorporated chemicals of concern into their tissues.

3. Incidental ingestion of contaminated sediments during normal feeding activities.

The following sections predict the risk to fish from exposure by these pathways.

5.2 POSSIBLE HAZARD

5.2.1 Possible Hazard from Direct Contact with Surface Water

Heavy metals in surface water would be dissolved and therefore may be more bioavailable to aquatic organisms. The hazard to fish from direct contact with dissolved metals in stream water may have two different components:

1. Acute and chronic hazards from exposure to the "typical" range of metals in Big Creek and Sutton Branch Creek.

2. Acute hazards associated with exposure to "pulses" of increased metal levels that have been observed in the creeks.

5.2.2. Possible Acute Hazard from "Pulse Events"

A major fish kill event in Big Creek was associated with the occurrence of a storm event leading to a ruptured dam in Annapolis (Schmitt 1997). Also, a phone conversation with Wanda Doolan, who is employed by MDNR and a park naturalist at Sam A. Baker Park, observed that during storm events Big Creek would turn a "milky" color. Annapolis is upstream of Sam A. Baker Park.

Conceptually, it is thought that storm events can lead to increases of metal concentrations in the river by two main mechanisms:

1. Increased flow in the river may lead to resuspension of tailings particles from sediments or bank deposits. This is expected to result mainly in an increase in total recoverable metals, with a smaller increase in dissolved metals.

2. Overland flow (run-off) of rain water or snow melt in areas of exposed tailings can result in very high concentrations of total and dissolved metals.

5.2.3 Possible Hazard from Ingestion of Benthic Organisms

Fish are known to feed on a wide variety of benthic organisms. According to Table 2b there is a diverse number of benthic organisms both in Sutton Branch and Big Creek. Very few data are available on the relative toxicity of metals in the diet to species of fish. Woodward et al. (1995a) found that brown trout and rainbow trout fry had susceptibility to metals from benthic organisms. The benthos collected at Annapolis Mine Site was not analyzed for metals, only diversity. However, Schmitt et al. (1997) found elevated concentrations of lead in fish in Big Creek. Therefore, lead is bioavailable to fish in Big Creek via some mechanism. The benthos alone may not be enough to impact the fish populations, but the additive impacts of contaminated benthos, sediment, surface water, and pulse events is a possible hazard for the fish populations in both Sutton Branch and Big Creek.

5.2.4 Possible Hazard from Incidental Ingestion of Sediment

It is not believed that fish intentionally swallow inorganic sediments, but a few reports were located which indicate that sand or small stones are occasionally found in the stomach content of trout (Papageorgiou et al. 1984) and suckers (Carl 1936, Macaphee 1960). Even though the amount of inorganic sediment ingested may be small, this could be a source of significant exposure because the concentration of metals in sediments is substantially higher than the concentration in benthic organisms.

5.2.5 Possible Total Hazard to Fish from All Exposure Pathways

The total hazard to fish from contact with chemicals of potential concern in the aquatic environment comes from direct contact with dissolved metals in surface water, ingestion of metals in prey species, and incidental ingestion of sediment. The combined exposures may contribute a chronic low-level stress on fish that would be manifest by effects such as decreased growth rate and/or decreased immunity to disease.

5.2.6 Possible Hazard to State Listed Rare and State-Watched Fish Species

As noted earlier, state-listed rare species include the southern brook lamprey and the State-watched silverjaw minnow, which may occur in Big Creek. There are no Federally Endangered or Threatened fish species that occur in Iron County, MO. Mussel surveys were not performed at the ALM to determine if there are any species in Big Creek. No mussels were found in Sutton Branch Creek. No toxicity data were located for these species, so no rigorous species-specific evaluation of hazard is possible.

Typical concentrations found in Sutton Branch and Big Creek are unlikely to be of concern, but if the lamprey and the minnow occur within this watershed, they are likely to be at risk of injury or acute mortality from intermittent pulses of elevated metals in the creeks. Although less certain, both species are likely to have similar risks of chronic stress from long-term average exposure to metals in water and/or the diet. This risk is likely to be greater in the vicinity of the ALM. It is important to note that these fish species spawn in the riffles of creeks, and most reports indicate that the fish do not generally migrate from the natal stream. The potential significance of this behavior is that older (migratory) fish are less sensitive to the toxic effects of metals than are fry, and it's usually the adult fish that are analyzed. Also, there is still a risk of acute toxicity and/or lethality associated with pulse events, since these events have been previously observed to cause acute lethality in adult fish. Moreover, repeated loss of migratory individuals from historic or current pulse events could ultimately result in a decrease or loss of this phenotypic behavior in the population.

5.3 EVIDENCE OF INCREASED EXPOSURE

Concentrations from Table 1, 1a, 4a, 4b, and 4c have demonstrated that the metals of concern are higher in surface water, sediment and soil from the impacted ALM areas when compared to background sites. The benthic macroinvertebrate diversity is also significantly different in Sutton Branch Creek below the PPE when compared to reference areas above the PPE. Fish below the PPE in Sutton Branch and at the confluence to Big Creek are exposed to higher concentrations of metals than fish in background streams. This conclusion is also supported by direct measurements of metal concentrations in whole fish and ALA-D activity in fish blood (Schmitt et al. 1984, 1992, 1993, 1997). Fish sampled in Big Creek were found to have higher tissue levels of cadmium, lead and zinc (Schmitt 1997) than fish from background sites.

5.4 SITE-SPECIFIC TOXICITY STUDIES

To date, there are no site specific toxicity studies performed with fish or any other aquatic organisms at the ALM.

5.4.1 Are Fish in Sutton Branch and Big Creek at an Increased Risk for Mortality?

In considering the likely basis for an increase in mortality of fish exposed in Sutton Branch and Big Creek, three alternative (but not mutually exclusive) options need to be considered:

1. Acute lethality from intermittent "pulses" of high concentrations of metals.
This hypothesis, although untestable with the current data, is considered to be plausible since a documented fish kill occurred after the rhyolite tailings dam broke in Annapolis (Schmitt 1997).

2. Chronic toxicity from exposure to average concentrations in water and/or the diet. Average concentrations of metals in creek water were not likely to be the primary factor responsible for any significant mortality. This conclusion is limited by the fact that fish (both juveniles and fry) may be exposed not only to metals in water but also to metals in natural food items. This dietary exposure, combined with the water exposure, could conceivably increase the total dose into a range capable of causing lethality.

Fish survival may be influenced by a variety of factors other than the concentration of metals in water. For example, low dissolved oxygen and/or elevated water temperature could conceivably increase mortality rates in fish. Although neither of these water quality parameters were monitored as part of this study, temperature and oxygen levels observed by EPA risk assessors during the spring suggests that these parameters were unlikely to have approached or exceeded levels that would be expected to cause acute mortality in fish.

3. Another potential non-chemical stressor is increased levels of mine waste particles eroding into the creeks after a storm event. Total Dissolved Solids (TDS) from the mine waste, coupled with the increased metals concentration associated with high TDS may contribute to "fish kills". Schmitt et al. (1992, 1997) noted that a massive fish kill occurred in Big Creek after a dam constructed of rhyolite tailings ruptured. Wanda Doolan, a park naturalist at Sam A. Baker State Park (15 miles downstream of Annapolis), noted the "milky" color of Big Creek as it flowed into the park after heavy storm events. Wanda Doolan filed a complaint May 1, 2003 with the Missouri Department of Natural Resources (Doolan 2003). This information, along with EPA observed low fish diversity in 2004, may suggest that pulse events of eroding mine waste contribute to the mortality of fish in Sutton Branch and Big Creek. However, acute lethality is not generally noted until TDS concentrations exceed 10,000 mg/L (ASCE 1992. Anderson et al. 1996) and Big Creek is not regularly monitored for TDS.

5.4.2 Effects of Longer-Term Exposure to Surface Water and/or Diet

A number of reports and studies indicate that exposure of fish to heavy metals in water and/or the diet can result in decreased growth (Seim et al. 1984, Roch and McCarter 1984, 1986, Bergman et al. 1993, Woodward et al. 1994, Woodward 1995, Marr et al. 1995). In addition to decreased growth, several biochemical and/or histological changes have been noted in these studies, including:

a) increased levels of metallothionein (proteins that bind metals)
b) increased lipid peroxidation in tissues (fat cell degeneration)
c) increased scale loss

This has led to the hypothesis (Bergman 1993, Lipton et al. 1995) that chronic exposure of trout to metals causes the occurrence of these "physiological impairments", and that the impact of these impairments is decreased growth in fish. Decreased growth is an effect of concern in fish since fecundity and reproduction rates tend to be correlated to size. Further, decreased growth has been found to be associated with decreased over-winter survival in young fish. That is, if surface water and/or dietary exposures to metals do result in decreased growth, this could account, at least in part, for decreased populations of fish observed in Sutton Branch and Big Creek.

5.6 WEIGHT OF EVIDENCE EVALUATION

Evaluation of the weight of evidence on a particular issue is a process that generally requires professional judgment. It is helpful to begin by summarizing all of the observations that bear on a particular issue, and then deciding how relevant and how convincing each observation is. That is, does the observation clearly imply that metals have caused a particular effect (e.g. acute lethality), or are there other credible interpretations that might account for the observation?

This basic weight of evidence approach is used below to seek conclusions to the following three key questions:

1. Do "typical" (non-pulse event) concentrations of metals in Sutton Branch and Big Creek reach concentrations that are likely to cause acute lethality in fish?

2. Do "pulses" of metals still occur in the creeks, and if so, do they reach concentrations high enough to cause acute lethality in trout or other fish?

3. Do "typical" concentrations of metals in Sutton Branch and Big Creek (including all routes of exposure) cause significant adverse effects on fish populations, and do typical concentrations cause growth inhibition?

Acute Lethality and Toxicity

Acute Lethality from Typical (Non-Pulse) Surface Water Concentrations

The question being considered here is whether or not the "typical" (non-pulse event) concentrations of metals in the creeks reach concentrations that are likely to cause acute lethality in fish. Observations on this question are discussed below.

1. Based on sediment data in Table 2a, the hazard of acute lethality in fish from the mixture of metals that occur in Sutton Branch Creek is dominated by lead and zinc. Hazards from other metals appear minimal, either alone or in combination. The hazard to fish from heavy metal contaminants in Big Creek sediment is also minimal.

2. At a hardness of 100 mg/L, the 96-hour LC50 value for an 0.4 g trout fry exposed to a mixture of metals (imitating a pulse event) in site water with zinc levels of 230 µg/L caused a 17% mortality (Bergman 1993).

3. Some acclimation studies in the laboratory have been performed (Marr et al. 1995); it seems likely that fish may become acclimatized to the elevated levels of metals in water. Because acclimation is often accompanied by biological costs such as reduced adaptive flexibility to other stresses or a metabolic cost such as decreased growth, it should not be viewed as beneficial. However, acclimation may provide increased tolerance to acute exposures from typical concentration values.

All of these findings support the conclusion that there is low risk of acute lethality to fish in the Sutton Branch and Big Creek from typical levels of metals.

The Hazard from Storm-Related Pulses or Other Intermittent Pulses in Surface Water Concentrations

The question being considered here is whether or not "pulses" of metals still occur in Sutton Branch and Big Creek, and if so, do they reach concentrations high enough to cause acute lethality in fish.

1. Historically, there was a clear association between a storm event and the occurrence of a fish kill in Big Creek (Schmitt 1997). This is thought to be due to a rhyolite tailings dam breaking and leaching metals into Big Creek. There are no other documented fish kills after storm events where the creeks turn "milky" in color.

2. Absence of reported fish kills is not evidence that acutely lethal events are not still occurring, since many fish kills (especially those that involve only small fish) probably go unnoticed and/or unreported.

4. Absence of lethal concentrations in routine surface water monitoring is not evidence that no pulses are occurring. Such routine monitoring has only a low chance of detecting a short pulse of metals.

5. Mine waste piles continue to erode into Sutton Branch and Big Creek, especially during heavy rains.

In conclusion, fish are at risk of succumbing to severe mine waste erosion when there is a major flood event.

Chronic Toxicity

Are "typical" concentrations of metals in Sutton Branch and Big Creek high enough to cause significant adverse effects on fish populations and are the concentrations high enough to cause growth inhibition? Findings related to this question are summarized and discussed below:

Hazard from Chronic Exposure to Surface Water

Based on Table 4a, heavy metal surface water concentrations in Sutton Branch and Big creek are basically all undetected. However, the lead in pore water at site 5 in Sutton Branch Creek (Table 4a) is 274 μg/L. This suggests that there are areas where fish populations may be exposed to high concentrations of lead. According to Schmitt et al. (1992,1997), results of a fish study done in Big Creek revealed high concentrations of lead, cadmium, and zinc in fish tissue.

Hazard from Dietary Exposure

Fish are known to feed on a wide variety of benthic organisms and there is a diverse number of benthic organisms both in Sutton Branch and Big Creek. The benthos alone may not be enough to impact the fish populations, but the additive impacts of contaminated benthos, sediment, surface water, and pulse events may be a possible hazard for the fish populations in both Sutton Branch and Big Creek.

Hazard from All Exposures Combined

EPA observed very few fish in Sutton Branch and Big Creek in April 2004. The habitat in both creeks is excellent for stonerollers (Catastomidae), a typical Ozarkian fish that feeds on the bottom of streams. It was unusual not to find the fish in either of the two creeks. Low fish density and diversity coupled with decreased macroinvertebrate diversity in Sutton Branch Creek below the PPE could be caused by high metal content in the sediments. Schmitt et al. (1992, 1997) studies found high levels of metals in fish tissue and fish blood which demonstrates that fish are assimilating the metals into their bodies. Contaminated surface and pore water may further contribute to fish being exposed to heavy metals.

Conclusion Regarding Chronic Hazard

Taken together, the data and information above are consistent with the hypothesis that lead and zinc in the aquatic environment (surface water, diet) is (are) imposing a low level chronic stress on fish. The most likely manifestation of this stress is decreased growth, but the magnitude of the effect cannot be stated with certainty, and data are not adequate to determine whether or not fish from Sutton Branch and Big Creek are actually smaller in size than expected. It is unknown to what degree this chronic stress contributes to the decrease in standing fish population, but it is considered likely that acute exposures to pulses or other high-concentration events are more likely to be important than chronic stresses, since even one fish kill from a pulse event could lead to significant reductions in the fish population.

6.0 HAZARDS TO AQUATIC MACROINVERTEBRATES

6.1 EXPOSURE PATHWAYS

A wide variety of different organisms may inhabit the bottom of a stream or river, including many different types of insects (mayflies, caddisflies, black flies, stoneflies, beetles, etc.), crustaceans (crayfish, isopods, amphipods), mollusks (snails, clams), and a few others. Collectively, these organisms are referred to as benthic macroinvertebrates. Benthic macroinvertebrates are often used to help evaluate the ecological status of a stream or river because:

1. They occur in most aquatic environments
2. They are relatively easy to collect and analyze
3. They live in intimate contact with the sediment and the water
4. They serve as an important source of food for fish
5. Some types of benthic organisms are especially sensitive to environmental pollution

Benthic macroinvertebrates may be exposed to tailings-derived contaminants in the aquatic environment by the following pathways.

- Direct contact with metals in river water (this pathway is most applicable to species which live on or close to the surface of the sediment substrate)
- Direct contact with metals dissolved into the interstitial water occupying the spaces between sediment particles (this pathway is most applicable to species that live buried within the sediment substrate)
- Ingestion of food web items (e.g., algae, diatoms, detritus, other benthic organisms) which have incorporated levels of metals into their tissues that are higher than in reference streams
- Incidental ingestion of fine sediment particles in association with normal feeding activities

The relative importance of these different exposure pathways is not known, and is likely to vary considerably among different species of benthic organisms. In addition, there may be considerable temporal variation in the relative importance of each pathway.

6.2 PREDICTED HAZARD TO BENTHIC ORGANISMS

6.2.1 Predicted Hazard from Contact with Surface Water

Benthic macroinvertebrates, such as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) spend their larval years living on the substrate in lotic streams. Macroinvertebrates are encompassed by surface water above and pore water below. Consequently they are continuously exposed to any surface and pore water contaminants. Other macroinvertebrates such as burrowing mayflies, Oligochaeta (worms), and several species of chironomids live in the interstitial spaces of sediments where pore water is located. Data on the concentration of metals in surface and pore water have been presented earlier (see Table 4a). In accordance with EPA recommendations (Prothro 1993), attention is focused on hazards from contact with dissolved metals, since dissolved metals are thought to be more predictive of hazard from direct contact than total recoverable metals.

Table 6 below, summarizes available water column toxicity data from the AWQC national database (EPA 1985b-d, 1987, 1996) for benthic species that either do occur or are reasonable surrogates for other species that do occur in Sutton Branch and Big Creek. An exception is the cladoceran, which is included on the list even though these organisms are mainly planktonic rather than benthic, and are not observed to occur in significant numbers in Sutton Branch or Big Creek. Cladocerans are retained because they are among the most sensitive of aquatic invertebrates to the effects of metals, and therefore can serve as a surrogate for other sensitive aquatic macroinvertebrates which are of concern in the ALM creeks, but for which EPA has not established standard toxicity values. For example, Ephemeroptera and Trichoptera are two sensitive macroinvertebrate orders and they are in the same feeding guild as the herbivorous cladoceran.

An important limitation to the toxicity values shown in Table 6 is that species-specific toxicity data are sparse or lacking for many important benthic macroinvertebrates found in Sutton Branch and Big Creek.

Table 6. Summary of species mean toxicity reference values for aquatic macroinvertebrates, (EPA 1985b-e, 1987, 1996).
H=Water Hardness * = no data available

	Species Mean Acute Value (µg/L) H=200mg/L				Species Mean Chronic Value (µg/L) H=200mg/L			
Species	**As**	**Cd**	**Pb**	**Zn**	**As**	**Cd**	**Pb**	**Zn**
Amphipoda	874	251	575	*	*	*	*	*
Cladoceran	4449	62	1805	1125	914	0.39	135	163
Chironomid	97000	5248	950652	*	*	*	*	*

Figures 8 & 9 in Appendix A compares data on total lead concentrations observed in Sutton Branch Creek both above (reference) and below the PPE. Figures 8 & 9 also include the pollution sensitive aquatic macroinvertebrates Ephemeroptera, Plecoptera, and Trichoptera (EPT index) and the pollution tolerant Chironomidae percentages.

Winner et al. (1980) found that macroinvertebrate community structure exhibits a predictable response to heavy metal contamination. He found that EPT was virtually eliminated in sections of streams contaminated with heavy metals. But, tolerant species such as chironomids and tubificid worms were abundant.

According to Figures 8 & 9 in Appendix A, Sutton Branch Creek followed this same community structure response when contaminated sections below the PPE were compared to upstream reference sections.

Potential Hazard from Pulse Events

As discussed previously, "pulses" in surface water concentration conditions exist in Sutton Branch and Big Creek. Specifically, bankside tailings deposits are known to accumulate crusts of metal-rich salt deposits during dry periods between storms, and surface run-off from such salt crusts can carry dissolved metals to the creeks. This type of event could result in acute toxicity to at least some benthic species.

6.2.2 Predicted Hazard from Contact with Sediment

Benthic macroinvertebrates that spend some or most of their life cycle within the sediment are believed to be exposed by the following pathways:

- Metal in Sediment
- Dissolved Metal in Pore Water
- Uptake and Adsorbed by Organism

Predictions Based on Total Metals in the Sediment

Sediment Concentration Data

Data on the concentration of metals in sediment have been presented earlier (see Section 4a). The EPA has not established national TRVs for total metals in sediment. However, MacDonald et al. (2000), developed a consensus based Threshold Effect Concentration (TEC) set of guidelines from all of the previous approaches. See Tables 6a and 6b below. The TEC and Probable Effect Concentration (PEC) numbers in the tables were taken directly from MacDonald et al (2000).

Table 6a. Sediment Quality Guidelines (SQG) for metals in freshwater ecosystems that reflect TECs (below which harmful effects are likely to be observed). MacDonald et al 2000. (DW= dry weight)

Substance	Threshold Effect Concentration TEC (Consensus Based)
Metals (mg/kg DW)	
Arsenic	9.79
Cadmium	0.99

Lead	35.8
Nickel	22.7
Zinc	121

Table 6b. SQGs for metals in freshwater ecosystems that reflect PECs (above which harmful effects are likely to be observed). MacDonald et al 2000.

Substance	Probable Effect Concentration PEC (Consensus Based)
Metals (mg/kg DW)	
Arsenic	33
Cadmium	4.98
Lead	128
Nickel	48.6
Zinc	459

An important characteristic of these TRVs is that they are based on sediment toxicity tests and field studies of bulk sediments contaminated with mixtures of chemicals, and the spectrum of toxic chemicals present in the sediment may vary from site to site and from sample to sample. Therefore, for sediment samples that are found to cause toxicity in exposed benthic organisms, it is not possible to know which metal (or combination of metals) is responsible for the observed effect.

Predicted Hazards Based on Site-Specific Total Metal Criteria

Figures 8 and 9 in Appendix A, present lead concentrations in ALM sediment and descriptive macroinvertebrate statistical results which are summarized graphically. Inspection of these data reveals that sediments below the PPE in Sutton Branch Creek are predicted to be of concern to benthic organisms, with the high levels of lead in sediment selected as the indicator of reduced EPT (pollution sensitive) and increased chironomid percentages (pollution tolerant) .

Predictions Based on Pore Water Measurements

A direct method for estimating the hazard to sediment-dwelling benthic organisms from metals in sediment is to measure the concentration of metals in the sediment pore water (water that collects in the interstitial spaces from the top of the sediment and downward approximately 10 cm). EPA Region 7 developed a pore water extractor based on instructions from Region 8's Standard Operating Procedure (SOP # EH-03) using a Micro-Push Point (EPA 2004). Results are located in Table 4a.

6.2.3 Hazard from Ingestion of Food Web Items

Data on the impacts to aquatic macroinvertebrates from ingestion of contaminated food items are sparse (Rainbow and Dallinger 1993, Timmermans 1993). Although the general consensus is that uptake from food is usually less than from water (Clements

1994), available data are sufficient to establish that the ingestion pathway can be an important source of exposure to some aquatic macroinvertebrates (Timmermans et al. 1992). Duddridge and Wainwright (1980) suggest that dietary exposures can be capable of limiting growth in at least some cases.

Based on the lack of data on the toxicity of metals in food chain items on aquatic invertebrate receptors, quantitative prediction of hazard using the traditional HQ and HI approach is not yet possible. To the extent that dietary exposures tend to be less important than water exposures in at least some species, failure to quantify the hazard from the ingestion pathway may not lead to a substantial underestimation of total hazard. However, the food pathway may be more important than the water pathway for some metals and/or some receptor species. Therefore, the inability to quantify hazard from ingestion exposures is a potential source of uncertainty that may tend to underestimate impacts of metal contamination on aquatic macroinvertebrate receptors.

6.2.4 Predicted Total Hazard to Benthics from All Pathways

Sections 6.2.1 and 6.2.2 (above) discuss the risk to organisms that are exposed via the surface water column, sediment and its associated pore water. It is important to emphasize that these exposure pathways may be additive. That is, at any one time a specific benthic organism may be exposed to surface water and/or sediment/pore water. It is dependent upon the species and their specific niche. Macroinvertebrates exposed to contaminated surface water and pore water will also be exposed to contaminants via the food pathway.

However, as discussed above, there are currently no quantitative methods available for estimating exposure and hazard from the oral pathway for benthic macroinvertebrates. As discussed in Section 6.2.3, the ingestion pathway may be a lesser source of exposure than water contact in some cases, but probably not in all cases. Thus, the inability to quantify and include this pathway is an important source of uncertainty.

6.3 EVIDENCE OF INCREASED EXPOSURE

Sampling results discussed above have demonstrated that the concentrations of metals of concern are higher in Sutton Branch Creek sediment below the PPE than above the PPE. This implies that aquatic macroinvertebrates below the PPE are more exposed to metals than those above the PPE. Data on metal concentrations in sediment samples that were co-located with the benthic diversity samples show a significant decrease in macroinvertebrate diversity below the PPE and an increase in metal tolerant chironomids (See Figure 8 and 9 in Appendix A).

6.4 SITE-SPECIFIC MACROINVERTEBRATE COMMUNITY STUDIES

In order to establish that benthic macroinvertebrates at the ALM have elevated exposure to metals, a benthic macroinvertebrate community study was performed in Sutton Branch and Big Creek by EPA's Ecological Risk Assessor in October 2003.

Four different stations, two on Sutton Branch Creek (above and below the PPE) and two on Big Creek (above and below Sutton Branch Creek confluence) were sampled for macroinvertebrates, sediment, water column, and pH. The locations and results for each of these stations are summarized below. See Figure 5 in Appendix A for a map of site locations.

Table 6c. Aquatic macroinvertebrate sites and co-located sediment and water column data, EPA, October 2003. Average pH in Sutton Branch Creek was 7.6 and 7.57 in Big Creek. Water column hardness range as $CaCO_3$ was 72.5-90.4 mg/L.

Site	Pb in water column (µg/L)	Pb in sediment (mg/kg)	EPT* Index	% Chiron**
Sutton Branch Creek 1000 ft Above PPE	non-detect	9.94	9	3
Sutton Branch Creek 1000 ft Below PPE	75.5	2600	5	24
Big Creek 1000 ft above Sutton Branch confluence	51.3	341	8	8
Big Creek 500 ft below Sutton Branch confluence	non-detect	387	8	8

* EPT- Ephemeroptera, Plecoptera, Trichoptera-pollution sensitive larvae
** Percent Chironomidae—pollution tolerant larvae

Qualitative Macroinvertebrate samples were collected from riffle area sediments with a D-net in October of 2003.

Measurement Endpoints

The EPA Region 7 Ecological Risk Assessor isolated all of the benthic macroinvertebrates present in each sample, classified each organism to the lowest taxonomic level practical (usually to genus or species), and counted the number of individuals in each group. From these raw data (i.e., the number of individuals in each taxon), the following endpoints were calculated:

Richness

Richness is the number of total benthic macroinvertebrates taxa per unit area at each station. Low values are judged to be one of the most sensitive indicators of environmental stress and metal pollution.

EPT Richness

As noted above, most mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) are sensitive to pollution, including metals pollution. This metric is the sum of the number of species in the EPT taxon.

<u>% Chironomidae</u>

The contribution a certain taxon has to the total number of individuals in a sample multiplied by 100. In this case the % Chironomidae is of interest because this family of macroinvertebrates is widely known to be tolerant of pollution including heavy metal contamination.

Results

Table 6d. Sutton Branch and Big Creek macroinvertebrate metric results, ALM, EPA, October 2003.

Site	Richness	EPT Index	% Chiron
Sutton Branch Creek 1000 ft Above PPE	19	9	3
Sutton Branch Creek 1000ft Below PPE	15	5	24
Big Creek 1000 ft above Sutton Branch confluence	16	8	8
Big Creek 500ft below Sutton Branch confluence	16	8	8

% Chiron= Percentage of Chironomidae in the sample

<u>Macroinvertebrate Taxa Richness</u>

This pattern of results is consistent with the hypothesis that taxa richness is reduced below the PPE in Sutton Branch Creek. Big Creek has reduced richness when compared to Sutton Branch Creek above the PPE; however, Big Creek's EPT Index is approximately the same as Sutton Branch Creeks above the PPE. The EPT Index is a more robust metric, when indicating water quality, than the overall taxa richness

<u>EPT Index</u>

A higher EPT Index correlates with a higher water quality value. Table 6d again shows a pattern of reduced water quality in Sutton Branch Creek below the PPE.

<u>% Chironomidae as Relative Abundance of Indicator Taxa</u>

Additional information on the impacts of environmental contamination on the benthic community may be obtained by focusing on the abundance of specific macroinvertebrate species or genera that are known to be metals-tolerant. Chironomid percentages are found to be higher in metal contaminated waters (Beltman 1999, Reynolds et al. 2002, Smolders et al. 2003, Winner et al. 1980) when compared to reference sites. The % chironomid below the PPE in Sutton Branch Creek is significantly higher when compared to % chironomid in Sutton Branch Creek above the PPE. Also see Figure 8 & 9 in Appendix A.

6.6 WEIGHT OF EVIDENCE EVALUATION

Exposure to Surface Water

Surface water exposure coupled with stresses from pulse events equals reductions in the abundance of sensitive taxa in Sutton Branch Creek below the PPE.

Exposure to Sediment/Pore Water

Benthic macroinvertebrates live in or on stream sediments and are constantly exposed to heavy metal contamination. According to Table 7c above, there are high levels of lead in sediment below the PPE in Sutton Branch Creek.

Direct Observations

Direct observations of benthic community structure and function at the ALM reveal:

- Sutton Branch Creek taxa richness is decreased by approximately 20% downstream of PPE when compared to upstream PPE.
- EPT richness is lower in Sutton Branch Creek downstream of PPE when compared to upstream PPE (by as much as 13%).
- Metal tolerant indicator species (Chironomidae) are increased in Sutton Branch Creek downstream of PPE when compared to upstream PPE (Figures 8 & 9, Appendix A);

These changes are consistent with the hypothesis that metals exposures are associated with these changes. The benthic macroinvertebrate community findings are supported by predictions that lead levels in surface water, sediment, and pore water are occasionally in a range of concern. These results support the hypothesis that reductions in sensitive species and other community changes are due, at least in part, to lead and/or other associated metals. Community-level abundance of benthic organisms is not reduced in Sutton Branch Creek above the PPE or in Big Creek.

Conclusion Regarding Hazard to Benthic Macroinvertebrates

All of the key findings summarized above are generally consistent with each other, and support the view that metals in the aquatic environment are contributing sufficient exposure to alter the composition of the benthic community in Sutton Branch Creek. The evidence indicates that eroding mine waste with it's associated high metal content likely contributes to a decrease in the number of species present, a decrease in sensitive species, and an increase in more pollution tolerant species. Metals impacts appear to be greater in Sutton Branch below the PPE than in Big Creek.

7.0 HAZARDS TO ALGAE

7.1 INTRODUCTION

Aquatic plants were basically non-existent in Sutton Branch Creek below the PPE. Above the PPE in shaded areas of Sutton Branch Creek there were mosses growing on submerged rocks. There was also no observed algae growth on rocks in Big Creek. Copper was detected in one sample from Sutton Branch Creek and copper is highly toxic to algae and other aquatic plants.

Aquatic plants may generally be divided into two broad categories: phytoplankton (free floating algae) and benthic (bottom-dwelling) plants. Benthic plants, in turn, may be categorized as mosses, rooted vascular plants, large filamentous algae, or as periphyton. Periphyton, (or aufwuchs) is an assemblage composed mainly of unicellular benthic algae along with other microflora (bacteria, fungi) that grow on or in close association with the surfaces of submerged substrates.

Benthic algae that occur in periphyton can be divided into two major groups: diatoms and non-diatoms. Diatoms are characterized by the presence of a cell wall composed of silica (the frustule). Non-diatoms, composed mainly green algae and blue-green algae, lack a silica-based frustule.

Algae, particularly diatoms, are useful biomonitors of water quality because they occur in very large numbers. They also have known environmental requirements and pollution tolerances which are unique to individual species. Further, algae are highly sensitive to physical and chemical factors (Weber 1997). Other advantages of using algae for bioassessment include (Plafkin et al. 1989, Bahls 1989):

- Algae are valuable indicators of short-term impacts due to their rapid reproduction rates and short life cycles.
- As primary producers, algae are most directly affected by physical and chemical factors.
- Sampling is easy, inexpensive, and creates minimal impact to resident algal populations.
- Relatively standard methods exist for the evaluation of functional and nontaxonomic structural characteristics (e.g., biomass and chlorophyll) of algal communities.
- Algal communities are sensitive to some pollutants (i.e. nutrients) which may not visibly affect other aquatic communities, or may only affect other communities at higher concentrations.

7.2 PREDICTED HAZARD TO ALGAE FROM METALS IN WATER

Exposure of algae to metals in water generally results in a decrease in diversity and productivity (Vymazal 1994). Sensitivity to metals and other pollutants may vary between species, but the general order of sensitivity is:

Diatoms > Green Algae > Blue-green Algae

The relative toxicity of different metals may also vary widely between species, the most common overall toxicity sequence is (Vymazal 1994):

Cu > Cd > Pb > Zn

Table 7 summarizes available water column toxicity data for each of the metals of concern to algal species that either do occur, or are reasonable surrogates for species that do occur, in Sutton Branch or Big Creek. All of the toxicity values shown in Table 7 are derived from the corresponding AWQC Documents prepared by EPA (1985b-e, 1987).

Table 7. Water column TRVs for algae species that should occur in Sutton Branch and Big Creek, taken from Clark Fork River Ecological Risk Assessment (EPA 1999). TRV Range is algae species specific for each metal. * = no available data

Algae Phylum	TRV Range (ug/L) As	Cd	Cu	Pb	Zn
Chlorophyta (Green Algae)	(48-4700)	(6.1-2500)	(300-8000)	(500-2500)	(20000)
Bacillariophyta (Diatoms)	*	*	(5-85)	*	(4300-20000)

There are a variety of environmental factors which are known to modify the toxicity of heavy metals on algae (Vymazal 1994). This includes agents which may bind dissolved metal ions (organic chelators, humic substances, suspended particles), and water quality parameters such as pH, hardness, ionic strength, temperature, salinity, light intensity, and oxygen level.

7.2.1 Conclusions/Uncertainty

There were no observed algae in Sutton Branch or Big Creek. Note that this is fairly unusual when compared to other Ozarkian creeks that usually have excessive algae growth. However, there is no evidence to suggest that metals are eliminating algae populations. Copper is used to destroy nuisance algae in reservoirs and lakes and there is evidence of elevated copper levels in the water column of Sutton Branch Creek (site 6, 9.12 μg/L, AWQS chronic levels are 9.0 μg/L at 100 hardness).

8.0 HAZARDS TO TERRESTRIAL PLANTS

8.1 OVERVIEW

Terrestrial plant communities within the ALM may be broadly categorized as either riparian or upland. The principal source of contamination in riparian area soils are deposits of tailings along the banks of Sutton Branch Creek. Some of these deposits are exposed, and some are buried beneath various depths of soil. Outside the riparian area, many soils that lie within the current and historic flood plain have been contaminated due

to past flood events, and some areas above the flood plain have been impacted by the standing chat piles.
The plant community within the riparian zone is characterized by species that either tolerate or require moist soils. A more detailed description of the plant communities characteristic of the riparian zone has been presented earlier (see Section 2.5.2). Upland soils are typically much drier and currently are wooded.

Plants are exposed to metals in soil principally through their roots. Exposure may also occur due to deposition of dust on foliar (leaf) surfaces, but this pathway is believed to be small compared to root exposure. Copper and zinc are considered to be essential or beneficial for plant growth (Kabata-Pendias 1992). However, excessive levels of these and other metals in soil may exert a variety of adverse effects on plants including reduced photosynthetic efficiency, reduced seed germination, and reduced rootmass formation. These phytotoxic responses may occur at the scale of the individual plant or may affect the entire plant community, resulting in areas of stressed and unhealthy vegetation. Stressed communities are often subject to invasion by weedy metals-tolerant species, which in turn can result in the disruption and displacement of an entire plant community that would otherwise be found in an affected area. In some locations, lethality to plants can result, and areas with little or no vegetative cover may occur. In the vicinity of the chat piles close to Sutton Branch Creek, areas of exposed tailings support only sparse, no vegetation or the metals tolerant weed horsetail (*Equisetum arvense*). Photographs of representative areas are presented in Appendix A (see Figures 6 and 7).

8.2 PREDICTED HAZARD TO PLANTS

The basic equation used for calculation of a Hazard Quotient (HQ) value for exposure of plants to metals in soils is:

$$HQ_{plant} = \frac{C_{soil}}{TRV_{plant}}$$

where:
C_{soil} = Concentration of metal in soil (mg/kg)
TRV_{plant} = Toxicity Reference Value (mg/kg) for soil exposure of plants.

Soil Concentration Values
Data on the concentration of metals in soils within the ALM have been presented previously (see Table 4c and 4d).

Phytotoxicity TRVs
A relatively large body of literature exists regarding metal phytotoxicity. These studies have shown that the toxicity of metals in soils varies widely between different plant species, and also depends on a large number of soil parameters including soil type, organic content, water content, soil condition, soil chemistry, and soil pH (Adriano 1986. CDM Federal (1996), Kabata-Pendias 1992, Efroymson et al. 1997). This variability is evident by inspection of Table 8, which summarizes phytotoxicity screening TRVs for

metals that have been recommended and used by different authors and groups. As seen, these values vary over an order of magnitude or more for each metal. For the purposes of performing a screening level phytotoxicity evaluation, HQ values were calculated based on Near, Mid and Far Site soil samples using the TRVs for each metal listed in Table 8. None of the TRVs are site-specific, but if all HQ values were below 1 based on the lowest TRV, it would be concluded that hazard of phytotoxicity is low. Conversely, the majority of HQ values based on the highest TRV were substantially higher than 1, it would be concluded that phytotoxicity was likely.

Table 8. Literature based phytotoxicity Toxicity Reference Values (TRVs). Phytotoxic effects include reduced photosynthetic efficiency, reduced seed germination, and reduced root mass formation.

	Phytotoxic Concentration in Soil			
Source	Arsenic	Cadmium	Lead	Zinc
*CDM Federal (1996)	224-315	8.6-40	179-250	196-240
Efroymson et al. (1997)	10	4	50	50
Kabata-Pendias (1992)	15-50	3-5	100-400	70-400

*CDM TRVs based on pH > 6.5, at the ALM the soil pH is above 6.5 for all samples

Table 8a. ALM Terrestrial plant range of Hazard Quotients (HQs) using Near, Mid and Far Pile soil samples collected by the EPA, April 2004. Table 8 TRV values (above) were used to develop the following HQs. CDM and Kabata-Pendiasand HQs are based on the low and high range TRVs. There is no range for Efroymson, et. al. 1997. *CDM TRVs based on pH > 6.5, at the ALM the soil pH is above 6.5 for all samples.

Site	**Arsenic**	**Cadmium**	**Lead**	**Zinc**
Near				
*CDM Federal (1996)	0.3-0.2	0.01-0.01	21.1--15.0	1.7--1.2
Efroymson et al. (1997)	5.9	0.3	473.6	37.5
Kabata-Pendias (1992)	3.9-1.2	0.2--0.1	315.7-94.7	25.0-7.5
Mid-Pile				
*CDM Federal (1996)	0.2-0.1	0.02-0.02	10.9-7.8	1.0-0.7
Efroymson et al. (1997)	3.7	0.5	244.3	22.3
Kabata-Pendias (1992)	2.5-0.7	0.3-0.1	162.8-48.9	14.9-4.5
Far				
*CDM Federal (1996)	0.1-0.05	0.03-0.02	0.7-0.5	0.3-0.2
Efroymson et al. (1997)	1.6	0.7	14.9	7.1

Kabata-Pendias (1992)	1.0-0.3	0.5-0.1	9.9-3.0	4.7-1.4

8.3 Conclusion

Screening Level HQ Table 8a shows the distribution of HQ values predicted on the basis of the lowest TRVs from Table 8 and the soil concentration data from Table 4c and 4d. Inspection of these results reveals the following points:

- When the lowest TRVs from Table 8 are used, predicted HQ values are quite large. The highest values are seen in Near Pile areas, where impacts would be expected in essentially all soil samples The largest contributor to the predicted hazard is lead, but hazards from arsenic, cadmium, and zinc are also predicted.

- When the highest TRVs from Table 8 are used, predicted HQ values are less, but are still in a range of concern for soil samples in Near Pile areas of the ALM. These results indicate that phytotoxicity is likely to be occurring, at least in the most contaminated soil areas near the pile. However, because none of the TRVs used in these calculations are based on studies or measurements using soils from the ALM they should be viewed as screening levels only. True levels of phytotoxicity could be either higher or lower.

9.0 HAZARDS TO TERRESTRIAL VERTEBRATES

9.1 INDICATOR SPECIES AND EXPOSURE PATHWAYS

The terrestrial ecosystem within the ALM supports a wide variety of vertebrate species that may be exposed to chemicals of concern in water, soil, or the food chain. Species that have been selected for quantitative evaluation in this assessment include:

- Shrew
- Otter
- Woodcock
- Canadian Goose
- Great Blue Heron

Shrew- The shrew was selected because it is a vermivore (worm consuming). At the ALM heavy metal mine site, lead (Pb) is the primary chemical of concern (COC). As discussed earlier in Section 3.3, lead does not biomagnify through the food chain. Instead, when organisms consume lead in their daily diet, lead replaces Ca^{2+} in the bone (Eisler 2000), lead inhibits the blood enzyme ALA-D (Schmitt 1997), and lead impacts smaller organisms the most and organisms at younger developmental stages (Eisler 2000).

Earthworms consume soil and inadvertently consume the heavy metals within that soil. Shrews may be getting a dose of heavy metals each time they consume an earthworm at the ALM and shrews may spend their entire lives within the boundaries of the ALM. Shrews are eaten by larger predators. However, larger predators, such as the fox, eat a variety of small mammals and plants and not just the shrew. The fox was not selected for quantitative evaluation because of the variety in its diet, its large foraging range (3 square miles), and the unlikelihood that the larger mammals would be impacted by heavy metals at the ALM.

Otter—The otter has been reintroduced into Missouri and is thriving. It was chosen for quantitative evaluation because it is a piscivore (fish consuming). Schmitt et al. (1997), found lead in the tissues of fish throughout Big Creek, therefore the otter was included in the HQ evaluations.

Woodcock—The woodcock was selected because it is an Avian vermivore. Woodcocks' do consume a variety of other invertebrates, but they prefer the earthworm. The ALM provides the perfect habitat for the woodcock, which consists of wooded areas with interspersed open areas, minimal human interference, and a water source (ODNR 2003). The woodcock is migratory and Missouri is at the southern edge of its nesting range and the northern edge of its wintering range. But, to be conservative we assumed an Area Use Factor (AUF) of 100 % for the woodcock and all the other receptors (Table 9). Further, a migratory factor of 0.5 was applied to the HQ.

Canadian Goose— The Canadian Goose was chosen because it is an herbivore. Several species of plants were analyzed at the ALM including horsetail *(Equisetum arvense)*. Horsetail is abundant and growing in the mine waste at the ALM. Canadian Geese are known to consume a variety of plants that are in local abundance including horsetail. For the purposes of this risk assessment, it was assumed that Canadian Geese ate 100% horsetail. Also, a migratory factor of 0.5 was applied to the HQ.

Great Blue Heron--- The Great Blue Heron is found throughout Missouri and was chosen because it is an Avian piscivore.

These species were selected to be representative of the wide range of other receptors that exist within the ALM, and to include examples of exposures to a range of different environmental media and food web items. Table 9c summarizes the exposure pathways that are judged to be relevant for each receptor.

9.2 PREDICTED HAZARD TO VERTEBRATES

9.2.1 Exposure Variables used in Exposure Equations

Exposure to receptors species chosen for evaluation is based on life histories, ingestion rates and exposure potential for each species. Exposure is dependent upon the concentration of the Chemical of Concern (COC) in the media to which the organism is exposed. Potential ingestion exposures to COC at the ALM site could be from the following sources:

- COC in the tissues of fish (C_F, mg/kg)
- COC in the tissues of earthworms (C_{EW}, mg/kg)
- COC in the tissues of plants (C_P, mg/kg)
- COC in drinking water (C_{SW}, mg/L)
- COC in incidentally ingested sediment (C_{SD}, mg/kg)
- COC in incidentally ingested surface soils (C_{SS}, mg/kg)

The basic equation used for calculation of an HQ value for exposure of terrestrial vertebrates to a chemical by ingestion of an environmental medium is:

$ADD = [(FD)(DW + ISS + ISD + F + EW + V + BF_{Pb})] \times (AUF)$

Where:

ADD = Average Daily Dose (mg/kg body weight / day)
DW = (drinking surface water ingestion dose) = $(C_{SW} \times NIR_W)$
ISS = *(incidental surface soil ingestion dose)* = $[C_{SS} \times (NIR_D \times FD_{ISS})]$
ISD = (incidental sediment ingestion dose) = $[C_{SD} \times (NIR_D \times FD_{ISD})]$
F = (fish ingestion dose) = $[C_F \times (NIR_D \times FD_F)]$
EW = (earthworm ingestion dose) = $[C_{EW} \times (NIR_D \times FD_{EW})]$
V = (vegetation ingestion dose) = $[C_V \times (NIR_D \times FD_V)]$
C_i = Site Concentration Data (See Table 9a Below)
BF_{Pb} = Bioavailability Factor of 0.5 for lead (Pb) only
NIR_D = Normalized Dietary Ingestion Rate (wet weight-g/g BW/day)
NIR_W = Normalized Water Ingestion Rate (wet weight-L/g BW/day)
HR = Home Range (ha)
AUF = Area Use Factor- is area specific since the receptors may not occupy the entire area given and the contaminant might not be uniformly distributed throughout the given area.
FD_i = Fraction Diet, a percentage
FD_{ISS} = Fraction of diet that is incidental surface soil, unitless
FD_{ISD} = Fraction of diet that is incidental sediment, unitless
FD_F = *Fraction of diet that is fish, unitless*
FD_{EW} = Fraction of diet that is earthworms, unitless
FD_P = Fraction of diet that is plants, unitless

Because all receptors are exposed to more than one environmental medium, the total hazard to a receptor from a chemical is calculated as the sum of HQs for that chemical across all media:

$$HQ = \sum HQ$$

If the effects of different chemicals on a receptor act on the same target tissue by the same mechanism, then the total hazard to the receptor may be estimated as the sum of the chemical-specific HQ values across chemicals:

$$HI = \sum HQ$$

At this site, it has conservatively been assumed that effects of all the metals on each of *the receptors are additive.*

Whenever possible, co-located values (CLV) were used in the exposure equations for the ALM. For example, a soil sample was taken within the same area as the earthworm sample. So, earthworm data and its associated soil sample data was used in the equation. Values for these variables are listed below in Tables 9, 9a, 9b, and 9c.

Table 9. Exposure variables used in Average Daily Dose (ADD) equations for receptor species, EPA, 2004.

	Variables									
Receptor Species	FD_{ISS}	FD_{ISD}	FD_F	FD_{EW}	FD_P	BF_{Pb}	NIR_D	NIR_W	HR (ha)	AUF[a]
Shrew	0.02[b]	0	0	0.98[c]	0	0.5[d]	0.62[e]	0.223[e]	0.22[f]	1
Otter	0	0.0006[e]	1[e]	0	0	0	0.24[e]	0.082[e]	400[e]	1
Woodcock*	0.10[g]	0	0	0.90[c]	0	0.5[d]	0.77[e]	0.1[e]	32.4[h]	1
Canadian Goose*	0.082[g]	0	0	0	0.90	0.5[d]	0.033	0.053	2830[i]	1
Great Blue Heron	0	0.002[j]	1	0	0	0	0.18[e]	0.045[e]	3.1[k]	1

* The total BW/day used in HQ evaluations were divided by 2 to account for migration activity

Notes:

a- Area Use Factor (AUF) is 1 at the ALM to be conservative.

b- Based on meadow vole data (Beyer, Connor & Gerould 1994). It was assumed that approximately 10 % of the dietary soil ingestion is from grooming activities and would be ingested (Dames and Moore 1995)

c- It was assumed that these animals eat only earthworms

d- Bioavailability Factor (BF) used for Pb Hazard Quotient (HQ) evaluations for shrew, woodcock, and Canadian goose (EPA, 2004)

e- EPA 1993

f- Platt 1976

g- Beyer et. al. 1994

h- Gregg 1984

i- Eberhardt et. al. 1989

j- Based on the blue-winged teal (*Anas discors*) data (Beyer et. al. 1994). A 10 % dietary soil ingestion from grooming activities was included in evaluation.

k- Great Blue Heron will range up to 3.1 km to forage in rivers and streams (Dowd & Flake 1985)

Table 9a. Numbers in bold are the site concentration data (Ci) used in ADD calculations for selected receptor species at Near, Mid, and Far Pile Areas, EPA, 2004. (see Figure 2 in Appendix A for Area locations)

NEAR PILE AREA

	Cadmium (mg/kg BW-day)			Lead (mg/kg BW-day)			Zinc (mg/kg BW-day)		
Medium	X	95% UCL	CLV*	X	95% UCL	CLV*	X	95% UCL	CLV*
Soil	3.05	4.89	**1.92**	1411	1625	**654**	375	621	**226**
Earthworm	n/a	n/a	**5.11**	n/a	n/a	**332**	n/a	n/a	**185**
Sediment	n/a	n/a	**1.62**	553	903	**962**	46.2	54	50
Plant**	2.68	n/a	**2.9**	1077	n/a	**1160**	214	n/a	**232**
Fish***	n/a	n/a	n/a	n/a	n/a	**4.57**	n/a	n/a	n/a
Water	**.10**	n/a	n/a	**.10**	n/a	n/a	**.10**	n/a	n/a

*Co-Located Value-See Section 9.2.1 (above) for a discussion on CLVs
**Horsetail plant is only located in the Mid-pile area and it was used for Near pile calculations.
*** Whole carcass (Schmitt et al 1993)

MID-PILE AREA

	Cadmium (mg/kg BW-day)			Lead (mg/kg BW-day)			Zinc (mg/kg BW-day)		
Medium	X	95% UCL	CLV	X	95% UCL	CLV	X	95% UCL	CLV
Soil	n/a	n/a	n/a	1411	1625	**756**	n/a	n/a	n/a
Earthworm	n/a	n/a	**1.08**	n/a	n/a	**20**	n/a	n/a	**75**
Sediment*	n/a	n/a	**1.62**	553	903	**962**	46.2	54	50
Plant	2.68	n/a	**2.9**	1077	n/a	**1160**	214	n/a	**232**
Fish**	n/a	n/a	n/a	n/a	n/a	**4.57**	n/a	n/a	n/a
Water	**.10**	n/a	n/a	**.10**	n/a	n/a	**.10**	n/a	n/a

*Near and Mid-Pile sediment are the same values ** Whole carcass (Schmitt et al 1993)

FAR PILE AREA

Medium	Cadmium (mg/kg BW-day)			Lead (mg/kg BW-day)			Zinc (mg/kg BW-day)		
	X	95% UCL	CLV	X	95% UCL	CLV	X	95% UCL	CLV
Soil	n/a	n/a	7.37	n/a	n/a	**149**	n/a	n/a	**71**
Earthworm	n/a	n/a	**1.08**	n/a	n/a	**5.15**	n/a	n/a	**75**
Sediment	n/a	n/a	ND*	25.2	n/a	n/a	10.8	n/a	n/a
Plant	n/a	n/a	**0.12**	n/a	n/a	**7.48**	n/a	n/a	**11**
Fish**	n/a	n/a	n/a	n/a	n/a	**4.57**	n/a	n/a	n/a
Water	**.10**	n/a	n/a	**.10**	n/a	n/a	**.10**	n/a	n/a

* Non-Detect ** Whole carcass (Schmitt et. al. 1993)

Table 9b. Receptor Toxicity Reference Values (TRV) used in HQ calculations in Table 9c. All TRVs were taken from Jasper County Baseline Ecological Risk Assessment (EPA 1998). See Section 9.2.4 (below) for a discussion on TRVs.

Receptor	**TRV**	
	NOAEL	**LOAEL**
Shrew	8	80
Otter	12.5	125
Woodcock	1.1	3.6
Canadian Goose	6	8
Great Blue Heron	3.85	38.5

The following Table 9c lists the HQ results in NOAELs and LOAELs for each receptor and for 3 of the heavy metals (Lead, Cadmium, and Zinc). Arsenic was not included in the table because arsenic values found at the ALM did not exceed screening benchmarks for soil toxicity to earthworms (60 mg/kg, Sample et al 1997), probable effect concentrations (PEC) in sediments (17 mg/kg, MacDonald et. al. 2000) (except for one Sutton Branch sediment sample at site 3 (see Figure 2 in Appendix A) 18.6 mg/kg Ar) or National Ambient Water Quality Criteria (NAWQC) (acute-360 µg/L, chronic-190 µg/L). Nickel was not included because its values also did not exceed screening benchmarks for toxicity to earthworms (200 mg/kg, Sample et al 1997), Probable Effect Concentrations (PEC) in sediments (48.6 mg/kg, MacDonald et al 2000) or NAWQC (acute-1400+ µg/L, chronic- 160+ µg/L).

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Near Pile

Class	Food Guilde	Receptor	Lead (mg/kgBW-day) NOAEL	Lead (mg/kgBW-day) LOAEL	Cadmium (mg/kgBW-day) NOAEL	Cadmium (mg/kgBW-day) LOAEL	Zinc (mg/kgBW-day) NOAEL	Zinc (mg/kgBW-day) LOAEL	Hazard Index (HI)* NOAEL	Hazard Index (HI)* LOAEL
Mammalia	Vermivore	Shrew	13.2	1.32	0.39	0 039	9 1	0.91	22.69	2.269
Mammalia	Piscavore	Otter	0.1	0 01	n/a	n/a	n/a	n/a	0.1	0 01
Aves	Vermivore	Woodcock	61.5	18.7	0 87	0.27	32.4	9.9	94.77	28 87
Aves	Herbivore	Canadian Goose	1.6	1 2	0.003	0.002	0.34	0 25	1 943	1.452
Aves	Piscavore	Great Blue Heron	0.82	0.08	n/a	n/a	n/a	n/a	0 82	0 08

Mid-Pile

Class	Food Guilde	Receptor	Lead (mg/kgBW-day) NOAEL	Lead (mg/kgBW-day) LOAEL	Cadmium (mg/kgBW-day) NOAEL	Cadmium (mg/kgBW-day) LOAEL	Zinc (mg/kgBW-day) NOAEL	Zinc (mg/kgBW-day) LOAEL	Hazard Index (HI)* NOAEL	Hazard Index (HI)* LOAEL
Mammalia	Vermivore	Shrew	1 5	0.15	0.45	0.0045	3 6	0 36	5.55	0 5145
Mammalia	Piscavore	Otter	0.09	0.009	n/a	n/a	n/a	n/a	0 09	0.009
Aves	Vermivore	Woodcock	13.8	4 2	0.21	0.065	13	4	27 01	8.265
Aves	Herbivore	Canadian Goose	1.52	1.14	0.003	0.0025	0.1	0.08	1 623	1 2225
Aves	Piscavore	Great Blue Heron	0.82	0 082	n/a	n/a	n/a	n/a	0 82	0 082

Far Pile

Class	Food Guilde	Receptor	Lead (mg/kgBW-day) NOAEL	Lead (mg/kgBW-day) LOAEL	Cadmium (mg/kgBW-day) NOAEL	Cadmium (mg/kgBW-day) LOAEL	Zinc (mg/kgBW-day) NOAEL	Zinc (mg/kgBW-day) LOAEL	Hazard Index (HI)* NOAEL	Hazard Index (HI)* LOAEL
Mammalia	Vermivore	Shrew	0.33	0.033	0.55	0.055	3.65	0.365	4.53	0.453
Mammalia	Piscavore	Otter	0.54	0.0054	n/a	n/a	n/a	n/a	0.54	0.0054
Aves	Vermivore	Woodcock	1.9	0.58	2.45	0.75	13.09	4	17 44	5 33
Aves	Herbivore	Canadian Goose	0.053	0.04	0.0036	0.003	0.112	0.08	0 1686	0 123
Aves	Piscavore	Great Blue Heron	0.82	0.082	n/a	n/a	n/a	n/a	0 82	0 082

*HI--The Hazard Index adds the metals Hazard Quotients (HQ), the NOAEL/LOAEL, together for a combined final number. This number will help determine the receptors risk to combined metal exposure as opposed to a separate risk from each of the metals.

(mg/kg BW (body weight)-day is wet weight)

9.2.2 Concentration Values

As shown in Table 9, terrestrial receptors may be exposed by ingestion of a wide range of environmental media. The results are discussed below.

Direct Measurements

Surface Water, Sediment, Soil

Direct measurements of metal levels in site media are available for surface water (creek water), soil and tailings, sediment, benthic organisms, fish and aquatic plants. The values for surface water, soil and sediment have been presented previously, as follows:

Surface water, Section 1 and 4, Table 1a, 4a, 4b
Sediment, Section 4, Table 4a, 4b
Soil, Section 4, Table 4c, 4d
Summary, Section 9, Table, 9a

Fish Tissue

As noted earlier in Section 5.0, ALM data on metal tissue burdens in fish were not collected by the EPA. Instead, the EPA used two Big Creek fish surveys (Schmitt et. al. 1993) and (Schmitt et al. 1997) which found elevated concentrations of lead in fish tissue (X = 4.57 mg/kg wet weight-whole carcass). Schmitt (et. al. 1984) also recorded greatly elevated whole lead body burdens of 9 to 18 mg/kg fresh weight in the Big River (0.3 mg/kg fresh weight in edible tissue is considered hazardous to human health) located in the lead belt slightly northeast of Annapolis.

<u>Site Specific Concentrations</u>

For other environmental media (terrestrial plants and soil invertebrates), used in the HQ calculations for the terrestrial receptors in Table 9. Site-specific data is as follows:

Table 9c. ALM Near, Mid, and Far Pile terrestrial plant, earthworm, and pill bug data, EPA, April 2004. Please see Figure 2 in Appendix A for Near, Mid, and Far Pile site map. Metal results are raw values in (mg/kg wet weight). Scientific names of plants can be found in Section 2.5.2. (ND=Non-Detect)

Site	Plant name	Ar	Cd	Pb	Ni	Zn
Near	Cedar	0.56	0.233	139	4.07	22.4
Near	Sumac	ND	0.68	23.9	1.87	103
Near	Dogbane	14.6	ND	602	18.5	64.7
Mid	Horsetail	34.5	2.68	1160	28.9	232
Far	Cedar	ND	0.12	1.08	ND	6.97
Far	Sumac	ND	ND	2.71	0.9	13.6
Far	Mint	ND	ND	7.48	0.5	10.9
Site	Invertebrate	Ar	Cd	Pb	Ni	Zn

Near	Earthworm (*Lumbricus terrestris*)	15.8	5.11	332	23.6	185
Near	Pill Bugs (*Armadillidium vulgare*)	7.44	6.78	110	5.05	228
Mid	Earthworm*	3.55	1.08	20	2.35	75
Far	Earthworm*	0.92	14.4	5.15	ND	73

*Earthworm Tissue Concentrations--Earthworms were collected and their tissues were analyzed for heavy metals without removing the soil from their gut (depuration). EPA R7 finds that terrestrial vermivores may ingest earthworms and gut soils as well as ingesting incidental surface soil.

9.2.3 Exposure Parameters and Intake Factors
Exposure parameters and intake factors for each receptor for each medium were derived from the Wildlife Exposure Factors Handbook (EPA 1993). In some cases, no quantitative data could be located, so professional judgment was used in selecting exposure parameters. See Table 9.

9.2.4 Oral Toxicity Factors and Relative Bioavailability Factors
The EPA has not yet developed standard oral dose-response values for wildlife receptors. Therefore, Toxicity Reference Values (TRVs) were developed for each chemical for each receptor based on the following two sources:

1. Jasper County Superfund Site Baseline Ecological Risk Assessment.1998. Prepared for EPA R7 Kansas City, KS by Black & Veatch, Philadelphia, PA. BVSPC No. 46500.0207.

2. Clark Fork River Ecological Risk Assessment. 1999. Prepared for EPA R8, Denver, CO by ISSI Consulting Group, Denver, CO.

Two different types of TRVs were developed for each chemical for each receptor. The first type is based on a reported exposure level (dose) that is not associated with any adverse effects in the receptor. This is referred to as the No Observed Adverse Effect Level (NOAEL) based TRV. The second type of TRV is based on a reported exposure level that causes an observable adverse effect, and is referred to as the Lowest Observed Adverse Effect Level (LOAEL) based TRV. This range of TRVs is one way to "bracket" the true threshold for adverse effects.

TRVs (both NOAEL-based and LOAEL-based) do not take bioavailability of metals into account. A bioavailability factor (BF) of 0.5 was added to the HQ due to the uncertainty of the exact amount of heavy metal assimilation into organisms' tissues. This adjustment factor of 50% is based on professional judgment, but is supported by evidence that metals in water or food exist in a readily bioavailable form. This concept has been used previously by EPA in the derivation of diet- and water-based Reference Doses for cadmium (IRIS 1998).

When no reliable toxicity data could be located for a receptor of concern, it was necessary to extrapolate toxicity data from studies using another type of receptor. In addition, in some cases available LOAEL TRVs were not available. To account for these data gaps, it is commonplace to multiply the NOAEL times 10 to derive the LOAEL TRV.

9.2.5 Results for Non-Threatened, Endangered or Sensitive Species

Inspection of Table 9c reveals the following main conclusions:

• Predicted hazards vary widely between different types of receptors. Highest hazards are predicted for the vermivores: shrews and woodcocks

• Hazards are highest at Near Pile locations, and tend to decrease as a function of distance from the mine waste piles.

• Even in Far Pile areas, hazards to some receptors (shrew and woodcock) are higher than 1.0. Because it is not expected that hazards should be high in Far Pile areas, these observations suggest that predicted HI values may be overly conservative in some cases.

9.2.6 Results for Threatened, Endangered or Sensitive Species

Section 2.5.6 lists Special Status Species that may occur at the ALM. HQs were not performed for these species, but all vermivores are considered to be at risk when foraging at the ALM.

9.3 WEIGHT OF EVIDENCE EVALUATION

As seen, some receptors are predicted to have little or no hazard of toxic effects from metals in the terrestrial environment, while others are predicted to have moderate to high hazard. For those receptors categorized as having moderate to high hazard, the most important sources of hazard are the following:

- The calculated hazards to the receptors in Table 9c which are above a level of concern in Far Pile areas suggests that some of the TRVs and/or some of the exposure assumptions may be unduly conservative. These predictions should be used to indicate which species are most likely to be impacted, and which media and metals are most likely to be of potential concern.

- In addition to these direct effects of chemicals on terrestrial receptors, it should also be remembered that phytotoxic effects of COCs (see Section 8) may have an indirect effect by reducing or altering the suitability of riparian zone and woodland habitat for some types of receptors, at least on a local scale.

9.4 UNCERTAINTY EVALUATION

As noted above, there is substantial uncertainty in many of the input parameters used to estimate exposure and risk to terrestrial wildlife receptors. The main sources of this uncertainty are summarized qualitatively in the following subsections.

9.4.1 Abiotic Media Concentrations

Nearly all of the calculations of receptor exposure and risk begin with measurements of the COCs in abiotic environmental media (soil, surface water, sediment). As noted in previous sections, because of the substantial variability in concentration values over time and/or space, there is uncertainty in the true concentration values at any particular site location.

9.4.2 Biotic Media Concentrations

Uncertainty is also introduced by the fact that the equations used to calculate HQs are based on limited data. The equations are based on a relatively small number of data points, which adds further uncertainty to the equations derived from those limited samples.

9.4.3 Wildlife Exposure Factors

Even if the concentrations of metals were known with accuracy in all abiotic and all biotic media (food web items), the actual intake of the COC by site wildlife receptors would still be uncertain because of the lack of site-specific knowledge of the actual intake rates. The food, soil, water and sediment intake (ingestion) rates used to estimate COC doses are derived from literature reports of intake rates by receptors at other locations. These rates may or may not serve as appropriate models for site-specific intake rates at this site. In addition, some wildlife receptor-specific intake rates are estimated by extrapolation from data on a closely related species. This introduces further uncertainty into the exposure and risk estimates.

9.4.4 Relative Absorption Efficiency

The toxicity of an ingested chemical depends on how much of it is absorbed from the gastrointestinal tract into the body. However, the actual extent of metal absorption from ingested media (soil, sediment, food, and water) is usually not known.

9.4.5 Toxicity Reference Values

One large source of uncertainty in the risk estimates for terrestrial wildlife receptors are the TRVs. This uncertainty arises from several different sources.

First, toxicological data are in most cases absent for each representative species, and extrapolation from the available toxicity data to the species of concern is needed. Because of the many physiological differences between species, this extrapolation introduces a large source of uncertainty to the risk estimates.

Second, in many cases the available toxicological data are not optimal for identifying a reliable chronic NOAEL or LOAEL. For example, the best available study may use a sub-chronic exposure duration, or may measure an endpoint that is not the most sensitive indicator of chronic toxicity.

In order to account for the data gaps and limitations, uncertainty factors are used to derive TRVs that are inherently conservative (they are more likely to overestimate than underestimate risk). For these reasons, confidence in the TRV (and in the HQ based on that TRV) should be interpreted in view of the size of the uncertainty factor used to estimate the TRV. An uncertainty factor of 0.5 was multiplied by the amount of COC in the soil and was added into the ALM receptor HQs in Table 9c.

9.5.6 Combination of Hazards

The basic approach used for estimating exposure and hazard to terrestrial receptors is to estimate the dose and the HQ for each COC separately, and then to add HQs across all COCs to derive a hazard index (HI). This technique assumes that the adverse effects of each COC are strictly additive and result in a cumulative injury. However, different chemicals act on different tissues of the body by different chemical mechanisms and in some cases the adverse effects are not actually cumulative and the summation across COC specific HQs may tend to overestimate the actual risk.

When two or more COCs interact on the same target tissue, the effects of combined exposure can sometimes be greater than additive (synergistic) and may sometimes be less than additive (antagonistic). The assumption of simple additivity is employed because of a general lack of data on chemical interactions. However, cases of synergy are relatively uncommon, and in the case of metals, cases of antagonism are sometimes noted. Thus, the assumption of additivity is more likely to overestimate than underestimate actual risk. Based on all of these considerations, the HQ and HI values calculated and presented in this section should be viewed as having substantial uncertainty. Therefore, any predictions of risk to site-specific wildlife species should be viewed as an indication that risk may exist, but further studies would be needed to determine if the predicted risk is actually associated with an adverse effect in the field.

10.0 HAZARDS TO TERRESTRIAL SOIL ORGANISMS

10.1 OVERVIEW

Soil organisms are defined as organisms that live in soil during an essential part of their life cycle. This includes both soil invertebrates (earthworms, pill bugs, etc.), and soil microbes (bacteria and fungi).

Soil organisms are important components of the terrestrial ecosystem not only because they are prey for other species, but also because they contribute to litter breakdown. Soil invertebrates fragment and partially solubilize organic matter, while soil microorganisms mineralize complex organic molecules to simple molecules that can be taken up by roots. Earthworms are probably the most important soil invertebrate in promoting soil fertility (Edwards 1992). Their feeding and burrowing activities break down organic matter, release nutrients, and improve soil aeration, drainage and aggregation. Earthworms are also important components of the diets of many higher animals (vermivores).

10.2 PREDICTED HAZARD TO SOIL ORGANISMS

Data on the concentrations of metals in soil organisms was presented earlier in Table 9a.

Soil Organism TRVs

Soil screening benchmarks for the protection of soil organisms have been developed by three different groups, including Oak Ridge National Laboratory (ORNL) (Efroymson et al. 1997), the National Institute of Public Health and the Environment (RIVM 1997), and the Canadian Council of Ministries of the Environment (CCME 1997). The values developed by each of these groups are summarized below:

Table 10. Soil screening benchmarks for the protection of soil organisms, units are in mg/kg dry weight.

Metal	ORNL Earthworm Benchmark	CCME Benchmark	RIVM Benchmark
Arsenic	60	20	40
Cadmium	20	3	12
Lead	500	375	290
Zinc	100	600	720

None of these TRVs are site-specific. If all HQ values were found to be below 1, it would be concluded that hazard to soil organisms is low and if a majority of HQ values were found to be higher than 1, it would be concluded that toxicity to soil organisms is likely.

Table 10a. ALM earthworm (*Lumbricus terrestris*) Screening Level Risk Calculation Hazard Quotients (HQs) using Near, Mid pile, and Far Pile samples collected by the

EPA, April 2004. Table 10 TRV values (above) were used to develop the following HQs. Since the earthworm data is in wet weight and the above TRVs are in dry weights, the invertebrate data was converted to dry weights (concentration multiplied by 0.20).

Site	Arsenic	Cadmium	Lead	Zinc	HI
Near					
ORNL	0.20	0.03	1.89	0.75	2.87
CCME	0.59	0.20	2.53	0.12	3.44
RIVM	0.29	0.05	3.27	0.10	3.71
Mid-Pile					
ORNL	0.13	0.05	0.98	0.45	1.60
CCME	0.38	0.35	1.30	0.07	2.19
RIVM	0.19	0.09	1.68	0.06	2.02
Far					
ORNL	0.05	0.07	0.06	0.14	0.33
CCME	0.16	0.49	0.08	0.02	0.75
RIVM	0.08	0.12	0.10	0.02	0.32

10.3 CONCLUSION

Predicted HQ and HI Values

Table 10a shows the distribution of HQ values predicted based on the literature-based TRVs for soil organisms. Inspection of these results reveals the following points:

• Predicted HQ values for soil organisms are greater than 1 for lead in Near Pile sites for all 3 TRVs. HI values exceed 1 for all Near and Mid-Pile TRVs. The largest contributor to the predicted risk is lead followed by Zinc, Arsenic, and Cadmium.

• When the highest TRVs are used, predicted HI values are much lower (Table 10a), but are still in a range of concern for lead in soil samples at Near and Mid-Pile areas. These results indicate that toxicity of metals in soils to soil organisms is likely to be occurring, at least in the most contaminated soil areas.

However, because none of the TRVs used in these calculations are based on studies or measurements using soils from the ALM, these predictions should be considered as screening level only. The true levels of toxicity to soil organisms could be higher or lower.

11.0 SUMMARY AND CONCLUSIONS

11.1 OVERVIEW

The ALM Site is in the location of a former lead (Pb) mine which reportedly operated during the 1920s to the 1940s. The mining activities included the excavation of ore bodies, the crushing and concentrating of the ore and the storage of the concentrated metals prior to off site shipment for smelting. The wastes from crushing and concentrating were disposed of on the surface of the property within a small ravine. It is believed that an estimated 1,173,000 tons of tailings have been disposed of on 10 acres of the site. Through the years the waste has eroded off the 10-acre pile down into the adjacent floodplain including Sutton Branch and Big Creek.

The principle reasons for concern at the ALM are as follows:

- EPA analytical data documenting metal contamination (arsenic, cadmium, lead, nickel, and zinc) in sediments and surface water above background concentrations.

- There is obvious and substantial contamination of Sutton Branch Creek and the adjacent floodplain with visible and buried tailings. Sutton Branch Creek is a small tributary to Big Creek but contains a diverse number of organisms including an endemic crayfish named the Big Creek crayfish (*Oronectes peruncus*). Big Creek is a perennial flowing waterbody and a Missouri Outstanding Resource Water that is ecologically and recreationally important.

- Tailings are known to contain a variety of different metals which may, if exposure is high enough, be toxic to a wide variety of different environmental receptors.

- At some locations, evidence of terrestrial phytotoxicity is readily detectable by simple visual inspection (nothing is growing on the mine waste).

- In June, 1992, there was a massive "fish-kill" event in Big Creek that occurred after the rupture of a dam (in Annapolis) constructed of rhyolite tailings (Schmitt 1997). This information suggests that fish may succumb to pulses of mine waste, so flood events of eroding mine waste are potentially hazardous to the health of fish.

11.2 WEIGHT OF EVIDENCE APPROACH

The EPA conducted several studies at the ALM including sediment, pore water, surface water, macroinvertebrates, soil, terrestrial invertebrates, and plant samples. Further, the EPA hired contractors to conduct several other/or similar studies at the ALM (see section 2.2 and Table 2). MDNR conducted sediment and surface water sampling at the ALM and two USFWS fish surveys were also performed in Big Creek within the boundaries of the ALM.

To take advantage of all this data, the process of assessing the impact of metals on each major component of the ecosystem (fish, benthic macroinvertebrates, algae, terrestrial plants, terrestrial animals, soil organisms) was performed using a weight of evidence approach, consisting of two major elements:

1. <u>Predictive Approach</u>
This approach is based on the comparison of site exposure levels to literature based exposure levels that are believed to cause no or minimal toxic effects. The ratio is referred to as a Hazard Quotient (HQ):

$$HQ = \frac{\text{Site Exposure}}{\text{Reference Exposure}}$$

HQ values less than 1 indicate effects are not expected, while values above 1 indicate effects may occur. In most cases, HQ values are not based on site-specific toxicity data, and do not account for site-specific factors that may either increase or decrease the toxicity of the metals compared to what is observed in the laboratory. Therefore, HQ values should be interpreted as estimates rather than highly precise predictions.

Because most receptors are exposed to more than one chemical by more than one route, HQ values may be added to yield an estimate of the total risk. The sum is referred to as the Hazard Index (HI). Adding HQ values across different chemicals assumes that the chemicals cause injury to the same tissues of the body, an assumption that is not always true. Thus, HI values are best interpreted as a screening level estimate of total hazard, and the true hazard is likely to be lower.

2. <u>Direct Observations of Receptor Diversity and Abundance</u>
A second approach for evaluating impacts of environmental contamination on ecological receptors is to make direct observations on the receptors in the field, seeking to determine whether any receptor population has unusual numbers of individuals (either lower or higher than expected), or whether the diversity (number of different species) of a particular category of receptors (i.e. plants, benthic organisms, birds) is lower than expected. The chief advantage of this approach is that direct observation of community status does not require making the numerous assumptions and estimates needed in the HQ approach. However, there are also a number of important limitations to this approach. The most important of these is that both the abundance and diversity of an ecological population depend on many site-specific factors (habitat suitability, availability of food, predator pressure, natural population cycles), and it is often difficult to know what the expected (un-impacted) abundance and diversity of an ecological population should be in a particular area. This problem is generally approached by seeking an appropriate "reference area" (either the site itself before the impact occurred, or some similar site that has not been impacted), and comparing the observed abundance and diversity in the reference area to that for the site. However, it is sometimes quite difficult to locate reference areas that are truly a good match for all of the important habitat variables at the site, so comparisons based on this approach do not always establish firm cause-and-effect conclusions regarding the impact of environmental

contamination on a receptor population.

As discussed above, methods available for evaluating potential impacts of environmental pollution on ecological receptors has advantages but also has limitations. For this reason, conclusions based on only one method of evaluation may be misleading. Therefore, the best approach for deriving reliable conclusions is to combine the findings of all methods for which data are available, taking the relative strengths and weaknesses of each method into account. If the methods all yield similar conclusions, confidence in the conclusion is greatly increased. If different methods yield different conclusions, then a careful review must be performed to identify the likely basis of the discrepancy, and to decide which method is more likely to yield the correct conclusion.

11.3 IMPACTS OF MINE WASTES ON THE AQUATIC ECOSYSTEM

Potential impacts of toxicity from mining-related contaminants (arsenic, cadmium, lead, *nickel, zinc) on the aquatic ecosystem within the ALM were evaluated in four parts.* Each of these four parts is summarized below.

1) Impacts to the Aquatic Community As a Whole

Potential hazards to the aquatic community, taken as a whole, were evaluated by comparing measured concentrations of metals in surface water with EPA chronic Ambient Water Quality Criteria (AWQC). This was done for dissolved metals. The only chemical that exceeds the AWQC value is lead (Table 2a). These results indicate that mining related contaminants in surface water (especially lead) may pose a hazard to the aquatic community. However, it is not possible to judge from these results which specific types of aquatic receptor may be at risk, or the nature or magnitude of that risk. For this reason, a more detailed evaluation was performed to investigate potential hazards to fish, benthic macroinvertebrates, and aquatic algae, as described below.

2) Impacts to Fish (Section 5)

Acute Lethality and Chronic Toxicity

It is helpful to evaluate the hazards of acute lethality and toxicity in fish from metals in surface water by considering two separate exposure scenarios:

a) Hazard from "typical" (non-storm pulse) conditions

b) Hazard from peak concentrations that occur during some storm events. The weight of evidence regarding each type of hazard is summarized and evaluated below.

Hazard from Typical (Non-Pulse) Surface Water Concentrations

The hazard to fish from the mixture of metals that occur at the ALM is dominated by lead. Between 1992 and 2004, concentrations of dissolved lead in Sutton Branch Creek measured under "typical" (non-storm event) conditions ranged from 1.6-75.5 ug/L. Big Creek water column concentrations ranged from <10 to 51.3 ug/L with a water hardness of approximately 100 mg/L.

Hazard from Storm-Related Pulses in Surface Water Concentrations

Historically, there has been at least one documented fish kill in Big Creek. This is thought to be due to a dam breakage of rhyolite tailings and may be considered to be a one time event. However, observed fish populations by EPA were diminished in Sutton Branch and Big Creek.

In recent years (1997-2004), no storm-related fish kill events have been reported within the ALM, and no data have been obtained to document the occurrence of acutely lethal concentrations of metals in Sutton Branch or Big Creek. However, absence of observed fish kills is not proof that fish kills are no longer occurring, and available monitoring data are not adequate to establish that short-term pulse events are not occurring. Because the basic source material remains in place, and because run-off waters from exposed tailings are known to contain very high levels of metals, it is concluded that the risk of acutely lethal pulses remains.

Chronic Toxicity

Fish receive chronic exposure to metals from three pathways :

- Direct contact with surface water
- Ingestion of metals in prey items
- Incidental ingestion of metals in sediment while feeding

Predicted Hazard from Chronic Exposure to Surface Water

Long-term exposure of fish to elevated levels of metals in surface water can result in decreased growth (Eisler 2000). Quantitative data on the concentration of metals that result in significant growth inhibition in fish are not available at the ALM and HQ calculations were therefore not performed.

Metals in Fish Tissue

Two Big Creek fish surveys were performed (Schmitt et al. 1993) and (Schmitt et al. 1997) and found elevated concentrations of lead in fish tissue (4.57 mg/kg wet weight-whole carcass) and high concentrations of cadmium (1.2 mg/kg wet weight-whole carcass). Schmitt (et al. 1984) also recorded greatly elevated whole lead body burdens of 9 to 18 mg/kg fresh weight in the Big River (0.3 mg/kg fresh weight in edible tissue is considered hazardous to human health) located in the lead belt slightly northeast of Annapolis.

Population Observations

The density and diversity of fish including suckers (Catastomidae) are lower in Sutton Branch and Big Creek than in similar Ozarkian streams. Even though it is possible that some of the differences observed may be attributable to habitat factors, there is still strong evidence that some of the differences are attributable to metals exposure.

Conclusion Regarding Chronic Hazard

Taken together, the information above is consistent with the hypothesis that metals in the aquatic environment (surface water, diet) is (are) imposing an intermittent low-level chronic stress on fish populations. It is unknown to what degree this chronic stress contributes to population-level effects (such as the decrease in standing fish populations). Based on the available literature, it is concluded that acute exposures to pulses or other high-concentration events are more important than chronic stresses in causing population-level effects, since even intermittent fish kills from pulse events could lead to significant reductions in fish populations. It is also concluded that decreases in fish populations may be due in part to other (non-metal) stressors. Available data are not sufficient to ascribe quantitative estimates to the relative importance of these factors, especially since the relative importance may vary widely from year to year.

3) Impacts to Benthic Macroinvertebrates (Section 6)

Benthic macroinvertebrates may be exposed to tailings-derived contaminants in the aquatic environment by the following pathways.

- Direct contact with metals in creek water (this pathway is most applicable to species which live on or close to the surface of the sediment substrate)
- Direct contact with metals dissolved into the interstitial water occupying the spaces between sediment particles (this pathway is most applicable to species that live buried within the sediment substrate)
- Ingestion of food web items (e.g., algae, diatoms, detritus, other benthic organisms) which have incorporated levels of metals into their tissues
- Incidental ingestion of fine sediment particles in association with normal feeding activities

The relative importance of these different exposure pathways is not known, and is likely to vary considerably among different species of benthic organisms. In addition, there may be considerable temporal variation in the relative importance of each pathway.

Exposure to Sediment/Pore Water

Benthic macroinvertebrates live in or on stream sediments and are constantly exposed to heavy metal contamination. According to Table 7c, there are high levels of lead in sediments below the PPE in Sutton Branch Creek.

Direct Observations

Direct observations of benthic community structure and function at the ALM reveal:

- Sutton Branch Creek taxa richness is decreased by approximately 20% downstream of PPE when compared to upstream PPE.
- EPT richness is lower in Sutton Branch Creek downstream of PPE when compared to upstream PPE (by as much as 13%).
- Metal tolerant indicator species (Chironomidae) are increased in Sutton Branch Creek downstream of PPE when compared to upstream PPE (Figures 8 & 9, Appendix A);

These changes are consistent with the hypothesis that metals exposures are associated with these changes. The benthic macroinvertebrate community findings are supported by predictions that lead levels in surface water, sediment, and pore water are occasionally in a range of concern. These results support the hypothesis that reductions in sensitive species and other community changes are due, at least in part, to lead and/or other associated metals. Community-level abundance of benthic organisms is not reduced in Sutton Branch Creek above the PPE or in Big Creek.

Conclusion Regarding Hazard to Benthic Macroinvertebrates
All of the key findings summarized above are generally consistent with each other, and support the view that metals in the aquatic environment are contributing sufficient exposure to alter the composition of the benthic community in Sutton Branch Creek. The evidence indicates that metals probably contribute to a decrease in the number of species present, a decrease in sensitive species, and an increase in more tolerant species. Metals impacts appear to be greater in Sutton Branch below the PPE than in Big Creek It is important to note that pulse events in Big Creek coming from Sutton Branch Creek may be stressing the benthos and other populations. Also, metals may not be the only stressor, but Sutton Branch Creek does not receive organic pollution and/or nutrient loading from point sources.

4) Impacts to Algae (Chapter 7)
Weight of Evidence Conclusion for Algae
There was no observed algae in Sutton Branch or Big Creek. Note that this is fairly unusual when compared to other Ozarkian creeks that usually have too much algae growth. However, there is no evidence to suggest that metals are eliminating algae populations. Copper is used to destroy nuisance algae in reservoirs and lakes and there is evidence of elevated copper levels in the water column of Sutton Branch Creek (site 6, 9.12 μg/L, AWQS chronic levels are 9.0 μg/L at 100 hardness).

Synthesis of Observations Across Aquatic Taxa
As discussed above, available data derived from various studies performed at the ALM indicate that:
a) algae are basically non-existent

b) the overall health of the benthic community is generally good, although some shifts in the composition of the benthic community (due to decreased abundance of sensitive species) are detectable in Sutton Branch Creek below the PPE

c) the average density of fish (especially Catastomidae) was observed to be decreased in Sutton Branch and Big Creek

11.4 IMPACTS OF MINE WASTE ON THE TERRESTRIAL ECOSYSTEM
Potential impacts of toxicity from mining-related contaminants (arsenic, cadmium, lead, nickel, and zinc) on the terrestrial ecosystem within the ALM were evaluated in three parts, as summarized below.

1) Impacts To Terrestrial Plants (Chapter 8)

The weight of evidence is strong that tailings materials present in the root zone of riparian area soils are phytotoxic to terrestrial plants. This conclusion is supported by all three types of data evaluation, including:

1) Laboratory tests of metals reveal high levels of lead in some plant tissues (Table 9a).

2) Direct observations in the field in the Near Pile areas with extensive visible mine waste. Phytotoxicity is evident due to little, no, or stressed plant growth.

3) Table 8a HQ ranges for plants via the soil reveals HQs greater than 1 at all 3 ALM areas (Near, Mid, and Far Pile).

2) Impacts To Terrestrial Animals (Chapter 9)
The calculated hazards to terrestrial vermivores in Table 9 are at a level of concern at the ALM in all areas. In addition, phytotoxic effects of COCs (see Section 8) may have an indirect effect by reducing or altering the suitability of riparian zone and woodland habitat for some types of receptors, at least on a local scale.

3) Impacts To Soil Organisms (Chapter 10)
The weight of evidence is strong that a potential hazard does exist to soil organisms (worms, microbes), at least in Near and Mid-Pile areas at the ALM. This is based on the predictive (TRV) analysis (Table 10a).

11.5 INTERACTIONS BETWEEN ECOSYSTEM COMMUNITIES

It is important to recognize that impacts of mine waste on one component of the ecosystem may also affect other parts of the ecosystem. Some types of potential interactions are discussed below, along with a qualitative evaluation of their potential significance.

• Decreases in fish population stemming from mining-related contamination could tend to decrease the abundance of piscivores (i.e., herons, eagles, otters, mink, etc).

• Decreases in terrestrial plants in mine waste areas and other locations of stressed or absent vegetation in the riparian zone may decrease the abundance of some types of terrestrial receptors that utilize that habitat for food or cover.

• Decreases in plant density along the creeks edge may reduce shade and cover for fish, and reduce the amount of organic detritus entering the river.

• Mine waste areas of sparse vegetation can serve as a source of sediment run-off into Sutton Branch Creek, increasing sediment loading and substrate embeddedness, which potentially impacts the availability of suitable spawning areas for fish and habitats for benthic macroinvertebrates and algae. This alteration in habitat may influence the nature of both the benthic and algal communities. The degree to which spawning is affected is not known, but could be significant.

11.6 SUMMARY

There is ample evidence that both the aquatic and the terrestrial environments within the ALM are contaminated by mining-related wastes, that living organisms within both ecosystems have elevated exposure to mining-related metals, and that the metals do cause adverse effects on at least some receptors in each ecosystem. Specific conclusions regarding the impact of these elevated exposures are summarized below:

Aquatic Ecosystem

- Algae appear to be non-existent.

- There is a decrease in taxa richness and in the abundance of some sensitive species of benthic macroinvertebrates in Sutton Branch Creek below the PPE and this is probably due, in part, to metals.

- The density of fish in Sutton Branch and Big Creek is lower than in typical Ozarkian creeks and metals are judged to be a contributor to this effect. The metals-related exposure pathway contributing to this decrease in population is not certain, but is more likely related to acutely lethal pulse events than to ambient levels of metals in surface water or the diet.

Terrestrial Ecosystem

- There is clear evidence of phytotoxicity to terrestrial plants in mine waste areas.

- There is good evidence that soil organisms (worms, microbes) are adversely impacted by soils from mine waste.

- The hazard to some terrestrial vermivores is predicted to be quite high in all areas of the ALM.

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APPENDIX A

Table A1. Annapolis Lead Mine Near and Mid Pile XRF Pb data (XRF was not performed at the Far Pile location), EPA 2004. See Figure 2 for a map of Near, Mid and Far pile areas within the ALM.

Near Pile XRF data, 415--15.24 x 15.25 meters (50ft^2) cells (mg/kg)

220	718	592	104	65	97	945	1263	1643	1323
611	50	94	86	776	885	589	532	1200	1093
84	91	79	105	1686	703	629	1877	2590	1423
525	92	103	600	1416	1786	1616	1417	1177	281
471	69	89	291	338	134	1453	1367	864	113
452	120	98	433	317	1530	1627	1310	1104	989
967	102	93	112	78	736	757	1133	133	176

238	66	709	1040	1157	1233	668	1283	1376	154
180	160	613	1127	682	1340	1450	109	315	568
678	530	487	285	584	1113	1977	1300	756	1296
1063	1055	770	523	743	968	812	1353	780	911
1216	1133	757	736	1280	1086	1127	1277	682	1340
1450	1075	1113	750	1145	1007	1313	1380	1303	1673
1247	1387	1560	1096	988	1383	1340	1150	1363	1216
831	1385	1220	1580	1476	502	1336	1160	1303	1273
1110	1200	572	2326	1896	1673	986	924	1173	1650
522	2530	7220	25033	3720	1726	1727	2380	1439	2193
3360	4660	1360	1726	513	98	300	913	1677	5123
2303	2426	1420	397	309	318	19300	5560	4130	3120
1517	740	665	640	3037	1070	1813	1135	1575	4860
8220	13966	9710	1213	230	2483	1906	1143	1206	913
619	371	963	3277	2353	2433	1433	1636	456	219
183	215	2883	3866	4316	878	175	189	204	1163
1506	214	185	1076	1716	1760	1710	1576	1193	1340
1280	879	202	125	159	192	218	227	405	1350
3853	5460	2290	913	208	177	169	164	128	651
170	255	1296	3446	3503	1011	2753	567	216	170
207	137	213	213	228	217	469	936	1406	1890
1280	1413	1393	871	430	218	138	187	262	194
244	313	330	755	1623	1386	615	176	197	152

144	540	100	629	2533	9340	11866	9263	7286	3683
2276	3256	878	2306	1616	446	676	755	735	1156
4500	775	2776	4563	1573	3080	378	468	921	1556
1680	183	126	132	111	111	113	163	995	758
249	154	204	133	130	178	514	480	134	124
126	145	97	208	574	138	114	105	128	2190
5276	3506	8866	961	598	415	314	265	2403	2586
3970	3036	2556	1966	1360	587	466	369	1273	1826
4793	4910	1626	1953	656	447	459	355	224	84
872	153	2333	339	5533	648	5246	2080	2313	865
362	743	1693	4623	4440	1058	329	153	163	1186
885	695	883	815	148					

Mean (X) = 1411 mg/kg, STD (σ) = 2220
95% CI = (1198, 1625), 95% UCL (1625)

Mid-Pile XRF data, 199—15.24 x 15.25 meters (50ft$^{2)}$ cells (mg/kg)

371	725	210	386	216	279	174	267	101	177
302	99	322	514	101	502	949	253	814	297
1666	2140	2636	2576	1823	1266	142	80	94	115
1583	1480	1783	2073	1276	1320	581	95	76	866
923	1213	1610	1218	1326	1313	1633	364	92	127
215	1240	1620	1403	1246	2176	1643	222	146	343
1756	1443	1263	1276	1350	979	200	1186	700	189
166	583	1153	398	1596	1068	247	343	113	213

596	1300	1633	1486	788	98	199	103	134	493
1663	501	777	148	120	140	145	850	344	438
740	113	106	89	129	723	756	1186	281	55
74	110	598	1256	987	1103	846	1048	970	210
137	242	111	1163	1363	990	903	962	876	1140
1156	957	1046	510	169	139	110	134	135	1220
1233	1136	1170	695	1386	949	821	117	184	98
722	1080	981	1726	822	820	819	1173	1017	114
105	147	1073	1053	525	97	105	1076	1090	1190
162	92	83	138	1323	1500	1440	429	101	101
465	1440	1266	1426	288	143	784	1263	2256	1436
1263	1383	699	363	384	930	305	176	1383	

Mean (X) = 756 STD (σ) = 591
95% CI = (639,840) 95% UCL (840)

ADDENDUM
DRAFT Annapolis Mine Superfund Site
Baseline Ecological Risk Assessment
Iron County, Annapolis, Missouri

REEVALUATION OF RISK TO TERRESTRIAL VERMIVORES, AQUATIC MACROINVERTEBRATES AND FISH
ANNAPOLIS MINE SITE, MISSOURI

July 2005

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Background

The Final Baseline Ecological Risk Assessment (Final BERA) for the Annapolis Mine Superfund Site evaluated risk to the biological function of terrestrial and aquatic ecosystems (EPA 2005). Risks were evaluated as conditions that would affect the production and maintenance of:

- Piscivorous vertebrates (Otter)
- Avian piscivores (Great Blue Heron)
- Aquatic invertebrate-eating receptor (Suckers)
- Vermivorous (earthworm consuming) vertebrates (Short-Tailed Shrew and Woodcock)
- Herbivorous vertebrates (Canadian Goose)

Ingestion exposure risks, expressed as a Hazard Index range (HI), indicated little to no risk to piscivores and herbivores. Fish (Suckers) were found to have assimilated lead into their tissues and blood and therefore have been found to be chronically impacted (Schmitt et al. 1984, 1992, 1993, 1997, 2002). Mine waste loading into Big Creek caused a documented fish kill which is an acute impact. Aquatic macroinvertebrates were found to have higher numbers of metal tolerant species and lower numbers of metal intolerant species in Sutton Branch Creek below the mine waste Point of Entry (PE) when compared to macroinvertebrate diversity above the PE. The Hazard Indices presented in the Final BERA for lead and zinc indicated that vermivores (Shrews and Woodcocks) were also at risk at the Annapolis Lead Mine site (EPA 2005).

The Annapolis Lead Mine Site was capped and the majority of post lead (Pb) soil levels are now (< 400mg/kg). This addendum contains a reevaluation of risks to vermivores, aquatic macroinvertebrates and fish.

Table 1. Annapolis Lead Mine Site reevaluated Hazard Quotient (HQ) receptor results using clean up levels of 400 mg/kg, EPA, July 2005. The earthworm tissue results used in the reevaluated HQ equations were taken from the Mid-Pile Area of the Annapolis Mine Site (*20 mg/kg Pb).

		Lead (Pb) (mg/kg body weight/day)			
Food Guilde	**Receptor**	**NOAEL**		**LOAEL**	
		Pre Cap	Post Cap	Pre Cap	Post Cap
Vermivore	Shrew	13.2	1.1	1.32	0.11
Vermivore	Woodcock	61.5	8.8	18.7	2.7

*R7 Ecological Risk Assessor hypothesized that the clean-up level of 400 mg/kg would reduce the Pb levels in earthworm tissue to Mid-pile levels. *Annapolis Mine Site Mid-Pile Pb soil levels were 756 mg/kg (EPA 2005).*

Discussion

Vermivores- According to Table 1, vermivores are still at risk of accumulating Pb into their tissues at 400 mg/kg. However, R7s Ecological Risk Assessor acknowledges that capping the Annapolis Lead Mine Pile has greatly reduced the risk to terrestrial wildlife by a factor of 12 for shrews and 7 for woodcock for a No Observed Adverse Effect level (NOAEL). Additionally, the background concentrations of lead in soils historically collected was 300 mg/kg (Sverdrup 1996) and is close to clean-up levels.

Aquatic Macroinvertebrates- Sutton Branch Creek sediments were visibly dominated with mine waste below the Point of Entry (PE). Lead concentrations found in sediments below the PE were as high as 4,800 mg/kg (Sverdrup 1996). The pile is approximately 350 feet east of Sutton Branch Creek and the mine waste enters Sutton Branch Creek via a natural ravine during storm events. The waste has been eroding into Sutton Branch and Big Creek for 60-80 years.

Winner et al. (1980) found that aquatic macroinvertebrate community structure exhibits a predictable response to heavy metal contamination. He found that sensitive macroinvertebrates such as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) was virtually eliminated in sections of streams contaminated with heavy metals. But, tolerant species such as chironomids and tubificid worms were abundant.

Sutton Branch Creek followed this same community structure response when contaminated sections below the PE were compared to upstream reference sections. However, now that the Annapolis Mine Waste Pile has been capped, erosion of mine waste into Sutton Branch has ceased. Therefore, given time and no further disturbance, R7 believes that macroinvertebrate diversity in Sutton Branch Creek below the PE will slowly recover. R7s Ecological Risk Assessor will monitor macroinvertebrate recovery for at least five years beginning in 2006.

Fish- The density and diversity of fish including suckers (Catastomidae) were observed to be lower in Sutton Branch and Big Creek than in similar Ozarkian streams. It is possible that some of the differences observed may be attributable to habitat factors, but there is still strong evidence suggesting that some of the differences are attributable to metals exposure and/or the turbidity of the mine waste itself.

The capped Annapolis Mine waste pile will no longer be eroding into Sutton Branch Creek. Consequently, given time, diverse plants will grow in the riparian corridor along Sutton Branch Creek, macroinvertebrates will slowly recover (and that recovery will be monitored), and in turn fish will come back to feed on the macroinvertebrates in a habitat more typical of Ozarkian streams. Fish sightings will be recorded during macroinvertebrate sampling events.

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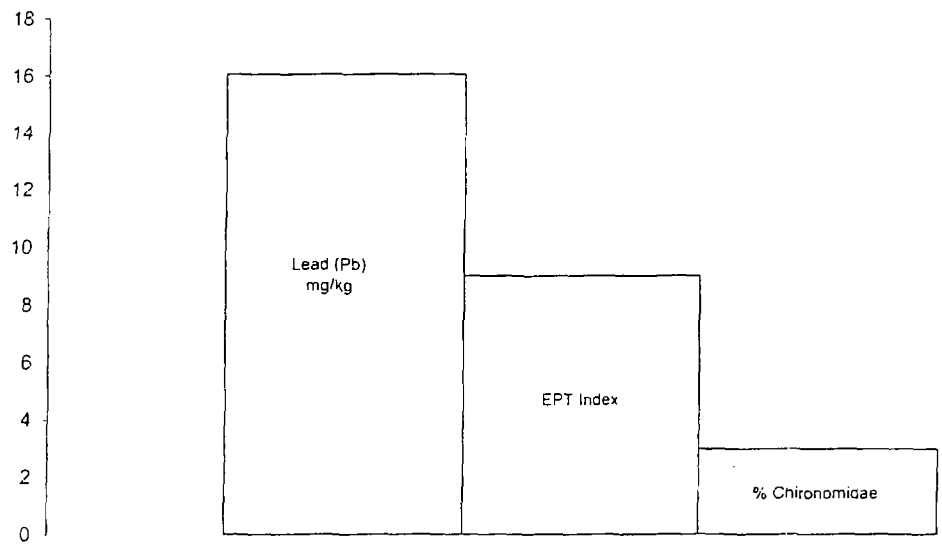

Figure 8. Average Lead (Pb) levels, Ephemeroptera, Plecoptera, and Trichoptera (EPT Index) (sensitive species), and % Chironomidae (tolerant species) in Sutton Branch Creek above the Probable Point of Entry (PPE), EPA, 2004.

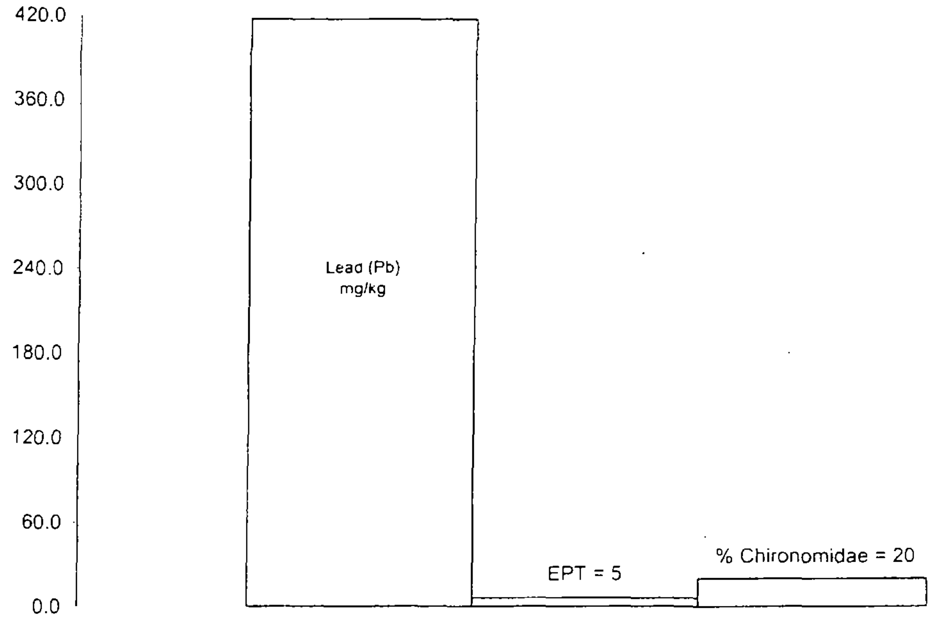

Figure 9. Average Lead (Pb), EPT, and % Chironomidae in Sutton Branch Creek below the PPE, EPA, 2004.

FEASIBILITY STUDY
ANNAPOLIS LEAD MINE SUPERFUND SITE
CERCLIS ID # MO0000958611
Operable Unit-3 (Town of Annapolis)
ANNAPOLIS, IRON COUNTY, MISSOURI

Prepared by

United States Environmental Protection Agency
Region VII
901 North 5th Street
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February 2007

I. Site Description

The Annapolis Lead Mine (ALM) is located approximately 0.75 mile north of Big Creek. The source area was a lead mine that operated between the years of 1919 and 1940. Mining activities at the site included the excavation of ore bodies, the crushing and concentrating of ore, and storage of the concentrated metals prior to off-site shipment for smelting. The crushing and concentrating wastes (tailings) were disposed of on the surface of the property within a ravine that is a tributary of Sutton Branch Creek. The resulting pile of tailings has been stabilized under an engineered cap and occupies approximately 10 acres of the site. Tailings' residue is present in the substrate of Sutton Branch Creek for approximately 0.75 mile downstream of the site where it merges with Big Creek. It has been estimated that 1,173,000 tons of tailings were deposited in the tailings pile area during the period of mining operations. The tailings pile and eroded deposits within Operable Unit 1 (OU 1) were the subject of a removal action which installed the cap prior to this Remedial Investigation (RI). The removal action resulted in the consolidation and covering of the tailings pile as well as the return of some of the outwash material to the pile prior to installation of the cap.

The mine and affected area are located approximately one mile east northeast of Annapolis, Missouri. Runoff from the former mine operation entered Sutton Branch Creek which flows downstream into Big Creek. The area affected by the mining wastes is considered rural/residential. OU 1 is defined as the Sutton Branch Creek floodplain from the probable point of entry to the confluence with Big Creek as well as the historic tailings pile and mine area and is approximately 200 acres in size. OU 2 is defined as Big Creek from the mouth of Sutton Branch Creek downstream to the confluence with the St. Francois River, which is a total of approximately 20 miles of stream. OU 3 is the soil in the town of Annapolis including church yards, school yards, and residential yards. OU 3 is the focus area of this Feasibility Study (FS).

II. Results of Remedial Investigation (RI)

The United States Environmental Protection Agency and the Missouri Department of Natural Resources screened 85 properties during the summer of 2006. Out of 85 properties, 83 were found to be below the screening level for lead of 400 parts per million. The soil lead screening level is the concentration of lead, if found in samples of residential surface soils, which would trigger further investigation. Metal contamination above the screening level for lead in Annapolis was found in one driveway and in one Soil Sampling Unit. The driveway will be remediated as part of a time-critical removal action during the summer of 2007. The Soil Sampling Unit will not be addressed. EPA has determined that a soil cleanup action on the elevated sampling unit is not necessary at this time. The primary factors contributing to this decision include:

- The lead soil concentration found in the southwest area of the property was only slightly above EPA's screening level.

- The area with the slightly elevated concentration was small and not currently a play area or likely to become a play area in the future.

- There was no pattern to the contamination in the community that would connect the property to the mine waste that is the subject of EPA's actions at the ALM site.

- The mean concentration of the lead across the property is well below the screening level.

Based on the data collected during the RI, EPA has concluded that OU 3 (soil in the town of Annapolis) is below levels of concern for human health and the environment. The minimal contamination in the town cannot be attributed to mine waste and is not considered a significant threat to human health and the environment.

III. Analyses of Alternatives

The primary objective of the FS is to ensure that appropriate remedial alternatives are developed and evaluated such that relevant information concerning the remedial action options can be presented and an appropriate remedy selected. The development and evaluation of alternatives shall reflect the scope and complexity of the remedial action under consideration and the site problems being addressed. Under 40 CFR 300.430(e)(3), the lead agency is required to develop a range of alternatives for source control actions. The ALM OU 3 site is not a source area and was found to have minimal lead contamination related to mine waste. The action that will occur at the ALM OU 3 site is a Removal Action of the contaminated driveway. Excluding the contaminated driveway, little or no improvement would be seen if remedial action alternatives were analyzed or implemented. Therefore, no further steps in the FS are necessary.