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FINAL
ECOLOGICAL RISK ASSESSMENT
FOR
CHEROKEE COUNTY, KANSAS, CERCLA SITE
BAXTER SPRINGS/TREECE SUBSITES

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EXECUTIVE SUMMARY

This report presents the findings of the Ecological Risk Assessment (ERA) conducted for the Baxter Springs/Treece (BS/T) Superfund Site, Cherokee County, Kansas. The BS/T subsites are part of the Picher mining field, which was one of the most productive lead and zinc mining areas in the United States. The ERA was developed in response to the Administrative Order of Consent between the EPA and the BS/T subsites Participating Group. This ERA uses an integrated ecosystems approach to determine: (1) if exposures are likely to result in a decreased ability of the ecosystem to function, and (2) if existing contamination of the subsites poses a hazard to key receptor populations and threatened and endangered (T&E) species and/or their critical habitat. Chemicals of Concern (COCs) for the BS/T subsites include the following heavy metals associated with mine wastes and with base metal ore deposits that are characteristic of the Tri-State Mining District: cadmium, cobalt, copper, iron, lead, manganese, mercury, nickel, silver, and zinc. All of these metals were not COCs for all media, however.

Environmental receptors were identified by considering the relevant exposure pathways and the potential or known occurrence of species exposed via those pathways. Key terrestrial species included upper level carnivores -- the barred owl, the red-tailed hawk, and the mink. Key aquatic receptors included benthic invertebrates and fish expected to occur within ephemeral streams of southeast Kansas. T&E species are not included on the list of key receptors, as they are given separate consideration.

Toxicity Reference Values (TRVs) were calculated for each COC for both aquatic and terrestrial receptors. A TRV is an exposure estimate that is not likely to result in chronic adverse effects to a given receptor group. Typically, an ecological risk assessment assumes that if the TRV is not exceeded, the species of interest will be protected (Suter *et al.*, 1983). Toxicity quotients (TQs) were then calculated for key receptors. The TQ approach is a commonly-used method of evaluating the possibility that aquatic and terrestrial populations onsite could be experiencing toxic effects.

Field survey data and other information available in the scientific literature applicable to the area were used as a comparison for interpreting the predictive (quantitative) results. TQs results were evaluated using these criteria: (1) TQs less than 1 represent no impact to exposed individuals, and (2) TQs equal to or greater than 1 indicate that the measured concentration exceeds a concentration that may have a chronic effect on some test species under a given set of experimental conditions. In this sense, TQs greater than 1 are an indication that chronic adverse impacts to exposed individuals are possible. The goal of the ERA is to evaluate the effects of the COCs on exposed populations. Chronic adverse effects to some individuals of a species may not result in measurable effects on populations. Accordingly, the procedure used to evaluate potential toxicity of the COCs to populations inhabiting the subsites is deemed conservative.

RISKS TO AQUATIC RECEPTORS

Field data reveal that only marginal, intermittent aquatic habitat is available in on-site streams. The availability of habitat in the streams is limited naturally and anthropogenically, primarily through the influence of mining and agriculture. Both subsites exhibit similar hydrologic characteristics with ephemeral, first- or second-order streams with low gradients channels. During dry periods the streams are essentially reduced to a series of small pools with little or no flow. Seepage and runoff from mill waste and areas disturbed by mining contribute cadmium and zinc, and to a lesser degree lead, to the surface water system. Tailings and chat can be eroded and washed into receiving channels were they occur as streambed sediments. The surrounding agricultural areas are sources of additional sediments, washed in from tilled and fallow fields. Agricultural areas are thought to be the major source of iron and lead in subsite streams. Stream habitat has been anthropogenically affected as a result of short-term augmentation of stream flow by seepage from chat piles and tailings ponds and channelization (removal of sediments) in upper Tar Creek. Channelization has also eliminated instream pool habitat. Some of the tailings ponds offer stable habitat for fish, as do temporary impoundments constructed by landowners on Willow Creek. In these stream segments, habitat has been artificially created. In general, where suitable stream habitat exists, a variety of fish species were observed to be both self-sustaining, naturally reproducing, and in good condition.

Toxicity Reference Values (TRVs) for aquatic receptors were estimated by employing accepted EPA (Stephan *et al.*, 1985) methodologies using Genus Mean Acute Values. The TRVs calculated represent concentrations of COCs at which chronic effects to the aquatic organisms likely to inhabit subsite systems would not be expected. Toxicity quotients were then calculated by dividing measured concentrations of COCs in water by respective TRVs to determine the potential for chronic effects to aquatic organisms. If the TQ based on the arithmetic mean concentration exceeded 1, then it was concluded that adverse effects to aquatic organisms are possible.

Results of the toxicity assessment, in combination with the field survey data, indicate that organisms inhabiting the Spring and Neosho Rivers, and Willow Creek are not expected to experience adverse effects from exposure to site-related metals. Infrequent ground-water discharge from mine openings in the Bruger area to Willow Creek contains sufficient concentrations of metals to potentially cause short-term acutely toxic conditions, especially with regard to zinc. TQs for seven of the 10 subsite tailings and subsidence ponds sampled indicated no potential chronic effects, however, some individual ponds contain levels of cadmium, iron, lead, and/or zinc above calculated TRVs. These ponds included BP-1 (Ballard Chat Wash), TP-5, and TP-7.

Results of the aquatic toxicity assessment indicate that chronic impacts to aquatic organisms resulting from elevated zinc and cadmium concentrations could be occurring in Spring Branch, while chronic impacts resulting from elevated zinc concentrations could be occurring in Tar Creek. The toxicity assessment results also indicate that the concentration of zinc in Tar Creek and Spring Branch could be affecting the species composition in these two ephemeral streams in that the maximum zinc concentrations exceeded levels acutely toxic to some of the more sensitive species that could inhabit these aquatic systems. While other factors such as bioavailability of the COCs and acclimation could reduce the toxicity of the observed zinc concentrations, field survey results suggest that species composition in these two streams may be affected by elevated zinc concentrations.

RISKS TO TERRESTRIAL RECEPTORS

Two exposure scenarios were used to assess risks to terrestrial receptors: the worst-case and the reasonable maximum exposure (RME) scenario. Potential routes by which terrestrial receptors could be exposed to mine-related metals include: (1) inhalation of fugitive dust; and (2) ingestion of soil, mine waste, vegetation, prey, and/or surface water. Dose estimates were calculated for key receptor species using the subsite surface water, soil, and mouse/fish body-burden data collected during the Remedial Investigation. Specific ecological endpoints (EEs) were identified which characterize the site-specific ecological system that may be affected by site-related metals and represent the actual environmental values to be protected. It is important to emphasize that no one ecological endpoint was used as a single, rigid standard; rather, both results of field surveys and the toxicity assessment represent endpoints were used in assessing potential effects due to metals toxicity.

TQs for terrestrial receptors were initially calculated using worst-case exposure parameters as a screen for (1) contaminant hot spots, and (2) receptors that don't require further evaluation. Worst-case exposure is defined in this case as the highest exposure that is reasonably expected to occur at a site. Since the worst-case scenario uses a combination of conservative (health-protective) assumptions and upper-bound (upper 95th percentile) data, it is expected to overestimate actual exposures. For biota, the worst-case approach implies that an organism spends its life in near-maximal contact with upper bound concentrations of all COCs in all media simultaneously. This can be unrealistic from the standpoint that (1) many receptors do not tend to stay in one area for very long, and (2) the maximum concentration of COCs in air, water, and soil are not geographically coincident. Furthermore, the use of single-value estimates, especially worst-case estimates does not provide adequate information to risk managers who must evaluate ecological risks. Consequently, a second, more plausible, exposure scenario was also used. For adequate protection of ecological receptors, knowledge of exposures typically encountered by key receptor organisms collectively is of greater value than estimates of upper bound exposures potentially affecting a few organisms. The reasonable maximum exposure (RME) scenario uses arithmetic mean concentrations and less conservative, more site-specific exposure assumptions to characterize ecological exposures. The RME scenario is more plausible since receptors do not spend all of their time at one specific location (e.g., at a hot spot or in areas where maximum or upper-bound concentrations occur). Therefore, the RME scenario was deemed to be more representative of expected exposure intensity. The exposure assumptions and exposure point concentrations used to quantify intakes for each scenario are outlined in Section 5.

Results show that exposure to cadmium, lead, and zinc are not expected to cause adverse effects in terrestrial predators, since RME TQs for the red-tail hawk and the barred owl are less than 1. Results of the toxicity assessment for the mink, however, indicate that chronic adverse effects from exposure to cadmium, lead, and zinc are possible since the worst-case and RME scenario TQs are slightly above 1 (RME TQs range from 1 to 3). These data indicated that terrestrial species who consume fish may experience adverse effects from exposure to cadmium, lead, and zinc.

A comparison of mean concentrations of metals in near-pile soils to the median concentrations reported to be phytotoxic indicates that zinc concentrations may be marginally phytotoxic, i.e., may cause some reduction in crop yield. Agricultural soil mean concentration TQs were all lower than 1, indicating that metal concentrations in agricultural soils are not phytotoxic. It is noted, however, that the phytotoxic reference data are based on agronomic plants and effects on the non-agronomic plant species which grow in near-pile soils are unknown.

No Federally-listed T&E species are known to occur within the Baxter Springs/Treece subsites. Nine state-listed T&E species have designated critical habitat within the subsites or have critical habitat that could be affected by the migration of site-related metals. TQs based on the upper-bound concentration of metals in on-site surface water bodies indicate that T&E exposure (using amphibians as surrogates) to site-related metals is not expected to cause adverse impacts in exposed individuals.

1.0 INTRODUCTION

This report presents the findings of the Ecological Risk Assessment (ERA) conducted for the Baxter Springs/Treece (BS/T) Superfund Site, Cherokee County, Kansas. A separate human health risk assessment has been performed and is submitted as a companion volume.

The U.S. Environmental Protection Agency (EPA), under the authority of the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA), has designated the Kansas portion of the Tri-State mining district (Kansas, Missouri, and Oklahoma) as the Cherokee County Superfund Site. The Cherokee County Superfund Site is divided into six subsites, which include the BS/T subsites. This Final Draft ERA was developed in response to the Administrative Order of Consent, Docket Number VII-90-F-0010, dated May 8, 1990, between the EPA and the BS/T Subsites Participating Group (The Group). The Group includes the following participating companies:

- AMAX, Inc.;
- Gold Fields American Corporation;
- ASARCO, Inc.;
- NL Industries, Inc.;
- Sun Company, Inc.;
- Eagle-Picher Industries, Inc.; and
- St. Joe Minerals Corporation.

CERCLA requires that actions selected to remedy hazardous waste be protective of human health and the environment. In addition, although many Superfund risk assessments focus on human health effects, there are some situations in which ecological receptors may be at greater risk (Bascietto *et al.*, 1990). This ERA is an investigation of potential threats to the environment from exposure to contaminants present at the subsites. It identifies potential exposure pathways from contaminants to environmental receptors inhabiting the area, characterizes the toxicological properties of the potential chemicals of concern (COCs), and quantifies the extent such exposures could contribute to ecological risk.

This ERA was performed using an integrated ecosystems approach in which the field observations/species survey data were used in combination with toxicity assessments. Used independently, neither method would likely provide a realistic nor complete risk characterization. This ERA combines the two methods to balance each other and determine if existing contamination of the subsites poses a hazard to key receptor populations and threatened and endangered species and/or their critical habitat. Ecological impacts were assessed using exposure-response data taken from the literature, cited below, in conjunction with measured environmental concentration data. Both terrestrial and aquatic receptors and habitats were evaluated. Guidance documents used to conduct this ERA include EPA's *Framework for Ecological Risk Assessment* (USEPA, 1992), EPA's *Ecological Assessment at Hazardous Waste Sites* (USEPA, 1989a), *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference Document* (USEPA, 1989b), *Risk Assessment Guidance for Superfund, Volume II, Environmental Evaluation Manual* (USEPA, 1989c). Input by EPA personnel was also utilized to prepare this ERA.

Risk assessment provides a mechanism for estimating risks in cases where risks may be judged to be excessive. It is a process that synthesizes available data on exposure and toxicity of metals and incorporates scientific and professional judgment to estimate the associated risk to the environment. Ecological risk assessment, the characterization of potential adverse effects resulting from exposure to environmental contaminants, involves four consecutive steps:

1. Data Collection and Evaluation: identifying contaminants of concern and defining the nature and magnitude of metals contamination in specific environmental media;

2. Exposure Assessment: determining the extent of ecological exposure to environmental contaminants;

3. Toxicity Assessment: determining the relationship between magnitude of exposure and the probability of occurrence of adverse effects; and

4. Risk Characterization: combining the first three steps to yield estimates of ecological risk.

1.1 ORGANIZATION

The first step in the ERA is to identify chemicals of concern, in this case, metals. This step is followed by evaluating potential exposure pathways and quantifying chronic daily intakes by key receptor species. To quantify exposures, exposure concentrations and receptor intakes must be estimated. The next step, toxicity assessment, identifies compounds likely to result in adverse effects in exposed populations. The final step, risk characterization, integrates field observations/data and information from the exposure and toxicity assessments to yield estimates of risk. Since uncertainty analysis is considered an important component of the risk assessment process, a qualitative discussion of uncertainty is included.

Section 1.0 is an introduction and overview. Section 2.0 provides a description of the terrestrial and aquatic environment of the BS/T Subsites. Section 3.0 discusses the extent of contamination onsite. Section 4.0 describes the screening process used to select the COCs. Section 5.0 details the exposure assessment process, including selection of key ecological receptors, describes the fate and transport of metals between and within various environmental media, identifies relevant pathways of exposure, estimates exposure point concentrations, and estimates dose for the key receptors. Section 6.0 outlines the ecological endpoints used to evaluate potential adverse impacts. Section 7.0 contains toxicity information on the COCs. Section 8.0 presents calculations of Toxicity Quotients. Section 9.0 discusses uncertainty associated with the risk assessment. Section 10.0 presents the risk characterization. Uncertainty stemming from 1) variability in the assumptions made and 2) data input values used and their effect on predicted exposures are also discussed briefly throughout the text.

1.2 SITE BACKGROUND

The Baxter Springs and Treece subsites were part of one of the most productive lead and zinc mining districts in the country. This district was known as the Picher mining field, and was centered near the town of Picher, Oklahoma, extending northward into southeastern Kansas. Peak production from the Picher field was recorded in 1925. Between 1921 and 1925, the field yielded 55 percent of the total zinc produced in the United States (McKnight and Fischer, 1970). Today, approximately 4 percent and 11 percent of the land areas are covered by mining/milling wastes or abandoned mill sites in the Baxter Springs and Treece subsites, respectively. The limestone formations underlying the subsites have been extensively mined and are honeycombed with workings ranging in depths from 200 to 500 feet below ground surface; in addition, over 100 open shafts, collapsed shafts, and subsidence features are present at the surface above the underground workings (McCauley *et al.*, 1983).

At one time, an estimated 75 million tons of coarse mill waste materials called chat, were accumulated in the subsites. The availability and abundance of chat gave rise to an industry devoted to processing this waste for construction materials, concrete aggregate, railroad ballast, road base, and blasting sand. Many of the unpaved secondary roads in southeastern Kansas and Missouri and northeast Oklahoma are surfaced with chat obtained from quarrying operations in the Picher field. Sears (1989) estimated that only about 6 percent of the chat originally found in mill waste piles within the subsites remains. One large chat quarrying company currently operates processing equipment in at least two locations within the subsites.

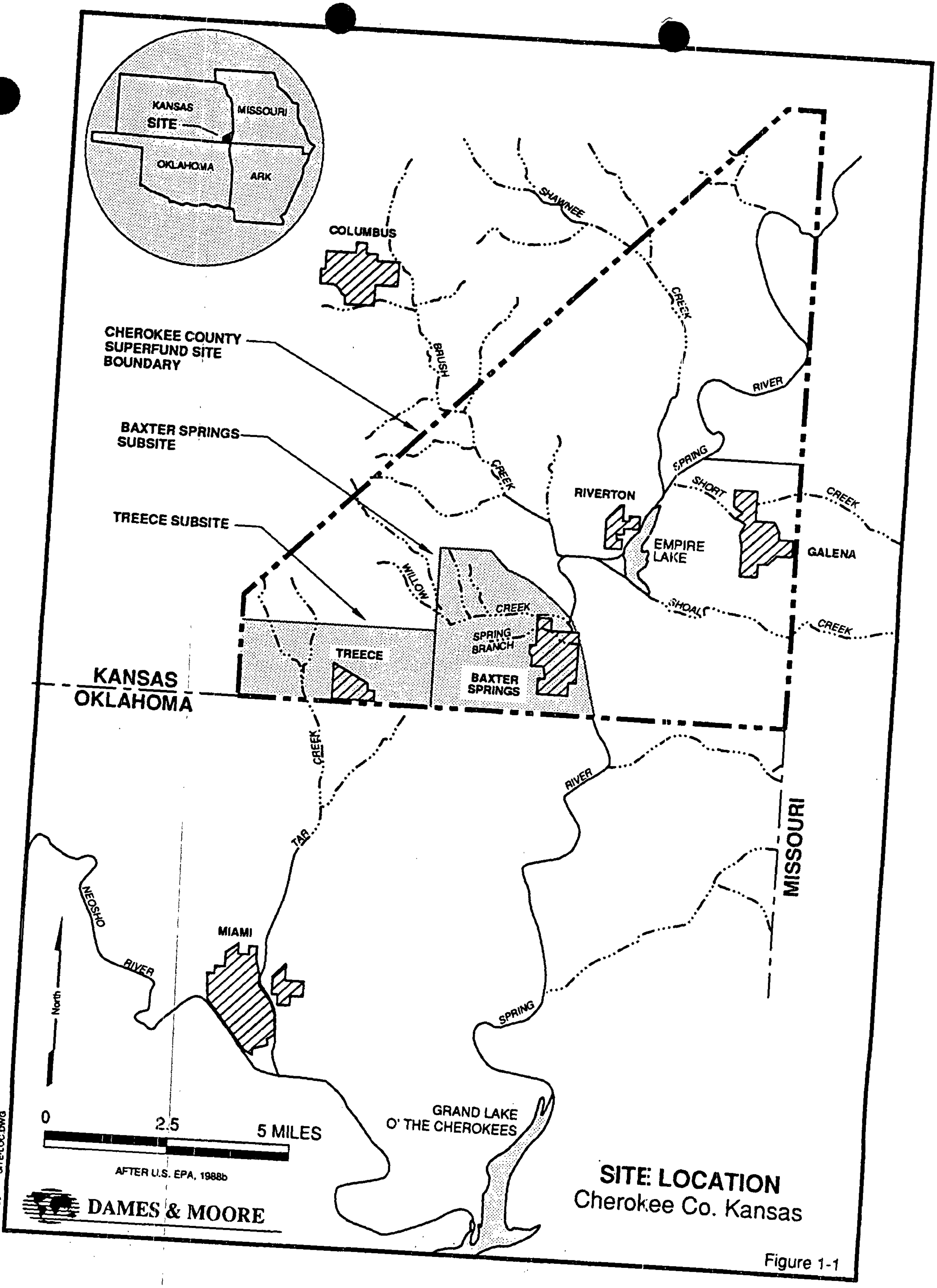
1.3 PHYSIOGRAPHY

The location of the subsites relative to surrounding landmarks is illustrated in Figure 1-1. In addition to the brief description below, the physiographic characteristics of the subsites are discussed in detail in the Final Draft Remedial Investigation (RI) Report (Dames & Moore, 1993a).

The Baxter Springs subsite covers approximately 17 square miles and includes the town of Baxter Springs, Kansas, with a population of 4,351 (1990 census). The subsite is drained by two ephemeral streams, Willow Creek and Spring Branch, and numerous small unnamed tributary drainages. Both creeks flow in an easterly direction across the subsite and enter the Spring River, which is an interstate perennial stream forming the eastern subsite boundary.

The Treece subsite covers approximately 11 square miles and includes the unincorporated town of Treece, Kansas, with a population of 172 (1990 census). Drainage within the subsite is primarily to the south with Tar Creek being the principal ephemeral surface water drainage. A smaller ephemeral branch, known as Tar Creek Tributary, joins this creek near the site boundary; another small drainage, Lytle Creek, flows south into Oklahoma from the eastern portion of the subsite. Lytle Creek joins Tar Creek south of the Oklahoma state line and Tar Creek continues southward before entering the Neosho River near Miami, Oklahoma. The Neosho River is the perennial surface water body potentially affected by source materials in the Treece subsite.

Both subsites lie primarily in the Osage Plains section of the Central Lowlands physiographic province characterized by gentle slopes and shallow low-gradient stream valleys with only occasional areas of topographic relief. Blue Mound, located in the Treece subsite, is the highest point in either subsite reaching an elevation of 970 feet above mean sea level. The extreme eastern portions of the Baxter Springs subsite lies within the Springfield Plateau section of the Ozark Plateau province characterized by hilly terrain with steep slopes and thin



SITE LOCATION
Cherokee Co. Kansas

Figure 1-1

2.0 STUDY AREA DESCRIPTION

The Final Draft Remedial Investigation (RI) for the BS/T subsites presents in detail the physical, geological, and hydrologic features and biotic characteristics of the subsites. This section of the ERA provides an overview of the study area features.

2.1. LAND USE AND CLIMATE

Land use characteristics of the subsites are generally characterized in the table below:

Subsite	Number of Acres Per Type of Land Use					Total Acres
	Urban	Cropland Pasture	Mine Related	Roads	Woodland	
Baxter Springs	1,000	6,800	440	60	2,580	10,880
Treece	50	5,000	755	50	1,185	7,040

Urban development (Baxter Springs and Treece) and light industry occupy approximately 1050 acres or six percent of the subsite acreage. Approximately 39 miles of chat-covered roads are present in the subsites, or about 0.6 percent of the total subsite acreage.

Cultivated crops are primarily wheat and soybeans with some grain sorghum, corn, barley, oats, and alfalfa. Tall fescue is the main tame grass grown for pasture or hay. Along the Spring River, agricultural land is used primarily for raising livestock and growing crops. Both beef and dairy cattle are raised in this area. Soils in croplands and pastures were sampled as a part of the RI and were termed Agricultural soils.

Vegetated non-agricultural areas (woodlands and bottomlands along streambeds) occupy approximately 3,765 acres or 21 percent of the total subsite area. Soil samples collected in these nonagricultural areas were termed A-Horizon samples and results were pooled with the Agricultural soil samples to form an Agricultural/A-Horizon (Ag/A) data set.

Mine and mill wastes and areas disturbed by these past activities affect approximately 1,195 acres or roughly 7 percent of the total subsite area. The various soils and waste materials included in this category are discussed in Section 2.2.

The prevailing winds in the subsites area are from the south with an average windspeed of 4.2 meters per second (9.3 mph). Kansas has a climate typical of the continental interior of North America with hot summers and cold winters. Annual mean temperatures range from about 58°F in the southeast to 52°F in the northwest. Monthly mean temperatures at Joplin, Missouri (15 miles to the east) range from 33°F in January to 80°F in July. Daily temperature variations of about 20°F occur year round. Temperatures of 90°F or higher occur an average of 49 days per year, while the average number of days with a temperature of 32°F or lower is 88 days per year. Average monthly precipitation ranges from 1.4 inches in January to 4.9 inches in June, with a mean annual total of 39.5 inches. About 75 percent of the rainfall occurs during the crop growing season (April to September), while an average of 12.3 inches of snow falls annually during the winter months (December through March). During 1991, the year the site was investigated, approximately 32 inches of rainfall occurred on the subsites.

Floods may occur throughout the year in southeastern Kansas. However, the greatest flood frequency is spring and summer during periods of potentially prolonged or torrential rain.

2.2 CHARACTERISTICS OF MINE AND MILL WASTES

2.2.1 Mine Waste

In general, historical mining operations generated two broad categories of waste: mine and mill waste. Mine waste is rock excavated from shafts and mine workings, and as such, most are cone-shaped piles of rock stacked near mine shafts. Most of the mine waste piles inspected consist of large blocks of limestone and jasperoid (silicified limestone) which lack vegetation and provide little habitat for wildlife. Mine waste piles cover about 18 acres or 0.1 percent of the subsite acreage.

2.2.2 Mill Waste

Mill wastes present at the surface are divided into two categories: chat and flotation tailings. Chat is silt to small gravel-sized crushed rock derived from gravitational mill processing. Flotation tailings are silt to fine sand-sized material derived from the froth flotation milling process. Flotation tailings have higher residual metal concentrations and are finer grained than chat.

There are an estimated 3.2 million yd^3 of chat remaining in the subsites (610,000 yd^3 in the Baxter Springs subsite and 2,625,000 yd^3 in the Treece subsite). The larger accumulations cover an estimated 324 acres in the subsites. Most of the chat piles now exist as remnants of larger piles (excavated chat), having been extensively quarried over the years and sold as various construction aggregates, railroad ballast, and roadway fill. Chat consists of siliceous chert, jasperoid, and limestone fragments that resulted from jigging and tabling (mechanical) separation processes. Chat comprises approximately 85 percent of the surface mill waste volume at the Baxter Springs subsite and 72 percent of mill waste volume at the Treece subsite.

Fine-grained flotation tailings were deposited in shallow bermed tailings impoundments. Site-specific grain size data show that these tailings range from about #35 mesh (0.0165 inches) to #400 mesh (0.0015 inches) or in the fine sand silt clay-size particle range. Generally, the dikes impounding the tailings are intact and, in many instances, hold water forming temporary ponds. These tailings are silt-like (fine-grained) and have a slow water infiltration rate and a high water-holding capacity. The estimated 291,000 yd^3 of tailings present in the Baxter Springs subsite cover about 80 acres, while the estimated 514,000 yd^3 of tailings present in the Treece subsite cover approximately 132 acres.

The larger deposits of chat are generally unsuitable for plant growth due to their insufficient organic matter, nutrient, and near-surface moisture. Although many of the excavated chat areas and some of the relict tailings impoundments exhibit sparse to moderate vegetative cover (usually perennial weeds or weeds mixed with grasses), areas covered by mine and mill waste do not allow for the establishment of diverse vegetation due to the physical limiting factors noted above. Experimental test plots established by the U.S. Bureau of Mines at the Galena subsite in chat (Norland and Veith, 1990) indicated that the native warm season grasses seeded to chat amended with organics had the best emergence of any species tested. Soapberry (*Yucca glauca var. mollis*), which is representative of species that typically occur in the revegetated strip-mined areas of southeastern Kansas (Stubbendieck *et al.* 1986), was observed by Dames & Moore field personnel growing in association with well-drained chat piles. The U.S. Bureau of Mines experimental test plot data for chat and the subsite observations of naturally revegetated excavated chat and tailings indicate the absence of acutely phytotoxic metal concentrations, at least for some species.

2.2.3 Soils Near Mine/Mill Waste Areas

Soils near mill waste accumulations were subdivided into two major categories based on their visible chat content. Soils in non-agricultural areas less than 300 feet from a mill waste pile that did not contain visible chat (or tailings) were termed near-pile soils. Near-pile soils are typically well vegetated.

Soils from obviously disturbed or affected areas that contained visible chat fragments (and possible tailings) were referred to as mill-site soils. Mill-site soils were collected from areas directly affected by past mining or milling activities, as indicated by the presence of concrete mill foundations, scrap, and other debris. Soils in these areas consisted of chat and soil mixtures that were sparse to moderately revegetated. The estimated areal extent of near-pile soils and mill-site soils are included with the acreage estimates for remnant chat areas and other disturbed areas totaling 261 acres in the Baxter Springs subsite and 380 acres in the Treece subsite. Thus, the total mine-related acreage shown in the land use table in Section 2.1 is made up of mine waste (18 acres), chat (324 acres), tailings (212 acres), and remnant/excavated chat areas (641 acres), totaling 1195 acres.

2.3 DESCRIPTION OF TERRESTRIAL HABITATS AND POPULATIONS ON SITE

The vegetation communities of the BS\T subsites are subdivided into four general topographic affiliations: 1) upland oak forest, 2) lowland maple forest, 3) floodplain cottonwood-mixed forest scrub, and 4) streamside willow scrub. Each topographic category is identified by the dominant species which form the principal communities (See RI, Table 4.8-1).

2.3.1 Vegetation Types

Generally, the BS/T subsites are representative of the Cherokee County portion of the Cherokee Prairie Region. Much of the land in the Cherokee Prairie Region is used for crops and pasture, but a mixture of scattered woodlots and riparian wooded areas remain (Newland *et al.*, 1964). Woodlands generally occur as irregular tracts and narrow bands located along streams and rivers but also as strips in upland drainageways and on the steeper upland slopes. Trees and hedgerows of trees also occur on most of the farmsteads; they were planted by the early farmers primarily to serve as farmstead and feedlot windbreaks and to minimize soil erosion.

Vegetation identification was conducted primarily in late March 1991, when shrubs and vines were leafing out. Identifications were made from leaf samples, nut and fruit collection, bark textures, and other visual characteristics. Plant identification references included Stephens, 1969 and Bare, 1979. See Section 4.8 of the RI for extensive review of site vegetation.

2.3.2 Terrestrial Wildlife

Wildlife species, relative abundance, and habitats occurring in the Baxter Springs and Treece subsites are typical of those occurring in the agriculturally-altered tall grass prairie/eastern woodland ecotone. Habitats are principally associated with cropland margins and non-tillable areas with native floral representation remaining only in the riparian bottoms and relic woodlots. As a result of extensive agricultural practices over the last century, diversity in the natural floral community is significantly limited in Cherokee County. While unvegetated mill waste deposits provide no habitat, the wildlife and floral communities in and around mine-related disturbance provide good diversity and abundance because these areas have not been subjected to agricultural or urban development.

Terrestrial predators of interest in the vicinity of the subsites primarily include members of the canid and mustelid families and predatory birds. Species that could potentially occur on site include coyote, red fox, gray fox, badger, long-tailed weasel, mink, striped skunk, bobcat, and spotted skunk. In addition, raccoon could occur in the area, although it is actually classified as an omnivore versus a carnivore. Field surveys documented the presence of fox (species unknown) and raccoon. Canids (species undetermined), raccoons, white-tailed deer, beavers, opossums, owls (species undetermined), and cottontail rabbits were observed within the subsites.

Other species which were observed or reported to occur [U.S. Soil Conservation Service (USSCS), 1985] include white-tailed deer, beaver, muskrat, Virginia opossum, eastern cottontail rabbit, white-footed mouse, short-tail shrew, hispid cotton rat, marsh rice rat, deer mouse, woodland vole, and eastern chipmunk. All species, excluding the woodland vole, eastern chipmunk, and the eastern cottontail rabbit were observed by field personnel. The three species not observed were listed as "expected to occur" based on suitable habitat availability.

Evidence of eight different raptors was documented during the October 1991 RI field surveys. Raptors were northern harrier, sharp-shinned hawk, red-tailed hawk (including a Harlan's hawk-color phase of red-tail), golden eagle, American kestrel, turkey vulture, great-horned owl, and barred owl. Other species expected to occur within the subsites include Cooper's hawk, rough-legged hawk, barn owl, and short-eared owl. All of these species are known to hunt over habitats similar to those present in subsites and all, except for the rough-legged hawk, golden eagle, and short-eared owl, may potentially breed in subsites.

Waterbirds observed in the subsites include Canada goose, mallard, wood duck, blue-winged teal, great blue heron, and possibly an egret (species unknown).

Over 100 species of songbirds could potentially occur on or near the subsites (Thompson and Ely, 1989). Since RI surveys occurred in early October, most observations were of year-round residents, migrants, and winter visitors. A total of 43 species were documented during the field surveys; eastern meadowlark and blue jay were the most commonly observed species, followed by Carolina chickadee, cardinal, American robin, starling, and common (or Northern) flicker.

A wide variety of reptiles and amphibians are known to occur within Cherokee County. Blanchard's cricket frog was common in areas of ponded water and along the creeks that transect the subsites. In addition, the southern leopard frog and bullfrog were observed. Three species of snakes were each observed in October 1991. These were the bullsnake, plains garter snake, and rough green snake. Each of these species could potentially use all project area habitats except mill waste piles and open water. Turtles observed were the Mississippi map, ornate box, red-eared, and stinkpot.

2.4 CHARACTERISTICS OF THE AQUATIC ENVIRONMENT

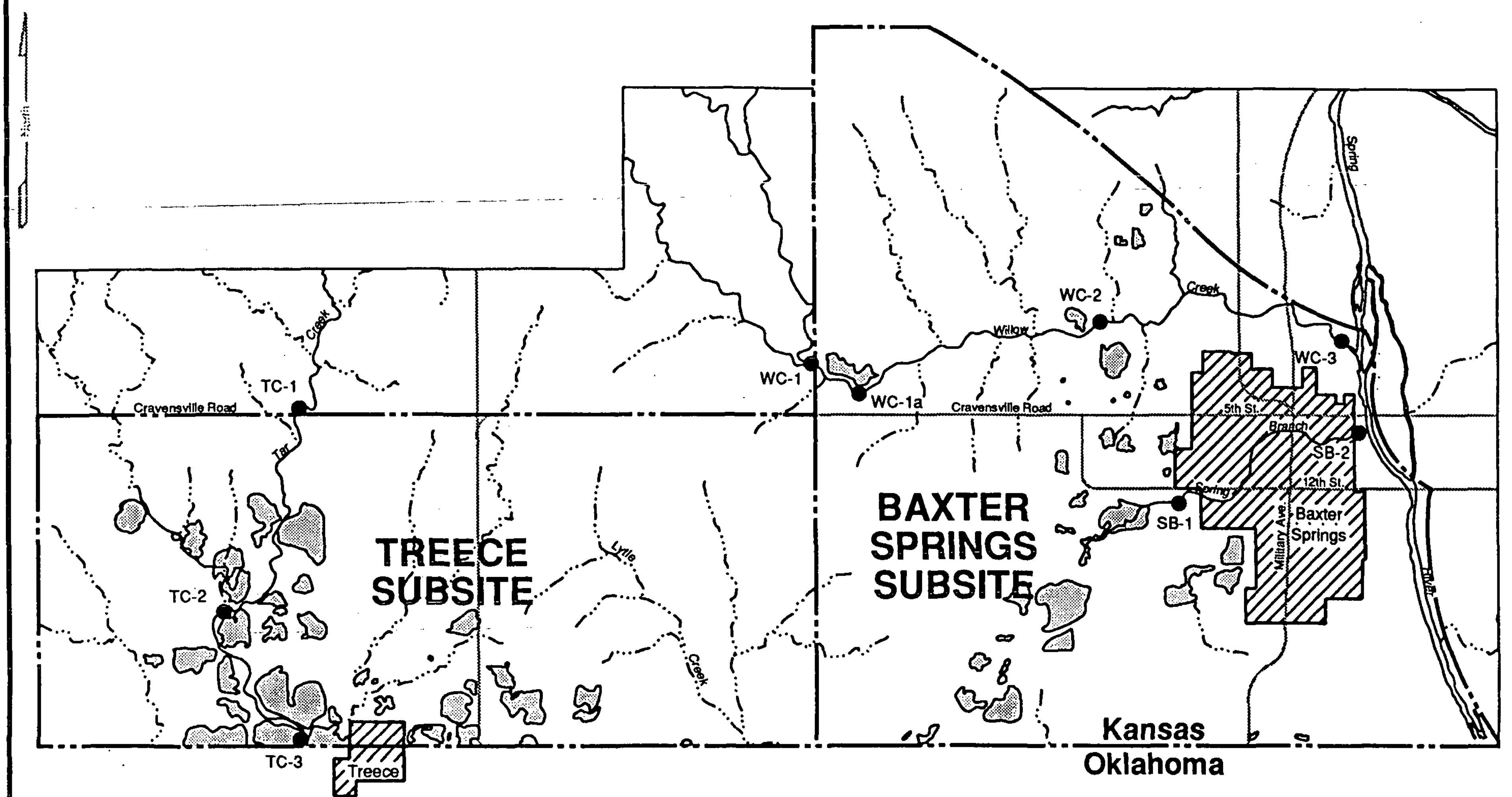
All watersheds in the Baxter Springs subsite are tributary to the Spring River while watersheds in the Treece subsite drain to Tar Creek, a tributary of the Neosho River (Figure 2-1). The Spring River is an interstate perennial river that flows through southeastern Kansas to Oklahoma, forming the eastern boundary of the Baxter Springs subsite. The Neosho and Spring Rivers are the major tributaries to Grand Lake O' The Cherokees in eastern Oklahoma.

Historically, streams within the on-site watersheds have been ephemeral and, as such, support limited aquatic life throughout the year. Overall habitat suitability of the on-site waters is limited. Physical limitations of the aquatic habitats include intermittent flows within the streams and associated elevated water temperatures and low dissolved oxygen during low flow periods. These flows may be augmented by the gradual release of precipitation retained within chat piles. With the exception of springs emanating from Mississippian limestone near the mouths of Willow Creek and Spring Branch (downstream of Highway 66), groundwater does not appear to play a major role in sustaining stream flow. While subsite streams do not flow during dry periods [observed by Dames & Moore in August and September 1991 and by Spruill (1987)], limited areas of ponded water remain in stream channels. Stream flow responds to seasonal precipitation patterns and can increase by an order of magnitude during rainfall events. As a result of increased rainfall runoff, sediment is moved from croplands and mill waste areas to receiving channels. Specific mill waste sediment sources include chat eroded from the larger piles located near stream channels and tailings eroded from breached tailings impoundments. The presence of shifting sediments in the stream channels is assumed to deteriorate habitat quality and to provide a source of leachable metals to surface waters.

2.4.1 Spring River Drainage

The Spring River is an interstate perennial river that flows through southeastern Kansas to Oklahoma and forms the eastern boundary of the Baxter Springs subsite (Figure 1-1;

3/93 AO-BSMP.DWG



Source: USGS Quadrangle Mapping;
Okla-Kans, Melrose, Baxter Springs, Picher, Neutral Quad Maps.

0 14000 28000 Feet

Approximate Scale

LEGEND

SUBSITE BOUNDARY

TAILINGS PILE

WC-1 MONITORING LOCATION AND NUMBER

DAMES & MOORE

Approximate Aquatic Biota Sampling Locations Cherokee County

Figure 2-1

Section 1.0). The Spring River watershed drains the southwestern portion of the Ozark Plateau in Missouri as well as portions of the Central Lowlands in southeastern Kansas and northern Oklahoma, which incorporates portions of the Tri-State mining district. The stream gage on the Spring River nearest the subsite is at Quapaw, Oklahoma, which is nine miles south of the City of Baxter Springs. At the Quapaw stream gage, the river drains an area of approximately 2,510 square miles. The average annual discharge between 1939 and 1990 of the Spring River at Quapaw is 2,078 cubic feet per second (cfs), with annual peak flows of 50,000 cfs to 100,000 cfs. The Spring River drainage is subject to periodic flooding caused by prolonged or intense periods of rainfall. A peak discharge of 190,000 cfs was recorded in May, 1943. Low flow periods occur during the summer. A minimum discharge of 5.8 cfs was recorded in July, 1954 (USGS, 1987).

Water quality in the Spring River is impacted, in general, by runoff from agricultural areas, treated sewage and coal mine drainage from Cow and Brush Creeks, and seepage/runoff from abandoned lead-zinc mines in the Short, Turkey, Center Creek, and Willow Creek watersheds [Kansas Department of Health and Environment (KDHE), 1980]. Historical KDHE water quality data for the Spring River show occasional exceedances of Federal chronic Ambient Water Quality Criteria (AWQC) for cadmium, copper, iron, lead, and zinc. Notwithstanding these occasional exceedances of Federal criteria, the river supports a varied warm-water fishery which includes largemouth bass, walleye, white bass, crappie, and catfish.

Empire Lake located on the Spring River approximately 10 miles upstream from Baxter Springs receives metals input from upstream sources, including lead-zinc mined areas in the Center Creek, Short Creek, and Turkey Creek drainages.

The Spring River system has historically received heavy metals input from mining activities throughout southeastern Kansas. Ferrington *et al.*, (1988) concluded that the Spring River benthic invertebrate community within the Empire Lake area has been affected by the presence of elevated levels of metals in the river causing reductions in standing crop densities and aquatic macroinvertebrates. Ferrington *et al.*, (1988) reviewed eight benthic invertebrate studies to establish general patterns of species richness and standing crop densities for lentic environments. He concluded that minimum standing crop density estimates should be on the order of 2,500 to 6,000 or more aquatic insects per square meter. This conclusion was based on the results of studies conducted relatively close geographically in Kansas, Nebraska, and the Texas-Oklahoma border, and more distant studies conducted in Florida, North Dakota, Iowa, and New York. Minimum standing crop densities estimated for those studies conducted relatively close geographically were also in the same range of 2,500 to 6,000 organisms per square meter. Ferrington *et al.* (1988) concluded that metals have reduced standing crop densities of aquatic macroinvertebrates. The standing crop density estimates for the Empire Lake study ranged from 173 to 822 organisms per square meter. Species diversity in Empire Lake, however, was greater in this area than in upstream locations and densities were comparable to those observed in Shoal Creek, which was the control site for the study. Furthermore, Ferrington *et al.* (1988) stated, "Based upon an analysis of the abundance patterns of major taxonomic groups, it must be concluded that the presence of high concentrations of metals in the sediments has not caused major shifts in community composition."

The KDHE (1980) report, *Water Quality Investigations of Lead-Zinc Mine Drainage Effects on Spring River and Associated Tributaries in Kansas 1978-1979*, indicated that most of the metals loadings to the Spring River and the resulting paucity of clean water invertebrates is due to releases from Short Creek, which contains relatively high heavy metals concentrations, especially zinc. Sampling results show that within the Spring River below Empire Lake, 53 percent of the benthic invertebrates sampled were clean water taxa. A similar study (USGS, 1992) conducted in the Spring River east of Baxter Spring showed that 49 percent were clean water taxa. These data indicate that the Spring River may be affected by metals loadings from Short Creek and other major tributaries (Figure 1-1), but the contributions to the Spring River from Willow Creek and Spring Branch (less than 1 percent) are small in comparison.

The USGS compiled available historic data for the upper Spring River basin (USGS, 1992). Short Creek at Galena represented less than 1 percent of the median flow in Spring River at Baxter Springs, but contributed 79 percent of the dissolved zinc load (calculated on an instantaneous load basis). Dissolved concentration of iron, lead, and manganese in the Spring River did not seem to be affected by lead-zinc mining in the Spring River basin. The Short Creek station also maintained the highest median dissolved copper concentration (0.24 mg/L at Galena). An almost total lack of benthic invertebrates at Center Creek (near Smithfield, Missouri) and Turkey Creek (near Joplin, Missouri) were attributed to elevated levels of zinc. Neither Willow Creek or Spring Branch were included in the USGS study.

Baxter Springs Subsite Drainages

The Baxter Spring subsite is primarily drained by Willow Creek and Spring Branch (see Figure 1-1, Section 1.0 and Figure 2-1). Although both streams are tributary to the Spring River, collectively they comprise less than 0.8 percent of the Spring River drainage basin above Quapaw, Oklahoma. Willow Creek has a total drainage area of approximately 16 square miles and flows from west to east through the subsite. The Baxter Springs subsite is located in the lower one-half of the watershed, occupying approximately 50 percent of the Willow Creek watershed. The entire Spring Branch watershed (3.3 square miles) is within the subsite, comprising another 20 percent of the total Baxter Springs subsite area. The remaining area of the Baxter Springs subsite is an upland area that drains south to southeast towards the Spring River.

The dominant land use of the Willow Creek watershed is agriculture. Evidence of past mining activity is confined to the lower reaches of Willow Creek, where chat piles and tailings ponds are present. Approximately 3 percent of the watershed surface area is covered with mill waste. Land use in the Spring Branch watershed includes the City of Baxter Springs as well as agriculture and mining. Urban areas are confined to the lower one-half of the Spring Branch watershed. Approximately 16 percent (140 acres) of the watershed surface area upstream of Baxter Springs contains evidence of past mine/mill activities.

Water quality in the streams relates directly to land use. The data indicate that the primary source of suspended solids in Willow Creek is surface runoff from croplands, pastures, or other sources unrelated to mining. With respect to regional conditions, it is important to note that total recoverable concentrations of lead were at or above chronic AWQC at the upstream, baseline (WC-1) station in all samples collected.

Willow Creek Surface Water Quality (total recoverable - mg/L)

Metal	Upstream (Mean)	Downstream (Mean)	Chronic AWQC[1]	Acute AWQC
Cadmium	0.0004	0.0015	0.0006-0.005	---
Iron	2.73	3.94	---	1.0
Lead	0.01	0.014	0.0012-0.0351	---
Zinc	0.15	0.63	0.054-0.524	---

[1] Adjusted for sample hardness.

Iron exceeded AWQC at the upstream and downstream stations with an arithmetic average of 2.7 mg/L and 3.9 mg/L, respectively. Runoff and seepage from mill waste apparently transports lead to the lower reaches of Willow Creek as evidenced by higher total recoverable lead concentrations at the two downstream sampling stations, WC-2 and WC-3. One set of samples analyzed for dissolved metals showed cadmium and zinc present in Willow Creek water samples primarily as dissolved metals. Levels of cadmium and zinc measured at the two downstream locations indicate that mill waste in the Willow Creek watershed contributes to in-stream concentrations of these metals. Concentrations of iron, lead, and suspended solids were usually highest during high flow periods resulting from rainfall runoff inputs and scouring of sediments; cadmium and zinc concentrations were usually highest during low flow periods when dilution from rainfall runoff was minimal.

Most of the downstream area of the Spring Branch watershed is urban in contrast to the Willow Creek watershed which is rural/agricultural throughout, while the upstream area is dominated by mill waste runoff/seepage. No baseline station was established in this watershed. Suspended solids concentrations and total recoverable iron concentrations in Spring Branch were minimal during both low- and high-flow sampling events. Total recoverable lead concentrations exceeded chronic AWQC only during the higher flow of the May sampling event. Cadmium and zinc total recoverable concentrations exceeded AWQC in all sampling rounds, regardless of flow conditions.

Spring Branch Surface Water Quality (total recoverable mg/L)

Metal	Upstream (Mean)	Downstream (Mean)	Chronic AWQC[1]	Acute AWQC
Cadmium	---	0.08	0.0015-0.0056	---
Iron	---	0.25	---	1.0
Lead	---	0.008	0.0049-0.0421	---
Zinc	---	9.9	0.141-0.591	---

1 Adjusted for sample hardness.

Stream discharge within the Willow Creek/Spring Branch system fluctuates dramatically particularly in response to seasonal changes in precipitation. Stream discharge in Willow Creek ranged from 0.01 cfs at WC-3 during the August sampling event to a high of 46.2 cfs during the May sampling event. Spring Branch discharge displayed similar variability where discharge during low flow was measured at 0.05 cfs and discharge at high flow was 16.7 cfs. The upper reaches of both streams become intermittent with isolated remnant pools during periods of low flow; flow in the furthest downstream reaches of each stream is supported by seepage from limestone bedrock.

2.4.2 Neosho River Drainage

The Neosho River drains the Central Lowlands of southeastern Kansas and portions of northern Oklahoma east and southeast of the Cherokee County subsites (see Figure 1-1; Section 1.0). The Neosho River is larger than the Spring River, draining 5,876 square miles (above Commerce, Oklahoma) with an average annual discharge of 4,428 cfs as measured at the Commerce gage. For the most part the drainage basin is founded on shale. The river is subject to periodic flooding and a peak discharge of 267,000 cfs has been recorded at the Commerce gage (USEPA Storet data).

The Neosho River receives flow from Tar Creek which, prior to its confluence with the Neosho River, flows through the Tar Creek Superfund Site (Picher mine field) located immediately south and downstream of the Treece subsite. Treece subsite surface-water sources contribute an estimated 8 percent of the annual zinc load in Tar Creek as measured where Tar Creek enters the Neosho River. Treece subsite ground-water sources contribute an additional zinc load to Tar Creek via mine openings in Oklahoma, estimated to be an additional 2 to 11 percent of the zinc load added to the Neosho River.

Water quality in the Neosho River is good, generally meeting AWQC except for iron which is regularly exceeded upstream of Tar Creek inputs. Chronic AWQC for zinc are occasionally exceeded as a result of contributions from Tar Creek (OWRB, 1983). Fish species composition and productivity in upper Grand Lake (five miles downstream from the mouth of Tar Creek) and within the Neosho River downstream from Tar Creek showed no apparent effects from mining activities, (Aggus *et al.*, 1983). Benthic macroinvertebrate density, particularly the dipteran *Chaoborus sp.*, increased in the Neosho River from the mouth of Tar Creek to upper Grand Lake as the river channel changes from a lotic to a lentic environment in this area. Tar Creek, however, was sufficiently contaminated from mining activities to result in low abundance and diversity of fish and benthic invertebrates where the creek enters the Neosho River at Miami, Oklahoma approximately 10 miles downstream from the Treece subsite (Aggus *et al.*, 1983). Aggus *et al.*, (1983) concluded that effects on the fish community diminish rapidly once water enters the Neosho River, primarily because of the ameliorating effects of increased water hardness in the Neosho River. Recent data concerning water quality and/or biologic conditions within the Neosho River were unavailable at this writing. It is assumed that conditions within the river are similar to that reported in 1983.

Treece Subsite Drainages

Tar Creek is the principal stream in the Treece subsite with a drainage area of approximately 8.6 square miles within Kansas (see Figure 1-1, Section 1.0 and Figure 2-1). The Tar Creek watershed is located southwest of the Willow Creek watershed and flow is from north to south through the Treece subsite to the Neosho River in Oklahoma. Approximately 50 percent of the Treece subsite is drained by Tar Creek. Two other watersheds, Tar Creek Tributary and Lytle Creek, drain the eastern portion of the Treece subsite before joining Tar Creek in Oklahoma. Tar Creek Tributary and Lytle Creek were considered unsuitable for supporting fish populations because of low stream flow. These watersheds cover a combined area of 7.7 square miles, and with Tar Creek, cover approximately 97 percent of the Treece subsite.

There are no known continuous stream flow records for Tar Creek or its tributaries within the Treece subsite; however, the USGS maintains a continuous-recording stream gage near the mouth of Tar Creek at Miami, Oklahoma, 10 miles downstream of the subsite. The drainage area above the Tar Creek station at Miami is 44.7 square miles. A maximum discharge of 3,100 cfs and a minimum discharge of 0.07 cfs have been recorded at this stream gage over the period of record from 1984 to 1990. Comparing the estimated Treece subsite annual discharge (5.3 cfs) to the average annual value for Tar Creek at Miami, Oklahoma (68.6) indicates that runoff from the Treece subsite contributes, on average, 8 percent of the flow in Tar Creek.

Since 1980, groundwater within the interconnected mines of the Picher field, including the mine workings in the Treece subsite, has overflowed to Tar Creek via shafts/drill holes near Picher and Commerce, Oklahoma. This mine water contains metals and sulphate and contributes an estimated 3,400 acre-ft/year to the flow in Tar Creek (Parkhurst, 1986). This represents about 12 percent of the average annual flow in Tar Creek as measured at Miami, Oklahoma. Neither mine shafts or cased holes in the Treece subsite have been observed to overflow to streams.

The Tar Creek watershed within the Treece subsite is similar to Willow Creek in terms of land use. Mining activity is generally confined to the lower one-half of the watershed within the Treece subsite, while the upstream area is dominated by agriculture. Water quality is influenced by runoff from agricultural areas and seepage and/or surface runoff from mill waste piles and tailings ponds. Suspended solids and total recoverable iron concentrations in Tar Creek were typically highest at the upstream (baseline) sampling station, TC-1, and decreased with distance downstream.

Tar Creek Surface Water Quality (total recoverable - mg/L)

Metal	Upstream (Mean)	Downstream (Mean)	Chronic AWQC[1]	Acute AWQC
Cadmium	0.001	0.02	0.0006-0.0038	---
Iron	10.8	1.3	---	1.0
Lead	0.03	0.05	0.001-0.023	---
Zinc	0.18	9.2	0.050-0.39	---

[1] Adjusted for sample hardness.

As with Willow Creek, these data indicate that suspended solids and associated iron are related to agricultural practices rather than to mining activities. Iron concentrations in Tar Creek generally exceeded acute AWQC in the May and December sampling events.

Dissolved solids, cadmium, lead, and zinc concentrations increased with distance downstream. Upstream concentrations of cadmium, lead, and zinc were at or near chronic AWQC levels but increased downstream during all sampling rounds as a result of runoff and seepage from mill waste areas.

The Tar Creek Tributary watershed has the same general land use and water quality characteristics as main-stem Tar Creek, i.e., runoff and seepage from mine waste impact water quality in surface waters. In the Lytle Creek watershed only 0.6 percent of the land surface is affected by mine-related operations and water quality is dominated by runoff from croplands. Acute AWQC for iron and chronic AWQC for cadmium, lead, and zinc were generally exceeded during all sampling rounds.

2.4.3 On-Site Ponds

The larger tailings and subsidence ponds were inventoried and a subset of these were sampled as a part of the RI (see Section 3.4 of the Final Draft RI). In general, tailings ponds were relatively shallow and covered up to 8.5 acres in surface area. Maximum depths of the ponds inspected ranged from approximately 4 to 12 feet in tailings ponds and 60 feet or deeper in subsidence ponds.

The ponds sampled exhibited neutral to slightly basic field pH ranging from 6.5 to 8.4. Specific conductance varied from 160 to 1,900 umhos/cm showing considerable variation in dissolved solids and hardness. Suspended solids concentrations were low or below detection limits in most ponds, indicating that metals detected in ponds are generally in the soluble or colloidal form as opposed to the particulate form.

All the ponds sampled represented undisturbed, post-mining conditions with the exception of Ballard Pond. Water in Ballard Pond is used in a chat-wash operation at the Ballard site. The Ballard Pond exhibited relatively high levels of metals. Additionally, the extreme water level fluctuations caused by the chat washing operations severely limits the opportunity for the establishment of a fishery.

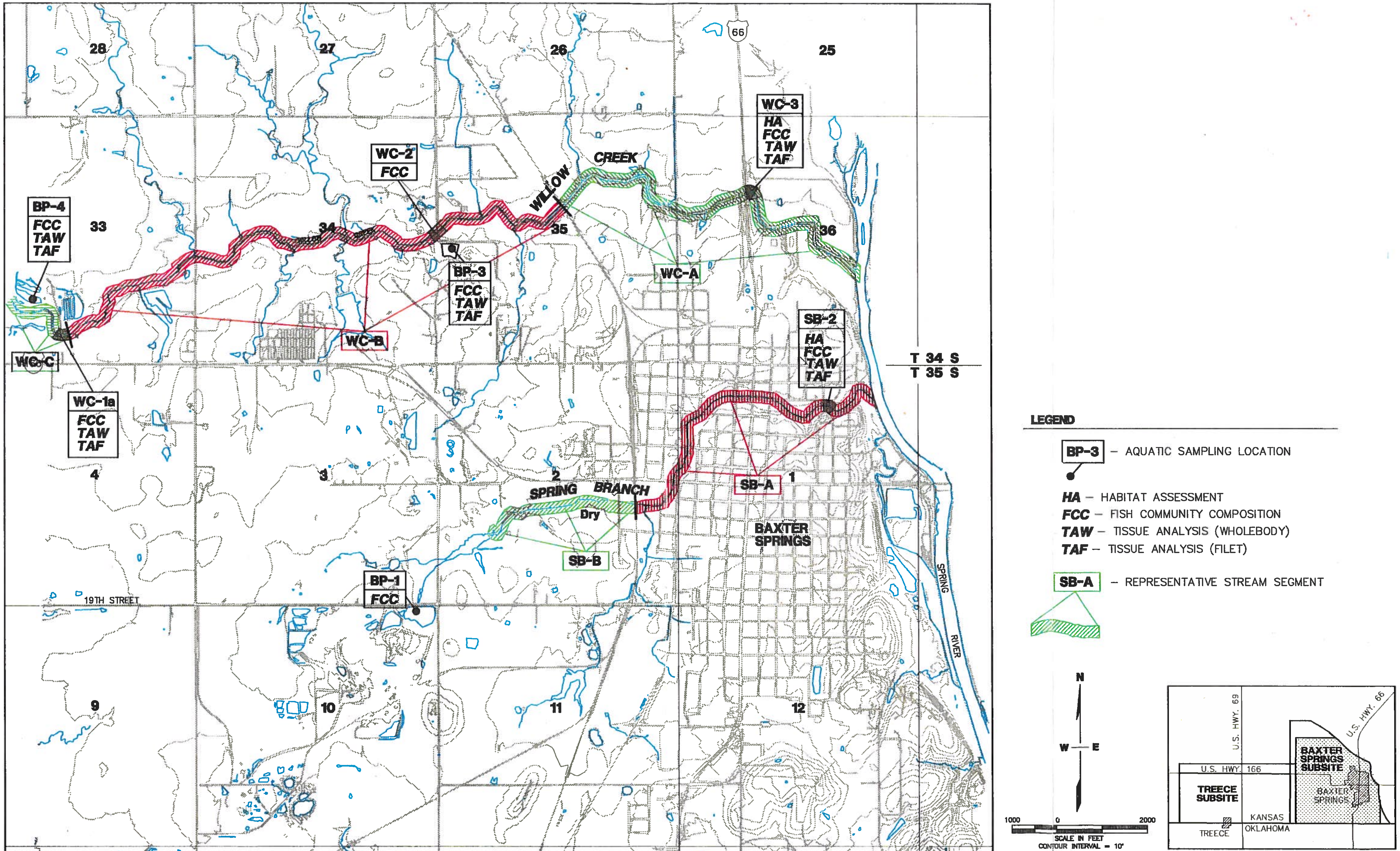
Metal concentrations in each of the ponds sampled were compared to acute AWQC. In five of the six ponds sampled during the aquatic biota investigation, metal concentrations (especially zinc) exceeded acute AWQC for one or more metals. However, fish were caught in four of the five ponds containing metals concentrations that exceed acute AWQC. Zinc concentrations in the ponds were generally higher than those observed in Willow Creek but less than average concentrations measured in Spring Branch, Tar Creek, and Tar Creek Tributary. Significantly, the physical limitations of the ephemeral streams (lack of water, fluctuating flow rates and water temperatures, high sediment load, etc.) present serious obstacles to aquatic life, and it is suspected that the deeper tailings and subsidence ponds offer a more stable, life-supporting environment than the on-site streams.

2.5 DESCRIPTION OF AQUATIC HABITATS AND POPULATIONS ONSITE

Figures 2-2 and 2-3 depict the location of aquatic habitats on site. Aquatic habitats occur in conjunction with: (1) Tar Creek and tributary watersheds in the Treece subsite; (2) the Willow Creek-Spring Branch watershed in the Baxter Springs subsite; (3) the stock ponds, flotation tailings ponds, and flooded subsidence pits that occur throughout the subsites; and (4) the Spring and Neosho Rivers that drain the subsites.

A small reservoir is located on the Spring River near Baxter Springs. This reservoir is inhabited by various game fish such as largemouth bass (*Micropterus salmoides*), spotted bass (*Micropterus punctulatus*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), bluegill (*Lepomis macrochirus*), and crappie (*Pomoxis* spp.) (USSCS, 1985). Interviews with local fishermen and a local bait shop owner in Baxter Springs, indicated that white bass (*Morone crysops*), hybrid white bass-stripers, and walleye (*Stizostedion vitreum vitreum*) migrate upstream from Grand Lake to the reservoir dam during spawning runs.

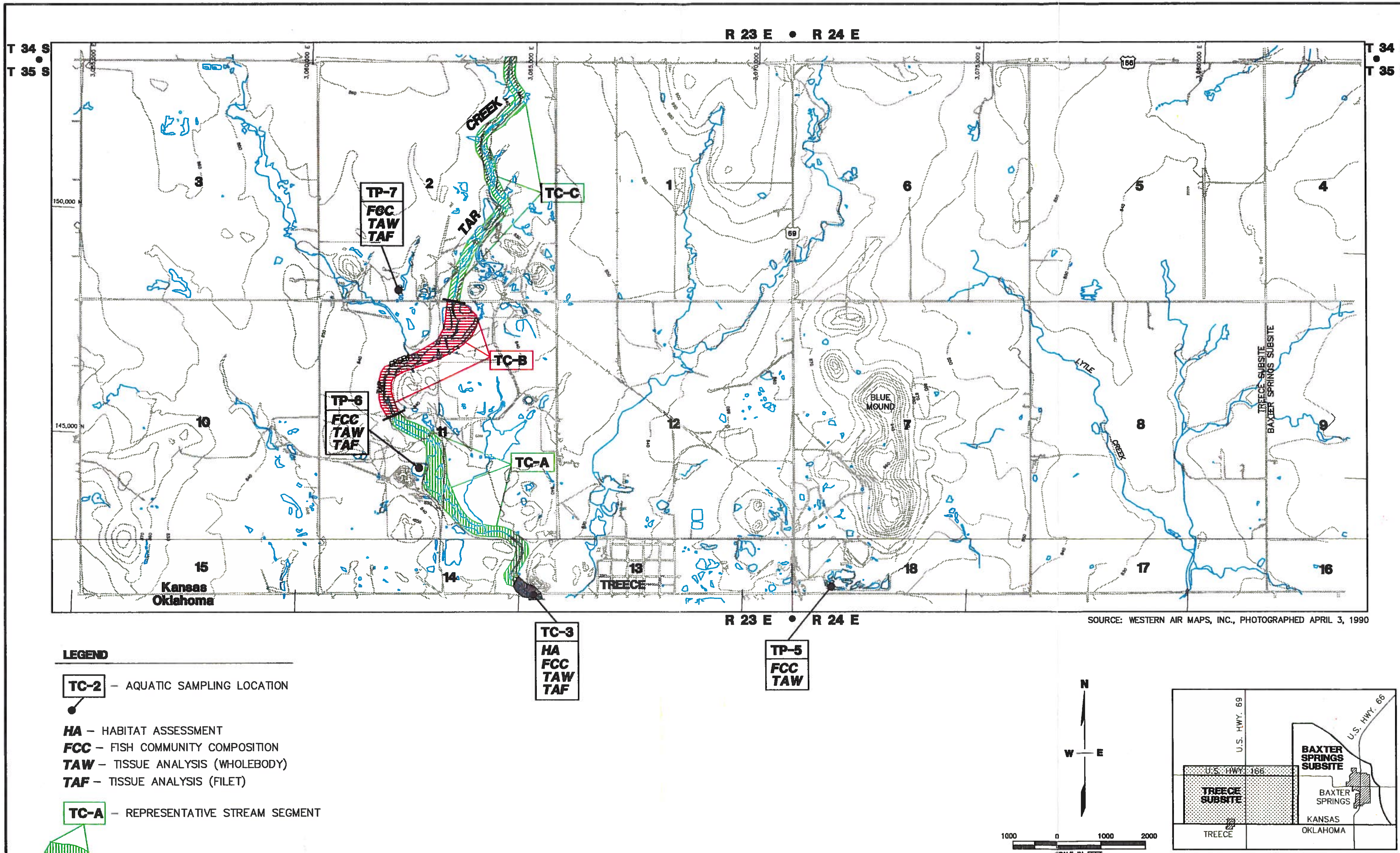
Fish potentially inhabiting the streams and ponds of the subsites include centrarchids (sunfish), ictalurids (catfish and bullheads), catostomids (suckers), and cyprinids (minnows). Forage fish such as shad and minnows are also expected to occur in small ponds and shallow



R 24 E

SOURCE: WESTERN AIR MAPS, INC. PHOTOGRAPHED APRIL 3, 1990

Dames & Moore	CHEROKEE COUNTY, KANSAS CERCLA SITE Baxter Springs / Treece Subsites *Baxter Springs Subsite* AQUATIC SAMPLING LOCATIONS	FILE NO. BAX-AS1 DATE 7/92 DRAWING NO. 2-2



SOURCE: WESTERN AIR MAPS, INC., PHOTOGRAPHED APRIL 3, 1990

Dames & Moore	CHEROKEE COUNTY, KANSAS CERCLA SITE Baxter Springs / Treece Subsites *Treece Subsite* AQUATIC SAMPLING LOCATIONS	FILE NO. TRC-AS1 DATE 7/92 DRAWING NO. 2-3

water onsite (CH2M Hill, 1986). Fish that would likely inhabit deeper water of the subsites, include smallmouth buffalo (*Ictiobus bubalus*), longear sunfish (*Lepomis megalotis*), redear sunfish (*Lepomis microlophus)*, green sunfish (*Lepomis cyanellus*), common carp (*Cyprinus carpio*) and other species typical of midwestern warmwater streams. Of these species, longear sunfish, redear sunfish, and green sunfish were collected by Dames & Moore personnel in on-site streams and ponds during the RI. In addition to these species, bluegill, spotted sucker, brook silverside, largemouth bass, black bullhead, pumpkinseed, white crappie, dace, and shiners were collected during the RI. Since many of the streams and ponds onsite are stocked, the presence of fish not indigenous to this area is possible.

The Baxter Springs/Treece subsite RI aquatic sampling program is the only known investigation of fish species occurrence within those segments of Tar Creek, Spring Branch, and Willow Creek that exist on site. However, the KDWP (1992a, personal communications with Mr. Larry Zuckerman, KDWP) indicated that only certain fish species would be expected to occur in ephemeral streams of southeastern Kansas. These species include yellow bullhead (*Ictalurus natalis*), black bullhead, green sunfish, blackstripe topminnow (*Fundulus notatus*), bullhead minnow (*Pimephales vigilax*), bluntnosed minnow (*Pimephales notatus*), red shiner (*Notropis lutrensis*), and mosquitofish (*Gambusia affinis*).

Although the Spring and Neosho Rivers are not technically within the boundaries of the subsites, potential impacts to aquatic organisms in these rivers are evaluated in this assessment, since they constitute the perennial receiving channels for subsite waters. The assessment of water quality within the rivers is based on existing data collected by KDHE (1980, 1987) and discharge data from the USGS (1987).

Both the Spring and Neosho Rivers are inhabited by diverse populations of fish and benthic invertebrates. Ferrington *et al.*, (1988) reported benthic invertebrates representing Ephemeroptera (mayflies), Odonata (dragonflies/damselflies), Megaloptera (alderflies), and Diptera (flies, mosquitoes, and midges) occurred within Empire Lake. Within the Spring River arm of Grand Lake O' the Cherokees, Aggus *et al.*, (1983) collected spotted gar, gizzard shad, carp, river carpsucker, smallmouth buffalo, channel catfish, flathead catfish, white bass, white crappie, and freshwater drum. The same species, excluding the white crappie and spotted gar were collected by Aggus *et al.*, (1983) from the Neosho River downstream from Tar Creek. In addition, longnose gar, shortnose gar, bigmouth buffalo, green sunfish, warmouth, orangespotted sunfish, bluegill, and longear sunfish were collected downstream from Tar Creek. Benthic invertebrates collected by Aggus *et al.* (1983) from the Neosho River and Spring River included oligochaetes and tubificids (aquatic earthworms), ephemeropterans (mayflies), Chironomids (midges) and Culicids (mosquitoes).

The Aggus *et al.*, (1983) study site of Tar Creek, within 3 miles of its confluence with the Neosho River, indicated that a variety of fish and benthic invertebrate species inhabit the stream. Fish species collected in Tar Creek included chubs, river carpsucker, channel catfish, killifish, *Gambusia sp.* (live bearers), brook silverside, white bass, green sunfish, bluegill, largemouth bass, and white crappie. Benthic invertebrates included mosquitoes, true flies, beetles, and true bugs.

The Kansas Department of Health and Environment (KDHE, 1992) collected benthic macroinvertebrates from Willow Creek northwest of Baxter Springs on May 18, 1979. The major groups of benthic macroinvertebrates represented in the samples include Coleopterans (water beetles), Dipterans (flies, mosquitoes, and midges), Ephemeropterans (mayflies), Hemipterans ("true bugs"), Odonates (dragonflies and damselflies), and Plecopterans (stoneflies).

2.6 SENSITIVE HABITATS AND SPECIES

Sensitive habitats that may occur within the subsites include habitats critical for the survival of threatened or endangered (T&E) species and wetlands, both of which are protected by various legislative acts and executive orders. Potential impacts were evaluated for T&E species that do or could occur within the subsites. T&E species lists and critical habitat descriptions were provided by the Kansas Department of Wildlife and Parks (KDWP, 1991).

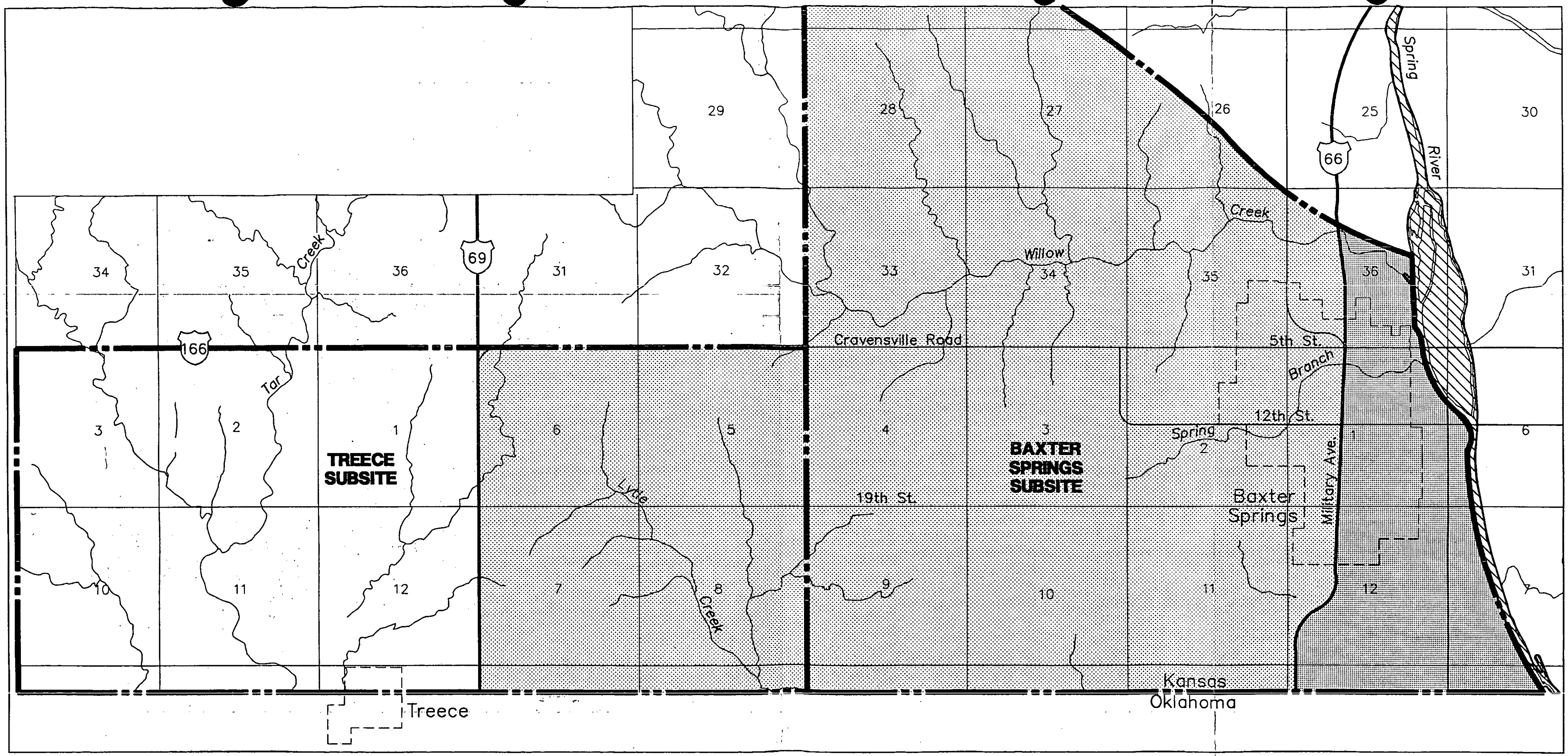
2.6.1 T&E Animal Species

No federally-listed T&E species are known to occur or have habitat within the areas impacted by mines (i.e., west of State Highway 66) within the Baxter Springs/Treece subsites.

The listing of T&E species for Kansas describes nine species which have critical habitat within the subsite (Table 2-1). The Kansas Non-Game and Endangered Species Conservation Act (K.S.A. 32:501-502) affords protection of habitat of species that are uncommon or not widely distributed in the state. Frequently, the species are common elsewhere but are uncommon in the state because of marginal habitat.

Critical habitat has been defined by Kansas Administrative Regulation as those areas documented as currently supporting self-sustaining population(s) of any threatened or endangered species of wildlife as well as those areas determined by Kansas Department of Wildlife and Parks to be essential for the conservation of any threatened or endangered species of wildlife. If a proposed project has potential to affect a listed species or its habitat, the project proponent must contact the Environmental Services Section, Kansas Department of Wildlife and Parks, Pratt, Kansas.

Of the nine listed species that have critical habitat within the subsites, only the northern spring peeper (*Hyla crucifer crucifer*) has critical habitat west of State Highway 66 in the general area where mine and mill deposits have been identified (Figure 2-4). The northern spring peeper is found in woodlands near small ponds, swamps, and the twilight zone of caves, east of State Highway 69 running north through Treece. This small frog prefers woodland ponds, with abundant emergent vegetation, marshes with standing trees, or shrubs during the March breeding season. Drainage and clearing of woodland wetlands has drastically reduced the peeper's preferred habitat.



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SCALE IN FEET

LEGEND

NORTHERN SPRING PEEPER

GREEN FROG

NORTHERN SPRING PEEPER
CAVE SALAMANDER
CENTRAL NEWT
EASTERN NARROW MOUTH TOAD
GRAYBELLY SALAMANDER
GROTTO SALAMANDER
NORTHERN RED BELLY SNAKE
DARK-SIDED SALAMANDER

CRITICAL HABITAT WITHIN SUBSITE BOUNDARIES, AS DESIGNATED BY KANSAS DEPARTMENT OF WILDLIFE AND PARKS, 1992.
GREEN FROG CRITICAL HABITAT (WETLANDS/SLOUGHS ALONG SPRING RIVER) MAY FALL WITHIN SUBSITE BOUNDARY (SEE APPENDIX E OF RI).

SUBSITE BOUNDARY

Source: USGS Quadrangle Mapping;
Kans, Baxter Springs, Neutral Quad Maps.

Dames & Moore

CHEROKEE COUNTY, KANSAS CERCLA SITE
Baxter Springs / Treece Subsites

CRITICAL HABITAT (KDWP, 1992)

FILE NO. HABITAT.DWG

DATE 6/92

DRAWING NO. **2-4**

The other eight species all have designated habitat east of State Highway 66 running north through Baxter Springs, near the Spring River where outcrops of Mississippian limestone provide the caves and springs they require. While their designated habitat is removed from the mine and mill waste deposits, individuals could be exposed to elevated metals concentrations via contact with runoff from upstream areas (i.e., Willow Creek and Spring Branch flow through the designated habitat area).

TABLE 2-1

STATE-LISTED THREATENED OR ENDANGERED SPECIES[1] OF THE BAXTER SPRINGS/TREECE SUBSITES

Species	Threatened	Endangered	Critical Habitat East of Highway 69[1]	Critical Habitat East of Highway 66[1]	Documented Occurrence East of Highway 66[1]
Cave Salamander		X		X	X
Central Newt	X			X	
Dark-Sided Salamander	X			X	X
Eastern Narrowmouth Toad	X			X	X
Greybelly Salamander		X		X	
Green Frog	X			X	
Grotto Salamander		X		X	
Northern Redbelly Snake	X			X	
Northern Spring Peeper	X		X	X	X

[1] KDWP threatened or endangered species listing, 1991.

2.6.2 Threatened and Endangered Plant Species

Three federally-listed T&E plant species occur in Kansas; Mead's milkweed (*Asclepias meadii*), running buffalo clover (*Trifolium stoloniferum*) and western prairie fringed orchid (*Platanthera praeclara*). Previous collections and the lack of suitable habitat indicate that these species are unlikely to occur within Cherokee County or the BS/T subsites. The State of Kansas has no native plant protection law and, therefore, no state-listed T&E plant species.

2.6.3 Wetlands

Wetlands are protected by Section 404 of the Clean Water Act and by Executive Order 11990, Protection of Wetlands, as well as by a variety of other acts and executive orders (e.g., Fish and Wildlife Coordination Act, 1958; Executive Order 11988, Floodplain Management, 1977; Emergency Wetlands Resource Act, 1986). Wetlands of the site are associated with streams, ponds, and marshes that have developed in subsidence pits and collapsed shafts as well as tailings and commercial chat-wash water recirculation ponds. Wetlands that exist onsite are shown in Appendix E of the RI. Although many of the wetlands (i.e., subsidence ponds and flotation ponds) onsite were established as a result of mining activities, the creation of these areas completes a potential pathway of exposure for organisms inhabiting or frequenting the wetlands. These wetlands potentially provide critical habitat for the state-listed northern spring peeper.

3.0 EXTENT OF CONTAMINATION ONSITE

Chemical and other data on the nature and extent of contamination onsite were collected during the RI in order to support the ERA. These data, which are described in detail in the Final Draft RI, are summarized below.

3.1 SOILS

The initial site inspection of the subsites revealed four basic soil types: (1) baseline (B-Horizon) soils; (2) soil in croplands, pastures, floodplains, and woodlots (i.e., agricultural and A-Horizon soil samples); (3) soils near the mill waste areas (i.e., near-pile soils); and (4) soils at mill sites. Near-pile, agricultural/A-Horizon, and baseline soil data were used to estimate dose/exposures to terrestrial and aquatic receptors and to identify COCs. Each of these soil types and its role in the ERA is described below. Mill-site soils generally lacked vegetative growth and associated habitat/forage. Terrestrial receptors were not assumed to ingest mill-site soils, since mill waste and associated mill-sites do not represent habitat over which receptors are likely to range. Therefore, mill-site soils were not used in the ERA because terrestrial receptors were assumed to inhabit near-pile and agricultural/A-Horizon soils.

3.1.1 Baseline Soils

Data on the concentration of metals in site baseline soil samples were necessary to determine if metals concentrations measured in on-site soils were elevated relative to local conditions. Site baseline soil samples were collected from the B-Horizon soil (the mineral soil horizon immediately below the A-Horizon) 14 to 24 inches below ground surface to minimize inclusion of near-surface metal accumulations in the litter or A-Horizon. Eight baseline soil samples were collected from diverse areas onsite (five from the Baxter Springs subsite and three from the Treece subsite). The geometric and arithmetic means and upper-bound (95th percentile) concentration of metals in baseline soils as well as information on the range, frequency of detection, and sample variance (standard deviation) are listed in Table 3-1. Upper-bound values were calculated using Equation 5-1 given in Section 5.3.

3.1.2 Agricultural/A-Horizon Soils

Agricultural and A-Horizon (Ag/A) soils from the subsites were characterized by sampling 15 locations in tilled and fallow fields, tame grass pastures, woodlands, and old fields. These samples represent typical subsite soils distant from mill waste accumulations. The geometric and arithmetic means and upper-bound (95th percentile) concentration of metals in Ag/A soils as well as information on the range, frequency of detection, and sample variance (standard deviation) are listed in Table 3-2. Compared to the upper-bound metals concentrations measured in B-Horizon baseline soils, Ag/A soils contain slightly higher levels of cadmium, chromium, cobalt, lead, manganese, nickel, vanadium, and zinc. Ag/A soil data were used to determine if metals levels in these soils are phytotoxic.

3.1.3 Near-Pile Soils

A-Horizon soils collected from areas near mine waste piles that were affected by past mining activities are defined as near-pile soils. Typically, near-pile soils were completely vegetated with a well-defined organic layer (humus and leaf litter). All near-pile sample stations were located within 300 feet of a mill waste accumulation due to the proximity of tilled fields (for sample locations see Section 3.2 of the Final Draft RI). At most sample stations, 0- to 2-inch and 10- to 12-inch core samples were collected. These data sets were combined because: (1) there was no statistical difference between the concentration of metals in 0- to 2-inch and 0- to 12-inch soil samples, and (2) the 0- to 12- inch data set was larger than the 0- to 2- inch set (n = 19 versus n = 12). This increase in sample size reduces statistical variability of the near-pile soil data. Since analyses showed there is no statistical difference between metals levels measured at these two depths, the combined data

Table 3-1. Concentration of Metals in B-Horizon Baseline Soils at the Baxter Springs/Treece Subsites

Metal	n	Range (mg/kg)	Frequency of Detection	Arithmetic Mean (mg/kg)	Geometric Mean (mg/kg)	Upper-Bound Value (mg/kg)	Standard Deviation (mg/kg)
Arsenic	8	ND[a] (1.9)-16	6/8 (75%)	7.1	5.2	11.1	5.9
Barium	8	70-370	8/8 (100%)	125	105.4	192.3	100.7
Beryllium	8	ND (0.15)-1.8	7/8 (88%)	1.0	0.8	1.3	0.5
Cadmium	8	ND (0.3-0.6)	0/8 (0%)	NA[b]	NA	NA	NA
Chromium	8	18-43	8/8 (100%)	30.3	28.7	37.1	10.2
Cobalt	8	4-11	8/8 (100%)	6.8	6.4	8.4	2.5
Copper	8	5-15	8/8 (100%)	10.5	9.8	13.1	3.9
Iron	8	18,000-71,000	8/8 (100%)	34,000	29,788	47,646	20,368
Lead	8	8.9-29	8/8 (100%)	17.4	16.3	21.7	6.5
Manganese	8	32-370	8/8 (100%)	133.5	98.9	209.8	113.9
Mercury	8	ND (0.05)	0/8 (0%)	NA[b]	NA	NA	NA
Nickel	8	6-18	8/8 (100%)	10.8	10.0	13.7	4.4
Selenium	8	ND (0.3)-0.9	1/8 (13%)	0.5	0.4	0.6	0.2
Silver	8	ND (0.5)-2	1/8 (13%)	0.7	0.6	1.0	0.5
Thallium	8	ND (0.3-0.6)	0/8 (0%)	NA[b]	NA	NA	NA
Vanadium	8	32-77	8/8 (100%)	51.1	49.1	61.4	15.3
Zinc	8	9-170	8/8 (100%)	43.9	28.8	79.2	52.7

[a] ND = not detected; value in parentheses represents one-half the reporting limit.

[b] Since all measured values were nondetectable, mean, upper-bound, and standard deviation values were not calculated.

Table 3-2. Concentration of Metals in Agricultural/A-Horizon (Ag/A) Soils at the Baxter Springs/Treece Subsites

Metal	n	Range (mg/kg)	Frequency of Detection	Arithmetic Mean (mg/kg)	Geometric Mean (mg/kg)	Upper-Bound Value (mg/kg)	Standard Deviation (mg/kg)
Arsenic	15	2.7-13	15/15 (100%)	5.6	5.0	6.8	2.6
Cadmium	15	ND (0.3)-5	5/15 (33%)	0.9	0.6	1.4	1.2
Chromium	8	13-55	8/8 (100%)	31.3	27	42.6	17
Cobalt	8	2.4-19	8/8 (100%)	10.8	8.4	15.6	7.2
Copper	15	3-14	15/15 (100%)	7.1	6.4	8.7	3.4
Iron	15	6800-74,000	15/15 (1005)	29,587	22,732	36,906	28,350
Lead	15	12-150	15/15 (100%)	42	32.1	58.3	35.9
Manganese	15	25-1300	15/15 (100%)	593	431.9	773.3	396.6
Nickel	8	2.6-23	8/8 (100%)	10.8	8.9	15.3	6.7
Selenium	8	ND (0.3)[a]-0.7	1/8 (13%)	0.4	0.3	0.4	0.1
Vanadium	8	25-88	8/8 (100%)	48.9	44.8	63.6	22
Zinc	15	23-830	15/15 (100%)	156.5	99.0	246.8	198.6

[a] ND = not detected; value in parentheses represents one-half the reporting limit.

will not significantly alter results. Furthermore, EPA agreed that using the 0- to 12-inch data was appropriate for the Human Health Risk Assessment (HHRA). To be consistent with the HHRA, the ERA also used the 0- to 12-inch data. B-Horizon data were collected from a depth of 14 inches or greater. The geometric and arithmetic means and upper-bound (95th percentile) concentration of metals in near-pile soils as well as information on the range, frequency of detection, and sample variance (standard deviation) are listed in Table 3-3.

Upper-bound concentrations of cadmium, chromium, cobalt, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc were higher in near-pile soils than concentrations of these metals measured in B-Horizon baseline soils. Upper-bound levels of arsenic, cadmium, cobalt, copper, iron, lead, manganese, nickel, selenium, and zinc were higher than the upper-bound concentration of these metals measured in Ag/A-Horizon soils.

Near-pile soil data were used to estimate doses for key terrestrial species and to determine if metals levels in these soils are phytotoxic. Terrestrial receptors were assumed to range or forage in near-pile areas rather than mine/mill waste areas because the mining-related areas including excavated chat, chat and tailings accumulations, and mill sites generally lack vegetation. In reality, only a small fraction of the subsites actually contains soils with metals at or above concentrations of near-pile soil levels, and most terrestrial receptors that inhabit the subsites are likely to range over a relatively large area. For example, if we assumed that a receptor spent equal time anywhere on the subsites, then the average exposure concentration would be an area-weighted average concentration of metals from mill waste, mill-site, near-pile and agricultural areas. However, it was conservatively assumed that animals range <u>exclusively</u> over near-pile soils which yields a higher metal concentration value.

Table 3-3. Concentration of Metals in Near-Pile Soils (0 to 12 in.) at the Baxter Springs/Treece Subsites

Metal	n	Range (mg/kg)	Frequency of Detection	Arithmetic Mean (mg/kg)	Geometric Mean (mg/kg)	Upper-Bound Value (mg/kg)	Standard Deviation (mg/kg)
Arsenic	19	3.7-17	19/19 (100%)	7.8	7.2	9.2	3.5
Barium	19	53-220	19/19 (100%)	106.6	100	124	43.6
Beryllium	19	0.4-1.7	19/19 (100%)	0.8	0.7	0.9	0.3
Cadmium	19	0.8-21	19/19 (100%)	4.5	2.7	6.6	5.2
Chromium	19	10-91	19/19 (100%)	34.2	29.7	41.7	18.9
Cobalt	19	4-42	19/19 (100%)	15.6	12.5	19.8	10.6
Copper	19	4.9-26	19/19 (100%)	9.9	9.2	11.8	4.7
Iron	19	13,000-112,000	19/19 (100%)	48,947	37,306	62,957	35,218
Lead	19	25-300	19/19 (100%)	88.1	72.1	113.9	64.8
Manganese	19	120-2100	19/19 (100%)	947.4	678.9	1286.5	852.4
Mercury	19	ND (0.01[a])-0.3	1/19 (5%)	0.1	0.3	0.1	0.05
Nickel	19	4-44	19/19 (100%)	14.5	10.7	19.4	12.2
Selenium	19	0.5-1.0	19/19 (100%)	0.6	0.6	0.7	0.2
Silver	19	1-4	19/19 (100%)	1.2	1.1	1.5	0.7
Thallium	19	0.3-2	19/19 (100%)	0.7	0.7	0.9	0.3
Vanadium	19	21-96	19/19 (100%)	53.2	48.4	62.5	23.3
Zinc	19	230-2900	19/19 (100%)	710.0	497.9	995.6	717.9

[a] ND= not detected; value in parentheses represents one-half the reporting limit.

3.2 SURFACE WATER AND SEDIMENT

Site-specific sampling and laboratory analysis of surface water (on-site streams included in the aquatic biota sampling program and ponds) and sediments was conducted during the RI. More specific information on surface water and sediment sampling methods and results is presented in the Final Draft RI. Sediment data were not directly used in the ERA, since potential toxic effects to aquatic receptors were evaluated by using site-specific surface water data. Surface water data were analyzed using the total recoverable metals method unless otherwise noted.

The arithmetic mean and upper-bound (95th percentile) concentration of total recoverable metals from main stem stations on the major ephemeral streams (Tar Creek, Willow Creek, Spring Branch) were used along with subsite pond results. The sample data along with information on sample variance, range, and frequency of detection are presented in Tables 3-4, 3-5, 3-6, and 3-7. Six (arsenic, barium, chromium, cobalt, selenium, and vanadium) of the 15 metals analyzed in on-site streams and ponds did not occur at detectable levels in any of the three streams. Those metals detected in one or more of the streams or ponds sampled include cadmium, copper, iron, lead, manganese, mercury, nickel, silver, and zinc.

Two tributaries of Tar Creek, Tar Creek Tributary and Lytle Creek, were sampled as a part of the RI studies. These data have not been used in this assessment. Although much smaller in magnitude, the water quality in Tar Creek Tributary is similar to that in Tar Creek and conclusions pertaining to the larger stream apply to its smaller tributary. Three water samples from Lytle Creek, on the other hand, indicated good water quality similar to the upstream (WC-1/TC-1) results (Figure 2-1). Very little mill waste is present in the Lytle Creek drainage within Kansas (0.6 percent) and the sampling results are reflective of "background" conditions. Since the stream was relatively unaffected by mine-related activities, it was excluded from the analysis.

Table 3-4. Concentration of (Total Recoverable) Metals in Tar Creek at Sampling Stations TC-2 and TC-3

Metal	n	Range (mg/L)	Frequency of Detection	Arithmetic Mean (mg/L)	Geometric Mean (mg/L)	Upper-Bound Value (mg/L)	Standard Deviation (mg/L)
Arsenic	6	ND (0.0025)[a]	0/6 (0%)	NA[b]	NA	NA	NA
Barium	6	ND (0.1)	0/6 (0%)	NA[b]	NA	NA	NA
Cadmium	6	0.007-0.04	6/6 (100%)	0.02	0.01	0.03	0.02
Chromium	6	ND (0.005)	0/6 (0%)	NA[b]	NA	NA	NA
Cobalt	6	ND (0.025)	0/6 (0%)	NA[b]	NA	NA	NA
Copper	6	ND (0.0025)-0.01	1/6 (17%)	0.005	0.004	0.008	0.004
Iron	6	0.4-2.5	6/6 (100%)	1.3	1.1	2.1	0.9
Lead	6	0.025-0.08	6/6 (100%)	0.05	0.05	0.07	0.03
Manganese	6	0.22-1.1	6/6 (100%)	0.6	0.5	0.9	0.4
Mercury	6	ND (0.0001)	0/6 (0%)	NA[b]	NA	NA	NA
Nickel	6	ND (0.02)-0.05	2/6 (33%)	0.03	0.03	0.04	0.01
Selenium	6	ND (0.0025)	0/6 (0%)	NA[b]	NA	NA	NA
Silver	6	ND (0.00005)-0.0021	4/6 (67%)	0.0004	0.0002	0.001	0.0008
Vanadium	6	ND (0.025)	0/6 (0%)	NA[b]	NA	NA	NA
Zinc	6	0.7-22	6/6 (100%)	9.2	4.8	17.2	9.7

[a] ND = not detected; value in parentheses represents one-half the reporting limit.

[b] Since all measured values were nondetectable, mean, upper-bound, and standard deviation values were not calculated.

Table 3-5. Concentration of (Total Recoverable) Metals in Willow Creek at Sampling Stations WC-2 and WC-3

Metal	n	Range (mg/L)	Frequency of Detection	Arithmetic Mean (mg/L)	Geometric Mean (mg/L)	Upper-Bound Value (mg/L)	Standard Deviation (mg/L)
Arsenic	8	ND (0.0025)[a]	0/8 (0%)	NA[b]	NA	NA	NA
Barium	8	ND (0.1)	0/8 (0%)	NA[b]	NA	NA	NA
Cadmium	8	ND (0.00025)-0.0023	7/8 (88%)	0.0015	0.0013	0.0019	0.0007
Chromium	8	ND (0.005)	0/8 (0%)	NA[b]	NA	NA	NA
Cobalt	8	ND (0.025)	0/8 (0%)	NA[b]	NA	NA	NA
Copper	8	ND (0.0025)-0.009	5/8 (63%)	0.005	0.004	0.007	0.004
Iron	8	0.2-11	8/8 (100%)	3.9	2.1	6.3	3.6
Lead	8	0.002-0.025	8/8 (100%)	0.014	0.011	0.02	0.009
Manganese	8	0.13-0.35	8/8 (100%)	0.21	0.20	0.26	0.07
Mercury	8	ND (0.0001)	0/8 (0%)	NA[b]	NA	NA	NA
Nickel	8	ND (0.02)-0.03	2/8 (25%)	0.02	0.02	0.03	0.004
Selenium	8	ND (0.0025)	0/8 (0%)	NA[b]	NA	NA	NA
Silver	8	ND (0.0001)-0.0006	4/8 (50%)	0.0002	0.0002	0.0003	0.0002
Vanadium	8	ND (0.025)	0/8 (0%)	NA[b]	NA	NA	NA
Zinc	8	0.9-1.0	8/8 (100%)	0.63	0.57	0.81	0.27

[a] ND=not detected; value in parentheses represents one-half the reporting limit.

[b] Since all measured values were nondetectable, mean, upper-bound, and standard deviation values were not calculated.

Table 3-6. Concentration of (Total Recoverable) Metals in Spring Branch at Sampling Stations SB-1 and SB-2

Metal	n	Range (mg/L)	Frequency of Detection	Arithmetic Mean (mg/L)	Geometric Mean (mg/L)	Upper-Bound Value (mg/L)	Standard Deviation (mg/L)
Arsenic	8	ND (0.0025)[a]	0/8 (0%)	NA[b]	NA	NA	NA
Barium	8	ND (0.1)	0/8 (0%)	NA	NA	NA	NA
Cadmium	8	0.023-0.16	8/8 (100%)	0.08	0.06	0.12	0.06
Chromium	8	ND (0.005)	0/8 (0%)	NA	NA	NA	NA
Cobalt	8	ND (0.025)	0/8 (0%)	NA	NA	NA	NA
Copper	8	ND (0.0025)-0.005	1/8 (13%)	0.004	0.003	0.006	0.003
Iron	8	ND (0.05)-0.6	5/8 (63%)	0.25	0.15	0.38	0.20
Lead	8	ND (0.0015)-0.022	6/8 (75%)	0.008	0.006	0.013	0.008
Manganese	8	0.03-0.23	8/8 (100%)	0.09	0.07	0.14	0.07
Mercury	8	ND (0.0001)-0.0004	1/8 (13%)	0.0001	0.0001	0.0002	0.0001
Nickel	8	ND (0.02)-0.05	5/8 (63%)	0.03	0.03	0.05	0.02
Selenium	8	ND (0.0025)	0/8 (0%)	NA	NA	NA	NA
Silver	8	ND (0.00005)	0/8 (0%)	NA	NA	NA	NA
Vanadium	8	ND (0.025)	0/8 (0%)	NA	NA	NA	NA
Zinc	8	3.2-20	8/8 (100%)	9.9	7.8	14.5	6.8

[a] ND = not detected; value(s) in parentheses represents one-half the reporting limit.

[b] Since all measured values were nondetectable, mean, upper-bound, and standard deviation values were not calculated.

Table 3-7. Concentration of (Total Recoverable) Metals in Subsite Ponds at the Baxter Springs/Treece Subsites

Metal	n[a]	Range (mg/L)	Frequency of Detection	Arithmetic Mean (mg/L)	Geometric Mean (mg/L)	Upper-Bound Value (mg/L)	Standard Deviation (mg/L)
Arsenic	9	ND (0.0025)[b]	0/9 (0%)	NA[c]	NA	NA	NA
Barium	9	ND (0.1)	0/9 (0%)	NA	NA	NA	NA
Cadmium	9	ND (0.00025)-0.032	7/9 (78%)	0.01	0.004	0.02	0.01
Chromium	9	ND (0.005)	0/9 (0%)	NA	NA	NA	NA
Cobalt	9	ND (0.025)	0/9 (0%)	NA	NA	NA	NA
Copper	9	ND (0.0025)-0.007	1/9 (11%)	0.005	0.005	0.006	0.001
Iron	9	ND (0.05)-1.7	2/9 (22%)	0.3	0.14	0.6	0.5
Lead	9	ND (0.0005)-0.10	4/9 (44%)	0.02	0.007	0.04	0.03
Manganese	9	0.04-0.88	9/9 (100%)	0.3	0.1	0.4	0.3
Mercury	9	ND (0.0001)-0.0005	2/9 (22%)	0.0003	0.0003	0.0004	0.0001
Nickel	9	ND (0.02)-0.03	2/9 (22%)	0.04	0.04	0.04	0.007
Selenium	9	ND (0.0025)	0/9 (0%)	NA	NA	NA	NA
Silver	9	ND (0.00025)-0.0046	6/9 (67%)	0.002	0.001	0.003	0.001
Vanadium	9	ND (0.025)	0/9 (0%)	NA	NA	NA	NA
Zinc	9	0.054-9.7	9/9 (100%)	1.8	0.5	3.7	3.1

[a] Ballard Pond data were not included (see text).

[b] ND = not detected; value(s) in parentheses represents one-half the reporting limit.

[c] Since all measured values were nondetectable, mean, upper-bound, and standard deviation values were not calculated.

For those metals detected in stream water, mean concentrations in downstream on-site sampling locations (i.e., stations TC-2/TC-3; SB-1/SB-2; and WC-2/WC-3 shown on Figures 2-1, 2-2, and 2-3) were compared to metal levels measured at the two upstream locations (i.e., TC-1 and WC-1). Since there was no suitable upstream sampling station for Spring Branch, as the entire drainage is located within the subsites, the mean concentration of metals in the Willow Creek upstream samples (i.e., WC-1 samples) was used for comparison. The WC-1 data set yielded slightly lower, more conservative numbers although t-test results indicated that there was no significant difference between the concentration of metals at upstream stations WC-1 and TC-1. Table 3-8 shows that mean concentration of metals in the downstream segment of Willow Creek (i.e., WC-2/WC-3) was comparable (within a factor of three) to the mean concentration of metals upstream (i.e., at WC-1) with the exception of cadmium and zinc. Mean cadmium and zinc levels measured at WC-2/WC-3 were four times higher than mean cadmium and zinc levels at WC-1. Mean cadmium and zinc levels measured in the downstream portion of Tar Creek (i.e., at stations TC-2/TC-3) were 20 and 50 times higher, respectively, than mean levels measured at TC-1, while mean cadmium and zinc levels measured in the downstream segment of Spring Branch (i.e., at stations SB-1/SB-2) were 200 and 66 times higher than mean upstream concentrations (i.e., mean of three WC-1 samples).

3.3 AIR

Both source-term air monitoring and emission modeling were used to assess concentrations of metals in air. Source-term monitoring was conducted over a seven-day period at potential fugitive dust source locations in both subsites. Analytical results (particulates, lead and other metals) are presented in Section 4.7 of the Final Draft RI. Concentrations of particulate matter less than 10 microns (PM_{10}) in size were below the 24-hour-standard of 150 $\mu g/m^3$ at all stations during the seven-day sampling period. The concentration of lead in the PM_{10} fraction was well below the existing quarterly standard of 1.5 $\mu g/m^3$. Arsenic, cadmium, and silica (as silicon) were not detected in any samples. The

Table 3-8. Comparison of Metals Levels Measured (Total Recoverable) at Downstream Versus Upstream Sampling Locations for Tar Creek, Willow Creek, and Spring Branch

Metal Detected in On-Site Streams	Mean Concentration at WC-1 (mg/L)	Mean Concentration at WC-2/WC-3 (mg/L)	Mean Concentration at TC-1 (mg/L)	Mean Concentration at TC-2/TC-3 (mg/L)	Mean Concentration at SB-1/SB-2 (mg/L)
Cadmium	0.0004	0.0015	0.001	0.02	0.08
Copper	0.004	0.005	0.01	0.005	0.004
Iron	2.7	3.9	10.8	1.3	0.25
Lead	0.01	0.014	0.03	0.05	0.008
Manganese	0.22	0.21	0.3	0.6	0.09
Nickel	0.02	0.02	0.02	0.03	0.03
Silver	0.0002	0.0002	0.0002	0.0004	0.0002
Zinc	0.15	0.63	0.18	9.2	9.9

concentration of copper in PM_{10} samples ranged from undetected to 0.6 $\mu g/m^3$, while concentrations of manganese, nickel, and zinc were at or below 0.3 $\mu g/m^3$.

For the emissions modeling, the highest concentration of metals in the fine particulate fraction (i.e., < 400 mesh or 38 microns) in chat and flotation tailings combined with the results of air dispersion modeling conservatively estimates the concentration of metals in ambient air as described in detail in Appendix B of the Human Health Risk Assessment. Since the modeling results were higher than the source-term air monitoring results, the modeling results were used to be conservative. Using the emissions estimates and our dispersion modeling, the resultant metal concentrations in ambient air (over the Treece subsite) were used in the ERA (Table 3-9).

Air monitoring was conducted at both subsites. Treece Subsite data were higher due to the greater preponderance of mill waste on that portion of the subsites. Source-term air samples were analyzed for arsenic, cadmium, cobalt, lead, manganese, nickel, and zinc. The results of the 1992 air monitoring program indicated that source-term PM_{10} and metals concentrations were well below both National Ambient Air Quality Standards and modeled concentrations. As a result, modeling data were used versus the monitoring data because the modeling data were felt to be more representative of site-wide ambient air conditions.

3.4 GROUNDWATER

Two major aquifer systems, referred to as the shallow and deep aquifers, underlie the subsites. The shallow aquifer is comprised of Mississippian limestones, which hosts the lead-zinc deposits that were mined at the subsites. Based on a residential water supply inventory, discussions with local water conservancy district personnel and field observations, the shallow aquifer within the subsites is not used for any beneficial purposes, including domestic, stock, or irrigation. Mine discharge to the surface within the subsites is limited to intermittent flows

Table 3-9
Estimated Concentration of Metals in Air Over the Treece Subsite[a]

Metal	Estimated Concentration in Air Over the Baxter Springs/Treece Subsites (ng/m^3)[b]
Arsenic	0.35
Cadmium	5.2
Copper	13
Lead	76
Manganese	10
Mercury	0.03
Zinc	910

[a] Concentrations were estimated using state-of-the-art, EPA-approved modeling practices as described in the Air Analysis Appendix B of the Human Health Risk Assessment (Dames & Moore, 1993b). The highest concentration of each metal detected in the fine particle fraction of either chat or flotation tailings was used to model air concentrations.

[b] Nanograms per cubic meter (10^{-3} micrograms).

from only one mine, the Bruger mine, northwest of Baxter Springs, to Willow Creek. The Bruger shafts did not discharge during the 1991 RI field activities, and as such neither flow measurements or water chemistry data are available. KDHE sampling results (1987) indicate relatively high metal concentrations in the discharge; cadmium 0.002 mg/L, lead 0.062 mg/L, and zinc 21.4 mg/L. A discussion of potential impacts on Willow Creek aquatic organisms resulting from the shaft discharge is presented in Section 8.0. Shallow aquifer contributions to the lower reaches of Willow Creek and Spring Branch (as seepage from limestones) is accounted for in subsequent analyses by incorporating data from downstream stations WC-3 and SB-2.

The deep aquifer occurs in Lower Ordovician sandy dolomite and represents the principal source of water for public, industrial, domestic, and agricultural supplies. Deep aquifer groundwater data were not used in the ERA, since very little groundwater is available for ingestion by terrestrial receptors, and sufficient quantities of surface water are readily available.

4.0 IDENTIFYING POTENTIAL CHEMICALS OF CONCERN

One of the first steps in the risk assessment process is to identify Chemicals of Concern (COCs). The COCs for the subsites were expected to be heavy metals associated with mine wastes and with base metal ore deposits characteristic of the Tri-State Mining District. A phased screening process was used to identify COCs in on-site media that may pose adverse impacts to environmental receptors (i.e., COCs for the human health and ecological risk assessments may be different). COCs were determined for the following media:

- Baxter Springs and Treece subsite ponds;
- Baxter Springs and Treece subsite streams;
- Agricultural plus A-Horizon (Ag/A) soils;
- Near-Pile (0 to 12 in.) soils; and
- Air.

It was not necessary to determine COCs for mill waste and sediment, since these two media were not required to estimate dose (and therefore potential adverse impacts) to terrestrial and aquatic receptors. Potential adverse effects to aquatic biota were evaluated by comparing the measured concentration of COCs in on-site waters to chronic toxicity data. Fish were the primary aquatic receptor, as they represent an integrator of possible impacts. If invertebrates and/or periphyton, which serve as a prey base for many fish species, have been adversely affected, it is reasonable to assume that fish would also be affected. In other words, a suitable ("normal") benthic population must exist to support an associated fish population. This ERA did indirectly assess impacts from sediments by collecting total recoverable and dissolved water samples and evaluating potential impacts to benthic invertebrates. Total recoverable concentrations reflect the concentration of COCs in suspended sediments that filter feeding benthic organisms are likely to ingest. Total recoverable concentrations were used in the toxicity assessment to evaluate potential adverse effects to aquatic biota. Similarly, COCs for mill/mine waste were not derived since these soils generally lack vegetation suitable for cover and foraging. Therefore, terrestrial receptors were assumed to be exposed to near-pile soils (or soils contaminated at comparable levels).

4.1 PHASED SCREENING PROCESS

Selection of COCs involved two phases. The first phase included the screening of metals after detection limits, detection frequencies, and blank samples were examined, while the second phase involved comparing estimated metals concentrations in various environmental media (e.g., air, soil, and water) to media-specific baseline data. Baseline data from on-site locations were used in these comparisons when available. If local baseline data were not available, any chemical detected in that medium was considered a COC. As a result of this process, metals likely to be site-related and detected at statistically significant concentrations were selected as COCs. In this sense, biota sensitivity to individual contaminants was not specifically considered in the selection of COCs. Biota were assumed to be exposed to all metals whose measured levels onsite were greater than those measured in the appropriate corresponding background sample. Table 4-1 presents the metals analyzed for in various media at the subsites.

Detection limits vary depending on the metal analyzed, the analytical instrument used, and the characteristics of the media being tested. Before any metals were eliminated, it was necessary to consider that detection limits can vary between sampling events. Therefore, some metals concentrations may not have been detected in samples from some sampling events due to varying detection limits. Metals not detected in any of the media samples were eliminated from further consideration.

Table 4-1. Metals Analyzed For in the Various Media Samples Collected at the Baxter Springs/Treece Subsites

Metal	Streams	Ponds	Air (based on monitoring)	Ag/A and Near-Pile Soils and Chat/Mill Waste
Arsenic	X	X	X	X
Barium	X	X		X
Beryllium				X
Cadmium	X	X	X	X
Chromium	X	X		X
Cobalt	X	X	X	X
Copper	X	X		X
Iron	X	X		X
Lead	X	X	X	X
Manganese	X	X	X	X
Mercury	X	X		X
Nickel	X	X	X	X
Selenium	X	X		X
Silver	X	X		X
Thallium				X
Vanadium	X	X		X
Zinc	X	X		X

Concentrations detected in blanks were then compared with concentrations detected in site samples. Sample results were considered positive only if the concentration in the site sample was five times the maximum concentration detected in any blank sample. No metals were eliminated from the list of COCs during this step, however, since all blank samples were at concentrations below the detection limits. Hence, blank sample data were not used to screen COCs.

Metals not eliminated during the first phase of the screening process were compared to local baseline concentrations. If the mean concentration of a contaminant in a given medium was statistically higher (at the 95 percent level of confidence) than its mean level in baseline samples as determined using a one-tailed t-test, that compound was considered a COC. If no baseline data for a given medium were available, it was conservatively assumed that any metal detected in that medium was a COC.

The one-tailed (versus a two-tailed) t-test was used in this evaluation because it is a more powerful tool for detecting statistically significant increases above a given mean baseline concentrations. The one- and two-tailed t-tests are virtually equivalent with the exception that the alternative hypothesis is defined differently in each case. In a one-tailed test, the alternative hypothesis is that metals concentrations are higher in the non-baseline data set while in a two-tailed t-test, the alternative hypothesis is that metals concentrations are either higher or lower. In a two-tailed test the significance level (i.e., the probability of a false positive decision) is usually divided equally between the two alternative hypotheses. In this case, the one- and two-tailed tests are computationally equivalent. However, a two-tailed test with a 5 percent significance level is equivalent to a one-tailed test with a only a 2.5 percent significance level. In general, for a given statistical test and fixed sample size, the power of the test (i.e., the ability to detect a real difference) is inversely related to the significance level selected. Therefore, the fixed 5 percent significance level offered by the one-tailed test makes it a more powerful tool.

4.2 COCs FOR SOIL

Ag/A and near-pile soil samples were analyzed for arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Table 4-2 shows that for Ag/A, mean sample concentrations were significantly greater than mean baseline (B-Horizon) concentrations for copper, lead, and manganese only. Baseline soil samples were taken from the B-Horizon layer, which is 14 to 24 inches below ground surface. Metals levels in samples taken from this depth are naturally occurring and have not been affected by historical mining activities. Eight baseline soil samples were collected onsite: three from the Treece subsite and five from the Baxter Springs subsite. Barium, beryllium, mercury, silver, and thallium were eliminated due to non-detection.

For near-pile soils (Table 4-3), mercury, silver, and thallium were not detected in any samples. Mean near-pile sample concentrations were significantly greater than mean baseline (B-Horizon) soil levels for cadmium, cobalt, lead, manganese, and zinc. Thus, these compounds are considered COCs for soil. The average concentration, standard deviation, t-values, assumptions used in the one-tailed t-test, and results of this t-test are shown in Table 4-3 for near-pile soils.

4.3 COCs IN SURFACE WATER

COCs for surface water were determined by considering each stream (Willow Creek, Tar Creek, Spring Branch) individually but all ponds were treated as a group. As a result, the three streams and the on-site ponds could have different COCs. Only those results of stream-water quality that corresponded with aquatic sampling locations were used. All aquatic biota mainstem sampling stream locations correspond to surface water quality sampling locations with the exception of aquatic biota sampling location WC-1a, which was situated between surface water quality sampling locations WC-1 and WC-2. All pond data (excluding Ballard chat-wash pond) were used.

Table 4-2. The Mean Concentration, Standard Deviation, Calculated and Table t Statistics, and Results of the t-Test for Agricultural/A-Horizon (Ag/A) Soils

Chemical	Mean B-Horizon Soils[a]	STD[b] B-Horizon Soils	Mean A-Horizon + AG[c] Soils	STD A-Horizon + AG Soils	Calculated t Value	Table t Value	Conclusion
Arsenic	7.1	5.9	5.6	2.6	-0.85	1.72	Accept[d]
Cadmium	0.4	0.1	0.9	1.2	1.16	1.72	Accept
Chromium	30.0	10.2	31.3	17	0.19	1.76	Accept
Cobalt	7.0	2.5	10.8	7.2	1.41	1.76	Accept
Copper	11.0	3.9	7.1	3.4	2.48	1.72	Reject[e]
Iron	34,000	20,368	29,587	21,841	-0.47	1.72	Accept
Lead	17.0	6.5	42.0	35.9	1.93	1.72	Reject
Manganese	134.0	113.8	593.0	396.6	3.17	1.72	Reject
Nickel	11.0	4.4	10.8	6.7	-0.07	1.76	Accept
Selenium	0.5	0.2	0.4	.01	-1.13	1.76	Accept
Vanadium	51.0	15.3	48.9	22.0	-0.22	1.76	Accept
Zinc	44.0	52.7	156.5	198.6	1.56	1.76	Accept

[a] B-Horizon soils were used to represent local baseline conditions.

[b] STD = standard deviation.

[c] AG = agricultural soils

[d] Accept denotes acceptance of the null hypothesis that the sample population mean is not greater than the B-horizon population mean.

[e] Reject denotes rejection of the null hypothesis that the mean sample population is greater than the B-Horizon population mean; hence that metal is a COC.

Data are in mg/kg dry weight.

water onsite (CH2M Hill, 1986). Fish that would likely inhabit deeper water of the subsites, include smallmouth buffalo (*Ictiobus bubalus*), longear sunfish (*Lepomis megalotis*), redear sunfish (*Lepomis microlophus)*, green sunfish (*Lepomis cyanellus*), common carp (*Cyprinus carpio*) and other species typical of midwestern warmwater streams. Of these species, longear sunfish, redear sunfish, and green sunfish were collected by Dames & Moore personnel in on-site streams and ponds during the RI. In addition to these species, bluegill, spotted sucker, brook silverside, largemouth bass, black bullhead, pumpkinseed, white crappie, dace, and shiners were collected during the RI. Since many of the streams and ponds onsite are stocked, the presence of fish not indigenous to this area is possible.

The Baxter Springs/Treece subsite RI aquatic sampling program is the only known investigation of fish species occurrence within those segments of Tar Creek, Spring Branch, and Willow Creek that exist on site. However, the KDWP (1992a, personal communications with Mr. Larry Zuckerman, KDWP) indicated that only certain fish species would be expected to occur in ephemeral streams of southeastern Kansas. These species include yellow bullhead (*Ictalurus natalis*), black bullhead, green sunfish, blackstripe topminnow (*Fundulus notatus*), bullhead minnow (*Pimephales vigilax*), bluntnosed minnow (*Pimephales notatus*), red shiner (*Notropis lutrensis*), and mosquitofish (*Gambusia affinis*).

Although the Spring and Neosho Rivers are not technically within the boundaries of the subsites, potential impacts to aquatic organisms in these rivers are evaluated in this assessment, since they constitute the perennial receiving channels for subsite waters. The assessment of water quality within the rivers is based on existing data collected by KDHE (1980, 1987) and discharge data from the USGS (1987).

Both the Spring and Neosho Rivers are inhabited by diverse populations of fish and benthic invertebrates. Ferrington *et al.*, (1988) reported benthic invertebrates representing Ephemeroptera (mayflies), Odonata (dragonflies/damselflies), Megaloptera (alderflies), and Diptera (flies, mosquitoes, and midges) occurred within Empire Lake. Within the Spring River arm of Grand Lake O' the Cherokees, Aggus *et al.*, (1983) collected spotted gar, gizzard shad, carp, river carpsucker, smallmouth buffalo, channel catfish, flathead catfish, white bass, white crappie, and freshwater drum. The same species, excluding the white crappie and spotted gar were collected by Aggus *et al.*, (1983) from the Neosho River downstream from Tar Creek. In addition, longnose gar, shortnose gar, bigmouth buffalo, green sunfish, warmouth, orangespotted sunfish, bluegill, and longear sunfish were collected downstream from Tar Creek. Benthic invertebrates collected by Aggus *et al.* (1983) from the Neosho River and Spring River included oligochaetes and tubificids (aquatic earthworms), ephemeropterans (mayflies), Chironomids (midges) and Culicids (mosquitoes).

The Aggus *et al.*, (1983) study site of Tar Creek, within 3 miles of its confluence with the Neosho River, indicated that a variety of fish and benthic invertebrate species inhabit the stream. Fish species collected in Tar Creek included chubs, river carpsucker, channel catfish, killifish, *Gambusia sp.* (live bearers), brook silverside, white bass, green sunfish, bluegill, largemouth bass, and white crappie. Benthic invertebrates included mosquitoes, true flies, beetles, and true bugs.

The Kansas Department of Health and Environment (KDHE, 1992) collected benthic macroinvertebrates from Willow Creek northwest of Baxter Springs on May 18, 1979. The major groups of benthic macroinvertebrates represented in the samples include Coleopterans (water beetles), Dipterans (flies, mosquitoes, and midges), Ephemeropterans (mayflies), Hemipterans ("true bugs"), Odonates (dragonflies and damselflies), and Plecopterans (stoneflies).

2.6 SENSITIVE HABITATS AND SPECIES

Sensitive habitats that may occur within the subsites include habitats critical for the survival of threatened or endangered (T&E) species and wetlands, both of which are protected by various legislative acts and executive orders. Potential impacts were evaluated for T&E species that do or could occur within the subsites. T&E species lists and critical habitat descriptions were provided by the Kansas Department of Wildlife and Parks (KDWP, 1991).

2.6.1 T&E Animal Species

No federally-listed T&E species are known to occur or have habitat within the areas impacted by mines (i.e., west of State Highway 66) within the Baxter Springs/Treece subsites.

The listing of T&E species for Kansas describes nine species which have critical habitat within the subsite (Table 2-1). The Kansas Non-Game and Endangered Species Conservation Act (K.S.A. 32:501-502) affords protection of habitat of species that are uncommon or not widely distributed in the state. Frequently, the species are common elsewhere but are uncommon in the state because of marginal habitat.

Critical habitat has been defined by Kansas Administrative Regulation as those areas documented as currently supporting self-sustaining population(s) of any threatened or endangered species of wildlife as well as those areas determined by Kansas Department of Wildlife and Parks to be essential for the conservation of any threatened or endangered species of wildlife. If a proposed project has potential to affect a listed species or its habitat, the project proponent must contact the Environmental Services Section, Kansas Department of Wildlife and Parks, Pratt, Kansas.

Of the nine listed species that have critical habitat within the subsites, only the northern spring peeper (*Hyla crucifer crucifer*) has critical habitat west of State Highway 66 in the general area where mine and mill deposits have been identified (Figure 2-4). The northern spring peeper is found in woodlands near small ponds, swamps, and the twilight zone of caves, east of State Highway 69 running north through Treece. This small frog prefers woodland ponds, with abundant emergent vegetation, marshes with standing trees, or shrubs during the March breeding season. Drainage and clearing of woodland wetlands has drastically reduced the peeper's preferred habitat.

Table 4-3. The Mean Concentration, Standard Deviation, Calculated and Table t Statistics, and Results of the One-Tailed t-Test for Near-Pile Soils (0 to 12 in.)

Chemical	Mean B-Horizon Soils[a]	STD[b] B-Horizon Soils	Mean Near-Pile Soils	STD Near-Pile Soils	Calculated t Value	Table t Value	Conclusion
Arsenic	7.1	5.9	7.8	3.5	0.4	1.71	Accept[d]
Barium	125.0	100.5	107.0	43.6	0.659	1.71	Accept
Beryllium	1.0	0.47	0.8	1.62	1.333	1.71	Accept
Cadmium	0.4	0.14	4.5	5.2	2.20	1.71	Reject[e]
Chromium	30.0	10.2	34.2	18.9	0.59	1.71	Accept
Cobalt	7.0	2.5	15.6	10.6	2.24	1.71	Reject
Copper	11.0	3.9	9.9	4.7	-0.58	1.71	Accept
Iron	34,000	20,368	48,947	35,218	1.12	1.71	Accept
Lead	17.0	6.5	88.1	64.8	3.06	1.71	Reject
Manganese	134.0	113.8	947.4	852.4	2.66	1.71	Reject
Nickel	11.0	4.4	14.5	12.2	0.78	1.71	Accept
Selenium	0.5	0.23	0.6	0.2	1.14	1.71	Accept
Vanadium	51.0	15.3	53.2	23.2	0.24	1.71	Accept
Zinc	44.0	52.7	710.0	717.9	2.59	1.71	Reject

[a] B-Horizon soils were used to represent local baseline conditions.

[b] STD = standard deviation.

[c] AG = agricultural soils

[d] Accept denotes acceptance of the null hypothesis that the mean baseline concentration is not greater than the B horizon mean concentration.

[e] Reject denotes rejection of the null hypothesis that the mean baseline concentration is greater than the B Horizon mean; hence, that metal is a COC.

Data are in mg/kg dry weight.

Surface-water samples collected from on-site streams and ponds were analyzed for arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, silver, vanadium, and zinc. Arsenic, barium, chromium, cobalt, selenium, and vanadium were not detected in any stream or pond samples. Manganese and nickel were eliminated from the lists of COCs for Tar Creek due to nondetection, while mercury was eliminated from the list of COCs for Willow Creek and Spring Branch due to nondetection.

As for soil contaminants, metals detected in on-site streams were included in the list of COCs for the streams only if the mean concentration of metals in baseline streamwater was significantly less than the mean concentration of metals in on-site streams as determined from a one-tailed t-test even if the measured concentration of that metal exceeded national Ambient Water Quality Criteria. The average concentration of metals in upstream samples TC-1 and WC-1 were used to represent local baseline levels for Tar Creek and Willow Creek, respectively. Since there was no suitable baseline data for Spring Branch, as the entire drainage is located within the site, the mean concentration of metals in the Willow Creek upstream samples (WC-1, n=3) was used to represent local baseline levels for Spring Branch. Upstream surface water sample results for Tar Creek were statistically similar but slightly higher. The use of Willow Creek upstream data is entirely appropriate in the formulation of a baseline data set for Spring Branch since watersheds are located in the same general area.

Data from on-site downstream sampling locations (i.e., TC-2 and TC-3 for Tar Creek, WC-2 and WC-3 for Willow Creek, and SB-1 and SB-2 for Spring Branch) were combined to represent site conditions. Thus, mean metal concentrations in TC-1 and WC-1 were compared to levels in TC-2/TC-3 and WC-2/WC-3, respectively, while mean concentrations in WC-1 were compared to mean levels in SB-1/SB-2. Sampling locations were combined for several reasons, primarily to increase sample size as 1) the ephemeral systems of the subsites are dynamic and susceptible to large fluctuations in flow rates and water chemistry, and 2) to increase the validity of the statistical evaluations required in the toxicity assessment by using as large a data base as possible. For instance, the downstream TC-3 station on Tar

Creek at the state line was sampled three times. To increase sample size and make the data more representative of Tar Creek main-stem on the whole, three sample results from TC-2 were added to form the Tar Creek main-stem data set (n=6); this data set was then compared to the upstream (baseline) results (n=3) at TC-1 to assess contributions from mill waste. The combination of sample results for TC-3 and TC-2 is valid because the water chemistry at these two downstream locations on Tar Creek is essentially the same (mean hardness 269 vs. 366 mg $CaCO_3$/L, mean cadmium 0.012 vs. 0.02 mg/L, and average zinc 8.8 vs 9.6 mg/L). This combined data set is more representative of the dynamic system that exists at the site. Since water chemistry is highly dependent on flow rate and since many of the biologic organisms of concern are mobile, it was appropriate to average the sampling data. Concentration estimates for the individual sampling locations as well as potential toxicity to aquatic receptors was evaluated for individual locations using the smaller data sets. These data are included in the Uncertainty Section (Section 9.0). Toxicity potential was evaluated by using the average hardness and average metal concentrations for each sample site (instead of combining the downstream data sets).

The average concentration, standard deviation, t-values, assumptions used in the one-tailed t-test, and results of the t-test are shown in Tables 4-4, 4-5, and 4-6 for Willow Creek, Tar Creek, and Spring Branch, respectively. These data show that for Willow Creek, mean on-site concentrations were significantly greater than mean baseline concentrations for cadmium, lead, and zinc. Thus, these metals were considered COCs for Willow Creek. Levels of manganese and zinc measured onsite in Tar Creek downstream water (i.e., TC-2 and TC-3) were significantly higher than concentrations measured at the baseline station, TC-1. Finally, concentrations of cadmium and zinc detected at sampling station SB-1 and SB-2 were significantly higher than concentration measured at the WC-1 baseline sampling station. The COCs for each stream are summarized in Table 4-7.

Table 4-4. The Mean Concentration, Standard Deviation, Calculated and Table t Statistics, and Results of the One-Tailed t-Test for Willow Creek

Metal	Mean Upstream[a] Value (mg/L)	STD[b] Upstream Value (mg/L)	n	Mean Site Value[c] (mg/L)	STD Site Value (mg/L)	n	Calculated t value	Table t Value	Conclusion
Cadmium	0.0005	0.0001	3	0.0015	0.0006	8	2.78	1.83	Reject[d]
Copper	0.0037	0.002	3	0.005	0.004	8	0.62	1.83	Accept[e]
Iron	2.73	1.72	3	3.94	3.57	8	0.55	1.83	Accept
Lead	0.0097	0.006	3	0.014	0.085	8	1.87	1.83	Reject
Manganese	0.22	0.021	3	0.211	0.07	8	-0.27	1.83	Accept
Nickel	0.02	0.0	3	0.02	0.0035	8	0.62	1.83	Accept
Silver	0.0002	0.0001	3	0.0002	0.0002	8	0.0	1.83	Accept
Zinc	0.15	0.03	3	0.63	0.27	8	3.01	1.83	Reject

[a] Upstream station is WC-1.

[b] STD = standard deviation

[c] Mean of levels measured at the downstream stations WC-2 and WC-3.

[d] Reject denotes rejection of the null hypothesis that the sample population mean is not greater than the upstream population mean; hence that metal is a COC.

[e] Accept denotes acceptance of the null hypothesis that sample population mean is not greater than the upstream population mean.

Table 4-5. The Mean Concentration, Standard Deviation, Calculated and Table t Statistics, and Results of the One-Tailed t-Test for Tar Creek

Metal	Mean Upstream[a] Value (mg/L)	STD[b] Upstream Value (mg/L)	n	Mean Site Value[c] (mg/L)	STD Site Value (mg/L)	n	Calculated t value	Table t Value	Conclusion
Cadmium	0.0013	0.0014	3	0.016	0.016	6	1.67	1.89	Accept[d]
Copper	0.0097	0.012	3	0.005	0.004	6	-0.936	1.89	Accept
Iron	10.83	15.77	3	1.33	0.88	6	-1.59	1.89	Accept
Lead	0.03	0.051	3	0.053	0.025	6	0.897	1.89	Accept
Manganese	0.31	0.41	3	0.62	0.39	6	1.91	1.89	Reject[e]
Nickel	0.02	0.0	3	0.028	0.013	6	1.04	1.89	Accept
Silver	0.0002	0.0002	3	0.0004	0.0008	6	0.413	1.89	Accept
Zinc	0.18	0.17	3	9.22	9.69	6	2.56	1.89	Reject

[a] Upstream station is TC-1.

[b] STD = standard deviation

[c] Mean of levels measured at the downstream stations TC-2 and TC-3.

[d] Accept denotes acceptance of the null hypothesis that the sample population mean is not greater than the upstream population mean.

[e] Reject denotes rejection of the null hypothesis that the sample population mean is not greater than the upstream population mean; hence that metal is a COC.

Table 4-6. The Mean Concentration, Standard Deviation, Calculated and Table t Statistics, and Results of the One-Tailed t-Test for Spring Branch

Metal	Mean Upstream[a] Value (mg/L)	STD[b] Upstream Value (mg/L)	n	Mean Site Value[c] (mg/L)	STD Site Value (mg/L)	n	Calculated t value	Table t Value	Conclusion
Cadmium	0.0005	0.0001	3	0.078	0.06	8	2.22	1.83	Reject[d]
Copper	0.0037	0.002	3	0.004	0.003	8	0.06	1.83	Accept[e]
Iron	2.73	1.72	3	0.29	0.19	8	-4.38	1.83	Accept
Lead	0.01	0.006	3	0.008	0.008	8	-0.33	1.83	Accept
Manganese	0.22	0.021	3	0.09	0.07	8	-3.15	1.83	Accept
Nickel	0.02	0.0	3	0.03	0.015	8	1.22	1.83	Accept
Silver	0.0002	0.0001	3	0.002	0.005	8	0.6	1.83	Accept
Zinc	0.15	0.03	3	9.9	6.8	8	2.4	1.83	Reject

[a] Upstream station is WC-1. There was no significant difference in t-test results if concentrations of metals in TC-1 samples were used.

[b] STD = standard deviation

[c] Mean of levels measured at downstream stations SB-1 and SB-2.

[d] Reject denotes rejection of the null hypothesis that the sample population mean is not greater than the upstream population mean; hence that metal is a COC.

[e] Accept denotes acceptance of the null hypothesis that the sample population mean is not greater than the upstream population mean.

Table 4-7. Summary of Chemicals of Concern (COCs) in Surface Water, Soil, and Air

	Agricultural/ A-Horizon Soils	Near-Pile Soils	Air (based on modeling)	Tar Creek	Willow Creek	Spring Branch	Ponds
METAL							
Cadmium		X	X		X	X	X
Cobalt		X					
Copper	X		X				X
Iron							X
Lead	X	X	X		X		X
Manganese	X	X		X			X
Mercury			X				X
Nickel							X
Silver							X
Zinc		X	X	X	X	X	X

Since baseline data for the ponds were not available, any metal detected was conservatively considered a COC. These metals include: cadmium, copper, iron, lead, manganese, mercury, nickel, silver, and zinc. The Ballard chat-wash pond was not included in this evaluation.

4.4 COCs IN AIR

Two sets of air data were generated for this analysis. The concentration of mine-related metals in air was estimated using EPA-approved modeling practices as described in Air Analysis (Appendix B) of the HHRA. The highest concentration of each metal detected in the fine particle fraction of either chat or flotation tailings was used to model air concentrations. Based on modeling results, COCs for air were cadmium, copper, lead, mercury, and zinc.

A second data set for air was generated from the air monitoring program conducted during May, 1992. Source-term samples were analyzed for arsenic, cadmium, cobalt, lead, manganese, nickel, respirable silica as silicon, and zinc. Modeling data were used in the ERA instead of the monitoring data because the modeling data better represented ambient air conditions and modeled values were higher than measured data.

5.0 EXPOSURE ASSESSMENT

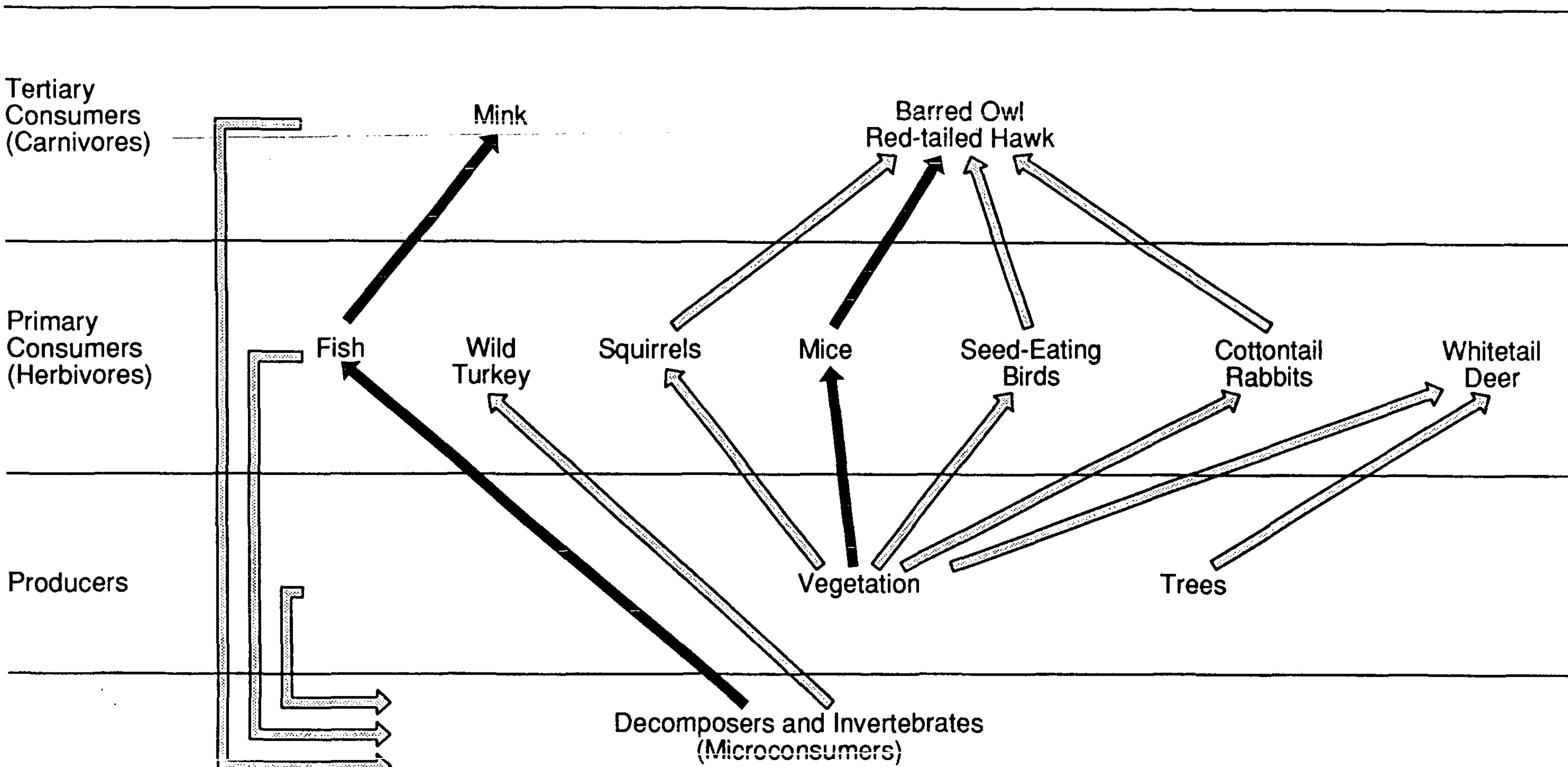
Determining if exposure to site-related contaminants may increase the incidence of adverse effects in exposed populations is an important step in the risk assessment process. The objective of this task is to estimate the magnitude, frequency, duration, and route of exposure to site-related chemicals by ecological receptors. Accomplishing this task involves completing the following steps:

- Selecting of key ecological receptors;
- Identifying potential pathways of exposure;
- Estimating exposure-point concentrations; and
- Estimating total contaminant intake by receptors.

5.1 SELECTING KEY ECOLOGICAL RECEPTORS

Environmental receptors are those organisms that may have been, are being, or may be exposed to metals contamination from the subsites. Environmental receptors are identified by considering the relevant exposure pathways and the potential or known occurrence of species exposed via those pathways. Species occurrence information was compiled from studies previously conducted in the Tri-State Mining District (Ferrington *et al.*, 1988; Jobe, 1988; Aggus *et al.*, 1983; and CH2M Hill, 1986) and from 1991 site-specific investigations conducted by Dames & Moore.

Since energy and matter flow through ecosystems in food webs, key species are those representative of the site food web. A food web describes the transfer of matter and energy from one trophic level or organism to another. Figure 5-1 depicts a simplified food web for the Baxter Springs/Treece subsites. Food webs can be delineated in hierarchies or trophic levels as follows:



LEGEND

- Pathway Used In Risk Assessment

SIMPLIFIED FOOD WEB

- Primary producers - photosynthetic plants;
- Primary consumers - eat plants (herbivore/granivores);
- Secondary consumers - eat herbivores (carnivores);
- Tertiary consumers - eat other carnivores (top carnivores);
- Decomposers - feed on dead or decaying organisms.

The following criteria were used to select the key species evaluated in the ERA:

- Species that are vital to the structure and function of the food web (i.e., principal prey species or species that are fundamental food items for principal prey species);
- Species that exhibit a toxicological sensitivity (vulnerability) to the COCs;
- Species that have unique life histories and/or feeding habits;
- Species that commonly occur within potentially-affected areas;
- Species that are representative of terrestrial, avian, riparian, and aquatic communities;
- Species that inhabit Cherokee County and are known or likely to occur within the subsites;
- Species composed of a mixture of avian, mammalian, and aquatic species designed to address a variety of life histories, feeding habits, and toxicological sensitivities; and
- Species for which toxicological data are readily available in the scientific literature.

Key receptors used in the risk assessment were selected to minimize the possibility that other species could be more exposed than the key species by focusing on higher-tropic level species and food chain effects. The primary focus of the environmental evaluation was on potential toxicity as well as associated impacts and food chain effects on primary and higher trophic level consumers, as they would provide sufficient data to assess the general condition of the ecosystem. The use of these selected organisms was therefore expected to adequately protect the majority of species potentially inhabiting the subsites, since these receptors are top predators.

Using these criteria, the following key species representing several different phyletic groups were identified:

- Barred owl;
- Red-tailed hawk;
- Mink;
- Benthic organisms, principally chironomids (midges) and tubificids (aquatic earthworms); and
- Fish, principally ictalurids (catfish) and centrarchids (sunfish).

Table 5-1 lists the rationale for selecting each species. Although each key species selected may not necessarily meet all of the criteria defined above, key species selected collectively would meet all criteria.

T&E species are not included on the list of key receptors, as they are given separate consideration. Potential impacts to T&E species must be evaluated if these species occur within site boundaries or there is reasonable potential for these species to occur in or use some portion of the site (i.e., critical habitat exists within the subsites or could be affected by the migration of COCs from the subsite). As discussed in Section 2.5.1, one state-listed amphibian has designated critical habitat within mining-impacted areas of the subsites and eight others have designated critical habitat downstream of the mining areas. This ERA focused on assessing individual-level impacts for those T&E species that could occur in mining-impacted areas or those species that could potentially be exposed to metals via contact with surface waters from the mining areas.

Table 5-1. Key Receptors Chosen to Model Potential Adverse Impacts Associated With Site-Specific Contamination and the Reasons for Selecting these Species.

Key Receptor	Rationale for Selection
Fish - Centrarchids (Sunfish) and Ictalurids (Catfish)	1. Secondary consumers that contact water and sediments directly (gills are in direct contact with water). 2. Ingests food items and sediment that have also been in prolonged contact with site-related metals. 3. Serves as integrators of aquatic exposures. 4. Substantial toxicology data available. 5. Substantial ecological and behavioral data are available. 6. Are ubiquitous in on-site surface water bodies.
Barred Owl & Red-Tailed Hawk	1. Secondary consumers that either frequent or reside within subsite boundaries. 2. Rely on small herbivorous mammals (e.g., white-footed mouse) as primary forage. 3. Known to occur onsite. 4. Representative of the terrestrial avian community.
Mink	1. Secondary consumer that is omnivorous and likely to forage on fish inhabiting subsite ponds and streams. 2. Exhibits a toxicological sensitivity to most metals of concern. 3. Substantial scientific data regarding ecology and behavior are available. 4. Known to occur within the subsites.
Benthic invertebrates Tubificids (Aquatic Earthworms) and Chironomids (Midges)	1. Secondary consumer that consumes periphyton and comes in direct contact with sediments that may contain site-related metals. 2. Represents a key food species for fish. 3. Substantial toxicological data are available. 4. Known to occur within Cherokee County and/or subsites. 5. Have gills that come in direct contact with water.

5.2 IDENTIFICATION OF POTENTIAL EXPOSURE PATHWAYS

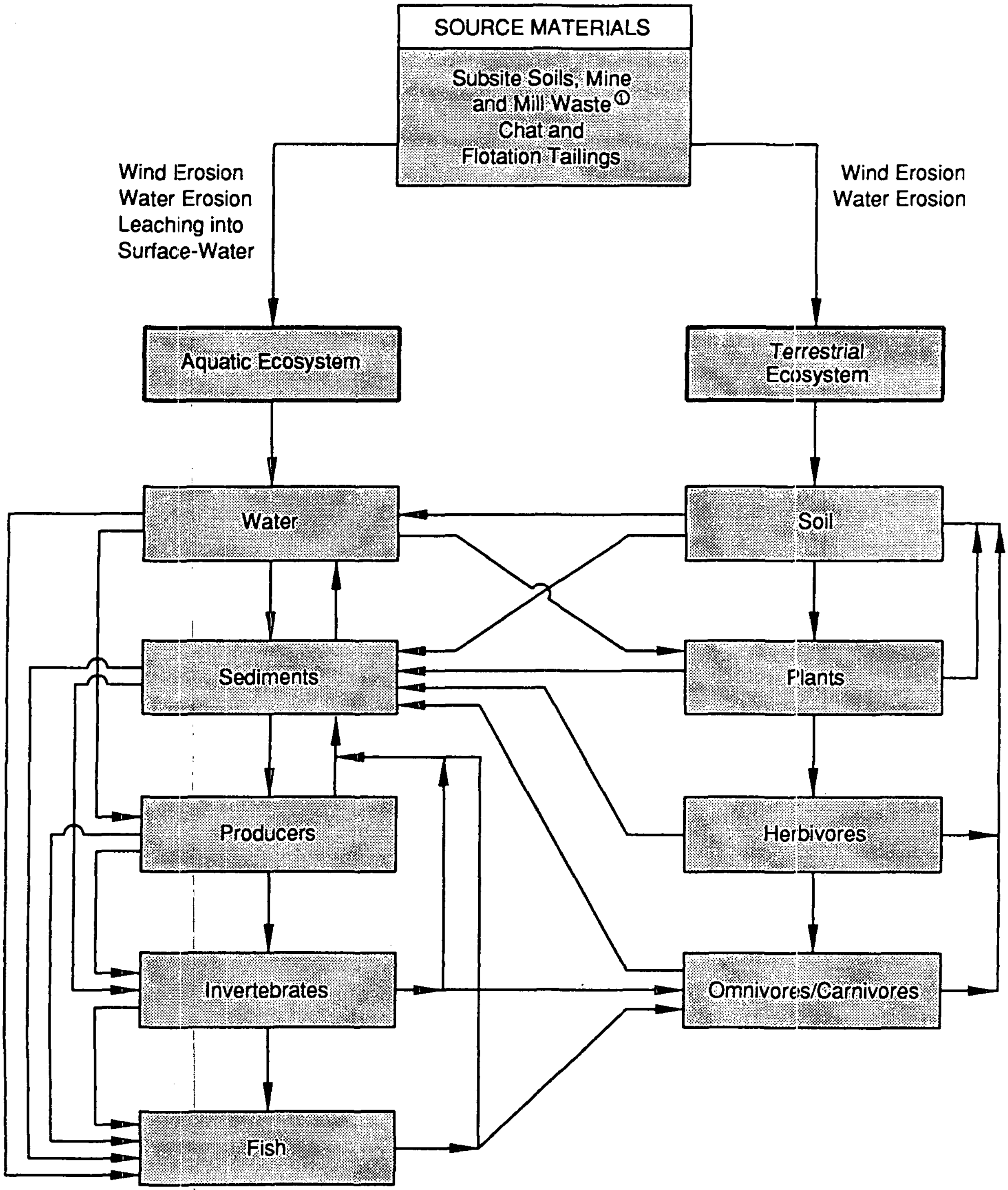
Potential exposure pathways are defined as the route media contaminants follow in order to reach potential receptors. For an exposure pathway to be considered complete, it must have a source, mechanism(s) of contaminant release, a retention and transport medium, a point of potential biota contact, and an exposure route at the contact point. Sources of metals in the subsites, as they pertain to ecologic receptors, include the mined/milled materials deposited on the subsite surface and subsite soils, and sediments. Figure 5-2, a simple site conceptual model for the Baxter Springs/Treece subsites, depicts relevant transport mechanisms.

While metals in subsite source materials may be transported to on- and off-site locations through a variety of mechanisms, they are primarily transported from source materials either in mass [erosion or cultural redistribution (i.e., chat hauling, regrading, etc.)] as windblown dust or waterborne sediments, or leached out as metallic ions and transported by water to soils and surface water. Overall, increased metals content in soils near mill waste piles was attributed to: (1) windblown transport of mill wastes as fugitive dust; (2) distribution of waste material or metallic ions to soils via seepage and/or runoff; or (3) redistribution to near-pile soils by physical means.

Since soils and mill wastes are relatively fine-grained, they represent potential sources of windblown dust. Agricultural fields, chat piles, unprotected tailings deposits, and chat-covered roads are the primary dust sources in the subsites. Neither mine wastes (development and waste rock piles), revegetated near-pile soils, or flooded tailings impoundments are significant sources of dust.

5.2.1 Terrestrial Pathways

Potential routes by which terrestrial receptors could be exposed to mine-related metals include: (1) inhalation of fugitive dust; and (2) ingestion of soil, mine waste, vegetation,



① - SEE TEXT FOR DESCRIPTION OF SOURCE MATERIALS

POTENTIAL PATHWAYS FOR HEAVY METAL CONTAMINATION OF TERRESTRIAL AND AQUATIC ECOSYSTEMS FROM MINE WASTES

DAMES & MOORE

Figure 5-2

prey, and/or surface water. While incidental ingestion of soil during grooming or foraging was evaluated, dermal absorption was not considered a complete exposure pathway for terrestrial receptors, since the COC metals are not readily absorbed through the skin. Inhalation of vapors was not quantified, as the metals of concern do not volatilize at ambient temperatures. Ingestion of vegetation was not evaluated since neither the owl, hawk, or mink consume substantial quantities of vegetation. Exposure pathways that were evaluated and the rationale for selecting these pathways are summarized in Table 5-2 for terrestrial receptors.

5.2.2 Aquatic Pathways

Exposure pathways relevant to the aquatic receptors present within on-site streams and ponds consist of two components: surface waters and sediments. Table 5-3 presents exposure pathways that were evaluated. Potential exposure routes for fish inhabiting on-site surface water bodies are limited to respiration (i.e., uptake of metals over the water/gill interface) and ingestion of contaminated food (prey) and sediment while foraging. Benthic invertebrates and bottom-feeding fish tend to take up metals by respiration, by feeding on algae attached to substrate particles, and by inadvertent ingestion of sediment during feeding. Therefore, some metals contained within the sediments may be retained by benthic invertebrates and consumers of these organisms, which are found in small ponds and shallow areas of streams (CH2M Hill, 1986). Exposure pathways evaluated in the ERA and the rationale for selecting these pathways are summarized in Table 5-3 for aquatic receptors.

5.3 ESTIMATING EXPOSURE POINT CONCENTRATIONS

The exposure scenario that frequently serves as a basis for risk management in ERAs is the worst-case exposure scenario, which is defined as the highest exposure reasonably expected to occur at a site (USEPA, 1989g). The intent of the worst-case scenario is to provide a conservative (health-protective) estimate of potential impacts to exposed organisms. Since the worst-case scenario uses a combination of conservative (health-protective)

Table 5-2. Exposure Pathways to be Evaluated for Terrestrial Receptors[a]

Pathway	Evaluated ?	Rationale
Dermal absorption of soil	No	Metals are not readily absorbed through integument.
Ingestion of soil	Yes	Incidental ingestion while foraging or grooming is possible.
Inhalation (vapors)	No	Metals of concern do not volatilize at ambient temperatures.
Inhalation (particulates)	Yes	Resuspension of particulates via wind may occur.
Ingestion of prey	Yes	Prey may accumulate metals.
Ingestion of vegetation	No	Neither the owl nor the hawk consumes substantial amounts of vegetation. Mink are opportunistic feeders that consume small mammals, birds, aquatic organisms, and eggs but are not known to consume vegetation (Burt and Grossenheider, 1976).
Ingestion of ground water	No	Access to ground water by terrestrial receptors is limited to springs. There are few springs onsite, and there is so much surface water available, there is no need for receptors to specifically seek out ground water. Hence, terrestrial receptors are assumed to drink surface water exclusively.
Ingestion of surface water	Yes	Access to surface water is likely and on-site water bodies may be contaminated from mill waste seepage and/or runoff.

[a] Every exposure pathway indicated in Figure 5-2 was not evaluated; only the major pathways were evaluated.

Table 5-3. Exposure Pathways to be Evaluated for Aquatic Receptors[a]

Pathway	Evaluated ?	Rationale
Ingestion of sediment	No	Incidental ingestion of suspended or bottom sediments while foraging was not expected to be a significant exposure pathway relative to the direct transfer across the gill membrane.
Dermal contact with surface water	No	It is unlikely that substantial amounts of metals would penetrate the dermal or chitinous layer of most organisms. Since aquatic organisms are likely to receive most of their metals dose from exchange across the gills, dermal uptake of metals was not quantified.
Ingestion of prey	Yes	Prey may accumulate metals of concern.
Uptake across the gill membrane	Yes	Aquatic organisms can accumulate metals at levels much higher than those measured in the surrounding water.

[a] Every exposure pathway indicated in Figure 5-2 was not evaluated; only the major pathways were evaluated.

assumptions and data, it is expected to overestimate actual risks. The rationale for evaluating the worst-case scenario, however, is that the risks estimated using this approach represent the highest risk to which any population living near the site is likely to be exposed. Thus, risks to any receptor are not likely to be greater than those estimated assuming worst-case conditions.

The exposure point concentration used to estimate worst-case exposures was the upper-bound (95th percentile) confidence limit on the arithmetic mean. Upper-bound concentrations were calculated using a one-sided confidence limit for the arithmetic mean (Gilbert, 1987) as demonstrated in the following equation:

$$UCL = \bar{x} + t_{1-\alpha, n-1} \frac{s}{\sqrt{n}} \qquad \textbf{(5-1)}$$

Where

$\bar{x}$ = the arithmetic mean;

s = the standard deviation;

n = sample size; and

t = the critical value of t for n-1 degrees of freedom at the 95 percent level of confidence.

It is important to recognize that if a small data set is highly variable, the upper-bound concentration, as derived using equation 5-1, may exceed the maximum value. In this case, EPA (1989d) recommends use of the maximum observed concentration as the worst-case estimate. To calculate upper-bound exposures, it was assumed the data were normally distributed per EPA (1989d) guidance.

Because of the uncertainty associated with estimating the extent of exposure to site-related metals, two exposure scenarios were developed. The use of single-value estimates, especially upper-bound analyses, does not provide adequate information to risk managers who must evaluate ecological risks. Consequently, a second, more plausible, exposure scenario was also used. The reasonable maximum exposure (RME) scenario uses arithmetic mean concentrations and less conservative exposure assessment assumptions to characterize ecological exposures. Both scenarios assume that all nondetectable values were equal to one-half the reporting limit. Details on the exposure assumptions used to quantify each scenario modeled are given in Section 5.4.

5.3.1 Exposure Point Concentrations for Air

The modeled air concentrations presented in Table 5-4 were used to estimate inhalation exposures by ecological receptors. These concentrations were estimated using the highest concentration of metals in the fine particulate fraction [#200 mesh (0.0029 inches) or finer] of either chat or flotation tailings. The concentration of metals in air over the Treece subsite was modeled, since it has a higher number of waste piles than the Baxter Springs subsite. The calculations used to estimate air concentrations are presented in Section 6.4.1 and Appendix B of the Human Health Risk Assessment (Dames & Moore, 1993b).

5.3.2 Exposure Point Concentrations for Surface Water

Although terrestrial receptors could ingest surface water and ground water (where it reaches the surface as springs), terrestrial receptors were assumed to consume surface water exclusively since (1) very little ground water relative to surface water is available for ingestion by terrestrial receptors, and (2) sufficient quantities of surface water are readily available. Data from on-site streams and ponds (i.e., Tar Creek, Willow Creek, Spring Branch, seven tailings ponds, and two subsidence pits) were used to estimate metal intake by terrestrial receptors from ingestion of surface water (Table 5-5). Only surface-water quality results from streams included in the aquatic biota investigation were considered in the

Table 5-4. Concentration of Selected Metals in Air Modeled Over the Treece Subsite[a]

Chemical	Concentration in Air (ug/m^3)
Cadmium	5.2×10^{-3}
Lead	7.6×10^{-2}
Zinc	9.1×10^{-1}

[a] Concentrations in air estimated using dust emissions estimates and air dispersion modeling. See Appendix B of the Human Health Risk Assessment for a detailed description of modeling methods. These concentrations were used to estimate exposures for the red-tailed hawk, the barred owl, and the mink. Although copper and mercury were identified as COCs for air, these data were not used to estimate exposure by terrestrial receptors, since levels of cadmium, lead, and zinc only were measured in mice and fish (see Section 5.3.3). Hence, only levels of these metals were included here.

exposure point assessment. These streams were assumed to represent the majority of waters ingested by potential receptors, since they could support diverse aquatic community. Upstream data (i.e., data collected from the control or reference sampling locations) were not used, since the objective of the ERA is to evaluate potential impacts to aquatic and terrestrial organisms from exposure to site-related metals. Upstream sampling locations were not impacted by site-related metals.

Ballard Pond data were not used because the pond is located within a commercial chat processing complex which does not constitute usable habitat for aquatic or terrestrial organisms. Chat washing at the Ballard operation has altered the water chemistry of this tailings pond by increasing metals concentrations above that observed in any other tailings pond in either subsite, specifically with respect to cadmium and lead. This increase is thought to stem from the continual leaching of metals from chat fines and from periodic input of make-up water from the nearby Ballard well, which draws ground water from the lower zone of the shallow aquifer. Physical conditions at this chat wash pond do not favor the establishment of aquatic populations. The pond water level was observed to fall precipitously during the summer of 1991, in part because of increased evaporation during the recycle process and seepage from the impoundment. Mine water from the Ballard well was pumped in to maintain the water level in the pond. After chat washing operations cease, the metals concentrations should drop as a result of dilution to levels similar to those observed in other tailings ponds. Physical limitations will limit the establishment of aquatic life in the pond in the future, as the pond will most likely dry out completely during the summer months.

5.3.3 Exposure Point Concentrations for Prey

It was assumed that the barred owl and the red-tailed hawk consumed only white-footed mice. This assumption is conservative and implies that all small mammals consumed by key terrestrial receptors would be contaminated at comparable levels. The upper-bound (95th percentile) and arithmetic mean concentration of lead, cadmium, and zinc in whole mice collected onsite were used to model intakes by the barred owl and the red-tailed hawk (Table

Table 5-5. Concentration of COCs in Subsite Surface Water

Metal	n	Range (mg/L)	Arithmetic Mean Concentration (mg/L)[a]	Upper-Bound Concentration (mg/L)[a]
Cadmium	31	ND (0.00025)-0.16	0.03	0.04
Copper	31	ND (0.0025)-0.01	0.005	0.007
Iron	31	ND (0.05)-11	1.5	2.4
Lead	31	ND (0.0005)-0.10	0.02	0.04
Manganese	31	0.03-1.1	0.3	0.43
Mercury	9[b]	ND (0.0001)-0.0005	0.0003	0.0004
Nickel	31	ND (0.02)-0.05	0.03	0.04
Silver	12[c]	ND (0.00005)-0.005	0.001	0.003
Zinc	31	0.054-22.0	5.0	7.1

[a] Concentrations are based on measured data from Tar Creek (n=6), Spring Branch (n=8), Willow Creek (n=8), and the ponds (n=9). These data were used to estimate exposures to the red-tailed hawk, barred owl, and the mink and to evaluate the toxicity of surface water to benthic invertebrates and fish.

[b] Detected in the ponds only.

[c] Detected in the ponds and in Tar Creek only.

ND = not detected. Value in parentheses represents one-half the reporting limit.

Table 5-6. **Concentration of Selected Metals In Whole Mice From All Samples Collected Sitewide**

Metal	n	Range (mg/kg)	Arithmetic Mean Concentration (mg/kg)[a]	Standard Deviation (mg/kg)[a]	Upper-Bound Concentration (mg/kg)[a]
Cadmium	10	0.03-3.3	0.8	1.2	1.5
Lead	10	0.2-5.9	2.2	2.2	3.7
Zinc	10	29.2-48.2	35.6	6.5	39.4

[a] Concentrations are based on measured data from Baxter Springs and Treece subsites, control site not included. These data were used to estimate exposures by the red-tailed hawk, barred owl, and the mink. All data are on a wet weight basis.

Table 5-7. Concentration of Selected Metals In Whole Body Fish Tissue From All Samples Collected Sitewide

Metal	n	Range (mg/kg)	Arithmetic Mean Concentration (mg/kg)[a]	Standard Deviation (mg/kg)[a]	Upper-Bound Concentration (mg/kg)[a]
Cadmium	31	0.01-1.0	0.2	0.2	0.3
Lead	31	0.07-22	3.5	5.1	5.1
Zinc	31	14.6-373	92.2	74.4	114.9

[a] Concentrations are based on measured data from Tar Creek, Spring Branch, Willow Creek, and the ponds. These data were used to estimate exposures to the mink. All data are on a wet weight basis.

5-6). The mink was assumed to consume 50 percent fish taken from on-site surface water bodies and 50 percent mice taken from all on-site sampling locations, excluding reference locations. This assumption is reasonable, given that its home range is 171 to 450 acres (Eisler, 1987), while the areas impacted by mine waste are 430 acres and 735 acres for the Baxter Springs and Treece subsites, respectively. The upper-bound and arithmetic mean concentration of lead, cadmium, and zinc in whole mice and fish taken from on-site water locations were used to model intakes by the mink (Tables 5-6 and 5-7, respectively). Whole body mice and fish samples were analyzed for cadmium, lead, and zinc only, as data available at the time of sampling and analysis suggested these chemicals were the most abundant and potentially toxic of the mining-related chemicals that might accumulate in ecological receptors. Thus, dose estimates for the three key terrestrial receptors were developed for cadmium, lead, and zinc only. The uncertainty associated with excluding other metals that were COCs in air, water, or soil is discussed in Section 9.0.

5.4 EXPOSURE (DOSE) ASSESSMENT

Determining if exposure to site-related contaminants may increase the incidence of adverse impacts in exposed populations is an important step in the risk assessment process. The objective of this task is to estimate the magnitude, frequency, duration, and route of exposure to site-related chemicals by ecological receptors. Accomplishing this task involves estimating total contaminant intake by potentially-exposed receptors for relevant pathways of exposure using the previously defined exposure-point concentrations.

5.4.1 Terrestrial Receptors

Each of the three key terrestrial receptors were evaluated separately. Potential exposures for these key species were determined based on the species' life history and feeding habits. Quantification of exposures involves using species-specific numerical exposure factors including body weight, ingestion rate, and fraction of prey, water, and soil/mine waste consumed from the contaminated area. Exposure factors used to model intakes by key

Table 5-8. Exposure Parameters Used to Model Intakes by Terrestrial Predators

Exposure Factor	Red-Tailed Hawk	Barred Owl	Mink
Body weight[a]	1.32 kg	0.74 kg	1.5 kg[b]
Food Ingestion Rate[a]	0.135 kg/day	0.09 kg/day	0.23 kg/day[b]
Water Ingestion Rate[c]	0.045 L/day	0.03 L/day	0.08 L/day
Inhalation Rate[d]	0.6 m^3/day	0.3 m^3/day	0.45 m^3/day
Soil Ingestion Rate	0.007[e]	0.005 kg/day[e]	0.002 kg/day[f]

[a] Source: Cedar Creek Associates, Inc., 1992. Personal Communications. Body weights were not measured during field studies; hence, body weight values listed in this table were obtained from a certified wildlife biologist.

[b] Value is for the male mink.

[c] Assumes that water ingestion rate is one-third of total daily food intake as reported for waterfowl by Kenaga (1973).

[d] Since species-specific data were not available, inhalation rates were derived from data available for the rat using an inhalation rate-to-body weight ratio. Rat inhales 0.2 m^3/day and weighs 0.5 kg. Although these values may seem subjectively determined, the reader should keep in mind that the concentration of COCs in air is small, as is the total contribution via inhalation to total daily intake of COCs by terrestrial receptors.

[e] Soil ingestion rate was assumed to be 10 percent of food ingestion rate, since raptors consume entire prey (i.e., pelt).

[f] Soil ingestion rate was assumed to be 2.8 percent of food ingestion rate, since the mink was assumed to consume primarily fish (i.e., incidental ingestion of soil while grooming was assumed to be 1 percent of food intake rate) (D. Mark Doolan, EPA, Region VII, personal communication, January 27, 1993).

species are presented in Table 5-8. Two different dietary scenarios (worst-case and RME scenarios) were evaluated for each key species.

5.4.1.1 Dose Estimates for the Red-Tailed Hawk and the Barred Owl

Potential routes of exposure for the red-tailed hawk and the barred owl include ingesting contaminated prey (the white-footed mouse as surrogate), inhaling subsite air, incidentally ingesting soil while foraging or grooming, and ingesting contaminated surface water. Intakes (mg/kg-day) were estimated using the following equation:

$$I = \frac{(CMxQMxFI) + (CAxQAxFI) + (CSxQSxFI) + (CWxQWxFI)}{BW} \qquad (5\text{-}2)$$

where:

- CM = the measured concentration metal in whole body mice (mg/kg);
- QM = the quantity of mice ingested (kg/day);
- CA = the predicted concentration of metal in air over Treece (mg/m^3);
- QA = the quantity of air inhaled (m^3/day);
- CS = the measured concentration of metal in near-pile soils (mg/kg);
- QS = the quantity of soil ingested (kg/day);
- CW = the measured concentration of metal in on-site surface water (mg/L);
- QW = the quantity of water ingested (L/day);
- FI = the fraction of material inhaled or ingested from the subsites; and
- BW = species-specific body weight (kg).

Values for QM, QS, QW, and QA are specified for each terrestrial species evaluated in Table 5-8, while the concentration of metals in air, surface water, whole body mice, and soil used to quantify intakes are listed in Tables 5-4 through 5-6 and Table 3-3, respectively. Although ingestion of vegetation is a potential exposure route for the raptors, the intake of metals from ingesting potentially-contaminated vegetation was not quantified, since the owl and hawk do not consume substantial amounts of vegetation. Dose calculations (spreadsheets) for all terrestrial receptors are included in Appendix A. The following assumptions were made to model exposures by the red-tailed hawk and the barred owl assuming worst-case conditions. Justification for the assumptions for modeling worst-case and RME intakes are based on best professional judgment corroborated by a technical review of the available literature and consultation with certified wildlife biologists.

- All mice samples (i.e., samples collected from throughout the subsites excluding reference sampling locations; n = 10) were used to model intakes. The upper-bound (95th percentile) concentration of the three metals evaluated in whole body mice samples from all on-site sampling locations (excluding reference locations) (Table 5-6) was used to model intakes by raptors.

- The red-tailed hawk and the barred owl were assumed to obtain 100 percent of their prey (mice) from impacted areas (i.e., fraction of prey consumed from the contaminated area is 1.0). This assumption is conservative since the home range of the red-tailed hawk is 210 to 1803 acres with a median value of 1000 acres (Johnsgard, 1990), while only 430 acres and 745 acres within the Baxter Springs and Treece subsites, respectively, have been impacted by mine waste. Thus, it is reasonable to assume that the hawk would forage only over mining-impacted areas. The home range of the owl is 1913 to 8200 acres (Johnsgard, 1990). The owl's relatively large home range indicates it is likely to forage over an area bigger than the mine-related areas onsite.

- 100 percent of the air inhaled by the red-tailed hawk and barred owl was assumed to be subsite air. Since it was assumed that air throughout the subsites contained equivalent levels of metals, FI values for inhalation correspond to those used to model prey ingestion intakes. Data used to estimate exposures via inhalation are the modeled concentrations in air over the Treece subsite (Table 5-4).

- Raptors were assumed to obtain 75 percent of their daily water requirements from ingesting prey. Hence, only 25 percent of the water taken in comes from ingestion of on-site surface water. This assumption is conservative, since it is unlikely that, on the average, raptors would obtain 25 percent of their total water intake from direct ingestion of water. The upper-bound concentration of metals in on-site surface water bodies presented in Table 5-5 were used.

- The fraction of soil incidentally ingested (e.g., while grooming) was assumed to be 10 percent (i.e., 90 percent of the soil taken in by raptors was assumed to originate from the ingestion of prey). This assumption is reasonable, since the whole body concentrations for the mice used to estimate raptor doses include the pelt and any associated soil adhering to the pelt. No hair was removed nor were the mice brushed or cleaned prior to laboratory tissue analysis. The upper-bound concentration of metals in near-pile soils listed in Table 3-3 were used to model intakes from incidental ingestion of soil. Terrestrial receptors were assumed to range exclusively over near-pile soils (or other soils contaminated at comparable levels). This assumption is conservative because the home ranges of the owl and the hawk are larger than the area covered by near-pile soils. For example, if a mean weight-average zinc concentration had been used, which takes into account the relative areal distribution of mill waste, near-pile soils and agricultural ground, the mean zinc concentration of home range soils would be 162.5 mg/kg versus the mean concentration of zinc in near-pile soils of 710 mg/kg.

The following assumptions were used to model the RME (reasonable likely exposure) scenario:

- The concentration of metals in mice tissue was the arithmetic mean level of all samples collected site-wide (excluding samples taken from reference locations) (Table 5-6).

- 75 percent of the raptor's forage consists of small mammals that originate from impacted areas (FI = 0.75). Because the home range of the hawk is larger than the area impacted by mine waste, it is feasible that the hawk could forage over nonmining-related areas. Given the large home range of the owl (1900 to 8200 acres) relative to mining-related areas (430 acres for Baxter Springs and 745 acres for Treece), it is reasonable to assume that the owl would obtain only a fraction of its prey from mining-impacted areas. These assumptions in combination with the use of arithmetic mean metals levels in prey represent logical RME conditions.

- 75 percent of the air inhaled by hawks and owls is assumed to be subsite air (see worst-case assumptions for justification) (Table 5-4).

- It was conservatively assumed that raptors obtain 90 percent of their daily water requirements from ingesting prey (i.e., FI for water ingestion = 0.10). In addition, arithmetic mean surface water concentrations were used to model RME intakes (Table 5-5).

- The fraction of soil incidentally ingested was assumed to be 0.1 (i.e., raptors obtain 90 percent of the soil taken in from ingestion of contaminated prey). The arithmetic mean concentration of metals in near-pile soils listed in Table 3-3 was used. (See worst-case assumptions for justification of the use of near-pile soil data.)

The assumption that mouse tissue was 100 percent of the predator's diet implies that any other prey consumed by predators contain cadmium, lead, and zinc at levels comparable to those measured in the whole body mice samples. White-footed mouse samples were used, since mice are an important food base for higher trophic level organisms (predators). The barred owl's diet consists primarily of small woodland rodents (e.g., white-footed mice) and the red-tailed hawk's diet consists of a variety of small rodents (Cedar Creek Associates, 1992). Dose estimates assumed 100 percent absorption and assimilation of metals.

Example Dose Calculation for the Red-Tailed Hawk

$$I = \frac{(CM \times QM \times FI) + (CA \times QA \times FI) + (CS \times QS \times FI) + (CW \times QW \times FI)}{BW} \qquad (5\text{-}2)$$

Worst-Case Intake of Lead = (3.7 mg/kg * 0.135 kg/day)
+ (0.04 mg/L * 0.045 L/day * 0.25)
+ (7.6×10^{-5} mg/m^3 * 0.6 m^3/day * 1.0)
+ (113.9 mg/kg * 0.014 kg/day * 0.1)
= 0.50 mg/day + 4.5×10^{-4} mg/day + 4.6×10^{-5} mg/day + 0.16 mg/day
= 0.62 mg/day / 1.32 kg = 0.50 mg/kg-day

Table 5-9 data show that both raptors are likely to ingest more zinc than cadmium or lead and that ingested quantities of cadmium and lead are similar.

5.4.1.2 Dose Estimates for the Mink

The mink was assumed to ingest subsite fish and mice, inhale subsite air, drink subsite surface water, and incidentally ingest near-pile soils while grooming. Again, calculated intakes were conservatively expressed as the amount of metal actually taken into the body using Equation 5-2 by assuming that the mink's diet is 50 percent fish and 50 percent mice. The measured concentration of metals in whole body fish (CF) and mice (CM) were used to

Table 5-9. Dose (Intake) Estimates for Terrestrial Receptors

Metal	Red-Tailed Hawk (mg/kg-day)	Barred Owl (mg/kg-day)	Mink (mg/kg-day)
WORST-CASE SCENARIO			
Cadmium	0.16	0.19	0.19
Lead	0.50	0.59	0.96
Zinc	5.15	6.07	16.5
RME SCENARIO			
Cadmium	0.07	0.08	0.11
Lead	0.26	0.31	0.63
Zinc	3.50	4.1	13.6

model intakes. Since mink are opportunistic feeders that consume small mammals, birds, aquatic organisms, and eggs but are not known to consume vegetation (Burt and Grossenheider, 1964), exposures to metals from the ingestion of potentially contaminated vegetation were not quantified. Dose calculations (spreadsheets) for all terrestrial receptors are included in Appendix A. Similar assumptions were used to estimate dose by the mink for the worst-case and an RME scenarios. The following assumptions were made to model a worst-case scenario:

- Mink were assumed to consume prey from the subsites only, which assumes that all fish and mice eaten are contaminated at levels comparable to those found in on-site fish and mice. Consumption of fish was assumed to account for 50 percent of the mink's total prey intake. This assumption is reasonable given the home range of the mink is 171 to 450 acres (Eisler, 1987), while the areas impacted by mine waste are 430 acres and 735 acres for the Baxter Springs and Treece subsites, respectively. Thus, it is reasonable to assume that the mink would forage exclusively on on-site prey (mice and fish).

- The upper-bound (95th percentile) concentration of metals used was the measured level in whole body fish and mice taken throughout the subsites (except reference sampling locations) (Tables 5-6 and 5-7). Consumption of mice was assumed to account for the remaining 50 percent of the mink's total prey intake.

- 100 percent of the air inhaled by the mink is assumed to be subsite air. Data used to estimate exposures via inhalation are given in Table 5-4.

- 50 percent of the water ingested by the mink is on-site surface water and 50 percent of the water ingested comes from ingesting on-site prey. This assumption is based on the premise that the mink would spend more time than other terrestrial predators (i.e., raptors) in or near surface water, since 50 percent of its diet is aquatic organisms. The upper-bound concentrations of metals in on-site surface water bodies presented in Table 5-5 were used.

- The fraction of soil incidentally ingested was assumed to be 10 percent, since it was assumed that only 50 percent of the mink's diet is terrestrial prey, which assumes that mink do not ingest large amounts of soil while grooming or foraging (i.e., the majority of soil ingested is that associated with their terrestrial prey). The upper-bound concentration of metals in near-pile soils listed in Table 3-3 were used.

The following assumptions were used to model intakes by the mink under a RME scenario:

- As for the worst-case scenario, mink were assumed to consume prey from the subsites only. For the RME scenario, however, the arithmetic mean concentration of metals in the whole body fish and mice from all samples collected site-wide (Tables 5-6 and 5-7) were used to quantify intakes. Since the mink consumes many fish over a lifetime, the concentration of metals in fish eaten by mink is more likely to approximate the arithmetic mean concentration of metals measured in on-site fish rather than upper-bound levels.

- 100 percent of the air inhaled by mink is subsite air. Data used to estimate exposures via inhalation are given in Table 5-4.

- 50 percent of the water ingested is on-site surface water (i.e., equal amounts of water needed by the mink come from ingesting prey). The arithmetic mean concentrations of metals in on-site surface water bodies presented in Table 5-5 were used.

- The fraction of soil incidentally ingested was assumed to be 10 percent. The arithmetic mean concentrations of metals in near-pile soils listed in Table 3-2 were used.

Example Dose Calculation for the Mink

$$I = \frac{(CM \times QM \times FI) + (CF \times QF \times FI) + (CA \times QA \times FI) + (CS \times QS \times FI) + (CW \times QW \times FI)}{BW} \qquad (5\text{-}3)$$

Worst-Case Intake of Lead

= (3.7 mg/kg * 0.23 kg/day * 0.5)
+ (5.1 mg/kg * 0.23 kg/day * 0.5)
+ (0.04 mg/L * 0.08 L/day * 0.5)
+ (7.6×10^{-5} mg/m^3 * 0.45 m^3/day * 1.0)
+ (113.9 mg/kg * 0.006 kg/day * 0.1)
= 0.4 mg/day + 0.6 mg/day + 2×10^{-3} mg/day + 5×10^{-5} mg/day + 0.07 mg/day
= 1.1 mg/day / 1.125 kg = 0.96 mg/kg-day.

Dose estimates for the mink are given in Table 5-9. These data show that all three receptors are likely to ingest more zinc than cadmium and lead, and that the mink is expected to ingest substantially more of all three metals, as a whole, than raptors. This finding is largely due to the fact that the mink has a higher food/water/soil intake-to-body weight ratio than the raptors and that the fish consumed by the mink, which are not eaten by the raptors, generally contain higher levels of metals than mice.

5.4.1.3 Evaluation of White-Footed Mouse Data

Dose (intake) estimates for the white-footed mouse were not calculated since body burden (tissue concentrations) were measured in mice collected from the Baxter Springs/Treece subsites. EPA had previously agreed (July 24, 1991 meeting) that focusing on primary and higher-level consumers would provide sufficient data to assess the general condition of the terrestrial ecosystem. The original purpose of collecting mice tissue samples was to estimate dose (intake) by higher trophic level organisms, not to quantify effects on mice. As a result, the mice body burden data are used here to qualitatively discuss potential impacts to mice since field data are not available to quantify intake parameters.

Body burden data for whole-body mice collected from the subsites were compared to body burden levels from reference areas to determine if metals levels in on-site mice are elevated relative to the concentration of metals measured in reference samples. The mouse data listed in Table 5-10 represent the total concentration of metals including the pelt and any associated dirt/soil. Since it is impossible to tell from the way the mice samples were analyzed how much of the reported concentration has actually accumulated in mouse tissue and how much was associated with the pelt and/or the gut, a comparison of the measured concentration of metals in mice with body burden levels reported to cause toxic effects in mice is not appropriate. The reference area within Cherokee County, was sampled during the RI field activities, while the non-Cherokee County sites are control samples collected from other mining areas (i.e., northern Idaho and central New Hampshire). In these cases, the mice were taken from control areas (not mine-impacted areas) within site boundaries.

Table 5-10. **Comparison of Concentrations of Metals in Whole-Body Mice Tissue Samples (Wet Weight) Collected from the Baxter Springs/Treece Subsites with the Metals Levels in Whole Samples Taken from Various Reference Locations**

Location	Cadmium Concentration (mg/kg)	Lead Concentration (mg/kg)	Zinc Concentration (mg/kg)
SAMPLE SITES WITHIN THE Baxter Springs/Treece SUBSITES			
Upper-bound value sitewide (n = 10)	2.8	6.6	90.3
Arithmetic mean concentration sitewide (n = 10)	1.7	4.5	73.9
REFERENCES SITES			
Cherokee County Control Site[a] (approximately 2.0 miles north of the City of Baxter Springs) (n = 3)			
-- Arithmetic mean	0.12	0.16	32.5
-- Upper-bound value	0.30	0.26	32.9
-- Range	0.02 - 0.27	0.10 - 0.24	32.1 - 32.7
Non-Cherokee County sites[b]			
-- Range[c]	<0.06 - 0.5	0.5 - 4.7	19.9 - 31.6

[a] These data were collected during the RI field activities.

[b] Schesinger, et al., 1974, Dames & Moore, 1989, and Smith and Rongstad, 1982.

[c] An arithmetic mean could not be calculated, since only a range of data were reported.

The data presented in Table 5-10 show that the arithmetic mean concentration of cadmium, lead, and zinc measured in whole-body mice samples taken from various sampling locations throughout the Baxter Springs/Treece subsites are, respectively, 14, 28, and 2 times higher than arithmetic mean levels measured in mice taken from the Cherokee County control site. The arithmetic mean concentration of lead measured in whole-body mice from the Baxter Springs/Treece subsites is within or lower than the range of concentrations in mice from non-Cherokee County areas. Note that the arithmetic mean could not be calculated for the non-Cherokee County reference sites, since only a range of concentrations was reported. These data show that mean levels of cadmium, lead, and zinc measured in mice from mining-impacted areas within the Baxter Springs/Treece subsites are elevated relative to levels measured in mice taken from the control (non-impacted) area of Cherokee County. Collectively, the whole-body data indicate exposure to metals but the sampling techniques used do not allow segregation of body burden (i.e., metals could be in muscle tissue, organs, gut, pelt, etc.).

5.5 EXPOSURES BY AQUATIC RECEPTORS

It was not necessary to predict doses for aquatic receptors, since whole-body analyses for fish were obtained as part of this study. Levels of cadmium, lead, and zinc in whole fish collected from on-site streams and ponds were compared to levels of these metals in fish from various reference areas (Table 5-11). Data on the various references areas were obtained from the EPA Storet database. Information as to whether these reference sites are located in a mining district similar to the Tri-State Mining District is not available in the Storet database. While the main purpose of collecting fish data was to evaluate food chain effects, these data were also used to determine if metals levels in on-site fish are elevated relative to the concentration of metals in reference samples. Table 5-11 shows that the arithmetic mean concentration of lead and zinc measured in whole-body on-site fish generally exceeds the range of values measured in fish from various reference sites while the mean cadmium concentration for on-site fish generally falls within the range of concentrations in fish taken from various reference locations in Kansas. These data indicate that on-site fish have been

Table 5-11. Comparison of Concentrations of Metals in Whole-Body Fish Samples (Wet Weight) Collected from the Baxter Springs/Treece Subsites with the Metals Levels in Whole Fish Samples Taken from Various Reference Locations

Location	Cadmium Concentration (mg/kg)	Lead Concentration (mg/kg)	Zinc Concentration (mg/kg)
SAMPLE SITES WITHIN THE Baxter Springs/Treece SUBSITES			
Upper-bound value sitewide (n = 33)	0.3	5.1	115.0
Arithmetic mean concentration sitewide (n = 33)	0.2	3.5	92.2
REFERENCE SITES[a]			
Neosho River at Chetopa (bottomfeeders)	<0.05 - 0.3	0.2 - 1.4	12.0 - 67.5
Neosho River at Chetopa (mixed species)	0.06	0.4	NR[a]
Kansas River at Lawrence, Kansas (bottomfeeders)	0.05 - 0.35	<0.1 - 0.2	53.7 - 71.0
Olathe Lake west of Olathe, Kansas (bottomfeeders)	0.02	2.8	41.0 - 66.0
Various Locations in Kansas (mixed species)	0.05 - 0.4	0.5 - 1.7	15.1 - 81.4

[a] EPA Storet database, 1992. Information on whether these reference sites are located in a mining district similar to the Tri-State Mining District is not available in the Storet database. Fish inhabiting surface water bodies within mine areas are expected to have higher metal body burdens due to the higher background levels of metals in these waters.

exposed to metals. Comparison of on-site results with the data from reference sites are not conclusive, however, since sample sizes, sampling methods, and laboratory preparation techniques for the reference site fish are not known. As with the mouse whole-body samples, the subsite whole-body fish samples were not collected for direct comparison to reference site data, but to estimate dose for upper trophic level organisms.

5.6 FIELD SURVEY RESULTS

Site-specific aquatic and terrestrial biota surveys were conducted by Dames & Moore personnel and Cedar Creek Associates, Inc. of Fort Collins, Colorado, during the Fall of 1991 to: 1) inventory terrestrial and aquatic species present, 2) evaluate the existing conditions with respect to habitat requirements of potential inhabitants, and 3) briefly search for unique habitats and other features of the landscape that may provide necessary life requisites for species of special concern (e.g., key indicator species and/or T&E species). A detailed description of the survey methods used during the biota investigations and results of those investigations are included in the Final RI.

5.6.1 Terrestrial Survey Methods

The terrestrial biota investigations emphasized the evaluation of higher level consumers (raptors and terrestrial predators) and primary consumers (songbirds and small rodents). This strategy was developed with EPA during the July 24, 1991 meeting. The focus was placed on primary and higher-level consumers as they would provide sufficient data to assess the general condition of the terrestrial ecosystem. General reconnaissance surveys were undertaken to evaluate relative populations and species composition on and about the Baxter Springs/Treece subsites and a control area for songbirds and small mammals.

Semi-quantitative surveys of raptors and predators involved traversing a 28-mile long transect within the Baxter Springs/Treece subsites to determine habitat occupied and distance from the nearest mine-related disturbance. Variable-width strip transects and traplines were established in four test areas and one control area to facilitate comparison of small mammal and songbird populations between test and control areas to detect disparities as a result of exposure to mine-related metals. White-footed mice (*Peromyscus leucopus*) were captured along these transects for laboratory analysis of body burden levels of site-related metals. Although a concerted effort was made to ensure that all five sites selected for comparative primary consumer investigations were established in similar, wooded habitats to minimize possible effects from habitat-related variables, one test site within the Baxter Springs subsite, BS-1, was somewhat anomalous. Detailed descriptions of the sampling/survey method used and of the five areas surveyed is given in Section 3.5 of the Final RI.

5.6.2 Terrestrial Survey Results

Wildlife species, relative abundance, and habitats occurring in the Baxter Springs/Treece subsites are typical of those occurring in the agriculturally-altered tall grass prairie/eastern woodland ecotone. Habitats are principally associated with cropland margins and non-tillable areas with native flora remaining only in the riparian bottoms and relic woodlots.

5.6.2.1 Predators

Terrestrial predators of interest in the vicinity of the subsites primarily include members of the canid and mustelid families. These animals did not appear to exhibit a distinct pattern of avoidance of or preference for mine-related disturbances. The plot of predator distance from mine-related disturbance shown in Appendix E of the Final RI forms a smooth curve from zero to several thousand feet, suggesting no obvious avoidance patterns.

5.6.2.2 Terrestrial Raptors

Of the 19 raptors that could occur onsite, red-tailed hawks were the most commonly observed species, followed closely by American kestrels and turkey vultures. Given the level of effort, apparent prey base, and actual habitat conditions, the number and diversity of observations seems high but may be normal for the area for the time of year (Cedar Creek, certified wildlife biologist). Table 4 (Appendix E of the Final RI) indicates the approximate distance of each raptor sighting from a mapped mine-related disturbance area. Plots of these data for the four most-often observed raptors indicates no distinct pattern of avoidance of or preference for these areas although this conclusion cannot be supported statistically. Distance between predator sightings and mine-related disturbances is only one of the many factors used to assess potential adverse impacts to terrestrial populations.

This lack of avoidance may be further corroborated by the fact that nine of the 80 sightings (11 percent) were of raptors over or very near chat piles. Conversely, one plausible explanation for this observation is that vegetation cover is usually quite thin in these disturbed areas, which provides raptors with a better view of the ground surface and the prey thereon. This hypothesis is supported by the fact that 34 of the 80 observations of raptors (43 percent) were over cultivated fields where a relatively unobstructed view of the surface is available.

5.6.2.3 Songbirds

The following five locations were evaluated to reveal disparities in songbird and small mammal communities.

SONGBIRD AND SMALL MAMMAL TEST SITES

Control	Area Located in a Non-Impacted Area	Entirely Wooded With the Exception of a Narrow Pipeline Corridor
Baxter Springs - 1 (BS-1)	Area close to mining-related contamination	A relatively narrow strip of second-growth introduced and invading woody plants bordering cultivated fields to the west and old fields to the east.
Baxter Springs - 2 (BS-2)	An area more distant from mining-related disturbances but well within the subsite boundaries.	Small woodlot of second-growth trees.
Treece - 1 (T-1)	Area close to mining-related contamination.	A mostly wooded lowland area.
Treece - 2 (T-2)	Area more distant from mining-related disturbances but well within the subsite boundaries.	Entirely wooded.

The control site was selected because of its distance from any mine-related disturbance and access could be readily attained. All four test sites (BS-1, BS-2, T-1, and T-2) and the control site represent wooded habitat. BS-1 has the most anomalous habitat of the five sites.

Of the more than 100 species of songbirds that could potentially occur on or near the subsites (Thompson and Ely, 1989), a total of 43 species were documented during the field surveys (Table 5, Appendix E of the Final RI), with the eastern meadowlark and the blue jay being the most commonly observed species. The highest number of songbirds were observed at the BS-1 sample location (average of 69 individuals per sample), which is more than twice the amount observed at any other sample location, while T-2 had the least number of songbirds with an average of 23 individuals per sample. Based on the professional judgment of the certified wildlife biologists conducting the survey, songbird density and diversity seemed to be within the realm of expectation for project area habitats when field observations were made (Fall, 1991). Given the relatively narrow width of wooded habitat at BS-1, the adjacent grassland and cultivated areas which did not restrict the biologist's vision, and the fact that over half of the observations were of non-woodland species (e.g., meadowlarks), it appears that songbird densities for BS-1 were higher than at the other sites, probably due to habitat-related variables.

5.6.2.4 Small Mammals

White-footed mice were by far the most commonly captured species with 49 total captures. BS-2 has the highest trap success rate with a total of 23 captures (11.5 percent), while BS-1 had a trap success rate of only 2 percent. The control, T-1, and T-2 sites had relatively consistent trap success rates of 8.5 percent, 7.0 percent, and 6.5 percent, respectively. The most plausible explanation for this difference in trap success rates is that the habitat of the BS-1 site is sufficiently different from the other four sites to cause anomalously low trapping results. For example, 17 white-footed mice were captured in the control area, which is not surprising since the white-footed mouse is a typical woodland species and the control area is primarily mature oak woodland with little brushy understory. Conversely, only four white-footed mice were captured in BS-1, which is principally a narrow strip of windbreak and brushy invader plants adjacent to an old field. The only apparent incongruity of such reasoning is that the woodland niche normally filled by the white-footed mouse was not filled by the deer mouse, a non-woodland resident. It would be expected that the old field adjacent to BS-1 would have supported deer mice that could then have been attracted to the trapline. However, there is no way to validate this hypothesis, as no deer mice were observed in the old field.

5.6.2.5 Other Wildlife

White-tailed deer (*Odocoileus virginianus*) is the only big game species occurring within the subsites, and habitats present provide year-round range. Although population estimates are not available for the Baxter Springs/Treece subsites, observations of deer during the various studies, especially the spotlighting activity, suggest that the population is near normal given habitat conditions. This opinion is based on the professional judgment of a certified wildlife biologist who has spent nearly a decade observing mid-western white-tailed deer populations. Further, no observable degradation or under utilization of forage seemed to be present to indicate that populations were either overly dense or too sparse given the availability of habitat.

Mr. Keith Sexon, Kansas Department of Parks and Wildlife Big Game Program Coordinator (KDWP, 1992), indicated that the whitetail deer populations in Kansas are stable with a slow increase in population numbers in southeast Kansas, which includes Cherokee County. Deer populations continue to increase in spite of the implementation of increased doe harvesting. The increase in numbers has been supported by landowner testimony, road kill data, and hunter success rates (harvest). According to Mr. Sexon, an additional sign of increasing populations is the increased frequency of twin and triplet whitetail births.

5.6.2.6 Terrestrial Species Diversity

The assessment of species diversity in the subsites was performed by determining whether noticeable differences between control (off-site) and on-site stations existed. Based on professional judgment and currently existing information previously presented, the terrestrial wildlife community in the Baxter Springs/Treece subsites appears to be normal in comparison with the control area and in relation to what might be expected in Cherokee County. There seem to be no obviously missing wildlife groups or poor population levels among those groups which could be considered most susceptible to heavy metal contamination (e.g., primary and tertiary consumers).

5.6.3 Aquatic Receptor Survey Methods

The quality of aquatic habitat and potential for aquatic species to inhabit streams within the subsites are important inputs to the exposure assessment of the environmental evaluation. Habitat analyses were conducted to document the availability and quality of aquatic habitat and, in turn, to assess the potential for these streams to support populations of fish that may complete a major pathway of metals exposure. Since similar data on the Spring and Neosho Rivers were not gathered during RI activities, the following sections report results of the biosurvey assessment that was completed for the on-site ponds and streams only. Also, because the Spring and Neosho Rivers receive metals input from numerous other upstream sources, the evaluation of the subsites influence on these rivers was not possible and, therefore, these rivers were not evaluated in detail.

5.6.4 Stream Habitat and Fish Community Assessment Results

Stream habitat was assessed during the Fall of 1991 for Tar Creek, Willow Creek, and Spring Branch using the habitat assessment methodology described in the Rapid Bioassessment Protocols for use in Streams and Rivers (USEPA, 1989). Due to the lack of flow in the upper reaches of the streams, only the lower reaches of each stream were sampled. Representative stream segments were identified based on stream channel morphology and stream flow for each stream. A stream segment of approximately 100 yards in length was evaluated near each downstream station (WC-3, TC-3, and SB-2) for the primary habitat parameters listed in column one of Table 5-12. Results of this assessment are presented in Table 5-12.

Pond and stream segments were evaluated to assess fish community composition and health by determining fish species diversity, size or age-class distribution, and condition factors (measure of plumpness or well-being). Ponds were sampled by a combination of gill nets, electrofishing (along the shoreline), and minnow traps. Minnow traps were essentially the most productive sampling methods used. Streams were sampled by electrofishing only. Representative samples of fish collected during this task were retained for fish tissue analysis. Fish sampling yielded fish from each aquatic sampling location except for Ballard Pond. Condition factors were calculated for the larger fish collected using references cited in Carlander (1977). A condition factor greater than one indicates that the species is experiencing normal isometric growth (i.e., length, breadth, depth, and weight are proportional (Everhart *et al.*, 1975)). A condition factor of 1 or greater indicates

Table 5-12. **Results of the Baxter Springs/Treece Subsites Stream Habitat Assessment**

Habitat Parameter	Stream Segment Near Willow Creek Station WC-3[a]	Stream Segment Near Spring Branch Station SB-2[a]	Stream Segment Near Tar Creek Station TC-3[a]
Bottom substrate/ available cover	0 (poor)	15 (good)	3 (poor)
Embeddedness	0 (poor)	8 (fair)	0 (poor)
Stream flow	0 (poor)	4 (poor)	2 (poor)
Channel alteration	3 (poor)	4 (fair)	2 (poor)
Bottom scouring and deposition	3 (poor)	4 (fair)	5 (fair)
Pool-to-riffle ratio	3 (poor-to-fair)	15 (excellent)	0 (poor)
Bank stability	10 (excellent)	8 (good)	8 (good)
Streamside cover	8 (good)	9 (excellent)	4 (fair)
Bank vegetative stability	10 (excellent)	8 (good)	8 (good)
OVERALL RATING	37 (poor-to-fair)	75 (fair)	31 (poor-to-fair)

[a] At each station a stream length of approximately 100 yards was evaluated for the primary habitat parameters listed in column one.

Table 5-13. Summary of Condition Factors by Species

	Within the Baxter Springs/Treece Subsites	Off-Site[a]
White crappie	1.2 - 2.9	0.55 - 2.3
Green sunfish	1.6 - 3.4	1.5 - 2.1
Bluegill	1.5 - 2.5	1.0 - 3.5
Warmouth	1.9 - 2.3	1.7 - 2.3
Largemouth bass	1.0 - 1.9	0.9 - 2.0
Pumpkinseed	2.1 - 2.6	1.9 - 2.0

[a] Condition factors for fish of similar length from locations near or similar to southeastern Kansas (Oklahoma, Iowa, and Illinois).

proportional growth; a condition factor less than one indicates that body weight is not in proportion (less than) body dimensions. Table 5-13 lists the condition factors for subsite fish and condition factors expected for fish of the same species and similar length existing in the same geographic area. Results of these assessments are briefly discussed for each individual on-site stream and ponds. Additional information regarding the results of these assessments are included in the Final RI.

5.6.4.1 Spring Branch

Segments of Spring Branch were accessed near water quality sampling locations SB-1, SB-2, and approximately 2000 ft. downstream from SB-1. Based on observations at these locations, Spring Branch was divided into two segments: SB-A, which extends from the Spring River upstream for approximately one stream mile; and SB-B, which includes the remaining length of the stream (Figure 2-2; Section 2.0). The stream channels of the upper reaches of Spring Branch (SB-B) were completely dry with an established stand of terrestrial vegetation (grasses) extending throughout the stream channel. Ponded or pooled water with little flow was characteristic of stream segments downstream (SB-A) to the Spring River. At sampling station SB-1 (Final RI presents details concerning the location of all stream stations and segments), bottom substrate consisted of approximately 40 percent rubble or gravel and appeared to have experienced sedimentation from upstream activities in that the rubble and gravel were approximately 60 percent covered by fine material. Flow at this study site was low (<0.1 cfs). This segment also exhibited moderate deposition of gravel and minor accumulations of silt in the pools. Streamside cover consisted primarily of shrubs with a tall canopy of trees. The pool-to-riffle ratio was estimated to be 3 with a variety of habitat and adequate depth in the pools to provide habitat. The overall quality of this segment was estimated to be fair, primarily based on the available substrate, which is conducive to benthic invertebrate productivity, the pool/riffle ratio, and streamside cover.

One location on Spring Branch (SB-2) was sampled for fish. Green sunfish ranged in length from 56 mm to 155 mm representing four age classes: age class I for the 55 to 64 mm fish, age class II for the 75 mm to 99 mm fish, age class III for the 105 mm to 134 mm fish, and age class V for the 155 mm to 159 mm fish. Fish age classes were determined by comparing on-site length-frequency histograms with existing species-specific age/length data (Carlander, 1977). Condition factors for all fish were greater than 1. Condition factors for green sunfish ranged from 1.7 to 2.1, while the condition factor for the individual yellow bullhead was 1.2.

5.6.4.2 Willow Creek

Willow Creek provides sporadic aquatic habitat throughout its length from a point immediately downstream of WC-1 to its confluence with the Spring River (Figures 2-1 and 2-2; Section 2.0). During the Fall 1991 field investigation, only one of six locations accessed on Willow Creek had noticeable flow. This location was in the area of station WC-2 where flow was estimated at < 1.0 cfs. This area had been observed by Dames & Moore personnel to be dry approximately 60 days before the Fall sampling effort. Based on these observations, Willow Creek was divided into three distinct segments: (1) segment WC-A, which extends from the Spring River upstream for approximately 1.5 stream miles; (2) segment WC-B, which begins approximately 1.5 miles upstream from the Spring River and extends another approximately 2.5 stream miles; and (3) segment WC-C, which begins at a point approximately four stream miles upstream of the Spring River and includes the remaining stream length.

The lower reach of Willow Creek (WC-A) receives recharge from limestone bedrock and appears to provide the best opportunity for establishment of a fishery based on the depth of water and apparent permanency of stream channel inundation. As previously mentioned, this segment of Willow Creek had no observable stream flow. The study site bottom substrate consisted of less than 10 percent rubble or gravel, which was at least 75 percent surrounded by fines or sediment. The channel lacked evidence of recent scouring, as it contained heavy deposits of fine material with a portion of the pools partially filled with silt. The pool/riffle ratio was greater than 25 with occasional riffles or bends providing habitat.

Bank failure or erosion was not evident, and over 80 percent of the stream bank was covered by vegetation or boulders. The streamside cover consisted primarily of shrubs with occasional trees and grasses. This segment provides poor-to-fair habitat based on bottom substrate/available cover, embeddedness, stream flow, channel alteration, and the pool-to-riffle ratio, all of which received a poor rating. In addition, the poor quality of the substrate may limit the productivity of benthic invertebrates (food organisms).

Three year classes were observed for the green sunfish collected at sampling station WC-3. These year classes probably represent age class I for the 55 to 84 mm fish (10 fish), indicating natural reproduction, age III for the 115 to 134 mm fish (three fish), and age V for the 160 to 179 mm fish (six fish). Condition factors calculated for fish sampled at WC-3 ranged from 1.8 to 3.4 for green sunfish, from 1.6 to 1.9 for longear sunfish, from 2.1 to 2.6 for pumpkinseed, while factors for the individual warmouth and spotted sucker sampled were 1.9 and 1.1. All fish appeared healthy with no indications of external parasites or physical or chemical stress.

The only other location on Willow Creek considered for habitat assessment was a site between WC-1 and WC-2. This site, which is designated as WC-1a on Figure 2-1, is generally described as an artificially created impoundment (no stream flow) providing adequate habitat. The habitat was created when an earthen-berm stream crossing was established (no culvert). Field personnel indicated that this stream-crossing was not present 60 days earlier.

The sampling of WC-1a resulted in a total of 56 fish including 36 bluegill, five green sunfish, five warmouth, three redear sunfish, three largemouth bass, one white crappie, one spotted sucker, one brook silverside, and one shiner. Bluegill ranged from 55 to 159 mm with possibly four year classes indicated. The year classes appear to include age 0 for the 55 to 84 mm fish, age I for the 90 to 114 mm fish, age II for the 115 to 134 mm fish, and age VI for the 150 to 159 mm fish. Again, species-specific age classes were determined by comparing on-site length-frequency histograms with existing species-specific age/length data

(Carlander, 1977). The reason that so many fish were present in the recently formed pond in Willow Creek at WC-1a is not readily known. One possible explanation is that since Willow Creek, both upstream and downstream from this pond, was lacking suitable flow to support fish, the fish are likely to have moved downstream during periods of reduced flow and became isolated in the pond.

The absence of fish at sampling station WC-2 was not surprising since this segment was dry 60 days prior to aquatic sampling. However, KDHE (1978) collected benthic organisms near this location and fish had been observed by field personnel when Willow Creek was flowing at WC-2.

5.6.4.3 Tar Creek

Tar Creek (within the Treece subsite) was divided into three segments based on stream channel characteristics (Figures 2-1 and 2-3; Section 2.0). These segments were labeled TC-A, which extends from the Oklahoma/Kansas state line upstream to a point near surface water monitoring station TC-2; TC-B, which extends from surface water monitoring station TC-2 upstream for approximately 0.75 stream mile; and TC-C, which covers the remaining upstream segment of Tar Creek.

During the Fall assessment, the lower reaches of Tar Creek (Segment TC-A) were observed to provide sporadic ponded water lacking flow. A 100-meter segment of Tar Creek immediately upstream from TC-3 was selected as a study site representative of lower Tar Creek. The water depth was approximately 1.5 meters (maximum) and was created by the partial damming effect of the road and culvert immediately downstream. The substrate consisted primarily of sand or silt with less than 10 percent rubble, gravel, or other stable substrate and, where larger particles existed, over 75 percent of the particles were surrounded by fine sediment. As with other areas observed in the Tar Creek system, this study site exhibited no flowing water. Channel alterations consisted of heavy deposits of fine materials. The pool/riffle ratio of the study site was essentially zero, because of the lack of any identifiable riffles due to lack of flow. The stream banks were moderately stable with only small areas of erosion within the study site. Over 80 percent of the stream bank was covered by vegetation, which was dominated by tall grasses. This segment provides only limited aquatic habitat primarily based on the stability of bank vegetation and streamside cover, which provides cover for fish. Conversely, the poor quality of the substrate parameters may limit benthic invertebrates (food-base organisms).

Segments TC-B and TC-C of Tar Creek were accessed near surface-water sampling locations TC-2 and TC-1, respectively. The stream channel in the area of TC-1 was dry with terrestrial vegetation established throughout the channel. The stream channel within the TC-2 area contained some small pools; however, no flowing water was observed and there was no observable evidence of conditions capable of supporting fishery resources. Aquatic habitat within these segments is unsuitable based on the absence of water within the stream channels.

Fish collected at sampling location TC-3 did not exhibit any overt signs of physical stress. Condition factors for fish greater than 20 mm in length ranged from 1.7 to 1.9 indicating good condition. All fish appeared healthy with no evidence of external parasites or stress. The fact that the fish inhabiting TC-3 are in good condition despite the limiting physical/habitat conditions probably reflects the fact that the fish have spent time in all segments of Tar Creek. As the stream dries up, fish are forced to migrate downstream. The fish sampled from TC-3 reflect the compression of upstream habitats during dry conditions.

5.6.4.4 Subsite Ponds

Four ponds within the Treece subsite (TP-3, TP-5, TP-6, and TP-7) and two ponds within the Baxter Springs subsite (BP-3 and BP-4) were sampled for fish tissue and to characterize community composition. All six ponds were inhabited by fish representing several age classes.

Treece subsite tailings pond TP-6 sampling resulted in a total of 52 fish consisting primarily of green sunfish (47) and unidentified sunfish (5). Eighty-five percent of the green sunfish collected were 55 to 89 mm in total length indicating a strong age I year class. Other year classes evidenced were age II (four fish) and age III (one fish).

Sampling of pond TP-7 resulted in a total of 78 fish representing five species. A majority (60 percent) of the fish consisted of bluegill. Other species included largemouth bass (6 percent), black bullhead (1 percent), warmouth (5 percent), and green sunfish (1 percent). Unidentifiable small sunfish comprised approximately 26 percent of the sample. The identified bluegill component of the sample exhibited a strong frequency of occurrence at the 40 to 69 mm length range indicating a strong age 0 year class. Other age classes represented were age I (six fish), age II (one fish), and age III (one fish).

TP-5 sampling resulted in a total of 85 green sunfish ranging in length from 29 to 87 mm. It should be noted that this pond was sampled using minnow traps only. The frequency of occurrence of these various lengths indicates possibly two age classes (age 0 and I) and strong natural reproduction. The apparent absence of additional age classes (i.e., age III and older) could also suggest premature mortality as a result of severe water level fluctuations and/or toxicity.

The Brewster Pond (BP-3) sampling results indicated an established population of green sunfish (77) ranging in length from 40 to 84 mm. Green sunfish in this size range (ages 0 and I) would provide forage for the white crappie inhabiting the Brewster Pond: three white crappie (350 to 370 mm) were collected. Black bullhead were also collected from the Brewster Pond.

BP-4 sampling results indicated strong naturally reproducing populations of green sunfish and bluegill. The length distribution of green sunfish indicates possibly two age classes; age 0 for fish of 40 to 44 mm and age I for fish of 50 to 89 mm. Two age classes may be represented for bluegill; age 0 for fish of 35 to 49 mm and age I for fish of 50 to 79 mm. Additional species collected included the black bullhead and larger green sunfish and bluegill.

In pond TP-7 bluegills sampled exhibited a strong frequency of occurrence at the 40 to 69 mm length range, indicating a strong age 0 year class and natural reproduction. Other age classes represented were age I (six fish), age II (one fish), and age III (one fish). Condition factors calculated for fish sampled in TP-7 ranged from 1.0 to 1.4 for largemouth bass, 1.9 to 2.3 for warmouth, 1.5 to 2.5 for bluegill, while the individual condition factors for the single black bullhead and green sunfish were 1.4 and 1.6, respectively. All fish appeared healthy with no evidence of external parasites or physical or chemical stress.

Age class data available for fish collected from subsite ponds appear to be limited to fish aged 3 years or younger, whereas some fish in the streams appear to be to as much as 6 years old. The differences in age composition between ponds and streams is most likely due to sampling methods and not necessarily to toxic effects from metals loads. Ponds were sampled by a combination of gill nets, backpack electrofishing along the shoreline, and minnow traps. The minnow traps were essentially the most productive method used, thus skewing the samples toward the smaller fish. Streams were sampled by electrofishing only, which provided more of an opportunity to collect a greater variety of fish representing different age classes and sizes.

6.0 SELECTION OF ECOLOGICAL ENDPOINTS

Ecological endpoints are characteristics of an ecological system that may be affected by site-related metals and as such epitomize the actual environmental values to be protected. Meaningful endpoints are those that characterize the relationship between contaminant levels (environmental concentrations) and potential adverse effects. Assessment endpoints are formal expressions of the actual environmental parameter to be protected, while measurement endpoints are a measurable or quantifiable characteristic that can be directly related to an assessment endpoint. For example, the assessment endpoint might be *could a significant reduction in population of a given species occur?* and the measurement endpoint might be to compare measured or estimated dose levels to toxicological data available in the literature for the same or a similar species to determine if frequent or gross mortality would be expected. Measurement endpoints can be measured in the field or laboratory (e.g., relative abundance measures) or can be summaries of relevant data reported in the scientific literature (e.g., LC_{50} values). Thus, measurement endpoints provide the means by which risk assessors can determine if ecological receptors have been significantly affected. This section discusses the ecological endpoints that were used to determine if adverse ecological impacts are possible for populations inhabiting the Baxter Springs/Treece subsites.

Ecological endpoints for individuals can include changes in typical death, growth, and fecundity rates or changes in tissue concentrations and behavior. Endpoints for populations include alterations in occurrence, abundance, behavior, reproductive performance, and age/size class structure. Community-level endpoints can be assessed by evaluating the numbers of species as well as species diversity and relative abundance indices. Ecosystems endpoints, as a whole, include biomass, productivity, or nutrient dynamics. Table 6-1 outlines specific ecological endpoints that were used to evaluate potential adverse effects associated with contamination at the Baxter Springs/Treece subsites. These endpoints are loosely divided into two categories: community-level and population-level endpoints. Community, or structural, endpoints were selected in an attempt to measure or assess changes in community/ecosystem structure or function. Population-level endpoints, which are

Table 6-1. Assessment and Measurement Endpoints Used to Evaluate Potential Adverse Effects to Ecological Receptors

	Assessment Endpoints		Measurement Endpoints
1.	Could a significant reduction in population of key aquatic and terrestrial species occur?	1.	Compare estimated or measured dose levels or exposure concentrations to toxicity reference values for the same or similar species. Toxicity reference values are designed to reflect potential reproductive, behavioral, and developmental effects, since these effects could influence population stability. Evaluate field data (e.g., fish condition factors and distance from mine-related disturbances for terrestrial receptors) as another means of assessing potential chronic effects.
2.	Could intake of metals result in chronic toxic effects in terrestrial or aquatic populations?	2.	Compare estimated dose levels with chronic toxicity reference values available in the literature for the same or similar species.
3.	Are gross signs of acute toxicity present?	3.	Individuals sampled have distinguishable signs of gross morbidity/stress.
4.	Do sufficient prey exist to support higher-level organisms?	4.	Determine if gross or frequent mortality of invertebrate aquatic organisms, which serve as the food base for many fish species, of the white-footed mouse, which is the food base for higher tropic level terrestrial predators, such as the raptors and the mink, and fish, which serve as the food base for omnivorous terrestrial predators, such as the mink, is expected by comparing exposure point concentrations or dose estimates with toxicological data available in the literature for the key species of concern.
4.	Has community structure been obviously impacted?	4.	Determine if specific areas are void of vegetation, aquatic, or terrestrial species. Determine if site community structure differs substantially from reference site community structure due to mining-related versus habitat-related differences.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	Types and numbers of plants and animals observed on-site vary significantly from the types and numbers expected to occur on-site as determined by biosurvey data and literature reports.
6.	Is water quality in on-site streams insufficient to support a diverse natural aquatic community?	6.	Compare site-specific water quality data to chronic toxicity reference values for species known or expected to occur in on-site water bodies.
7.	Have significant habitat modifications occurred?	7.	Use biosurvey results to compare on-site habitat to nearby undisturbed (reference) habitat.

Table 6-1. Assessment and Measurement Endpoints Used to Evaluate Potential Adverse Effects to Ecological Receptors (Concluded)

	Assessment Endpoints		Measurement Endpoints
8.	Are significant reductions in reproductive fitness of key terrestrial and aquatic species probable?	8.	Compare estimated or measured dose levels to toxicity reference values for reproductive effects obtained for the same or a similar species.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	Evaluate the probability of broad toxic effects exists as determined from a comparison of chronic toxicity reference values for the same or similar species to dose estimates.
10.	Are soils phytotoxic?	10.	Comparison of minimum and median toxic concentrations reported in the literature to measured levels of metals in subsite soils.

typically more useful in an ERA (Suter, 1990), attempt to define impacts at lower levels (i.e., population or organismal).

Separate endpoints were defined for aquatic and terrestrial populations, since they have different modes of exposure to and contact with contaminated media. Quantitative ecological endpoints for aquatic receptors are primarily directed toward fish, as they serve as an integrator of a variety of possible impacts. The reasoning for focusing on higher-level organisms (i.e., fish) was that if aquatic invertebrates and/or periphyton were sufficiently affected from exposure to site-related metals, then those effects would be passed onto fish, since fish rely on lower-level organisms as their primary food base. In other words, a viable, self-sustaining benthic population must exist to support an associated fish population. Although this ERA focused primarily on potential effects to fish, the possibility that benthic organisms could also be adversely affected was also evaluated. The main difference is that benthic populations were not sampled during the RI, since the original intent of the RI aquatic sampling program was to collect fish samples and to evaluate food chain effects.

Similarly, ecological endpoints for terrestrial receptors were intended to evaluate potential adverse effects on higher-level organisms (predators) and to evaluate food chain effects. Evaluating food chain effects involves assessing whether effects on one population could cause effects on other populations that either feed on the first population or are prey for another population. The ecological endpoints outlined in Table 6-1 were designed to protect populations and to ensure the long-term integrity of the ecosystem. In this context, loss of some individuals is acceptable if there is a reasonable assurance that the entire population will not be adversely impacted. This approach is supported by Barnthouse and Suter (1986) who state that "ecological risk assessments used in decision making should be based, to the greatest extent possible, on objective estimates of ecological damage (e.g., probabilities of <u>population</u> extinction or reductions in abundance of plants and animals)."

These endpoints provide a benchmark for comparative purposes. Ecologic endpoints are only relative guidelines for prioritizing potential effects and should not be utilized as rigid standards (personal communication, EPA Region VIII, Chief Toxicologist, Chris Weis, July, 1992). Hence, no one ecological endpoint can or should be used to determine if adverse effects to exposed populations are likely. Both results of field surveys and the toxicity assessment represent classes of endpoints that were useful in assessing potential effects due to metals toxicity. The validity/strength of the ecological endpoints as they were used in the ERA is discussed in the Uncertainty Section (Section 9.0). Tables 9-1 through 9-7 discuss the specific field and laboratory data used to evaluate the ecological endpoints aquatic and terrestrial receptors. No one ecological endpoint was necessarily given more weight or was deemed more valid than another per se, but data corresponding to all endpoints were evaluated to yield an accurate picture of potential impacts on the ecosystem.

7.0 TOXICITY ASSESSMENT

Toxicity assessment evaluates the nature and extent of adverse effects from exposure to site-related chemicals. It consists of a hazard evaluation and a dose-response assessment. The hazard evaluation involves a comprehensive review of toxicity data for multiple species to identify the severity of toxic properties associated with the COCs. Once the potential toxicity of a chemical has been established, the next step is to determine the amount of chemical exposure that may result in adverse ecological effects. Thus, the toxicity assessment evaluates the increased likelihood of adverse ecological effects as a result of exposure to site-related metals. The approach used in this ERA was to evaluate available toxicological data and to estimate <u>chronic</u> toxicity values for ecological receptors. Estimating chronic versus acute effects was considered appropriate in this case for these reasons:

- Since environmental receptors have been exposed to site-related metals for more than 100 years, acute effects that receptors may have experienced have probably already occurred.
- Since no recent continuous spills or releases have occurred, current receptors have most likely adapted to their long-term exposure to site-related metals.
- Chronic effects are a better measure of long-term impacts than acute effects.

Sources of the toxicity values used in this ERA (to evaluate potential adverse effects to plants and terrestrial and aquatic organisms) include Fish and Wildlife Service (FWS) Contaminant Hazard Series Synoptic Reviews, the Agency for Toxic Substance and Disease Registry (ATSDR) Toxicological Profiles, EPA Health Effects Assessment and Water Quality Criteria documents, and other current toxicological literature. Specific references are listed in the various data tables included in this section.

The first step in the toxicity assessment is to calculate toxicity reference values (TRVs). A TRV is an exposure estimate for a receptor group, including sensitive subgroups, establishing levels of exposure not likely to cause appreciable deleterious effects from chronic exposure.

Typically, ecological risk assessment assumes that if the TRV is not exceeded, the species of interest will be protected (Suter *et al.*, 1983). The next step in the toxicity assessment process (which is described in detail in Section 8) is to use the derived TRVs in order to calculate toxicity quotients (TQs) for all key receptors of concern. The calculated TQs represent one commonly used method of evaluating the possibility that aquatic and terrestrial populations onsite could be experiencing chronic effects that may not be readily observable in the field. Finally, the third component of the toxicity assessment discusses the uncertainty inherent in evaluating potential adverse effects of exposure to site-related metals in exposed populations. The discussion of uncertainty presented in Section 9.0, also analyzes other data, including literature reports and biosurvey results that may influence the toxicity of site-related metals. It should be emphasized that numerous site-specific factors affect the actual toxicity of metals. The factors that could ameliorate the toxicity of metals include adaptation, acclimation, and the bioavailability of metals in the natural environment. Conversely, other factors could increase the toxicity of site-related metals. For example, exposure to a mixture of COCs by key receptors could increase the toxicity of individual metals synergistically. Similarly, various environmental processes could alter the metal into a form that is more soluble and therefore more bioavailable.

7.1 DERIVATION OF TOXICITY REFERENCE VALUES

Human health toxicologists are routinely confronted with the problem of attempting to represent human toxicology (deriving health-protective toxicity values) based on animal studies. Ecotoxicologists must also address such concerns. Although extensive aquatic toxicity information is available, deriving meaningful toxicity reference values for terrestrial organisms is difficult because of the absence of data, especially for avian species. Nevertheless, it was necessary to develop appropriate toxicity values for avian, mammalian, and aquatic receptors in the ERA.

The derivation of TRVs depends on whether the desired toxicological data are available for the endpoints and species of concern. Typically, there is imperfect correlation between the concentration and the endpoint of concern so these data must be adjusted in some manner to account for scientific data gaps. For example, if toxicity data for the endpoint, species, and metal of concern are not available, data must be adjusted using various scientific methods available in the literature to determine the TRVs. Sections 7.2 through 7.5 outline the specific method used to derive TRVs for aquatic receptors, terrestrial vertebrates, plants, and T&E species, respectively.

7.2 TOXICITY REFERENCE VALUES FOR KEY AQUATIC RECEPTORS

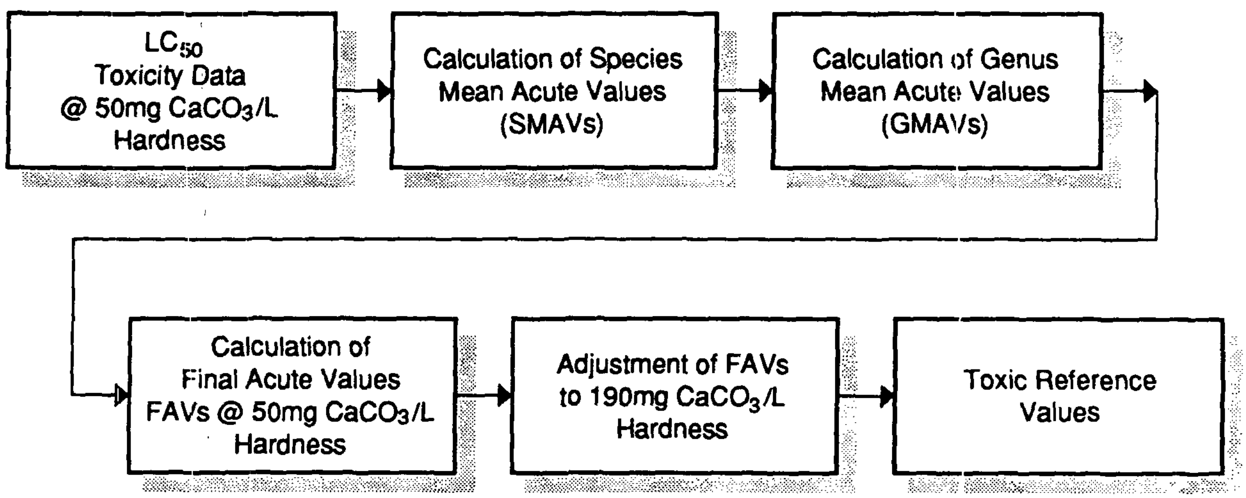
Toxicological benchmark data that specify an LC_{50}, the concentration causing death in 50 percent of exposed individuals, is the most frequently measured endpoint in aquatic toxicity testing and has been used by EPA to establish national Ambient Water Quality Criteria (AWQC). These benchmark data (LC_{50}) were used for a variety of species known or suspected to occur to derive chronic TRVs for all aquatic organisms (i.e., both vertebrates and invertebrates) in subsite surface water bodies. The term chronic refers to the duration of the test (not to the observed effect) and is defined as an exposure duration greater than 10% of the organism's lifespan (Suter *et al.*, 1987). The implication of developing chronic TRVs is that the toxicant or its effects accumulate in the organism over time, resulting in effects that are not observed following brief exposures (Suter *et al.*, 1987).

In this assessment, EPA-recommended procedures described in Stephan *et al.* (1985) were used to predict chronic TRVs from acute LC_{50} data for metals of concern for key aquatic organisms. To be protective of all potential receptors, EPA usually calculates national AWQC using toxicity test results for a variety of organisms, including the most sensitive species (defined as more susceptible to contaminant exposure than most others). As these sensitive species may or may not be present at the site, these criteria may be too stringent (or otherwise inappropriate) for application here. Not all the species used in the calculation of the national criteria are expected to occur in the ponds and intermittent streams within southeast Kansas. Therefore, and according to EPA approved procedures, the site-specific values more representative of the species occurring or potentially occurring within the site were developed.

It was not necessary to calculate site-specific TRVs for four COCs: copper, iron, manganese, and silver. The upper-bound concentration of copper and silver measured in all subsite surface waters (0.007 mg/L and 0.003 mg/L, respectively) is less than the national chronic AWQC for the protection of freshwater aquatic life of 0.012 mg/L and 4.1×10^{-3} mg/L, respectively, at 100 mg $CaCO_3$/L. Federal AWQC are not available for iron and manganese. Levels of manganese ranging from 1.5 mg/L to 1000 mg/L are reported to be tolerable to aquatic organisms. Since the upper-bound levels of manganese measured in surface water bodies do not exceed 0.9 mg/L, manganese is not expected to cause adverse impacts in exposed populations. Iron was not considered a COC for subsite streams since the average on-site concentrations did not significantly differ from upstream locations. Iron is a COC for ponds since any metal detected was conservatively considered a COC (see Section 4.3). Iron toxicity reference value is considered 1.0 mg/L. The diagram in Figure 7-1 depicts the steps involved in calculating the site-specific TRVs.

Site-specific TRVs were calculated using EPA methodology (Stephan *et al.*, 1985). EPA developed this method for deriving numerical values for the protection of aquatic organisms. These site-specific water quality values are based on toxicity data from a diverse range of species, including highly sensitive species known or suspected to occur in on-site waters. Sensitive species are those that cannot tolerate high concentrations of contaminants. In other words, they are likely to experience adverse effects sooner than more tolerant species. Federal AWQC are generally calculated by EPA using toxicity test results for a variety of species, including salmonids and cladocerans, that are generally quite sensitive to elevated metals concentrations. Only those species or genera considered representative of ephemeral streams of southeast Kansas were selected for the site-specific toxicity assessment. A listing of these species for each metal of concern is provided in Appendix A.

Various site-specific factors, such as water hardness and the sensitivity of organisms known or suspected to occur onsite, can strongly influence (usually lessen) the toxicity and bioavailability of some metals. EPA recognizes that water hardness ameliorates the toxic effects of many metals to varying degrees and provides a method of adjusting toxicity data to account



SITE SPECIFIC TRV CALCULATION FLOW CHART

DAMES & MOORE

Figure 7-1

for this fact. As a result, a standard Federal AWQC value may be more stringent than is necessary to protect aquatic life in a given surface water body, although there are AWQC hardness correction factors for those metals influenced by hardness. For example, subsite streams and ponds have, on the average, a hardness of 190 mg $CaCO_3$/L. The site-wide mean hardness of 190 mg $CaCO_3$/L was selected, since this is the geometric mean hardness of all on-site water bodies. Since hardness levels varied not only between water bodies but within a given water body, the geometric mean hardness for all on-site surface water bodies was used.

In determining the Federal AWQC, EPA evaluates numerous toxicity data representing varying hardness values. Since hardness has long been known to affect the toxicity of metals, Federal AWQC were adjusted to a hardness of 50 mg $CaCO_3$/L to facilitate comparison between metals with criteria on a common scale. As such, the Federal AWQC are intended to protect all aquatic organisms in all surface water bodies. However, in so doing, these criteria may actually be overly protective for some water bodies. In other words, the AWQC may be lower (more stringent) than is truly necessary to maintain viable aquatic life in a given water body. The basis for establishing the typically conservative AWQC to be more protective of aquatic life is to err on the side of conservatism.

The mean site-wide hardness is quite high relative to the AWQC promulgated by EPA, adjusted for a hardness of 50 mg $CaCO_3$/L. Hence, relatively high water hardness measured in on-site surface water bodies is likely to substantially ameliorate the toxicity of the metals of concern. Furthermore, subsite streams and ponds are inhabited or potentially inhabited by species known to be relatively tolerant of elevated metals concentrations (e.g., warm water species including centrarchids and ictalurids). Therefore, the use of AWQC based on more sensitive species and adjusted for a hardness of 50 mg $CaCO_3$/L would yield overly protective TRVs for the subsite streams and ponds. To calculate TRVs that more accurately reflect actual site conditions, toxicity data were used for species known or suspected to occur onsite given habitat limitations (i.e., ephemeral warm water streams). These data were then applied to the approach developed by EPA (Stephan *et al.*, 1985) to determine site-specific water quality values. The methods and data used to calculate TRVs for aquatic organisms are outlined in detail in Appendix A and summarized below.

The first step in the EPA approach for deriving site-specific TRVs is the calculation of a Species Mean Acute Value (SMAV) and a Genus Mean Acute Value (GMAV) for each metal of concern. EPA calculated the SMAVs (the geometric mean of all appropriate species acute values) and GMAVs (the geometric mean of all appropriate SMAVs within a particular genus) for the respective metals. The use of GMAVs provides a representation of those species within a given genera with respect to tolerance and sensitivity.

The metal-specific GMAVs were used by EPA to determined Final Acute Values (FAVs) for each metal. First, GMAVs were ranked from high (most tolerant) to low (most sensitive). Specifically, ranks were assigned to the GMAVs for a given metal ranging from "1" for the most sensitive genera (i.e., the genera with the lowest GMAV) up to "n" for the most tolerant genera (i.e., the genera with the highest GMAV), where "n" represents the number of GMAVs calculated for that metal. The cumulative probability for that metal was calculated as Rank/(n+1). The four GMAVs with a cumulative probability closest to 0.05 were selected to calculate the FAV for that metal. FAVs were then calculated using the four selected GMAVs and cumulative probabilities for each metal of concern according to the set of equations (7-1) in Appendix A.

This procedure was used to calculate site-specific FAVs; however, only those toxicity data available in the AWQC documents for organisms known or suspected to occur within the subsite streams and ponds were used. Hence, the GMAVs used to calculate the FAV were taken from the AWQC document for a given metal for warm water species potentially occurring in intermittent streams of southeastern Kansas. Species selection was based on data developed by KDWP (1992b,c), the Kansas Department of Health and Environment (KDHE) (1992), and results from the remedial investigation field efforts conducted in 1991. According to the KDWP (1992b,c), aquatic vertebrates most likely to occur in ephemeral streams of southeastern Kansas include yellow bullhead, black bullhead, green sunfish, black-striped topminnow, bullhead minnow, bluntnose minnow, red shiner, and mosquitofish. Species, in addition to those listed by the KDWP, that were collected during the 1991 field investigations include several species of centrarchids (pumpkinseed, bluegill, warmouth, and largemouth bass, and black crappie), and cyprinids (minnows). Benthic invertebrates were not sampled during RI field activities since fish were considered an integrator of possible impacts and represented the primary aquatic receptor of concern. However, benthic invertebrate data are available for a site near WC-2 on Willow Creek (KDHE, 1980). The aquatic invertebrate species collected were mayflies, stoneflies, aquatic beetles, flies, mosquitoes, midges, dragonflies, damselflies, and water bugs.

The GMAVs were then ranked from 1 to n (from highest sensitivity to lowest). Therefore, a new cumulative probability was calculated for each metal based on site-specific data. The four lowest GMAVs (most sensitive Genera) were selected to calculate the site-specific FAV using equation 7-1. Toxicity data presented in the AWQC documents not representative of species known or suspected to occur onsite were excluded from the calculations of site-specific FAVs.

Toxicity values for species known or suspected to occur onsite and the specific calculations and toxicity data used to derive site-specific FAVs are presented in Appendix A. The FAVs for each metal are summarized in Table 7-1. The FAVs, which at this point are based on the default hardness of 50 mg $CaCO_3$/L, are adjusted to reflect the mean hardness of all on-site waters (190 mg $CaCO_3$/L) using the metal-specific equations (7-2 and 7-3; Appendix A) obtained from the respective AWQC documents.

The FAV was then converted to a final chronic value (FCV), since the goal of the toxicity assessment is to derive chronic TRVs for all ecological receptors. FAVs were converted to FCVs using the final acute-to-chronic ratios calculated by EPA and included in the AWQC documents. An acute-to-chronic ratio was calculated by EPA for a given metal for each species for which both acute and chronic data were available from the same test. A geometric mean acute-to-chronic ratio was calculated for the subsite surface waters using the individual acute-to-chronic ratios (provided by EPA) for the species known or suspected to occur onsite (i.e., for

Table 7-1. **Adjusted FAVs, Chronic Values, and TRVs for Aquatic Receptors**

Metal[a]	Final Acute Value (FAV) (µg/L)	Pooled Slope[b]	ln (Y-Intercept) based on 50 mg $CaCO_3$/L	FAV Adjusted to a Hardness of 190 mg $CaCO_3$/L (µg/L)	Acute-to-Chronic Ratio[b]	Final Chronic Value (µg/L)	Toxicity Reference Value (mg/L)
Cadmium	34.451	1.128	-0.873	155	NA[d]	7.92	0.008
Mercury	17.004	NA[c]	NA[c]	17	3.73	4.56	0.005
Nickel	4049.879	0.846	5.000	12,530	17.99	696.48	0.7
Lead	256.152	1.273	0.566	1401	51.29	27.32	0.03
Zinc	1013.954	0.847	3.607	3142	2.21	1423.21	1.4
Iron	NA[e]	NA	NA	NA	NA	NA	1.0

[a] Copper, manganese, and silver were eliminated, since upper-bound concentrations of these metals measured in on-site surface water bodies did not exceed chronic AWQC or other values reported to be toxic to aquatic life (see Section 7.2).

[b] Source: Metal-specific EPA Ambient Water Quality Criteria documents.

[c] Hardness does not affect the toxicity of mercury; hence the FAV for mercury does not need to be adjusted for hardness.

[d] An acute-to-chronic ratio is not available for cadmium (USEPA, 1984a). See text for alternative calculations.

[e] A site-specific TRV for iron was not calculated, since the Federal AWQC value of 1.0 mg/L was sufficient.

the same species used to derive the FAV for that metal). A geometric versus arithmetic mean value was used in order to be consistent with the EPA's AWQC methodology.

The FCV was derived for each metal of concern (excluding cadmium) by dividing the FAV (adjusted for a hardness of 190 mg $CaCO_3$/L) by its geometric mean acute-to-chronic ratio. Acute-to-chronic ratios were not calculated for cadmium by EPA. Hence, an alternative method was used to derive an equivalent cadmium TRV for the subsites. The adjusted FAV listed in Table 7-1 (as calculated from the GMAVs for species known or suspected to occur onsite) and the predictive regression equations developed by Suter *et al.* (1987) and Suter (1986) were used to derive a site-specific TRV for cadmium based on the concept of Maximum Allowable Toxicant Concentration (MATC). The MATC is the standard chronic test endpoint for aquatic toxicity testing. It is the calculated or approximated threshold for statistically significant effects on growth, reproduction, or survival. The MATC is defined as the effects threshold at or below which adverse chronic effects are not expected to occur (Suter, 1992). The MATC is the geometric mean of the highest no-observed-effect concentration and the lowest concentration causing a statistically significant effect on growth, reproduction, or survival in a life-cycle toxicity test (Mount and Stephan, 1969) and was considered equivalent to the FCV derived using methodology in Stephan *et al.* (1985). The acute-to-chronic ratios used for each metal and their corresponding FCVs are listed in Table 7-1.

Regression equations that provide estimated MATCs have been developed for aquatic vertebrates (Suter *et al.*, 1987) and for aquatic invertebrates (Suter, 1986). The regression equation for aquatic vertebrates (Suter *et al.*, 1987) yielded a more conservative (lower) MATC than the equation for invertebrates (Suter, 1986), and was used to develop a MATC for cadmium that would be protective of all aquatic organisms. This adjusted MATC, which is listed in Table 7-1, was used as the TRV for cadmium.

In summary, the site-specific criteria for each metal of concern was calculated by following a step by step approach as described early in this section. This approach involved the use of several data evaluated and calculated by EPA. LC_{50} levels for certain species and genera were used by EPA to calculate species and Genus Mean Acute Values, the latter of which was used in this document to calculate Final Acute Values (FAVs) for each metal of concern. The FAVs were then adjusted for the site-specific surface water hardness of 190 mg $CaCO_3$/L to more accurately reflect the toxicity of the various metals considering hardness generally ameliorates metal toxicity. Final Chronic Values (FCVs) were then calculated for all metals, except cadmium, using the acute to chronic ratios provided in the respective ambient water quality criteria documents. In the case of cadmium, a Maximum Allowable Toxicant Concentration (MATC) was calculated according to Suter et al. (1987) and the resulting product was considered the FCV. The FCVs (converted to mg/L) were identified as the Toxicity Reference Values (TRVs).

7.3 TOXICITY REFERENCE VALUES FOR TERRESTRIAL RECEPTORS

For terrestrial wildlife, there is imperfect correlation between the species and endpoint (effect) measured and the species and endpoint of concern such that available toxicological data must be adjusted to account for these scientific data gaps. For example, toxicological data measuring a chronic effect in mink are not available, while data for other related species were available. Use of toxicity data for surrogate species (i.e., for species other than the species of concern) and for effects other than chronic effects introduces uncertainty into the toxicity assessment calculations. The magnitude of that uncertainty depends upon (1) the degree of taxonomic difference between the key and test species; (2) the conditions under which the toxicity data were established (e.g., test duration); and (3) the endpoint of interest (e.g., chronic LOAEL) and the endpoint measured (e.g., death). In human health risk assessment, these disparities are often dealt with by applying order-of-magnitude uncertainty factors (UFs). The application of multiple order-of-magnitude UFs has its historical roots in the calculation of "acceptable daily intakes" (ADIs) to determine safe levels of contaminants in food (Lewis *et al.*, 1990). For example, in human health risk assessment, an UF of 10 is usually applied to account for variation in sensitivities among individuals (i.e., to protect sensitive subpopulations). An additional UF of 10 is used to account for the interspecies variability between humans and other animals. A third factor of 10 is often applied to extrapolate a subchronic (or acute) value to a chronic value. Since the UFs are simultaneously applied, the presumptive assumption is that everything will go wrong every time at once. To illustrate this point, consider an ecologic example: the most sensitive life stage of the most sensitive species will be exposed to the most concentrated effluent at low-flow conditions while debilitated by stress (Barnthouse and Suter, 1986). Clearly, the aforementioned example represents an extremely conservative scenario that is unlikely to consistently occur in an ecologic system. Thus, under the original method, UFs were not "best estimates" of the true No-Observable-Adverse-Effect-Level (NOAELs) but were upper-bound estimates that yielded a highly conservative value.

In this assessment, uncertainties associated with using toxicological data were offset by applying various uncertainty factors to the laboratory or literature endpoint values to better approximate chronic effects with a margin of safety considered adequate to protect terrestrial populations using the equation:

$$\text{Toxicity Reference Value} = \frac{\text{Benchmark Value}}{(\text{UF1} \times \text{UF2})} \qquad (7\text{-}1)$$

where: UF1 is an adjustment factor to account for anticipated differences in the susceptibility between the key species of concern and the test population (i.e., to account for <u>interspecies</u> variability).

UF2 is used to extrapolate data from acute or subchronic studies to chronic toxicity estimates. This factor is predicated on the expectation that continued exposure at a given dose rate will produce increasing effects that may not be evident after subchronic exposure. Again, subchronic exposures are defined as those lasting less than 10 percent of the animal's lifetime (Suter *et al.*, 1987).

Thus, subchronic or acute data for surrogate species were used only after they had been adjusted (through the application of specific uncertainty factors) to reflect chronic effects for the key species of concern.

Even though cobalt and manganese were identified as COCs in near-pile soils, copper and mercury were COCs in air, and copper, iron, manganese, mercury, nickel, and silver were identified as COCs in subsite ponds in addition to cadmium, lead, and zinc, terrestrial receptors were assumed to be exposed to cadmium lead, and zinc only. At the time mice and fish tissue samples were collected, cadmium, lead, and zinc were suspected to be the most prevalent and potentially toxic site-related metals. Hence, tissue samples were analyzed and dose estimates for terrestrial receptors are presented for these three metals only. The uncertainties associated with excluding other COCs is discussed in Section 9.0.

Lowest-Observable-Adverse-Effect-Level (LOAELs), or the lowest concentration that causes a significant adverse effect in test organisms, available for various avian and mammalian species used as surrogates for raptors and the mink are presented in Tables 7-2 and 7-3, respectively. Data on the toxicity of cadmium, lead, and zinc to the red-tailed hawk and the barred owl was not available, so data for surrogate avian species, including quail, chickens, doves, pigeon, ducks, kestrels, chicks, and mallards, had to be used (Table 7-2). The reported LOAELs based on oral administration varied by a factor of three for zinc to a factor of 370 for cadmium. These data show that there is a tremendous amount of variability in the response of various laboratory avian species following exposure to lead. Similarly, studies evaluating the toxicity of metals to the mink were not available, so data on the toxicity of cadmium, lead, and zinc to rats, mice, rabbits, sheep, pigs, and dogs were used (Table 7-3). For mammals, published LOAELs varied by a factor of 470 for cadmium, by a factor of 33 for lead, and by a factor of 7 for zinc. These data show that there is a tremendous amount of variability in the response of various mammalian and avian species following exposure to some metals.

Table 7-2. Toxicity of Metals to Avian Species[a]

Test Organism	LOAEL[b] (mg/kg-day)	Number of individuals tested	Duration of Exposure	Effect(s) Observed	Reference
CADMIUM					
Japanese quail (young) (Coturinex coturinex)	75	8	42 days	Ventricular hypertrophy; anemia; testicular and bone marrow hypoplasia	Richardson et al., 1974
Chickens	75	NA[c]	NA	Reduced body weight; anemia	Freeland and Cousins, 1973
1-week-old Wood Ducks (Aix spousa)	100	7	90 days	Severe kidney lesions	Mayack et al., 1981
Mallard ducklings (Anas platyrhynchos)	14.6[d]	36	90 days	Significantly lower hemoglobin levels; sever kidney degeneration	Cain et al., 1983
1-year-old Mallards (Anas platyrhynchos)	210	40	90 days	Decreased kidney weight; suppressed egg production	White et al., 1978; White and Finley, 1978
Hens	50	NA	NA	Reduced egg production	Anke et al., 1970
LEAD					
Mallards (Anas platyrhynchos)	5	NA	21 days	Depressed blood ALAD activity;	Demay et al., 1982
1-year-old Mallard Drakes (Anas platyrhynchos)	21.6 (as lead acetate)	20	12 weeks	40% reduction in blood ALAD activity	Finley et al., 1976
Chickens	300	NA	NA	Decreased growth	Demay et al., 1982
Mallard (Anas platyrhynchos)	151	NA	NA	Depressed blood ALAD activity; some deaths	Dieter and Finley, 1978
Mallard (Anas platyrhynchos)	25 (as lead nitrate)	NA	12 weeks	Depressed blood ALAD activity	Finley et al., 1976
Japanese quail (Coturinex japonica)	25 (as tetraethyl lead)	NA	NA	Death	Eisler, 1988

Table 7-2. Toxicity of Metals to Avian Species (Continued)

Test Organism	LOAEL[b] (mg/kg-day)	Number of individuals tested	Duration of Exposure	Effect(s) Observed	Reference
American kestrel (Falco sparverius)	50 (as lead powder)	NA	5 months	Blood ALAD reduced 80%	Franson et al., 1983 (cited in Eisler, 1988)
American kestrel nestlings (Falco sparverius)	25[e]	NA	10 days	Reduced blood ALAD	Hoffman et al., 1984 (cited in Eisler, 1988)
Chickens (Gallus sp.)	1850 (as lead acetate)	NA	4 weeks	Growth rate suppressed 47%	Franson and Custer, 1982 (cited in Eisler, 1988)
Ringed Turtle Dove (Streptopelia risoria)	75 (as lead acetate)	NA	single oral dose	Some deaths; kidney damage	Kendall and Scanlon, 1982 (cited in Eisler, 1988)
ZINC					
7-week-old Mallard Ducklings (Anas platyrhynchos)	3000	50	45 days	Decrease in gonadal weight; paralysis; anemia	Gasaway and Buss, 1972
2-week-old Chicks	1000[f] (as zinc oxide)	88	28 days	Pancreatic lesions	Dewar et al., 1983
3-day-old Peking Ducklings	2500 (as zinc sulfate)	130	56 days	Cellular atrophy; pancreatic lesions	Kazacos and van Fleet, 1989

[a] These data were used to derive toxicity reference values for the red-tailed hawk and the barred owl.

[b] LOAEL = lowest-Observable-Adverse-Effect-Level. All doses were administered orally.

[c] NA = data not available

[d] The LOAEL for the Mallard ducklings was selected since (1) they represent an early (and probably more sensitive) life stage and (2) the associated test was of the longest duration even though the mallard (an Anseriform) is not as closely related to the raptors (Falconiformes) as the quail (a Galiform).

[e] The LOAEL for the American kestrel was selected since (1) the kestrel belongs to the same Order as the raptors (both are Falconiformes) and (2) the test species represents an early (and probably more sensitive) life stage (nestling).

[f] The LOAEL for the chicks was selected since (1) all three values are for young individuals that probably represent a more sensitive life stage, and (2) chickens (Galiformes) are more closely related to raptors than the mallard or the duck (Anseriformes).

Table 7-3. Toxicity of Metals to Terrestrial Mammals[a]

Test Organism	Duration of Exposure	LOAEL[b] (mg/kg-day)	Reference
CADMIUM			
Mice	4 weeks	0.95[c]	Ogoshi et al., 1989
Rats	82 - 90 weeks	1.8	Fingerle et al., 1982
Rats	2 months	31	Wilson et al., 1941
Rats	NA	10	Schroeder and Mitchener, 1971
Rabbits	6 months	15.5 (in drinking water)	Stowe et al., 1972
Lamb	NA	60	Doyle et al., 1974
Pigs	NA	450	Cousins et al., 1973
LEAD			
Dog	180 days	3[d]	Clark, 1979
Dog	12 weeks	100	Clark, 1979
Dog	12 weeks	100 (as lead acetate)	Maxfield et al., 1972
Rabbit	N/A[d]	>5	Barth et al., 1973
Pregnant sheep	59 days	9 (as lead acetate)	James et al., 1966
Rat	NA	25 (as soluble lead salt)	Schroeder and Mitchener, 1971
Rat	20 months	5	Kopp et al., 1980a,b
ZINC			
Mice	5 - 14 months	38[c]	Aughley et al., 1977
Rats	NA[e]	50	Fahim et al., 1975 (cited in Venugopal and Luckey, 1978
Rats	73 days	250	Kinnamon, 1963

[a] These data were used to derive toxicity reference values for the mink.

[b] LOAEL = Lowest-Observable-Adverse-Effect-Level.

[c] This LOAEL was selected since both rats and mice belong to the order Rodentia (i.e., one species is not more closely related than the other), and the mice value was lower than the rat value.

[d] The LOAEL for the dog was selected since dogs are more closely related to the mink than rabbits or rats (both the dog and the mink belong to the Order Carnivora), and this value was the lowest value.

[e] NA = data not available

The best study was selected using the following criteria suggested by Dr. Glen Suter of Oak Ridge National Laboratory (personal communication, December 2, 1992). Ideally, tests involving animals from several different life stages or tests conducted on early life stages would be used. If these data are not available (e.g., only tests done on adults are available), then tests of long duration where the route of administration is via the food and where the test species is closely related to the key species of concern should be selected. Species that are closely related (taxonomically) can serve as surrogates for those species expected to occur on site if insufficient toxicity data exists for those key species. It is, therefore, possible to select one or more of the commonly tested species as surrogates for species found at a site during the assessment of toxicity (USEPA, 1992). The intent of surrogate species selection is to minimize physiological differences between species potentially occurring on site and those from which there is an abundance of toxicity data. Hence, it was not necessary to arbitrarily select the lowest benchmark value available.

Using these guidelines, the data noted (see footnotes) in Tables 7-2 and 7-3 were used to calculate TRVs (Table 7-4) for the raptors and the mink, respectively. If not specifically stated, all LOAELs were assumed to be based on oral administration, since this is the standard route of administration of terrestrial toxicity tests. For the raptors, the LOAEL for mallard ducklings (14.6 mg/kg-day) was selected from the available cadmium data since (1) the associated test was of the longest duration, and (2) ducklings represent an early and probably more sensitive life stage even though the mallard (an Anseriform) is not as closely related to the raptors (Falconiformes) as the quail (Galiformes). Similarly, the LOAEL for the American Kestrel or sparrow hawk nestlings (25 mg/kg-day) was selected from the available lead data since (1) the test species represents an early life stage, and (2) the kestrel and the eagle are both Falconiformes. Finally, the LOAEL for the chicks (1000 mg/kg-day) was selected from the available zinc toxicity data since (1) the two-week old chicks probably represent a sensitive life stage, and (2) chickens (Galiformes or grouse-like birds) are more closely related to raptors than the mallard (Anseriform). For the mink, the LOAEL for mice was selected over the LOAEL for rats for cadmium (0.95 mg/kg-day) and zinc (38 mg/kg-day) since (1) both rats and mice belong to the order Rodentia (i.e., one species is not more closely related to the mink than the

Table 7-4. Toxicity Reference Values for Terrestrial Receptors

Metal	Literature Value Selected (mg/kg-day)	Uncertainty Factor	Toxicity Reference Value (mg/kg-day)
RED-TAILED HAWK			
Cadmium	14.6	9[a]	1.6
Lead	25	9	8.3
Zinc	1000	9	111
BARRED OWL			
Cadmium	14.6	9	1.6
Lead	25	9	8.3
Zinc	1000	9	111
MINK			
Cadmium	0.95	9	0.1
Lead	3	9	0.3
Zinc	38	9	4.2

[a] UF1 (interspecies extrapolation) = 3 and UF2 (acute-to-chronic extrapolation) = 3, since the test duration was not longer than 10% of the animal's lifetime.

belong to the order Rodentia (i.e., one species is not more closely related to the mink than the other), and (2) the mice value was lower (more conservative) than the rat value. The LOAEL for the dog (3 mg/kg-day) was selected from the available toxicity data for lead since (1) dogs are more closely related to the mink than rabbits or rats (both the dog and the mink are Carnivores), and (2) the test for the dog was of the longest duration.

Once the single "best" toxicity data had been selected, it was necessary to adjust these values by applying UFs to reflect the key species of concern and to reflect chronic exposures (i.e., to apply an interspecies and an acute-to-chronic extrapolation factor, if necessary). Lewis *et al.* (1990) evaluated numerous studies regarding toxicological extrapolations from animals to humans in an effort to determine more representative UFs. Although the focus of Lewis *et al.* (1990) is on animal-to-human extrapolations, their theories and conclusions apply equally to animal-to-animal extrapolations, since the terrestrial species evaluated in ERAs are often more closely related to the species for which toxicological data are available than humans. TRVs for terrestrial receptors were derived for each metal using Equation 7-1 and the following values for UF1 and UF2:

UF1: Lewis *et al.* (1990) reported that UF1 may range from <1 to 10, with values of one to three being most likely. A value of 10 indicates a high susceptibility in the key species versus the test population, while a value of less than or equal to one indicates the test organism was more sensitive than the key receptor species. A value of 3 was used for UF1.

UF2: Weil and McCollister (1963) found that in generating a full chronic benchmark value from a subchronic one, a downward adjustment of the subchronic value by a factor of three was adequate 73 percent of the time. Extrapolating subchronic to chronic data 73 percent of the time is considered reasonable for this assessment, since the goal of this ERA is to protect populations versus individuals. Therefore, a value of 3 was used to extrapolate subchronic data to estimate corresponding values for chronic exposures.

In addition, since the data upon which these extrapolations are based are lognormally distributed, and three is a mid-point of a lognormal distribution (Dourson and Stara, 1983) the medial value of 3 was selected. The chronic TRVs for raptors and the mink are presented in Table 7-4.

7.4 TOXICITY REFERENCE VALUES FOR AMPHIBIANS

Potential adverse effects to T&E species were evaluated since the northern spring peeper has designated critical habitat within mine-impacted areas within the subsites and Willow Creek and Spring Branch flow through designated critical habitat for several other T&E species, primarily amphibians. Toxicological data for the specific T&E species of interest (i.e., the species that could occur onsite or the species whose critical habitat could be affected by site-related metals) were not available, so data for surrogate amphibians were used. Table 7-5 lists the toxicity data for amphibians that were used to quantify potential adverse impacts to subsite T&E species. As for terrestrial receptors, it was necessary to apply an interspecies and acute-to-chronic extrapolation factors to the available benchmark data to obtain chronic TRVs for the species of concern.

Although more conservative UFs are typically used for T&E species, the T&E species of concern are state-listed not Federally-listed. The primary reason that the State of Kansas listed these amphibians is to protect the westernmost range where they could occur, which happens to extend into Cherokee County. Since their population numbers are not threatened, but their habitat is limited, UF1 and UF2 were each set at three, which was considered sufficiently conservative in this case.

7.5 PHYTOTOXICITY OF SOILS

It is not uncommon for metals in soil to be phytotoxic at concentrations less than those that may result in human health or other ecological effects. Plants were assumed to be exposed to metals in soil, as reported on a total metals basis. This is standard practice, since total elemental concentration is the form reported by standard EPA-approved analytical methods. It should be noted, however, that the given concentration of total metal will not be equally available to all plants growing in different soils. The actual phytotoxicity of soils depends on the pH and other physiochemical properties of the soil, the chemical form of the metal, and the sensitivity of local plant species.

Table 7-5. Toxicity Reference Values for Amphibians

Amphibian Species	Life Stage	Exposure Duration	Range of LOAELs[a] (mg/L)	Mean LOAEL (mg/L)	Uncertainty Factor Applied	Toxicity Reference Value (mg/L)
CADMIUM						
Xenopus laevis	tadpole	15-75 hrs.	80 - 100	90	9[b]	10
COPPER						
Rana pipiens	tadpole	life	0.06 - 0.16	0.11	3[c]	0.04
LEAD						
Rana nigromaculata	eggs	to first cleavage	70	70	9[b]	7.8
MANGANESE						
Triturus cristatus carnifex	N/A[d]	19 - 23 weeks	5	5	3[c]	1.7
MERCURY						
Rana nigromaculata	tadpole	life	0.4 - 0.8	0.6	3[c]	0.2
ZINC						
Rana pipiens	adult	15 days	155	155	9[b]	17.2

[a] LOAEL = lowest observable adverse effect level

[b] UF1 (interspecies extrapolation) = 3, and UF2 (acute-to-chronic extrapolation) = 3 since the test duration was not longer than 10% of the animal's lifetime.

[c] UF1 (interspecies extrapolation) = 3, and UF2 (acute-to-chronic extrapolation) = 1 since the test duration was at least 10% of the animal's lifetime.

[d] Data not available.

REFERENCE: Harfenist et al., 1989.

The median concentration of metal in soil reported to be phytotoxic in either laboratory or field studies was obtained from the literature. These data, which are presented in Table 7-6, were used to derive TQs for plants. The data listed in Table 7-6 were compiled from numerous sources and represent effects ranging from reduced growth to severe injury and death of vascular plants. Use of crop yield reduction test data are the primary means of evaluating phytotoxicity of metals (i.e., LOAELs or NOAELs are not generally listed for test species but could be inferred from the available data). The phytotoxicity data used in the ERA are concentrations which researchers estimated would cause some reduction in crop yield (e.g., zinc at soil concentrations between 60 mg/kg and 960 mg/kg caused yield reduction in various agronomic species ranging between 26 and 98 percent). Since the mean value is more susceptible to the number of tests the researchers conducted and the metals concentration used, the median phytotoxic concentration was used. The mean value has the effect of skewing the data in the direction of most research results. For example, if the researchers conduct studies primarily with high metal concentrations resulting in high crop yield reductions, then the resulting mean metal concentration would be high and would not be a good indicator of the average or low toxicity value for that metal. Hence, the median value was used, as it was considered more representative of the range of data shown to be phytotoxic.

7.6 TOXICITY PROFILES FOR THE COCs

Table 7-7 presents general information and contaminant-specific discussion on toxic effects of the COCs evaluated in the ERA for the BS/T subsites. Three categories of effects are considered: phytotoxicity, toxicity to terrestrial receptors, and toxicity to aquatic organisms. The values presented in Table 7-7 are not concrete in that organisms and populations may adapt to higher concentrations without indications of adverse effects, and some organisms and certain life stages may be more susceptible than others.

Table 7-6. Toxicity of Metals to Plants

Metal	Median Phytotoxic Concentration[a] (mg/kg)
Cadmium	40
Cobalt	25*b*
Copper	20.3
Lead	250
Manganese	1500*b*
Zinc	240

[a] The phytotoxic data were compiled from numerous sources and represent effects ranging from reduced growth to severe injury and death of vascular plants. See Section 8.4 for additional information.

[b] Minimum value was used because a range of data and, hence, a median value was not available.

Table 7-7. Summary of Potential Toxic Effects of the Potential Chemicals of Concern for the Baxter Springs/Treece Subsites

Metal	Phytotoxic Effects	Toxic Effects to Terrestrial Organisms	Toxic Effects to Aquatic Organisms
Cadmium	Non-essential element for plants and may be phytotoxic at low levels (Friberg et al., 1971). Cadmium concentrations in plants of 130 ppm for leaves and 898 ppm in roots causes wilting and leaf burn (Adriano, 1986). Plant levels of 25 ppm can cause reduced productivity in some species (Kabata-Pendias and Pendias, 1984). A high cadmium-to-zinc ratio reduces available zinc and increases cadmium bioavailability. A synergistic relation exists between cadmium and lead.	Non-essential element for animals (Friberg et al., 1971). Cadmium ingestion of 50 µg/g for 10 days by rats caused anemia and renal injury (Friberg et al., 1971). Cadmium has a high toxicologic potential in mammals enhanced by its ability to bioaccumulate in mammalian tissues and to a poor or nonexistent homeostatic mechanism (Venugopal and Lucky, (1978). Renal damage is the classic symptom of chronic cadmium poisoning (Venugopal and Luckey, 1978).	Toxicity affected by hardness, pH, organics. Acute values range from 1.0 µg/L for rainbow trout to 28,000 µg/L for a mayfly (USEPA, 1984a). Chronic values range from 0.15/µg/L in Daphnia to 156 µg/L for Atlantic Salmon (USEPA, 1984a). Cadmium bioconcentrates.
Cobalt	Cobalt is an essential element for plants.	Cobalt is essential for animal nutrition. Rats can tolerate up to 200 ppm, sheep up to 10 ppm, cattle up to 20 ppm without experiencing adverse effects (Venugopal and Luckey, 1978)	N/A[a]
Copper	Copper is an essential nutrient for plants. Soil concentrations greater than 36 mg/kg may cause leaf chlorosis in plants (Adriano, 1986).	Copper is an essential nutrient for animals. Dietary exposure of 10 to 15 ppm may cause blood disorders in small mammals (Smith and Rongstand, 1982; Blus et al., 1987). Copper tends to accumulate in the liver.	Copper is an essential nutrient for aquatic life but toxic in low concentrations. Acute toxicity ranges from 16.7 µg/L for northern squawfish to 10,240 µg/L for a stonefly (USEPA, 1984b). Chronic toxicity ranges from 3.9 µg/L for brook trout to 60.4 µg/L for northern pike (USEPA, 1984b). Toxicity affected by hardness, alkalinity, pH, total organic carbon. Copper toxicity primarily rated to Cu^{+2} ion and ionized hydroxides.
Iron	NA[b]	NA[b]	The primary form of concern in the aquatic environment are ferrous or bivalent (Fe^{++}), and ferris or trivalent (Fe^{+++}) icons.

Table 7-7. Summary of Potential Toxic Effects of the Potential Chemicals of Concern for the Baxter Springs/Treece Subsites (Continued)

Metal	Phytotoxic Effects	Toxic Effects to Terrestrial Organisms	Toxic Effects to Aquatic Organisms
Lead	Lead is a non-essential element for plants (Demay et al., 1982). Lead interacts synergistically with cadmium and antagonistically with calcium, phosphorus, and sulfur.	Lead is a non-essential element for animals (Demay et al., 1982). Lead concentrates in animal liver and kidneys at relatively high ingestion (138 mg/kg) and may cause blood disorders in small mammals (raccoons) (Demay, et al., 1982).	Toxicity affected by hardness, pH. Acute toxicities range from 142.5 µg/L for an amphipod to 235,900 µg/L for a midge (USEPA, 1984c). Chronic toxicity ranges from 12.3 µg/L to 128.1 µg/L for a cladoceran (USEPA, 1984c). Lead bioconcentrates.
Manganese	Manganese is an essential micronutrient in plants. Manganese may be toxic to plants at concentrations around 1.0 mg/L in irrigation water applied to soils with pH <6.0 (USEPA, 1986).	Manganese is an essential micronutrient for animals. Manganese not known to be toxic to animals. It is the least toxic of the essential metals (Venugopal and Luckey, 1978). Mammals can tolerate concentrations up to 1000 ppm without experiencing adverse effects (Venugopal and Luckey, 1978).	Manganese is an essential micronutrient for aquatic life. Manganese ions are rarely found above 1.0 mg/L (USEPA, 1986). Manganese generally not considered to be toxic for aquatic life.
Mercury	N/A[b]	N/A[b]	Toxicity affected by chemical speciation of Hg. Chronic values for methyl mercury range from <0.07 µg/L for brook trout and Daphnia to 1.1 µg/L for Daphnia tested in Hg II (USEPA, 1984d). Mercury bioconcentrates. Acute values for Hg II range from 2.2 µg/l for Daphnia pulex to 20,000 µg/l for aquatic insects (USEPA, 1984d).
Nickel	N/A[b]	N/A[b]	Toxicity affected by pH, hardness. Acute toxicity ranges from 1,101 µg/L for a cladoceran to 43,240 µg/L for a fish. Chronic toxicity ranges from 14.8 µg/L with Daphnia to 526.7 µg/L for fathead minnows.

Table 7-7. Summary of Potential Toxic Effects of the Potential Chemicals of Concern for the Baxter Springs/Treece Subsites (Continued)

Metal	Phytotoxic Effects	Toxic Effects to Terrestrial Organisms	Toxic Effects to Aquatic Organisms
Silver	N/A[b]	N/A[b]	Acute values for invertebrates range from 0.25 µg/L for Daphnia magna to 4,500 µg/l for Gammarus pseudolimnaeus (USEPA, 1980). Acute value for fish range from 3.9 µg/l for Pimephales promelas in safe drinking water to 280 µg/L for Onchorynus gairdneri in hard water (USEPA, 1980). No apparent relationship between water hardness and chronic toxicity (U.S. EPA, 1980). Chronic effects range from 0.12 µg/L for rainbow trout to 29 µg/L for Cladoceran (USEPA, 1980)
Zinc	Zinc is an essential element for plants (Adriano, 1986). Zinc, calcium, and cadmium are especially antagonistic. Zinc concentrations in plants of 500 ppm causes leaf chlorosis and stunted growth (Kabata-Pendias and Pendias, 1984). Zinc interacts antagonistically with cadmium, calcium, copper, and phosphorus.	Zinc is an essential element for animals (Adriano, 1986). Animals have relatively high tolerance to zinc because of efficient homeostatic mechanisms that form organic ring complexes or cause the element to be excreted (Smith and Rongstad, 1982).	Zinc is an essential element for aquatic life. Acute values range from 50.7 µg/L for Daphnids to 88,960 µg/L for a damselfly (USEPA, 1987). Chronic values range from 46.73 for Daphnids to >5,243 µg/L for a caddisfly (USEPA, 1987). Toxicity affected by hardness, pH, organics.

a Cobalt is not a COC for on-site surface water; hence, potential toxic effects to aquatic organisms were not evaluated.

b Mercury, nickel, and silver are not COCs for on-site soils; hence, potential toxic effects on plants and terrestrial receptors were not evaluated.

8.0 TOXICITY ASSESSMENT: CALCULATION OF TOXICITY QUOTIENTS

Toxicity quotients (TQs) are a general method for assessing the environmental hazards of chemicals. The TQ method is the direct arithmetic comparison of a concentration from a laboratory toxicity test with an expected or measured environmental concentration (Barnthouse and Suter, 1986). TQs are defined as the exposure point concentration (mg/L) for aquatic receptors, the chronic daily intake (mg/kg-day) for terrestrial receptors, or soil concentrations (mg/kg) for plants divided by chronic Toxicity Reference Values (TRVs). The TQ approach is routinely used by EPA as the simplest, quantitative method available for estimating risks to ecological receptors (USEPA, 1992). TQs were calculated the key aquatic, terrestrial, and T&E species; in addition, data on the toxicity of on-site soils to plants were also evaluated. Results of the TQ assessment are discussed for each of these receptor groups in subsequent sections of this chapter.

Before discussing the specific TQ results, the criteria used to evaluate TQs is defined and an explanation of how the TQ results were interpreted for the study is provided. TQs results were evaluated using these criteria: (1) TQs less than 1 indicates that adverse effects in exposed individuals or populations are not probable, and (2) TQs greater than 1 indicate that adverse chronic impacts are possible. A TQ of 1 indicates that the measured concentration equals a concentration that caused some impact in a certain test species under a given set of experimental conditions. In this sense, TQs greater than or equal to 1 can be used to indicate that chronic adverse impacts to exposed individuals are possible. Given that the goal of this ERA is to evaluate potential adverse effects on exposed populations, a TQ of 1 may be overly conservative. Within a given ecosystem, loss of some individuals in a receptor group is acceptable if the entire population will not be adversely impacted. Barnthouse and Suter (1986) state that "ecological risk assessments used in decision making should be based, to the greatest extent possible, on objective estimates of ecological damage (e.g., probabilities of population extinction or reductions in abundance of plants and animals)." Hence, a TQ greater than 1 does not necessarily mean that effects observed in the laboratory are likely to occur in the field or that exposures in the field are significant enough to cause population-level effects. Furthermore, there is not a linear (one-to-one) relationship between increasing TQ values and the probability of adverse effects. For example, a TQ of 2 does not imply that adverse effects are twice as likely to occur in a given system as a TQ of 1. The intent of a TQ approach is to provide a continuous quantitative scale so that evaluation and rank ordering of potential impacts can be assessed.

The TQ approach was used to evaluate the potential for adverse effects to receptors within the same ecologic group. As stated in Section 5.0, selection of key receptor species was designed to minimize the possibility that other species might be more exposed than the selected key species. Ecosystem effects were evaluated by analyzing potential impacts to various trophic levels from primary producers (plants) to higher-trophic level organisms (tertiary consumers). The focus on primary and higher-level consumers provides sufficient data to assess the general condition of all trophic levels. Thus, by using toxicological data for the key receptors species selected for this site, it is appropriate to conclude that other species within the same ecological group are unlikely to be adversely affected. For example, if the red-tailed hawk, barred owl, and the mink (the key higher-level terrestrial species selected for this site) are not expected to experience adverse effects, then it is reasonable to assume that other less exposed terrestrial wildlife are unlikely to experience adverse effects. However, these criteria do not apply to T&E species since they are given more stringent individual consideration in the ERA. Potential effects to T&E species are discussed in Section 8.3.

Finally, it is important to emphasize that results of the toxicity assessment represent one of the many sources of data used to determine if exposure to site-related metals by ecological receptors is likely to cause adverse chronic effects. Results of the toxicity assessment should not be used as the sole means to evaluate potential chronic effects to exposed receptors. Field biosurvey data, other pertinent information available in the scientific literature, and results of the toxicity assessment were used <u>in combination</u> to yield a more realistic picture of potential impacts to exposed populations. Field data were used to provide tangible evidence of potential adverse (both acute and chronic) effects. Toxicity assessment results and literature data were used to augment the field data to sense subtle chronic adverse impacts that may not be readily observable in the field. The validity and strength of the various types of data used to evaluate potential adverse effects are discussed in Section 9.

8.1 TOXICITY OF METALS TO AQUATIC RECEPTORS

The potential toxicity of metals to aquatic receptors was evaluated using a balanced approach. This approach involved comparing the arithmetic mean concentrations of metals measured in on-site surface water bodies with the TRVs (in Table 7-1) and comparing upper-bound concentrations of metals measured in on-site surface water bodies with the Criterion Maximum Concentrations (CMC) identified by EPA.

Using the mean total recoverable concentration data yields a realistic portrayal of actual on-site exposures for these reasons:

1. Many aquatic organisms can tolerate infrequent pulses of high metals concentrations (Mulvey and Diamond, 1991).
2. Metals concentrations tend to vary temporally with changes in rainfall runoff amounts (Remedial Investigation).
3. Arithmetic mean concentrations best reflect long-term exposure to varying conditions (USEPA, 1989d).
4. TQs based on arithmetic mean concentration data are more indicative of potential population-level effects than upper-bound TQs, and the goal of this assessment is to determine the potential for risk and the relative magnitude of risk to receptors and areas.

Example TQ Calculation:

The TRV for zinc was estimated to be 1.4 mg/L (Table 7-1). Using the measured arithmetic mean concentration of zinc in Tar Creek (9.2 mg/L from Table 3-8), the TQ for zinc in Tar Creek would be 9.2 divided by 1.4 or 6.

The resulting TQ of 6 suggests that individuals inhabiting Tar Creek may be experiencing chronic adverse effects. Although the possibility of population-level impacts cannot be definitely ascertained, these results suggest that population-level effects on aquatic organisms are possible.

Results of the toxicity assessment using the arithmetic mean total recoverable concentration data for aquatic organisms (Table 8-1) show that aquatic populations inhabiting Willow Creek, the Spring River, and the Neosho River are not expected to experience chronic adverse effects from exposure to the metals of concern since the TQs based on mean concentrations and TRVs for these metals are less than 1. Results of the toxicity assessment indicate that adverse chronic effects from exposure to arithmetic mean levels of cadmium and zinc by aquatic populations inhabiting Spring Branch and the subsite ponds and from exposure to zinc by aquatic organisms inhabiting Tar Creek are possible (mean TQs range from 1 to 10).

Stephan et al. (1985) has established criterion for the protection of aquatic organisms and their uses. This criterion is based, in part, on a Criterion Maximum Concentration (CMC) which is equal to one-half the Final Acute Value (FAV). EPA further defines the CMC as that one-hour average concentration which should not be exceeded more than once every three years on the average (Stephan et al., 1985). Comparison of field data collected and analyzed during the remedial investigation efforts with the CMC indicate that TQs equaling or exceeding 1 imply potential toxicity to aquatic organisms within Spring Branch, Tar Creek, and subsite ponds.

Toxicity quotients equaling or exceeding 1 based on the upper-bound concentration and the CMC (i.e., upper-bound concentration divided by CMC) indicate that acute toxicity is possible from exposure to elevated levels of cadmium and zinc in Spring Branch and from exposure to elevated levels of zinc in Tar Creek and the subsite ponds. Despite this apparent acute toxicity, fish were present in the streams. Although few fish (eight centrarchids) were collected from Tar Creek, numerous fish of varying genera (50 fish representing centrarchids, ictalurids, and cyprinids) were collected in Spring Branch. The biosurvey results indicate that there may be site-specific factor(s) affecting the actual toxicity of these metals. These factors

Table 8-1. Summary of Toxicity Quotients Based on Toxicity Reference Values for the COCs in On-Site Streams and the Spring and Neosho Rivers for Aquatic Organisms

Water Body and COCs	FAV Adjusted to a Hardness of 190 mg $CaCO_3$/L (mg/L)	Upper-Bound Concentration[a] (mg/L)	Criterion Maximum Concentration (mg/L)	Toxicity Quotient Based on CMC and Upper-Bound Concentration	Toxicity Reference Value (mg/L)	Arithmetic Mean Concentration[a] (mg/L)	Toxicity Quotient[b] Based on TRV and the Mean Concentration
SPRING BRANCH (All Stations, N = 8)							
Cadmium	0.155	0.12	0.078	2	0.008	0.08	10
Zinc	3.14	14.5	1.57	9	1.423	9.9	7
TAR CREEK (Downstream Stations Only, N = 6)							
Manganese[c]	NA	NA	NA	NA	1.5	0.6	0.4
Zinc	3.14	17.2	1.57	11	1.423	9.2	6
WILLOW CREEK (Downstream Stations Only, N = 8)							
Cadmium	0.155	0.002	0.078	0.03	0.008	0.0015	0.2
Lead	1.401	0.02	0.071	0.3	0.027	0.01	0.4
Zinc	3.14	0.81	1.57	0.5	1.423	0.6	0.4
SPRING RIVER (NEAR BAXTER SPRINGS)[(d)]							
Cadmium	0.155	0.004	0.078	0.05	0.008	0.003	0.4
Lead	1.401	0.04	0.071	0.6	0.027	0.02	0.7
Zinc	3.14	0.5	1.57	0.3	1.423	0.4	0.3
NEOSHO RIVER (NEAR THE MOUTH OF TAR CREEK)[(e)]							
Cadmium	0.155	0.002	0.078	0.03	0.008	0.002	0.3
Zinc	3.14	0.6	1.57	0.4	1.423	0.3	0.2

[a] Data are total recoverable values.

[b] Toxicity quotients are the concentration in water divided by the chemical-specific toxicity reference value; toxicity quotients were rounded to one significant figure.

[c] Levels of manganese ranging from 1.5 mg/L to 1000 mg/L are reported to be tolerable to aquatic organisms, hence 1.5 mg/L was used for the TRV (USEPA, 1986).

[e] KDHE Storet database; see Appendix B.

[f] OWRB, 1983; see Appendix B.

may include the bioavailability of the metals (i.e., dissolved concentrations) and acclimation or adaptation of the species present in subsite surface waters.

Centrarchids (Sunfish Genus *Lepomis*), which are represented in both Tar Creek and Spring Branch, are one of the more zinc tolerant species (Genus Mean Acute Value equals 10.5 mg/L, Appendix A). Tar Creek downstream station results showed mean concentrations of 9.2 mg/L and an upper-bound concentration of 17.2 mg/L (Table 3-4). At these levels, more sensitive species would be excluded (i.e., minnows, snails, isopods). For Spring Branch, the mean and upper-bound zinc concentrations were 9.9 and 14.5 mg/L, respectively (Table 3-6) but a variety of genera were represented. An explanation of this apparent inconsistency may be that the fish were collected near the mouth of Spring Branch, below inputs from urban areas of Baxter Springs (the city's storm water runoff and limestone bedrock seepage). These inputs may dilute the creek water and supply organics or minerals which would act to reduce the bioavailability of metals. This is exemplified by the fact that the total recoverable mean zinc concentration at the downstream SB-2 station was roughly half of that at SB-1 (7.5 vs. 14 mg/L). With toxicity ameliorated to some degree in the lower segment of Spring Branch, some of the more sensitive species have become established (e.g., ictalurids and cyprinids). In summary, the results of the field data indicate that acute toxicity is not occurring in these streams for select groups of tolerant aquatic organisms.

TRVs and TQs for each pond were calculated using pond-specific hardness and concentration data (Table 8-2). Note that the resultant TQs are less than or equal to 1 for seven of the ten ponds sampled. All TQs for Ponds BP-2, BP-3, BP-4, BP-5, TP-1, TP-6 and TP-9 were below unity. The Ballard Chat Wash Pond (BP-1) exhibited the highest TQ (10 for cadmium). Other TQs exceeding unity included 2 for zinc (TP-5) and for iron (TP-7); and 7 for lead (TP-7).

8.2 TOXICITY OF METALS TO TERRESTRIAL RECEPTORS

A tiered approach was used to determine if existing concentrations of COCs in on-site environmental media are likely to cause adverse effects in exposed terrestrial populations.

Table 8-2

Summary of Toxicity Quotients for COCs and TRVs
Adjusted for Individual Pond Hardness

POND BP-1 (CHAT WASH) HARDNESS = 1,110 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.35	0.034	10
Iron	ND (0.3)	1.0	0.3
Lead	0.12	0.259	0.5
Manganese	0.24	1.5	0.2
Mercury	0.0003	0.005	0.1
Zinc	7.90	6.33	1

POND BP-2 (TAILINGS) HARDNESS = 130 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.0005	0.006	0.08
Iron	ND (0.05)	1.0	0.05
Lead	ND (0.0005)	0.017	0.03
Manganese	0.04	1.5	0.03
Mercury	ND (0.0001)	0.005	0.02
Zinc	0.083	1.03	0.08

POND BP-3 (TAILINGS) (FISH OBSERVED) HARDNESS = 840 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	ND (0.00004)	0.027	0.02
Iron	ND (0.2)	1.0	0.2
Lead	ND (0.002)	0.183	0.01
Manganese	0.16	1.5	0.11
Mercury	ND (0.0001)	0.005	0.02
Zinc	0.32	5.0	0.06

Table 8-2 (Continued)

Summary of Toxicity Quotients for COCs and TRVs Adjusted for Individual Pond Hardness

POND BP-4 (TAILINGS)
(FISH OBSERVED)
HARDNESS = 92 mg $CaCO_3$/L

COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	ND (0.0003)	0.004	0.07
Iron	0.3	1.0	0.30
Lead	ND (0.0005)	0.011	0.05
Manganese	0.06	1.5	0.04
Mercury	0.0005	0.005	0.10
Zinc	0.054	0.77	0.07

POND BP-5 (SUBSIDENCE)
(FISH OBSERVED)
HARDNESS = 470 mg $CaCO_3$/L

COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.005	0.017	0.30
Iron	ND (0.1)	1.0	0.10
Lead	ND (0.003)	0.087	0.04
Manganese	0.28	1.5	0.19
Mercury	ND (0.0001)	0.005	0.02
Zinc	1.30	3.06	0.42

POND TP-1 (SUBSIDENCE)
(FISH OBSERVED)
HARDNESS = 560 mg $CaCO_3$/L

COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.024	0.019	1
Iron	ND (0.1)	1.0	0.1
Lead	ND (0.002)	0.108	0.02
Manganese	0.22	1.5	0.15
Mercury	ND (0.0001)	0.005	0.02
Zinc	2.60	3.55	0.73

Table 8-2 (Continued)

Summary of Toxicity Quotients for COCs and TRVs
Adjusted for Individual Pond Hardness

POND TP-5 (TAILINGS) (FISH OBSERVED) HARDNESS = 740 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.032	0.024	1
Iron	ND (0.2)	1.0	0.2
Lead	0.025	0.154	0.16
Manganese	0.62	1.5	0.41
Mercury	0.0004	0.005	0.08
Zinc	9.70	4.49	2

POND TP-6 (TAILINGS) (FISH OBSERVED) HARDNESS = 680 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.022	0.023	1
Iron	ND (0.1)	1.0	0.1
Lead	0.013	0.139	0.09
Manganese	0.04	1.5	0.03
Mercury	ND (0.0001)	0.005	0.02
Zinc	0.39	4.18	0.09

POND TP-7 (TAILINGS) HARDNESS = 110 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.007	0.005	1
Iron	1.7	1.0	2
Lead	0.10	0.014	7
Manganese	0.88	1.5	0.6
Mercury	ND (0.0001)	0.005	0.02
Zinc	1.20	0.89	1

Table 8-2 (Concluded)

Summary of Toxicity Quotients for COCs and TRVs
Adjusted for Individual Pond Hardness

POND TP-9 (TAILINGS) (FISH OBSERVED) HARDNESS = 320 mg $CaCO_3$/L			
COCs	Concentration (mg/L)	TRV (mg/L)	TQ
Cadmium	0.0025	0.012	0.2
Iron	ND (0.05)	1.0	0.05
Lead	0.031	0.053	0.6
Manganese	0.10	1.5	0.07
Mercury	ND (0.0003)	0.005	0.05
Zinc	0.45	2.21	0.2

Parentheses denote constituent not detected at reporting limit shown. Arsenic, barium, chromium, cobalt, selenium, and vanadium were not detected. Nickel, copper, and silver are not shown because upper-bound values are lower than the chronic AWQC, adjusted for individual pond hardness.

TRV was not calculated for iron. The acute AWQC of 1.0 mg/L was substituted. This AWQC is not hardness dependent.

Levels of manganese ranging from 1.5 mg/L to 1000 mg/L are reported to be tolerable to aquatic organisms (USEPA, 1986), hence, 1.5 mg/L was used as TRV.

ND = Not detected; value in parentheses is one-half reported limit.

First, toxicological data available in the literature were adjusted using the application of UFs to obtain TRVs for key terrestrial species. Next, TRVs were compared to the worst-case intake estimate for each key species. If the resulting TQ (i.e., the worst-case TQ) was less than 1, adverse chronic impacts to exposed terrestrial populations are not likely. Finally, if the worst-case TQ equaled or exceeded 1, then Reasonable Maximum Exposure (RME) intake estimates were compared to the TRV to calculate an RME TQ. Worst-case exposure estimates were used as a screening-level approach to derive conservative TQs. Since worst-case dose estimates assume that an individual is exposed to upper-bound concentration of metals in various environmental media, the probability that an individual would receive a dose equivalent to the worst-case dose is extremely small. It is not realistic to conclude that terrestrial receptors would consistently come in contact with the upper-bound concentration of metal given that most receptors tend to have relatively large home ranges. Therefore, the RME exposure scenario is more representative of expected exposure intensity.

Table 8-3 lists the worst-case and RME dose estimates and TQs for the raptors and the mink. The focus on primary and higher-level consumers is based on the rationale that results of these investigations would provide sufficient data to assess the general condition of the terrestrial ecosystem. Results shows that exposure to cadmium, lead, and zinc is not expected to cause adverse effects in raptors, since all worst-case TQs for the red-tailed hawk and the barred owl are less than 1 (worst-case TQs range from 0.05 for zinc to 0.2 for lead). Conversely, exposure to cadmium, lead, and zinc could cause adverse effects in the mink and animals with similar food base, since the worst-case and RME TQs are slightly above 1. Worst-case TQs range from 2 for cadmium to 4 for zinc, while RME TQs range from 1 to 3.

8.3 TOXICITY OF METALS TO AMPHIBIANS

For T&E species, protection of individuals and/or their critical habitat is essential. To fulfill this objective, a more conservative approach was used for T&E species than was used for the other terrestrial and aquatic species of concern. Potential adverse effects were assumed possible if the upper-bound concentration of metal in on-site surface waters exceeded

Table 8-3 Toxicity Quotients for the Key Terrestrial Receptors

Receptor and Metal	Toxicity Reference Value (mg/kg-day)	Worst-Case Dose (mg/kg-day)	Worst-Case Toxicity Quotient[a]	RME[b] Dose (mg/kg-day)	RME Toxicity Quotient[a]
RED-TAILED HAWK					
Cadmium	1.6	0.16	0.1	0.07	NA[c]
Lead	2.8	0.50	0.02	0.26	NA
Zinc	111	5.15	0.05	3.5	NA
BARRED OWL					
Cadmium	1.6	0.19	0.1	0.08	NA
Lead	2.8	0.59	0.02	0.31	NA
Zinc	111	6.1	0.05	4.1	NA
MINK					
Cadmium	0.1	0.19	2	0.11	1.0
Lead	0.3	0.96	3	0.63	2
Zinc	4.2	16.5	4	13.6	3

[a] Toxicity quotients are the dose divided by the toxicity reference value. TQs were rounded to one significant figure.

[b] Reasonable maximum exposure scenario.

[c] Calculated only if the worst-case TQ was greater than or equal to 1.

the metal-specific TRVs listed in Table 8-4. This approach is conservative, since it is unlikely that T&Es would remain in prolonged contact with surface water contaminated at the upper-bound level. Conversely, this approach also assumes that direct contact with surface water is the sole and primary means of T&E exposure. The data in Table 8-4 show that exposure to site-related metals is not expected to cause chronic toxic effects in exposed T&Es even when upper-bound concentration data are used to calculate TQs. With the exception of zinc in Tar Creek, TQs based on the upper-bound concentration of total recoverable metals in surface water were less than 1 (upper-bound TQs ranged from 0.0002 for cadmium in Willow Creek and the Neosho River to 0.8 for zinc in Spring Branch). The upper-bound TQ for zinc in Tar Creek is one, which indicates that adverse chronic effects are possible in amphibians exposed to Tar Creek water.

8.4 PHYTOTOXICITY OF SOILS

Metals concentrations in near-pile and Ag/A soils were used to estimate whether these two soil types were expected to be phytotoxic. The phytotoxicity of mine and mill wastes (includes excavated chat and mill-site soils) was not evaluated in this report since physical limitations associated with these materials apparently limits or precludes vegetative growth regardless of metals content (i.e., lack of moisture, nutrients, and organic matter). To evaluate the potential phytotoxicity, metal toxicity data for agronomic species were obtained (Table 8-5). The phytotoxicity data are concentrations that researchers estimated caused some reduction in crop yield (e.g., zinc at soil concentrations between 60 mg/kg and 960 mg/kg caused a 43 to 98 percent reduction in yield in various agronomic species, see Appendix C). A mean value was calculated but was assumed to be more susceptible to the number of tests the researchers conducted and the concentration of metal used in a given study and has the effect of skewing the data in the direction of most research results. For example, if the researchers conduct studies primarily with high metal concentrations resulting in high crop yield reductions, then the resulting mean metal concentration would be high and would not be a good indicator of the average or low toxicity value for that metal. Hence, the median value was used, as it was considered more representative of the range of data shown to be phytotoxic. Upper-bound and mean metal concentrations in Ag/A and near-pile soils were

Table 8-4. Summary of Toxicity Quotients Based on Toxicity Reference Values for the COCs in On-Site Streams and Ponds and the Spring and Neosho Rivers for Amphibians

Water Body and COCs	Toxicity Reference Value (mg/L)	Upper-Bound Concentration[a] (mg/L)	Toxicity Quotient[b] Based on the Upper-Bound Concentration
SUBSITE PONDS			
Cadmium	10	0.02	0.007
Copper	0.04	0.006	0.2
Iron	NA[c]	0.6	---
Lead	7.8	0.04	0.01
Manganese	1.7	0.4	0.2
Mercury	0.2	0.0004	0.002
Nickel	NA[c]	0.04	---
Silver	NA[c]	0.003	---
Zinc	17.2	3.7	0.2
SPRING BRANCH			
Cadmium	10	0.12	0.01
Zinc	17.2	14.5	0.8
TAR CREEK			
Manganese	1.7	0.9	0.5
Zinc	17.2	17.2	1
WILLOW CREEK			
Cadmium	10	0.002	0.0002
Lead	7.8	0.02	0.003
Zinc	17.2	0.8	0.05
SPRING RIVER (near Baxter Springs)			
Cadmium	10	0.004	0.0004
Copper	0.04	0.015	0.4
Lead	7.8	0.05	0.01
Zinc	17.2	0.5	0.03
NEOSHO RIVER (downstream of the mouth of Tar Creek)			
Cadmium	10	0.002	0.0002
Zinc	17.2	0.6	0.03

[a] Data are total recoverable values.

[b] Toxicity quotients are the concentration in water divided by the chemical-specific toxicity reference value.

[c] Not applicable.

Table 8-5. Phytotoxicity of Metals in Soil - Agronomic Species

Metal[1]	Minimum Phytotoxic Concentration[2] (ppm, dry weight)	Median Phytotoxic Concentration (ppm, dry weight)
Cadmium	5[3]	40
Cobalt	25[4]	NA
Copper	3.15[5]	20.3
Lead	100[6]	250
Manganese	1,500[7]	NA
Zinc	60[8]	240

1 Listed metals are COCs for Ag/A or near-pile soils, see Tables 4-2 and 4-3 in text.

2 From data tables in Appendix A; minimum values are lowest concentrations which resulted in measurable yield reduction.

3 5 ppm soil concentration caused a 16.5-27.9 percent yield reduction in various crops (four tests).

4 25 to 50 ppm concentration was toxic to rice (Kitigishi and Yamane, 1981).

5 3.15 ppm soil concentration caused 22 percent yield reduction in soybeans (one test).

6 100 ppm soil concentration caused 15.9 percent yield reduction in oats (one test).

7 Minimum phytotoxic level (Shacklette and Boerngen, 1984).

8 60 ppm soil concentration caused effects ranging from either a yield increase, or no yield reduction, to a 42.5 percent yield reduction in slash pine seedlings (eleven tests).

compared to the median concentration reported to be phytotoxic in either laboratory or field studies (Table 8-6). If the resulting upper-bound TQ for a given metal was less than 1, existing levels of that metal in site soils were not expected to be phytotoxic. If the upper-bound TQ was greater than or equal to unity, then the TQ was calculated using the arithmetic mean concentration of that metal in on-site soils. If the TQ based on the arithmetic mean equaled or exceeded 1, then existing levels of that metal were deemed potentially phytotoxic.

Table 8-6 shows that TQs based on the upper-bound concentration of COCs in near-pile soils did not exceed 1 for all COCs except zinc (upper-bound TQ equals 2). These data indicate that existing levels of zinc in near-pile soils may be phytotoxic. No phytotoxic data were located in the literature for species likely to establish in near-pile soils. Thus, use of the median phytotoxic concentration shown to reduce yields in the more sensitive agronomic species is probably conservative when applied to near-pile soils. Similar results were obtained for Ag/A soils; TQs for all COC were less than 1. These data indicate that existing levels of COCs in Ag/A soils are not expected to be phytotoxic, since all mean TQs did not exceed unity.

It should be noted that the zinc applied in the laboratory/field tests was in a sulfate form which is soluble and readily available to the plants. In contrast, plant-available fractions of the total zinc concentrations in near-pile samples (as determined from on-site monitoring data) ranged from 2 to 55 percent, with a mean of 16.9 percent compared with the near 100 percent for laboratory test species. If the mean plant-available figure of 16.9 percent is applied to the upper bound concentration in near-pile soils (996 mg/kg) the resultant TQ becomes 0.7. While data on plant-available concentrations that cause toxic symptoms are limited [corn was affected by 450 to 1400 ppm plant-available zinc, and cowpeas by 180 to 700 ppm plant-available zinc (Adriano, 1986)] the relatively low fraction of plant-available zinc in near-pile soils suggests that zinc concentrations may not be phytotoxic, or at worst only phytotoxic to some species. The revegetation study currently in progress at the Galena subsite (Norland, 1991) illustrates the importance of plant-availability as grasses and other tolerant species are grown in chat, amended with organics, exhibiting total zinc concentrations ranging between 10,588 and 16,338 mg/kg.

In summary, the large gap that exists between the toxicological data available from laboratory tests and subsite plant species/soil conditions makes the prediction of a phytotoxic level, applicable to near-pile soils, very difficult. Given what we know, however, the information would indicate minimal impacts to the naturally occurring, established vegetation in the near-pile areas. The concentration of zinc in near-pile soils could potentially reduce crop yields but little if any effect on the native species is expected. Potential changes could include minor reductions in biomass yield or changes in species composition. None of the changes mentioned are likely to have any measurable effect on the terrestrial ecosystem of the subsites. Based on field assessments, the vegetation in near-pile areas provides cover and food for certain wildlife although they may not be preferred as forage sites, and are unlikely to be used extensively by livestock if other pasture is available.

9.0 UNCERTAINTY ANALYSIS

An uncertainty analysis takes into account the inherent variability in measured and estimated parameters, allowing decision makers to evaluate risk estimates in the context of the quality and reliability of the assumptions and data used in the assessment. Considerable uncertainty invariably attends the evaluation of ecological risks. Principal sources of uncertainty include the contaminant database, the exposure assessment, and the toxicological database. Additional uncertainties involved in site characterization are also likely to influence risk estimates. The extent to which the following major sources of uncertainty may over- or underestimate the exposure (and hence risk) to ecological receptors at the BS/T subsites is discussed in this section:

- Determination of the nature and extent of contamination;
- Exposure assessment;
- Ecological assessment and measurement endpoints;
- Derivation of toxicity reference criteria.

9.1 UNCERTAINTY ASSOCIATED WITH DETERMINING THE NATURE AND EXTENT OF CONTAMINATION ONSITE AND THE SELECTION OF CHEMICALS OF CONCERN

The uncertainties involved in measuring chemical concentrations in environmental media can be substantial. Major sources of uncertainty in environmental sampling and analysis include handling procedures; sampling location, number, and density; selection of COCs; analyte extraction; sample dilution; analytical detection limits and handling of non-detects; analyte interference; and instrument limitations. Even with strict quality assurance and control measures, there is no assurance that the environmental samples taken are fully representative of the site. Uncertainties associated with identifying chemicals of concern include those connected with sampling environmental media and those related to the use of small data sets in the statistical evaluation of data. These uncertainties are expected to have a low to moderate potential to over- or underestimate risk.

9.2 UNCERTAINTY ASSOCIATED WITH THE EXPOSURE ASSESSMENT

The exposure assessment includes a number of major sources of uncertainty, including receptor selection, exposure point concentration and intake rate estimation, bioavailability of contaminants, and the distribution of the receptors and stressors. Because this assessment is deterministic (i.e., parameters are estimated as single values rather than distributions), uncertainty cannot be described quantitatively. The following discussion therefore focuses on (1) identification of sources of uncertainty in exposure estimation, and (2) their qualititative or relative importance in interpretation of results.

9.2.1 Selection of Key Receptors

The selection of key receptors used in the risk assessment was designed to minimize the possibility that other species could be more exposed than the key species themselves by focusing on higher-tropic level species and food chain effects, as they would provide sufficient data to assess the general condition of the ecosystem. The use of these selected organisms was therefore expected to be adequately protective of the majority of species potentially inhabiting the subsites, since these receptors are top predators.

Given the lack of toxicological data specific for the terrestrial species likely to occur onsite, it is possible that some sensitive species may have been overlooked (i.e., some species more sensitive than those chosen were not evaluated). The lack of inclusion of sensitive species would tend to underestimate risks, although this source of uncertainty is not expected to substantially under- or overestimate risks, since the mink is generally considered sensitive to the effects of metals exposure. Furthermore, it is possible that other, lower-trophic level receptors, such as the white-footed mouse, could be more exposed than the raptors and the mink, since mice have closer and more frequent contact with contaminated soils. Accurate dose estimates for the white-footed mice were not possible, since: (1) the concentration of site-related metals was not measured in the vegetation and seeds on which mice primarily forage; (2) body burden data were measured with the pelt on, making it impossible to differentiate between metals loads in tissues and that associated with pelt; and (3) tissue levels for various organs were not obtained, so that the true tissue burden could not be segregated from the amount of metals present in the gut. These limitations did not allow a comprehensive evaluation of potential impacts on terrestrial prey species (using the white-footed mice as a surrogate). Thus, potential impacts to all ecologic groups making up the terrestrial ecosystem were not quantified, which means that ecosystem-level effects may be underestimated.

Similarly, toxicological data specific to aquatic species likely to occur onsite is not complete and some sensitive species may have been overlooked. However, based on the data available from the Kansas Department of Parks and Wildlife and results of aquatic biota sampling, most major groups of aquatic organisms have been taken into consideration during the selection of key aquatic receptors. It is possible that other receptors including Gammarus could occasionally occur onsite and, therefore, be exposed to on-site contamination and lack of inclusion of such receptors would underestimate risks. In general, amphipods are not adapted for withstanding drought conditions and other adverse environmental conditions (such as those occurring onsite) (Pennak, 1989).

9.2.2 Approaches to Dose Estimation for Terrestrial Receptors

A simple dose-based approach using site-specific tissue data was developed to evaluate potential exposures by key terrestrial receptors to COCs under both worst-case and reasonable maximum exposure (RME) scenarios. It was assumed that upper-bound and arithmetic mean environmental concentrations were representative of the range of concentrations to which ecologic receptors are likely to be exposed. Since the actual exposure concentrations to which these receptors may actually be exposed cannot be precisely determined, both worst-case and RME scenarios were evaluated to provide decision makers with a better perspective on the range of risks likely to be encountered by terrestrial receptors from exposure to site-related chemicals.

9.2.2.1 Worst-Case Scenario

Dose estimates for terrestrial receptors were first calculated using worst-case exposure parameters as a screen for (1) contaminant hot spots, and (2) receptors that don't require further evaluation. Worst-case exposure is defined in this case as the <u>highest</u> exposure reasonably expected to occur at a site. The primary objective was to arrive at an exposure estimate that will fall within the high end of the actual probabilistic exposure distribution (i.e., between the 90th and 99.9th percentiles). Exposure point concentrations were defined as the upper 95 percent confidence limit on the arithmetic mean, and worst-case exposure parameters are generally set at around the 95th percentile of their individual distributions.

For biota, the worst-case approach implies that an organism spends its life in near-maximal contact with upper-bound concentrations of all COCs in all media simultaneously. This can be unrealistic from the standpoint that (1) many receptors do not tend to stay in one area for very long, and (2) the maximum concentration of COCs in air, water, and soil are not geographically coincident. For example, the maximum concentration of cadmium was measured in Spring Branch, while the maximum concentration of lead and zinc were found in Tar Creek. It is not physically possible for a receptor to be exposed to the maximum concentration of a single chemical in all media simultaneously, much less to the maximum concentration of all COCs in all media concurrently.

Because this approach is highly unlikely to underestimate exposure, it is useful as a screen. However, because it is likely to project exaggerated risks to biota, primarily due to negligence of the critical effects of spatial and temporal variations in exposure, it is not appropriate for assessment of actual ecological risk. It is important to note that, in contrast to human health risk assessment, where every individual must be protected, the fundamental unit for ecological risk analysis is the <u>population</u> (USEPA, 1992). That is, adverse effects on or even loss of a portion of the receptor population (unless the species is specially protected) is considered unlikely to significantly diminish its viability or disrupt the community or ecosystem of which it is a part (USEPA, 1992). Thus, because the frequency of worst-case exposure in a population is by definition low, reliance on worst-case analysis alone adds unquantifiable but probably significant uncertainty to the exposure assessment in the form of (1) overestimation of risk, and (2) lack of information on exposure to the majority of receptors.

9.2.2.2 RME Scenario

For adequate protection of terrestrial receptors, knowledge of exposures typically encountered by key receptor organisms collectively is of greater value than estimates of upper-bound exposures potentially affecting a few organisms. Dose estimates and TQs were therefore developed for the more plausible RME scenario. The RME scenario uses arithmetic mean concentrations and somewhat less conservative exposure assessment assumptions to characterize ecological exposures. The RME scenario is more plausible since receptors do not spend all of their time at one specific location (e.g., at a hot spot or in areas where maximum or upper-bound concentrations occur). In fact, the home range (the area over which an organism ranges and forages) of most predatory birds and animals, such as the red-tailed hawk, barred owl, and the mink, tend to be large relative to the size of the subsites. Thus, it is more likely that receptors would be exposed to varying concentration levels, which are best represented by using some measure of central tendency (e.g., mean or median concentration data). The RME approach thus explicitly addresses one of the major sources of error in the worst-case scenario -- the failure to account for spatial integration of exposure as organisms move throughout their ranges -- by averaging exposure concentrations over the key receptors' home ranges.

To provide a perspective on the extent to which worst-case assumptions may have overestimated dose rates, terrestrial receptors were assumed to spend an equal amount of time on Ag/A and near-pile soils, and the arithmetic mean area-weighted concentration of metals in these soils was used to calculate dose estimates. Mean levels of cadmium, lead, and zinc in near-pile soils and the area-weighted mean concentration of these metals are shown in Table 9-1. As shown in this table, the mean soil exposure concentrations used in the RME scenario are about 2 to 5 times higher than the mean area-weighted concentrations. Corresponding dose estimates and TQs did not change appreciably (Table 9-1). Therefore, this source of uncertainty is expected to have a low impact on risk estimates.

Table 9-1. **Toxicity Quotients for Key Terrestrial Receptors Using the Area-Weighted Average Concentration of COCs in Ag/A and Near-Pile Soils**

Receptor and Metal	Arithmetic Mean Concentrations in Ag/A Soils (mg/kg)	Arithmetic Mean Concentration in Near-Pile Soils (mg/kg)	Arithmetic Mean Area-Weighted Concentration in Near-Pile and Ag/A Soils (mg/kg)	Toxicity Reference Value (mg/kg-day)	RME[a] Dose (mg/kg-day)	RME Toxicity Quotient[b]
RED-TAILED HAWK						
Cadmium	0.9	4.5	0.9	1.6	0.06	0.04
Lead	42	88	42.5	8.3	0.21	0.03
Zinc	157	710	162.5	111	2.9	0.03
BARRED OWL						
Cadmium	0.9	4.5	0.9	1.6	0.08	0.05
Lead	42	88	42.5	8.3	0.25	0.03
Zinc	157	710	162.5	111	3.5	0.03
MINK						
Cadmium	0.9	4.5	0.9	0.1	0.11	1
Lead	42	88	42.5	0.3	0.61	2
Zinc	157	710	162.5	4.2	13.4	3

a RME equals reasonable maximum exposure scenario.

b Toxicity quotients are dose divided by the toxicity reference value. TQs were rounded to one significant figure.

NOTE: Arithmetic Mean Concentrations for Ag/A and Near-Pile Soils are multiplied by 0.99 and 0.01, respectively and summed for area-weighted concentration.

9.2.2.3 Intake Values Used

Intake estimates used for the terrestrial receptors are based upon data available in the scientific literature and information obtained from a certified wildlife biologist. As mentioned above, the use of point estimates to define these values hinders quantitative evaluation of both their individual uncertainties and contributions to overall uncertainty. However, these impacts are expected to be relatively small.

Inhalation rates were extrapolated from data available for the rat. Although inhalation factors were subjectively determined, they are not expected to greatly affect intake estimates, since the concentration of COCs in air and, hence, the total daily intake of COCs via inhalation is small (Table 9-2). Inhalation accounted for less than 1 percent of the total daily intake of COCs for all three key terrestrial receptors evaluated. Since soil ingestion provides a greater and more variable proportion of total daily intake (2 to 6 percent for the mink and 5 to 24 percent for the raptors; Table 9-2), this parameter may have a greater potential to over- or underestimate risk than inhalation rates.

Food ingestion rates have the greatest effect on intake estimates, since ingestion of prey accounted for 93 to 96 percent of the mink's intake and 75 to 95 percent of the raptor's total intake (Table 9-2). Uncertainties associated with all intake factors could result in either an over- or an underestimation of the true dose rate, since these values are not known exactly. However, the conservative intake values chosen for this ERA would tend to result in over- rather than underestimation of dose.

9.2.2.4 Assimilation Efficiency

The efficiency of assimilating of metal burdens in prey ingested by terrestrial predators represents another source of uncertainty associated with dose estimates for terrestrial receptors. Most metals tend to accumulate in the liver and kidneys of mammals and birds (Lande, 1977; Chen *et al.*, 1977), with the exception of lead, which accumulates in bone tissue (Demay *et al.*

Table 9-2. Contribution of Inhalation and Ingestion of Water, Soil, and Prey to Total Daily Intake by Terrestrial Receptors[a]

	Intake via Inhalation (mg/kg-day)	Percent of Total Daily Intake	Intake from Water (mg/kg-day)	Water As A Percent of Total Intake	Intake from Soil (mg/kg-day)	Soil As A Percent of Total Intake	Intake from Prey (mg/kg-day)	Prey As A Percent of Total Intake
RED-TAILED HAWK								
Cadmium	2.4×10^{-6}	<1	3×10^{-4}	<1	0.01	6	0.15	94
Lead	3.5×10^{-5}	<1	3×10^{-4}	<1	0.12	24	0.38	75
Zinc	4.1×10^{-4}	<1	0.06	<1	1.1	21	4.0	78
BARRED OWL								
Cadmium	2.1×10^{-6}	<1	4×10^{-4}	<1	0.01	5	0.18	95
Lead	3.1×10^{-5}	<1	4×10^{-4}	<1	0.14	24	0.45	76
Zinc	3.7×10^{-4}	<1	0.07	1	1.2	20	4.8	79
MINK								
Cadmium	2.1×10^{-6}	<1	0.001	<1	0.004	2	0.18 (0.15 / 0.03)[b]	96 (80 / 16)
Lead	3.0×10^{-5}	<1	0.001	1	0.06	6	0.90 (0.38 / 0.52)	93 (39 / 54)
Zinc	3.6×10^{-4}	<1	0.24	1	0.5	3	15.7 (4.0 / 11.8)	95 (24 / 71)

[a] Based on the worst-case scenario.

[b] Numbers in parentheses represent the contribution from ingestion of mice and fish, respectively.

1982). In their study Demay *et al.* (1982) found that only 10 percent of the lead measured in whole-body mice samples was actually bioavailable to predators, since the bones ingested by most raptors are eliminated before digestion. While owls do regurgitate the bones and fur, all raptors do not, however. Dose estimates presented in Section 5.0 do not account for the fact that lead burdens in prey may not be available to predators. If the red-tailed hawk and the barred owl regurgitate bones and fur, the dose estimates presented in Table 5-9 are likely to overestimate the amount of lead to which these receptors are exposed.

To provide a perspective on the uncertainty associated with dose estimates for lead, intakes were calculated for the raptors assuming that only 10 percent of the lead measured in subsite mice was actually assimilated (i.e., an assimilation factor of 0.1 was included in the intake equation for lead only). Results show that worst-case TQs for lead dropped by a factor of three (Table 9-3). These data show that not accounting for assimilation is likely to overestimate risks.

9.2.2.5 Bioavailability of Metals

The bioavailability of metals is well known to be decreased by association with soils and sediments, including those in Tar Creek (McCormick and Burks, 1987). Because of this interaction with soil particles, only a fraction of the metals ingested in contaminated soil is actually available to be absorbed (and, therefore, exert toxic effects). The bioavailability and toxicity of metals varies with the physical and chemical form of the metal. Metals sorbed to particulates or those that exist in a complexed form are generally less bioavailable than metals in the dissolved form. The lack of correction of intakes for the reduced bioavailability of soil-associated metals in this assessment tends to overestimate exposure and risk estimates.

9.2.2.6 Absorbed Versus Administered Dose

Most toxicological data available for terrestrial organisms are based on administered versus absorbed doses. The administered dose approach is conservative in that it does take into

Table 9-3. Toxicity Quotients for Key Terrestrial Raptors Using a 10 Percent Assimilation Factor for Lead[a]

Receptor and Metal	Toxicity Reference Value (mg/kg·day)	Worst-Case Dose (mg/kg·day)	Worst-Case Toxicity Quotient[b]	RME[c] Dose (mg/kg·day)	RME Toxicity Quotient
RED-TAILED HAWK					
Cadmium	1.6	0.16	0.1	0.07	NA[d]
Lead	8.3	0.16 (0.50)[e]	0.02 (0.06)	0.11	NA
Zinc	111	5.2	0.05	3.5	NA
BARRED OWL					
Cadmium	1.6	0.19	0.1	0.08	NA
Lead	8.3	0.19 (0.59)	0.02 (0.07)	0.13	NA
Zinc	111	6.1	0.06	4.1	NA

[a] Assumes that raptors regurgitate bones and hair; therefore, only 10% of the total body burden of lead associated with mice is assimilated by predators.

[b] Toxicity quotients are dose divided by the toxicity reference value. TQs were rounded to one significant figure.

[c] RME equals reasonable maximum exposure scenario.

[d] RME TQs were calculated only if the worst-case TQ was greater than or equal to 1.

[e] Numbers in parentheses indicate values without incorporation of 10% assimilation factor.

account the fact that some fraction of the administered dose may not be absorbed across lung, skin, or gastro-intestinal tract barriers. If 100 percent absorption of ingested or inhaled contaminants does not occur, then the dose that actually effects a target organ or system would be lower, and dose estimates given in Section 5.0 would overestimate the true risk to terrestrial receptors. It is generally agreed, however, that the administered dose approach is reasonable and provides results that are protective of environmental health.

9.2.3 Exposure Estimation for Aquatic Receptors

Dose-based estimates for aquatic organisms was not calculated in this ERA. Instead an EPA-approved approach that accounts for water column concentrations of metals and their toxicity to aquatic organisms (Stephan *et al.*, 1985) was adopted for calculating site-specific TRVs. This widely-accepted approach is expected to account for the major pathway of toxicity to aquatic organism (i.e., assimilation across gill membranes). While other pathways of exposure, including ingestion of sediments and prey, does add to the overall dose and hence potential toxicity of metals to aquatic organisms, generally, ingestion of contaminated sediments while foraging is not a significant exposure pathway relative to the direct transfer of chemicals across the gill membrane. This assumption is predicated on the fact that most metals are bioavailable to aquatic life only when they are dissolved in the water rather than sorbed to bottom or suspended sediments (DiToro *et al.*, 1991). O'Donnell *et al.* (1985) referred to sediment as "a relatively large compartment with low availability," versus water, "a smaller compartment of higher availability." Sediments may release metals to the interstitial water thus providing a source of metals uptake. This uptake would probably be equivalent to uptake from the water column, however (O'Donnell *et al.*, 1985).

The Stephan *et al.* (1985) approach includes the incorporation of chronic toxicity values (as opposed to the singularly more conservative NOAELs and LOAELs) for the four most sensitive aquatic species expected to occur within a particular site. By using the most sensitive species, both the NOAEL/LOAEL and the toxicity attributable to the ingestion of food organisms or sediments are compensated. The extent of compensation, however, cannot be estimated.

9.3 UNCERTAINTY ASSOCIATED WITH ECOLOGICAL ENDPOINTS

In most cases, assessment endpoints cannot be directly measured, so measurement endpoints are identified that can be quantitatively related to an assessment endpoint. Various measurement endpoints were selected to identify stressor effects on prey and predator populations including the fish condition factor, predator distance from mine-related areas, results of the toxicity assessment, and field survey results.

Fish condition factors were used as an indicator of the general well-being of fish collected onsite considering their relative length and weight. It is recognized that these factors are subject to the influence of numerous parameters other than direct influences of metals toxicity, including food availability, disease, general stress, and water temperature. Although the fish appeared to be in good condition based on condition factor results, other chronic effects directly associated with metals toxicity or with other physical limiting factors (e.g., habitat limitations) might not be detected by this approach. That is, condition factors greater than one, such as those observed in on-site fish, do not conclusively indicate that adverse chronic effects such as suppressed growth or reproduction are not occurring.

Another aquatic measurement endpoint considered was the age classes of fish inhabiting on-site ponds. Eighty-five percent of the green sunfish collected from subsite ponds were apparently of Age I year class, suggesting that natural reproduction was occurring. Conversely, the lack of fish in older age classes could indicate higher than normal mortality, possibly attributable to various stresses, including harvesting, predation, and metals toxicity. Since different sampling methods were used with varying success, it cannot be concluded that the observed decrease in age classes is indicative that the water is having a toxic effect. It is possible that the observed decreases in age classes is indicative of the incomplete sampling approach used rather than a toxic effect.

9.3.1 Uncertainty Associated with the Field Data

Both field surveys and exposure assessment results supply endpoints that are useful in assessing potential effects due to metals toxicity. While the limited field investigations conducted did not reveal adverse effects, the possibility that subtle chronic effects might be occurring could not be ruled out. In order to make the best use of the most data, toxicity assessment and field survey results were both used in this ERA to determine if adverse effects on ecological receptors were evident or possible. This approach is in accordance with EPA (1992) guidance, which states that, "generally speaking, field data, monitoring data, and toxicity testing of contaminated media are more useful and reliable than literature estimates." Uncertainties associated with the toxicity assessment are discussed below.

Plots of predator distance from mine-related areas formed a smooth curve, suggesting that predators did not exhibit an obvious pattern of avoidance of mine-related areas. The distance between predator sightings and the nearest mine waste area may be influenced by factors other than mining, such as the presence of towns, roads, etc. Although the fact that there was no obvious pattern of avoidance, as measured by predator distance from mine-related areas, mining stresses could cause avoidance in some animals, attract others, and have no effect on yet another group. The distance between an animal and the nearest disturbed mined area may not be a valid endpoint for all (particularly more mobile) species.

9.3.2 Uncertainty Associated with the Toxicity Assessment

There is considerable uncertainty associated with the calculation and interpretation of TQs for all receptor groups evaluated in this ERA. Although TQs less than 1 generally indicate that observable effects are not probable, there is no regulatory guideline for an acceptable ecological TQ level. A particularly difficult area of interpretation is when TQs are only slightly higher or lower than one. In these cases, other relevant information, such as field data and the estimated uncertainty bounds on the dose and toxicity values, need to be considered before final conclusions concerning potential adverse effects can be made.

A TQ of 1 indicates that the measured concentration exceeds a concentration that caused some impact in a certain test species under a given set of experimental conditions. Within a given ecosystem, loss of some individuals in a receptor group is acceptable if the entire population will not be adversely impacted. Barnthouse and Suter (1986) state that "ecological risk assessments used in decision making should be based, to the greatest extent possible, on objective estimates of ecological damage (e.g., probabilities of population extinction or reductions in abundance of plants and animals)." Hence, a TQ greater than 1 may not necessarily mean that effects observed in the laboratory are likely to occur in the field or that exposures in the field are significant enough to cause population-level effects. Furthermore, there is not a linear (one-to-one) relationship between increasing TQ values and the probability of adverse effects. For example, a TQ of 2 does not imply that adverse effects are twice as likely to occur in a given ecologic group as a TQ of 1. The intent of the TQ approach is to provide a continuous quantitative scale so that evaluation and rank ordering of potential impacts can be assessed. Thus, given that the goal of this ERA is to assess population-level effects, the use of an arbitrary threshold of 1 for evaluating the potential for adverse effects on exposed receptor populations may overestimate population-level risks, although the degree to which potential risks are overestimated cannot be determined.

9.3.2.1 Terrestrial Receptors

There are two main sources of uncertainty associated with calculating TQs for terrestrial receptors: uncertainty associated with dose estimates and the uncertainty associated with derivation of TRVs. Uncertainty associated with dose estimates was discussed in Section 9.2.2. Uncertainty is associated with the derivation of the TRVs used to calculate TQs, since toxicological benchmark data were not available for the key species of concern. Considerable uncertainty exists in the toxicological values used to calculate TQs for terrestrial receptors both from the original studies (e.g., accuracy of observations recorded, variability in dose, exposure conditions, and routes of application, etc.) and from the application of uncertainty factors. Standardized uncertainty factors have not been developed for the purpose of deriving protective TRVs for ecologic receptors. Adjustment factors addressing some of the same sources of uncertainty considered in the derivation of human reference doses have therefore been developed for use in this ERA. This approach is intended to ensure that the TRVs derived are adequately protective of wildlife.

The application of UF1 to the benchmark value assumes that the key receptor species are more sensitive than the test organism. This assumption could over- or underestimate TQs, since there is no evidence to determine if the hawk, owl, and mink are equally sensitive, less sensitive, or more sensitive than laboratory test animals. UF2 was used to extrapolate data from acute or subchronic studies to chronic toxicity estimates. This factor is predicated on the expectation that continued exposure at a given dose rate will produce increasing effects that may not be evident after subchronic exposure. Chronic exposures are defined as those lasting at least 10 percent of the animal's lifetime. In cases where the experimental exposure duration was less than 10 percent of the animal's lifetime, UF2 was set equal to three. If the exposure duration was greater than 10 percent of the animal's lifetime, UF2 was set equal to one. This approach may be conservative in that there is no evidence to conclude that when exposure duration was greater than 10 percent of lifetime that UF2 should not be negative. While the precise impact these assumptions have on the resulting TQs cannot be accurately quantified, the evidence presented here suggests that the resulting TQs may be conservative.

It is possible that additional uncertainty factors could be considered in deriving TRVs for terrestrial receptors. For example, an additional uncertainty factor, UF3, could be used to account for potential differences between laboratory test conditions and actual field conditions, while UF4 could be used to extrapolate toxicity results obtained for a single species to those suitable for a community or ecosystem. Typically, UFs ranging from 1 to 10 are used to extrapolate from laboratory to field conditions. Originally, UF1 and UF2 were set to three based on the scientific rationale provided in Section 7.3. A value of 10 probably represents a very, perhaps overly, conservative estimate. To provide a perspective on how changing the UFs used to derive TRVs for terrestrial receptors could alter TQs, each of the four multiplicative uncertainty factors (UF1 through UF4) was set equal to three based on the assumptions outlined in Section 7.3. These new TRVs are shown in Table 9-4. Using the same "best" toxicity data

Table 9-4. Revised Toxicity Reference Values for Terrestrial Receptors Using Four Uncertainty Factors Each Set Equal to Three

Metal	Literature Value Selected (mg/kg-day)	Uncertainty Factor	Toxicity Reference Value (mg/kg-day)
RED-TAILED HAWK			
Cadmium	14.6	81[a]	0.2
Lead	25	27[b]	0.9
Zinc	1000	81	12.4
BARRED OWL			
Cadmium	14.6	81	0.2
Lead	25	27	0.9
Zinc	1000	81	12.4
MINK			
Cadmium	0.95	81	0.01
Lead	3	81	0.04
Zinc	38	81	0.5

[a] UF1 (interspecies extrapolation) = 3, UF2 (acute-to-chronic extrapolation) = 3, since the test duration was not longer than 10% of the animal's lifetime, UF3 (laboratory to field conditions) = 3, and UF4 (single organism to population) = 3.

[b] UF1 (interspecies extrapolation) = 3, UF2 (acute-to-chronic extrapolation) = 3, since the test duration was not longer than 10% of the animal's lifetime, UF3 (laboratory to field conditions) = 3, and UF4 (single organism to population) = 3.

Table 9-5. Revised Toxicity Quotients for Key Terrestrial Receptors Using Four Uncertainty Factors Each Set Equal to Three

Receptor and Metal	Toxicity Reference Value (mg/kg-day)	Worst-Case Dose (mg/kg-day)	Worst-Case Toxicity Quotient[a]	RME[b] Dose (mg/kg-day)	RME Toxicity Quotient
RED-TAILED HAWK					
Cadmium	0.2	0.16	0.8	0.07	NA[c]
Lead	0.9	0.507	0.6	0.26	NA
Zinc	12.4	5.2	0.4	3.5	NA
BARRED OWL					
Cadmium	0.2	0.19	1	0.08	0.4
Lead	0.9	0.59	0.7	0.31	NA
Zinc	12.4	6.1	0.5	4.1	NA
MINK					
Cadmium	0.01	0.19	19	0.11	11
Lead	0.04	0.96	24	0.6	15
Zinc	0.5	16.6	33	13.6	27

[a] Toxicity quotients are dose divided by the toxicity reference value. TQs were rounded to one significant figure.

[b] RME equals reasonable maximum exposure scenario.

[c] RME TQs were calculated only if the worst-case TQ was greater than or equal to 1.

listed in Table 7-4, revised chronic TRVs and TQs for the raptors and the mink are shown in Table 9-5. These results show that adding two additional UFs would increase TQs by a factor of eight to 11. Worst-case TQs for the raptors remain below 1, with the exception of that for cadmium and the owl, which equals 1. The RME TQ for cadmium and the owl is 0.4. Worst-case TQs for the mink range from 19 for cadmium to 33 for zinc, while RME TQs range from 11 to 27. The above calculations indicate that the use of two uncertainty factors to estimate risks to communities versus individuals may underestimate risk if all UFs are positive. It is possible, however, that values for some UFs could be negative, which would tend to overestimate risk estimates.

9.3.3.2 Aquatic Organisms

The primary source of uncertainty in calculating TQs for aquatic receptors is the derivation of the TRVs. Since an acute-to-chronic ratio is not available for cadmium, the estimated FAV for cadmium was used to predict the chronic maximum allowable toxicant concentration (MATC) (which is equivalent to the FCV) using Suter *et al.*'s (1987) regression model. Use of the MATC as the chronic endpoint is conservative, since it is defined as the effects threshold at or below which adverse chronic effects are not expected to occur (Suter, 1992). Thus, it is possible that concentrations above the MATC level could be tolerated without significant population-level effects. The exact concentration that would not be expected to cause population-level effects cannot be precisely calculated; however, the definition of an MATC suggests that TQs greater than 1 do not necessarily imply the occurrence of adverse effects in exposed populations. On the other hand, as with any predictive regression equation, the calculated MATC could be higher or lower than the true value. Therefore, the resulting TQs calculated for cadmium could under- or overestimate the potential that aquatic organisms might experience adverse effects. The degree to which the TQ may under- or overestimate such effects, however, cannot be quantified.

For the remaining metals, there is uncertainty associated with and conservatism built into the derivation of the FAV used to calculate TRVs for aquatic receptors. For example, the FAV

Table 9-6. Sensitive Species Used to Derive TRVs for the Metals of Concern

Cadmium	Lead	Nickel	Mercury	Zinc
Fathead Minnow	Snail	Mayfly	Midge	Snail
Isopod	Fathead Minnow	Fathead Minnow	Crayfish	Snail
Banded Killifish	Bluegill	Pumpkinseed	Guppy	Isopod
Snail	Guppy	Guppy	Crayfish	Fathead Minnow

is based on the four most sensitive species that are known or suspected to occur onsite. The species used to estimate TRVs for each metals are listed in Table 9-6. Centrarchids (sunfish), ictalurids (catfish), tubificids (aquatic earthworms), and chironomids (midges), which are most likely to occur in subsite waters, are not represented for cadmium or zinc. The fact that the species used for criteria derivation are generally more sensitive than subsite species will tend to overestimate risk estimates.

Furthermore, in calculating the fathead minnow *(Pimephales promelas)* GMAV's for cadmium, the EPA selected the most sensitive life-stage of the fathead minnow. Although data from 23 toxicity tests are presented in the Ambient Water Quality Criteria for Cadmium (USEPA, 1984a), only six were selected by EPA to calculate GMAVs. All six data were obtained from toxicity tests involving fathead minnow fry. The toxicity data ranged from 0.01 mg/L to 0.05 mg/L (geometric mean = 0.026 mg/L). The remaining 17 data were obtained from other life-stages, including adult, and ranged from 0.63 mg/L to 73.5 mg/L (geometric mean = 4.62 mg/L). These data for life stages other than the most sensitive life stage (fry) were "not used in calculations because data were available for the more sensitive life stage" (USEPA, 1984a). Therefore, the estimation of the cadmium FAV would appear to be quite conservative. On the other hand, the FAVs derived for the other metals could underestimate the potential toxicity of subsite waters to early life stages, since these FAVs are not based solely on data for sensitive life stages.

The calculation of TRVs for aquatic organisms was derived from the EPA guidance document for calculating site-specific TRVs (Stephan *et al.*, 1985). In so doing, the Genus Mean Acute Values (calculated by EPA) were used for the four most sensitive genera potentially occurring on the site. The primary invertebrate species selected included rotifers, crayfish, midges, and snails. Amphipods were excluded from this list because these organism are not adapted for withstanding drought conditions and other adverse environmental conditions (i.e., such as those occurring onsite) (Pennak, 1989). Based on the Stephan *et al.* (1985) approach, by incorporating Gammarus in the calculations, the only aquatic TRV that would be affected is lead. The on-site stream segments where the TRV for lead could substantially exceed one would be Willow Creek and Tar Creek, where background concentrations of lead exceed on-site concentrations. Therefore, the inclusion of Gammarus in the calculation of site-specific TRVs are not expected to substantially affect the overall conclusions of aquatic risk assessment.

Hardness

TRVs for aquatic organisms inhabiting subsite waters were calculated using a single site-wide hardness value (geometric mean equals 190 mg $CaCO_3$/L) instead of values measured at the individual stream stations. This approach significantly reduced the calculations necessary to derive a TRV, thus simplifying the report. TRVs vary directly with hardness for most metals, and the site-wide hardness value was somewhat less than most main-stem stream water hardnesses. Evaluation of TQs on a station-by-station basis could be used to evaluate localized effects, rank specific stream segments on an ecological risk basis, or to derive in-stream compliance standards. TRVs and TQs for each mainstem sample station within each stream were calculated using the specific hardness and concentration data for that station. These data are presented in Tables 9-7 through 9-9 and summarized in Table 9-10.

When the mean site-wide hardness value was used for Tar Creek (Table 8-1), only the TQ for zinc exceeded 1 (upper-bound and mean TQs = 11 and 6). When the station-specific hardness and concentration data were used (Table 9-7), TQs for zinc ranged from 0.4 at TC-1 to 5 at TC-2. TQs for all other COCs in the downstream (mining-affected) stations were less than or equal to 1, except for cadmium at TC-3 (2) and zinc at TC-3 (4). The fact that the TQs for lead at TC-1, the station upstream of mining-related inputs, also exceeds 1 indicates that nonmining-related sources of lead are influencing the water quality of Tar Creek. Calculations based on station-specific hardness and concentration data indicate that levels of cadmium in the downstream sections of Tar Creek may also cause adverse effects in aquatic organisms in addition to zinc.

Similar results for obtained for lead in Willow Creek (Table 9-8). The TQs for lead at the upstream station (WC-1) and at Station WC-2 were both 1, indicating that levels of lead in

Table 9-7. TRVs and TQs for Tar Creek Aquatic Receptors Using Station-Specific Hardness and Concentration Data

STATION TC-1

Metal	FAV Adjusted to 41 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration	Mean TQ
Cadmium	31.9	NA	2.2	0.002	0.001	0.5
Mercury	17	3.73	4.56	0.005	ND (.0001)	.02
Nickel	3,832	17.99	213	0.21	ND (0.02)	0.09
Lead	199	51.29	3.88	0.004	0.03	7.5
Zinc	856	2.21	387	0.387	0.18	0.4

STATION TC-2

Metal	FAV Adjusted to 268 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration	Mean TQ
Cadmium	229	NA	10.5	0.010	.012	1
Mercury	17	3.73	4.56	0.005	ND (0.0001)	.02
Nickel	16,814	17.99	934	0.93	.027	0.03
Lead	2,171	51.29	42.3	0.04	.031	0.8
Zinc	4,199	2.21	1,900	1.9	8.8	5

STATION TC-3

Metal	FAV Adjusted to 366 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration mg/L	Mean TQ
Cadmium	325	NA	13.6	0.014	0.022	2
Mercury	17	3.73	4.56	0.005	ND (0.0001)	.02
Nickel	21,886	17.99	1,216	1.21	0.03	0.02
Lead	3,229	51.29	62.9	0.06	0.075	1
Zinc	5,467	2.21	2,473	2.4	9.6	4

ND - not detected; value in paranthesis represents one-half the reporting limit

Table 9-8. TRVs and TQs for Willow Creek Aquatic Receptors Using Station-Specific Hardness and Concentration Data

STATION WC-1

Metal	FAV Adjusted to 73 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration (mg/L)	Mean TQ
Cadmium	53	NA	3.61	0.004	0.0004	0.1
Mercury	17	3.73	4.56	0.005	ND (0.0001)	0.02
Nickel	5,595	17.99	311	0.31	ND (0.02)	0.06
Lead	415	51.29	8.1	0.008	0.01	1
Zinc	1,396	2.21	631	0.63	0.15	0.2

STATION WC-2

Metal	FAV Adjusted to 103 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration (mg/L)	Mean TQ
Cadmium	78	NA	4.8	0.005	0.002	0.4
Mercury	17	3.73	4.56	0.005	ND (0.0001)	0.02
Nickel	7,487	17.99	416	0.42	ND (0.02)	0.05
Lead	643	51.29	12.5	0.013	0.013	1
Zinc	1,868	2.21	845	0.85	0.64	0.8

STATION WC-3

Metal	FAV Adjusted to 245 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration (mg/L)	Mean TQ
Cadmium	207	NA	9.8	0.010	0.001	0.1
Mercury	17	3.73	4.56	0.005	ND (0.0001)	0.02
Nickel	15,585	17.99	866	0.86	0.02	0.02
Lead	1,937	51.29	37.8	0.04	0.015	0.4
Zinc	3,892	2.21	1,761	1.76	0.62	0.4

ND = not detected; value in parenthesis represents one-half the reporting limit.

Table 9-9. TRVs and TQs for Spring Branch Aquatic Receptors Using Station-Specific Hardness and Concentration Data

STATION SB-1

Metal	FAV Adjusted to 354 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration (mg/L)	Mean TQ
Cadmium	314	NA	13.3	0.013	0.12	9
Mercury	17	3.73	4.56	0.005	0.0001	0.02
Nickel	21,295	17.99	1,184	1.18	0.037	0.03
Lead	3,095	51.29	60.3	0.06	.011	0.2
Zinc	5,315	2.21	2,405	2.4	13.9	6

STATION SB-2

Metal	FAV Adjusted to 425 Hardness (ug/L)	Acute-to Chronic Ratio	Final Chronic Value (ug/L)	TRV (mg/L)	Station Concentration (mg/L)	Mean TQ
Cadmium	385	NA	15.4	0.015	0.054	4
Mercury	17	3.73	4.56	0.005	ND (0.0001)	0.02
Nickel	24,836	17.99	1,380	1.38	0.028	0.02
Lead	3,906	51.29	76.2	0.076	0.004	0.05
Zinc	6,205	2.21	2,808	2.8	7.5	3

ND = not detected; value in parenthesis represents one-half the reporting limit

Table 9-10. Summary of Toxicity Quotients for COCs Based on Final Acute Values Adjusted for Average Hardness at Each Sampling Stations where TQs Exceed 1

Sample Station and COCs[a]	FAV Adjusted to Average Hardness (mg/L)	Toxicity Reference Value (mg/L)	Arithmetic Mean Concentration[b] (mg/L)	Toxicity Quotient Based on TRV and the Mean Concentration
SPRING BRANCH, SB-1 (UPPER STATION, n = 3)				
Cadmium	.314	0.013	0.12	9
Zinc	5.3	2.4	13.9	6
SPRING BRANCH, SB-2 (LOWER STATION, n = 5)				
Cadmium	.385	0.015	0.05	4
Zinc	6.2	2.8	7.5	3
TAR CREEK, TC-1 (UPSTREAM STATION, n = 3)				
Lead	199	0.004	0.03	7.5
TAR CREEK, TC-2 (MID-REACH STATION, n = 3)				
Cadmium	.229	0.01	0.01	1
Zinc	4.2	1.9	8.8	5
TAR CREEK, TC-3 (LOWER STATION, n = 3)				
Cadmium	.325	0.01	0.02	2
Zinc	5.5	2.5	9.6	4

[a] Data are total recoverable values, see RI Appendix D.

[b] COCs shown are those whose TQ exceeded 1 using station-specific hardness to adjust FAV.

Willow Creek are dominated by nonpoint sources unrelated to mining. TQs for all remaining COCs in Willow Creek were less than 1, which indicates that adverse effects to aquatic organisms inhabiting this stream are not probable.

Results based on the site-wide mean hardness for Spring Branch indicated that adverse effects from exposure to upper-bound and mean concentrations of cadmium and zinc were possible (Table 8-1); upper-bound TQs for cadmium and zinc were 2 and 9, respectively, while mean TQs for cadmium and zinc were 10 and 7, respectively. When site-specific hardness and concentration data were used (Table 9-9), mean TQs for cadmium at Station SB-1 and SB-2 were 9 and 4, respectively, while the mean TQs for zinc were 6 and 3, respectively. These results show that using the site-wide mean hardness value overestimated risks for exposure to cadmium at SB-2 and to zinc at both downstream stations. Potential risks to organisms at Station SB-1 from exposure to cadmium were underestimated. The overall conclusion remains the same; potential adverse effects from exposure to cadmium and zinc are possible.

9.4 OTHER FACTORS AFFECTING METALS TOXICITY TO AQUATIC RECEPTORS

Scientific investigations have determined that several factors may affect the actual toxicity of metals. The quality of surface waters within the BS/T subsites seems to support this theory as exemplified in the following discussions. Measured concentrations of some site-related metals at some locations, especially zinc, suggest that resident fish populations should be nonexistent or severely stressed, yet actual field data demonstrate the presence of aquatic organisms. The site-specific TRV for zinc adjusted for a hardness of 190 mg $CaCO_3$/L is 1.4 mg/L, while the arithmetic mean concentration of zinc in on-site surface water bodies ranges from 0.6 mg/L in Willow Creek to 9.9 mg/L in Spring Branch (3 to 50 times the site-specific value).

Despite these relatively high levels of zinc, multiple species of fish representing various age classes are known to exist in on-site streams. The fact that these exceedances coincide with apparently normal natural populations may indicate that the site-specific TRVs developed in Section 7.0 may be overprotective for this ecosystem, and/or that other factors may be influencing the toxicity and/or bioavailability of the metals present in on-site surface water bodies. Such factors include (but may not be limited to) the sensitivity of the species present onsite, the metals' speciation status, evolutionary or acquired tolerance, antagonism among the metals, water hardness, and the frequency of occurrence of toxic conditions. Conversely, other factors could increase the toxicity of site-related metals. For example, exposure to a mixture of COCs by key receptors could increase the toxicity of individual metals synergistically. Similarly, various environmental processes could alter the metal into a form that is more soluble and therefore generally more bioavailable.

9.4.1 Acclimation and Tolerance

Tolerance can be achieved by physiological acclimation during low levels of exposure and/or by genetically based mechanisms. Physiological tolerance is not inherited, and such individuals lose their tolerance when transferred to unpolluted environments, while genetic tolerance is inherited by offspring regardless of whether they are reared in polluted or nonpolluted environments (Mulvey and Diamond, 1991). An organism's tolerance to environmental stressors is greatly affected by the environmental conditions previously experienced, i.e., prior exposure to the stressors (Chapman, 1985). Acclimation of fish populations resulting in increased tolerance to concentrations of zinc exceeding AWQC has been documented in the literature (Spehar, 1978; Chapman, 1978, 1985; Sinley *et al.*, 1974; Rahel, 1981). Melancon and Miller (1984) conducted in situ bioassays at Prickly Pear Creek, Montana, and reported decreased mortality in resident brook trout and hatchery brook trout exposed to effluent spiked with zinc and copper that were allowed to acclimate 7 to 10 days in the Creek. Chapman (1985) reported 90 percent of early life stage chinook salmon previously acclimated to 0.51 mg/L zinc for five months survived a 96-hr exposure to 1.4 mg/L zinc (the LC_{50}). Similar acclimation to elevated zinc concentrations was reported by Sinley *et al.* (1974). In a 21-month test, rainbow trout exhibited up to a four-fold increase in their tolerance to zinc when exposed as eggs to concentrations of zinc ranging from 0.01 to 0.55 mg/L. Spehar (1976) reported that adult flagfish (Cyprinodontidae, *Jordanella floridae*) showed a three-fold increase in tolerance to zinc when exposed as eggs and fry. These results demonstrate that acclimation to zinc during early life stages can result in an increased tolerance to zinc by some species.

The fact that the myriad of fish species inhabiting on-site streams and ponds have been exposed to zinc levels above AWQC levels during early life stages suggest that these populations may have acclimated to the relatively consistent (long-term) presence of elevated levels of zinc in on-site surface water bodies. Since the potential for acclimation by resident fish was not directly evaluated, the degree to which acclimation may ameliorate the toxicity of metals in subsite surface water bodies cannot be determined. Furthermore, while the studies cited herein were not conducted using fish species that are known or suspected to occur onsite, they do indicate that acclimation by some species can increase resistance to the toxicity of some metals.

Elevated metals concentrations have existed at the BS/T subsites at varying intensities for the past 80 to 100 years. Populations of aquatic organisms that have continued to exist and reproduce within the subsites throughout these years have been subjected continually to these conditions and, as a result may have acquired a certain level of tolerance. Tolerance can be broadly defined as "the ability of an organism to cope with the stress associated with exposure to metal concentrations that are inhibitory or lethal to nontolerant individuals" (Mulvey and Diamond, 1991). Although the presence of various fish species in surface water bodies with elevated metals concentrations suggests that such genetically-acquired tolerance to metals of the subsites may be occurring, site-specific studies to examine this phenomenon were not conducted. If fish at the site have developed tolerance (acquired or inherited), then the TRVs used in this assessment will lead to overestimation of risk.

9.4.2 Metals Speciation and Bioavailability

Metal speciation and bioavailability are also important in determining the toxicity of metals to aquatic organisms. It is widely accepted that metals have highly variable toxicity due to their interactions with other materials present in the water (Chapman, 1985). The formation of less toxic metal complexes can account for the diminution of metals toxicity in natural waters (Chapman, 1985). High concentrations of some metals can be tolerated by aquatic organisms if the metals are bound or complexed to particulates in the water. Factors affecting metal speciation and chemical form include pH, hardness, alkalinity, suspended solids content, the presence of organic and inorganic ligands in the water, and oxidation reduction potential. For example, at a pH of 6.0, zinc exists as free ion (98 percent) and as zinc sulfate (2 percent), while at a pH of 9.0, zinc occurs mainly as a monohydroxide ion (78 percent), as zinc carbonate (16 percent), and as free ion (6 percent) (USEPA, 1987). Generally, waters with higher alkalinities tend to result in the formation of insoluble zinc carbonate and hydroxide compounds that are not readily absorbed by most aquatic species (USEPA, 1987). Furthermore, Allen *et al.* (1980) reported that the toxicity of zinc was not related to total metal concentration but to the predicted free (ion) metal concentration. While the references cited herein did not specifically address the BS/T subsites, they do suggest that speciation and/or complexation of metals could ameliorate their toxicity there as well. Since the speciation of metals in subsite waters was not determined, the degree to which this phenomenon may influence metals toxicity cannot be determined.

9.4.3 Effects of Exposure to Mixtures of Metals

The cumulative effects of simultaneous exposure to multiple metals by terrestrial and aquatic receptors was evaluated, since it is possible that the cumulative effects of simultaneous exposure to multiple metals may result in the mixture being more toxic than exposure to a single metal. Cadmium tends to act antagonistically with zinc in many plants and animals (both terrestrial and aquatic), while lead tends to act synergistically with cadmium and zinc. For example, Weis and Weis (1991) found that the presence of zinc increased the viable hatch of herring eggs in systems with 5.0 mg/L cadmium. Although all metals (i.e. copper, manganese, iron, etc.) within the terrestrial and aquatic ecosystems may exhibit cumulative effects on organisms, it is assumed that cadmium, lead, and zinc represent the majority of toxic affects.

If the effects of cadmium and lead and lead and zinc are assumed to be additive, the cumulative worst-case TQ (Table 8-3) for the raptors (determining by summing the individual TQs for cadmium and lead and lead and zinc) would still be well below 1. The cumulative worst-case TQ for the mink would be 7. If all COCs were assumed to be additive, cumulative mean TQs would range from less than 1 for the raptors to 9 for the mink. Similarly, the cumulative worst-case TQs for aquatic receptors exposed to cadmium, lead, and zinc inhabiting subsite ponds (Table 8-2), onsite streams, the Spring River (Table 8-1), and the Neosho River (Table 8-1) would be 11.5 (BP-1), 1.7, 1.4, and 0.5, respectively.

Conversely, if cadmium and zinc do act antagonistically, TQs for the mink (determined by subtracting the TQs for cadmium and zinc) could be equal to 1. If cadmium and zinc are assumed to be antagonistic, the TQ for Spring Branch would be 3.

Uncertainty is also associated with the fact that only cadmium, lead, and zinc were evaluated for key terrestrial receptors even though additional compounds were defined as COCs. Whole body fish and mice samples were analyzed for cadmium, lead, and zinc only, as data available at the time of sampling suggested that these chemicals were the most abundant and potentially toxic of the mining-related chemicals that might accumulate in ecological receptors. No other metals were detected above background concentrations in subsite streams. The presence of addition metals including cobalt, iron, manganese, mercury, nickel, and silver in subsite ponds suggests that terrestrial receptors could ingest other metals in addition to cadmium, lead, and zinc if subsite ponds were used as a drinking water source. Since terrestrial receptors are expected to obtain only a small amount of their total daily water needs from subsite ponds given the other sources of water, the dose estimates presented in Section 5.0 are not expected to be substantially underestimated.

Mercury and copper were identified as COCs for air, since these compounds were detected in the fine particle fraction of either chat or flotation tailings upon which the modeling estimates are based. Exclusion of copper and mercury as COCs in air is not expected to substantially alter dose estimates, since the inhalation pathway was minor relative to ingestion of prey. It is possible that mice could have accumulated cobalt and manganese from subsite soils as well. Exclusion of cobalt and manganese from prey is not expected to substantially increase risk estimates, since manganese and cobalt are not particularly toxic to mammals. Some mammals can tolerate up to 1000 mg/kg manganese per day without experiencing adverse effects, while rats can tolerate from 25 to 250 mg/kg-day of cobalt (Venugopal and Luckey, 1978). Nevertheless, assuming that terrestrial receptors were exposed to cadmium, lead, and zinc only may underestimate their true risk.

9.5 PHYTOTOXICITY

The phytotoxicity data used in the ERA are concentrations that researchers estimated caused some reduction in crop yield. The mean value was assumed to be more susceptible to the number of tests the researchers conducted and the concentration of metal used in a given study, which has the effect of skewing the data in the direction of most research results. For example, if the researchers conduct studies primarily with high metal concentrations resulting in high crop yield reductions, then the resulting mean metal concentration would be high and would not be a good indicator of the average or low toxicity value for that metal. Hence, the median value was used, as it was considered more representative of the range of data shown to be phytotoxic.

Use of the median phytotoxic concentration could under- or overestimate the true phytotoxic potential of subsite soils. To provide a perspective on the uncertainty associated with using the median value, the minimum and median phytotoxicity numbers are shown in Table 9-11. These data show that the median phytotoxic concentration is about two to eight times higher than the minimum phytotoxic concentration for cadmium, copper, lead, and zinc.

Minimum and median phytotoxic concentration data were compared to the mean and upper-bound concentration of COCs in Ag/A and near-pile soils (Section 8.0). Use of Ag/A soil data is appropriate since crops are currently being grown in Ag/A soils and the species used in most of the phytotoxicity studies are agronomic. Results of direct comparison of the phytotoxicity data with near-pile soil concentrations are more uncertain, since toxicity data are not available for onsite non-agronomic plant species. If the minimum phytotoxic concentration is compared to the upper-bound concentration of COCs in near-pile soils, TQs exceed 1 for zinc (17). If the mean concentration of COCs is used, TQs are greater than 1 for zinc (12). Use of the median phytotoxic concentration results in TQs for zinc less than 5.

Table 9-11. Phytotoxicity of Metals in Near-Pile Soils - Agronomic Species

Metal	Near-Pile Soils				
	Minimum Phytotoxic Concentration (mg/kg dry weight)	Median Phytotoxic Concentration (mg/kg day weight)	Upper-Bound/Mean Concentration (mg/kg dry weight)	Upper-Bound/Mean Toxicity Quotient Based on the Minimum Phytotoxic Value[a]	Upper-Bound/Mean Toxicity Quotients Based on the Median Phytotoxic Value[b]
Cadmium	5	40	6.6/4.5	1/0.9	0.2/0.1
Cobalt	24	NA[c]	19.8/15.6	0.8/0.7	NA
Lead	100	250	113.9/88.1	1/0.9	0.5/0.4
Manganese	1500	NA	1287/947	0.9/0.7	NA
Zinc	60	240	996/710	17/12	4/3

[a] Toxicity quotient equals the concentration in near-pile soils divided by the minimum phytotoxic concentration. All values are rounded to one significant figure.

[b] Toxicity quotient equals the concentration in near-pile soils divided by the median phytotoxic concentration. All values are rounded to one significant figure.

[c] Not applicable.

Potential toxic effects of metals on plants must be put into perspective by considering the following points: (1) TQs based on the median zinc phytotoxic concentration were not substantially elevated above 1; (2) near-pile soils are typically vegetated with a well-defined organic layer that should reduce the plant-available concentration; (3) near-pile soils represent less than 4 percent of the total subsite land area; and (4) toxicity data are not available for non-agronomic species that are likely to inhabit near-pile soils. Naturally-occurring species, and especially invader (weedy) species in mineralized areas, tend to be more tolerant of heavy metals than most agronomic plants (Adriano, 1986). Moreover, plant-available fractions of the total zinc concentration in near-pile samples determined from onsite monitoring data collected during the RI ranged from 2 to 55 percent, with a mean of 16.9 percent. Data on plant-available concentrations that cause toxic symptoms are limited, as most studies use total recoverable concentrations to evaluate toxic effects. The difference in heavy metal tolerance between agronomic species and invader species is reflected in their distribution on the site. Near-pile areas contain invader or weedy species that include lambsquarter (*Chenopodium album*), sheep sorrel (*Rumex acetosella*), redtop (*Agrostis gigantea*), broomsedge (*Andropogan virginicus*), switchgrass (*Panicum virgatum*), common ragweed (*Ambrosia artemisiifolia*), and pigweed (*Amaranthus spp.*). These data taken as a whole indicate the existing levels of zinc in near-pile soils are not expected to cause adverse population effects to the weedy species that currently occupy the area, but may be marginally phytotoxic to agronomic species should they be planted in near-pile areas..

10.0 RISK CHARACTERIZATION

Risk characterization involves estimating the magnitude of potential health risks and making summary judgments about the nature of potential adverse impacts to ecological receptors. This section evaluates potential adverse effects to key terrestrial and aquatic species associated with exposure to COCs at the subsites using two integrated approaches. Risks were assessed by comparing the measured or estimated exposure levels (i.e., dietary intake or exposure concentration) with chronic toxicity values as described in Sections 7.0 and 8.0. Field survey and other data available in the literature were used as a comparison for interpreting the predictive (quantitative) results. Thus, both the general biota survey (field) results and the quantitative toxicological comparisons were used to obtain a realistic assessment of potential impacts.

10.1 AQUATIC RECEPTORS

The aquatic habitat of the BS/T subsites can be categorized into two general types of surface waters: (1) tailings ponds and collapsed subsidence pits, and (2) ephemeral streams. While both water systems can be affected by rainfall and drought, stream habitats are particularly vulnerable to the effects of high and low flow conditions. During low flow periods, for example, elevated water temperatures and low oxygen levels can occur, while during high flow periods, silty substrate can shift and increase suspended solids levels in the streams. Heavy siltation can also be caused by the erosion of material from the surrounding agricultural and nonvegetated mill waste areas. When temporary in-stream impoundments (created to provide a source of drinking water for agricultural livestock) fail, heavy sediment loads can be introduced into the streams. Both of these conditions may stress local aquatic populations. However, where suitable stream habitat exists, a variety of fish species were observed to be both self-sustaining and in relatively good condition.

Similarly, the deeper on-site tailings and subsidence pit ponds provide a somewhat more stable habitat and aquatic biota surveys indicated a diversity of fish species in apparent good condition. Overall, the aquatic survey results revealed that exposed fish populations in most on-site surface water bodies were not experiencing acute impacts, as no obvious signs of chemical or physical stress, such as those noted in the FWS *Hazard Reviews*, were evident in the fish collected. It was necessary, however, to evaluate the possibility that aquatic organisms inhabiting subsite streams and ponds could be experiencing chronic effects from exposure to site-related metals. Chronic effects were assessed by comparing measured surface-water concentrations with chronic toxicity reference values (TRVs), i.e., application of the TQ approach, in conjunction with the site-specific biosurvey data and other information/data available in the scientific literature. Results of all methods and data sources are discussed individually for all surface waterbodies.

10.1.1 Spring and Neosho Rivers

For the Spring River toxicity assessment, historical water quality data for the reach located east of Baxter Springs was used. Water chemistry in this reach is dominated by contributions from upstream tributaries. Mean TQs for aquatic populations were less than 1 for all three COCs: cadmium, lead, and zinc. The TQ values presented in Table 8-1 indicate that adverse acute and chronic impacts are not expected. It is noted that the TQs are based on toxicity reference values calculated for aquatic species that could occur in ephemeral streams of southeast Kansas. A recalculation of the criteria based solely on species that occur or could occur in the river would probably somewhat modify the TQs.

The ecological endpoint assessment summary for the Spring River is presented in Table 10-1. Biota in the Baxter Springs reach of the river were not surveyed for this study, and without support from field observations, the toxicity assessment was used to evaluate potential impacts. Existing literature appears to support the TQ numbers for the Baxter Springs reach, in that a variety of fish species reside in the river and clean-water benthic organisms are present (Ferrington et al, 1988; KDHE, 1980). Nonetheless, the aquatic organisms in portions of the Spring River and upstream tributaries <u>are</u> impacted by mining-related runoff and seepage, specifically with respect to cadmium and zinc concentrations

Table 10-1. Ecological Endpoint Assessment Summary for the Spring River

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic populations possible?	1.	Based on results of toxicological evaluation (TQs) and contribution of subsite water to the Spring River, influence from on-site discharge is not expected to cause a significant reduction in populations of key aquatic species.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	Intake of site-related metals is not expected to add an appreciable increment of toxicity above that impact emanating from mining and other activities upstream to which aquatic organisms are exposed. TQs for the COCs onsite (cadmium, copper, lead, and zinc) were all less than 1. As a result, the flow contribution of surface water from the Baxter Springs subsite is low and does not contribute significantly to metals concentrations in the Spring River.
3.	Are gross signs of acute toxicity absent?	3.	Aquatic organisms were not sampled from the Spring River during the RI, as the Spring River system historically received heavy metals input from mining activities throughout southeastern Kansas. Although Ferrington et al. (1988) indicated that adverse impacts to aquatic biota inhabiting the Spring River may be occurring, the Spring River receives a much greater proportion of heavy metals inputs from other mining areas throughout southeastern Kansas and southeastern Missouri than from the Baxter Springs Subsite.
4.	Has community structure been obviously impacted?	4.	Aquatic organisms were not sampled from the Spring River during the RI, as the Spring River system historically received heavy metals input from mining activities throughout southeastern Kansas. Although Ferrington et al. (1988) indicated that adverse impacts to aquatic biota inhabiting the Spring River may be occurring, the Spring River receives a much greater proportion of heavy metals inputs from other mining areas throughout southeastern Kansas and southeastern Missouri than from the Baxter Springs Subsite.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	N/A
6.	Is water quality in sufficient to support a diverse natural aquatic community?	6.	Based on TQ results, water quality within the Spring River appears to be sufficient to support a diverse aquatic community.
7.	Have significant habitat modifications occurred?	7.	N/A
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	8.	Based on TQ evaluations, no significant reductions in reproductive fitness of key aquatic species is expected.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	N/A
10.	Are soils phytotoxic?	10.	N/A

primarily from Short Creek near Galena (USGS, 1992). Based on the results of the toxicity assessment and the minor flow contributions of subsite water to the Spring River, influence from on-site discharge is not expected to cause a significant reduction in populations of key aquatic species.

The toxicity assessment for the Neosho River, below Tar Creek inputs, is based on a very limited data base of six water samples (OWRB, 1983). TQs for the COCs cadmium and zinc were below 1 indicating an apparent lack of chronic impacts. Again, these results should be viewed as estimates only since they are based on limited and outdated water chemistry data and little field data are available. In support of this conclusion, Aggus *et al.* (1983), indicated that aquatic organisms inhabiting the Neosho River are not impacted by metals from Tar Creek and that any effects on the aquatic community within the Neosho River diminish rapidly once Tar Creek water enters the Neosho River, primarily because of the ameliorating effects of increased water hardness within the Neosho River (Table 10-2).

10.1.2 Tar Creek

Mean TQs for zinc and manganese in Tar Creek equal 6 and 0.4, respectively (Table 8-1) suggesting that adverse chronic impacts to aquatic organisms are possible from exposure to elevated zinc levels. A comparison of the mean and upper bound zinc concentrations for Tar Creek with the Criterion Maximum Concentration for zinc indicates that acute effects are also possible (TQ=11). Results of field investigations indicate that fish numbers in the lower segment of Tar Creek were low relative to other streams within the subsites, probably as a result of the combination of marginal habitat and elevated zinc concentrations. The only fish collected from Tar Creek were representative of the family Centrachidae (sunfishes) which are relatively tolerant of elevated metals concentrations as compared to representatives of other families (i.e., Ictalurids-catfish and Cyprinids-minnows). All fish collected form Tar Creek had condition factors greater than 1.

Table 10-2. Ecological Endpoint Assessment Summary for the Neosho River

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic populations possible?	1.	According to Aggus et al. (1983), aquatic organisms inhabiting the Neosho River are not impacted by metals from Tar Creek. Effects on the aquatic community within the Neosho River diminish rapidly once Tar Creek water enters the Neosho River, primarily because of the ameliorating effects of increased water hardness of the Neosho River.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	An evaluation of metals toxicity to aquatic organisms known or suspected to inhabit the Neosho River showed that neither of the chemicals detected in river water above background levels (cadmium and zinc) presented potential toxicity to these organisms, as all mean TQ values were less than 1.
3.	Are gross signs of acute toxicity absent?	3.	Biosurveys of the Neosho River were not conducted during the RI, since this river receives metals input from several other sources. Fish species composition and productivity within the Neosho River downstream from Tar Creek showed no apparent effects from mining activities (Aggus et al., 1983).
4.	Has community structure been obviously impacted?	4.	Community structures of the Neosho River has not been impacted primarily because of the ameliorating effects of increasing water hardness in the Neosho River (Aggus et al., 1983).
5.	Have significant community level transformations occurred in plant and animal systems?	5.	Although aquatic organisms may be severely stressed in Tar Creek because of the mine discharge (including Picher Field) upstream, aquatic communities are not impacted within the Neosho River, primarily because of the ameliorating effects of increased water hardness within the Neosho River.
6.	Is water quality sufficient to support a diverse natural aquatic community?	6.	Based on TQ results and data reported by Aggus et al. (1983), water quality within the Neosho River appears to be sufficient to support a diver aquatic community.
7.	Have significant habitat modifications occurred?	7.	N/A
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	5.	Based on TQ evaluations, no significant reproductive fitness of key aquatic species is expected to occur within the Neosho River.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	N/A
10.	Are soils phytotoxic?	10.	N/A

The ecological endpoint assessment summary for Tar Creek is presented in Table 10-3. Aquatic habitat in most of Tar Creek is marginal based on the periodic absence of water within the stream channels. During dry periods, Tar Creek within the Treece subsite is reduced to a series of ponds within the channel, which provide limited habitat. The overall stream habitat was rated poor-to-fair using EPA's Stream Habitat Assessment Methodology (USEPA, 1989e).

During the 1991 RI sampling program, no signs of acute toxicity were observed. Fish collected at the downstream sampling location (TQ-3) were in good condition with no evidence of stress (lesions, abnormal accumulations of mucous, or parasites). The fish species sampled during the 1991 RI sampling program were representative, although not all inclusive, of fish species expected to inhabit intermittent drainages within southeast Kansas. Fish species collected within the lower reach of Tar Creek included green sunfish, redear sunfish, warmouth, and an unidentified sunfish. As indicated by the low numbers of fish collected within the lower reaches of Tar Creek and the results of the toxicity evaluation (TQs), water quality does not appear sufficient to support a diverse natural aquatic community although limited suitable habitat for such support is also impacting the potential for a diverse natural aquatic community. Based on the 1991 RI sampling results, the numbers of fish collected do not support any conclusions as to the natural reproduction of aquatic species. Fecundity assessments were not conducted as part of the RI program. TQ results indicate that reproductive effects are possible.

Tar Creek has been impacted by mining and agricultural influences. Agricultural influences are limited primarily to the adverse effects associated with sedimentation of stream pools from runoff of fallow and tilled fields. Mining impacts are, however, both adverse and beneficial. The adverse impacts are associated with sedimentation of stream pools from surface runoff and unvegetated mine affected areas and the chemical (metals) toxicity associated with surface runoff and ground water recharge of the streams. Beneficial effects of mining include the short-term augmentation of stream discharge from water retained within the chat piles and tailings ponds. The retained water is slowly released by subsurficial

Table 10-3. Ecological Endpoint Assessment Summary for Tar Creek

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic populations possible?	1.	Physical habitat within Tar Creek limits the capacity of the stream to support aquatic life. Fish sampled from Tar Creek during the 1991 RI sampling program indicate reduced numbers of fish within the lower segment of Tar Creek as compared to other streams within the subsite. Species and numbers of fish collected included green sunfish (2), redear sunfish (3), warmouth (1), and an unidentified sunfish (2). Condition factors of the fish collected were greater than 1.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	Results of the toxicity assessment (TQs) indicate possible adverse acute and chronic effects from exposure to zinc (i.e., TQs exceed 1).
3.	Are gross signs of acute toxicity absent?	3.	During the 1991 RI sampling program, no signs of acute toxicity were observed. Fish collected were in good condition with no evidence of stress (lesions, abnormal accumulations of mucous, or parasites).
4.	Has community structure been obviously impacted?	4.	Based on the limited aquatic habitat that was rated as "poor to fair," aquatic community impacts resulting from mining activities are considered minimal. The entire Tar Creek system within the Treece subsite has intermittent streamflow at best.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	The fish species sampled during the 1991 RI sampling program were representative, although not all inclusive, of fish species expected to inhabit intermittent drainages within southeast Kansas. Fish species collected within the lower reach of Tar Creek included green sunfish, redear sunfish, warmouth, and an unidentified sunfish.
6.	Is water quality sufficient to support a diverse natural aquatic community?	6.	As indicated by the low numbers of fish collected within the lower reaches of Tar Creek and the results of the toxicity evaluation (TQs), water quality does not appear sufficient to support a diverse natural aquatic community although limited suitable habitat for such support is also impacting the potential for a diverse natural aquatic community.
7.	Have significant habitat modifications occurred?	7.	Habitat within the upper segments of Tar Creek has not been altered as a result of mining activities; however, the stream flow within this segment of Tar Creek is naturally intermittent. Stream segments of the lower reaches of Tar Creek within the Treece subsite have been altered by mining activities but, as with the upper reaches, also experiences intermittent stream flow that severely limits the availability of aquatic habitat.
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	8.	Based on the 1991 RI sampling results, the numbers of fish collected do not support any conclusions as to the natural reproduction of aquatic species. Fecundity assessments were not conducted as part of the RI program. TQ results indicate that reproductive effects are possible.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	Upper-bound TQs calculated for T&E species (amphibians) were less than 1, indicating that chronic adverse effects are unlikely.
10.	Are soils phytotoxic?	10.	N/A

seepage which augments the base stream flow. Tar Creek has been channelized several times in response to ROD requirements thus removing additional pool habitat and sedimentation resulting from both mining and agricultural activities.

Inconsistencies between observational field data and calculated TQs (which indicated potential chronic and acute affects) could also be due to several factors that could ameliorate the toxicity of elevated metals levels onsite. These factors include metals speciation, high alkalinity and hardness, acclimation and evolutionary tolerance of indigenous species to metals, and bioavailability of metals. The water chemistry in Tar Creek is based, to some degree, on the interaction between alkalinity, hardness, and metal speciation. The actual bioavailable concentration of zinc produced by this interaction is unknown; however, it is possible that bioavailable zinc is significantly lower than measured total recoverable zinc. In addition, acclimation and/or adaptation of fish populations to zinc has been documented in the literature (Chapman, 1978, 1985; Melancon and Miller, 1984). The cumulative impact of all of these mitigating factors may account for the field data indicating that the species currently in residence do not exhibit an apparent pattern of acute toxicity. However, certain biota that are more sensitive to elevated concentrations of zinc and which could be expected to exist in Tar Creek may have been excluded due to both the physical and chemical limitations.

10.1.3 Willow Creek

Mean TQs for COCs in Willow Creek (cadmium, lead, and zinc) were below 1 (Table 8-1), which indicates that adverse effects on aquatic organisms are not likely. Evaluation of ecological endpoints relevant to Willow Creek are shown in Table 10-4. Observational field data for Willow Creek (Table 10-4) reveal that sporadic aquatic habitat is available throughout the creek's length from a point immediately downstream of WC-1 to its confluence with the Spring River. Although most of Willow Creek upstream of WC-2 is reduced to a series of intermittent ponds during dry periods, one location (WC-1a) between sampling stations WC-1 and WC-2, an artificially-created impoundment with no stream flow, provides poor-to-fair

Table 10-4. Ecological Endpoint Assessment Summary for Willow Creek

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic populations possible?	1.	Physical habitat within Willow Creek limits the capacity for the stream to support aquatic organisms. A diverse population of fish was sampled during the 1991 RI sampling program. Species and numbers of fish collected included largemouth bass (3), white crappie (1), green sunfish (24), redear sunfish (3), longear sunfish (5), bluegill (36), pumpkinseed (3), warmouth (6), spotted sucker (2), brook silverside (1), and various species of dace (2). Although actual population estimates were not conducted, the results of the 1991 sampling program indicate populations of key aquatic species have not been significantly reduced.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	An evaluation of TQs for the metals of concern within Willow Creek indicate that adverse acute and chronic effects are not expected (TQs for all COCs were less than 1).
3.	Are gross signs of acute toxicity absent?	3.	During the 1991 RI sampling program, no signs of acute toxicity were observed. Fish collected were in good condition with no evidence of stress (lesions or abnormal accumulations of mucous).
4.	Has community structure been obviously impacted?	4.	Based on the limited aquatic habitat that was rated as "poor to fair," the aquatic community structure is not obviously impacted by subsite activities. The factors limiting the quality of aquatic habitat include shifting bottom substrate, increased embeddedness, intermittent stream flow, channel alterations (resulting from agricultural related activities), and bottom scouring and deposition.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	The fish species sampled during the 1991 RI sampling program were representative of fish species expected to inhabit intermittent drainages within southeast Kansas. Species collected included largemouth bass, white crappie, green sunfish, bluegill, warmouth, longear sunfish, redear sunfish, pumpkinseed, spotted sucker, brook silverside, and various dace species.
6.	Is water quality sufficient to support a diverse natural aquatic community?	6.	As indicated by the sampling of numerous fish species within Willow Creek, water quality of Willow Creek appears to be sufficient to support a diverse natural aquatic community, especially since the evaluation of TQs indicates that chronic effects are not likely.
7.	Have significant habitat modifications occurred?	7.	Habitat modifications have not occurred as a direct result of mining activities within Willow Creek; however, habitat has been altered occasionally by activities associated with area agriculture. Willow Creek occasionally is impounded by earthen dams creating temporary habitat. However, the dams are frequently breached causing increased sedimentation and siltation downstream. Willow Creek aquatic habitat is naturally limited primarily due to intermittent flows.
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	8.	Based on the 1991 RI sampling results, the green sunfish are reproducing naturally as indicated by the presence of three age classes. Numbers of other species collected were insufficient to determine reproductive fitness. Fecundity assessments were not conducted as part of the RI program. TQs indicate that reproductive effects are not expected.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	TQs calculated for T&E species show that adverse effects are not expected.
10.	Are soils phytotoxic?	10.	N/A

habitat which is limited to bank vegetation and streamside cover. Fifty-six fish representing nine species were taken from this area. All fish sampled appeared healthy with no indications of external parasites or physical or chemical stress.

Thus, the results of the toxicity assessment, biosurvey data, and available reference data are consistent and indicate that aquatic organisms inhabiting Willow Creek are not likely to experience adverse chronic effects. An exception is the occasional mine water discharge associated with the Bruger shafts near WC-2. Mine water discharge from the Bruger shafts on Willow Creek near WC-2 contains zinc at concentrations that could be acutely toxic to resident aquatic organisms. Historic sampling results (KDHE, 1987) suggest zinc concentrations of more than 21 mg/L may be present in the discharge, and depending on the amount of flow in Willow Creek available for dilution, Bruger shaft inputs could have short-term impacts on the aquatic system. These infrequent surges are likely to cause acute toxicity, possibly more severe than in Tar Creek where fish may have become acclimated or adapted to continuously elevated concentrations of zinc.

10.1.4 Spring Branch

Mean TQs based on the mean concentration of cadmium and zinc COCs for Spring Branch were 10 and 7, respectively, shown in Table 8-1 indicating that adverse chronic effects to aquatic organisms inhabiting Spring Branch are possible. Comparison of the mean and upper-bound cadmium and zinc concentrations to the Criterion Maximum Concentrations indicates that acute effects are also possible. Conversely, aquatic surveys of Spring Branch indicate that the diversity and number of fish present are similar to that expected for intermittent drainages within southeast Kansas (Table 10-5). The fish collected in Spring Branch appeared to be in good condition and did not display evidence of physical or chemical stress. The green sunfish collected represented four age classes indicating possible normal reproduction amongst this particular population of green sunfish. Numerous site-specific factors influence the actual toxicity to these resident species. These factors include adaptation and acclimation of the fish, and dilution in the lower segment of Spring Branch.

Table 10-5. Ecological Endpoint Assessment Summary for Spring Branch

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic species possible?	1.	A diverse population of fish was sampled during the 1991 RI sampling program. Species of fish collected included predominantly green sunfish along with yellow bullhead, chubs, and plains topminnows. The green sunfish sample represented four age classes (I, II, III, and IV). Condition factors of the green sunfish were all above 1.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	An evaluation of TQs for the metals of concern within Spring Branch indicated possible acute and chronic effects on aquatic organisms from exposure to cadmium and zinc (i.e., mean concentration TQs were > 1).
3.	Are gross signs of acute toxicity absent?	3.	During the 1991 RI sampling program, no signs of acute toxicity were observed. Fish collected were in good condition with no evidence of stress (lesions or abnormal accumulations of mucous).
4.	Has community structure been obviously impacted?	4.	Based on the limited aquatic habitat which was rated as "fair," the aquatic community structure has not been obviously impacted as a result of subsite activities.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	The fish species sampled during the 1991 RI sampling program were representative of fish species expected to inhabit intermittent drainages within southeast Kansas. Species collected included green sunfish, yellow bullhead, plains topminnow, and various chubs.
6.	Is water quality sufficient to support a diverse natural aquatic community?	6.	As indicated by the sampling of numerous fish species within the lower reaches of Spring Branch, it appears to support a diverse natural aquatic community. Conversely, results of the toxicity assessment suggest that adverse acute and chronic effects are possible, which could limit species numbers and diversity.
7.	Have significant habitat modifications occurred?	7.	Habitat is naturally limited primarily due to the intermittency of stream flow. Physical or anthropogenic changes have been made historically to the stream. These changes have resulted in decreased habitat and sedimentation. Results of the TQ assessment indicate that metals levels in Spring Branch due to mining activities (i.e., mill and mine waste) may cause adverse effects in aquatic receptors from exposure to cadmium and zinc.
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	8.	Based on the 1991 RI sampling results, the green sunfish are reproducing naturally as indicated by the presence of four age classes. Numbers of other species collected were insufficient to determine reproductive fitness. Although fecundity assessments were not conducted, TQs indicate that reproductive effects are possible.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	TQs calculated for T&E species show that adverse chronic effects are not expected.
10.	Are soils phytotoxic?	10.	N/A

Additionally, a shift in species composition may have historically occurred in favor of more tolerant aquatic species.

The aquatic habitat available in Spring Branch was rated as fair (Table 10-5). Unlike Tar Creek, the Spring Branch drainage is entirely contained within an area impacted by mining and streamflow is supported, at least over the short term, by seepage from a large chat-wash pond (Ballard Pond) in the upper part of the basin. It is believed that much of the dissolved cadmium present within the stream originates from this industrial pond.

As discussed in Section 10.1.2 (Tar Creek), a variety of factors may be present that could ameliorate the potential toxicity of observed zinc (and cadmium) concentrations, thereby explaining the apparent discrepancy between the calculated TQs and the observational information. Again, these factors include bioavailability, hardness, and acclimation. While the derived TQ values are similar to Tar Creek, Spring Branch differs from Tar Creek in several respects: 1) the Spring Branch watershed is much smaller than Tar Creek within Kansas (3.3 vs. 8.6 square miles) with mill waste dominating water quality in the upstream reach; 2) mean total recoverable cadmium concentrations in Spring Branch are approximately four times the mean concentration in Tar Creek, probably due to seepage from the Ballard Pond which is kept full artificially with pumped ground water; 3) as a result of Ballard Pond contributions and ground-water seepage from limestones in the downstream reach, flows are maintained over longer periods in Spring Branch; and 4) as a result of the sustained flow during dry periods and the rocky/pebble substrate in the downstream, limestone bedrock reach, habitat for fish is noticeably improved over Tar Creek. The factors listed above may account for some of the differences observed in fish numbers and diversity, between Spring Branch and Tar Creek.

10.1.5 Subsite Ponds

Results of the toxicity assessment show that all mean COC TQs for ponds (BP-2, BP-3, BP-4, BP-5 and TP-1, TP-6 and TP-9) were less than or equal to 1. The ponds constituted seven of the 10 ponds sampled (Table 8-2). The Ballard Chat Wash Pond (BP-1) exhibited the highest TQ (10 for cadmium). The remaining TQs exceeding unity included 2 for both zinc (TP-5) and iron (TP-7); and 7 for lead (TP-7). These data indicate that adverse effects to exposed populations are possible.

A summary of ecological endpoints relevant to the aquatic community inhabiting subsite ponds is given in Table 10-6. Seven of the ten ponds inventoried contained fish, primarily green sunfish. Of these seven ponds, six were sampled to collect fish species. They contained fish representing several age classes, which indicates that some reproduction was occurring. Condition factors indicate that the fish were healthy with no obvious evidence of external parasites or chemical stress.

Three of the subsite ponds inventoried did not contain fish. Tailings pond TP-7 contained total recoverable iron and lead concentrations at levels well above the toxicity reference values 1.0 mg/L (TQ=2) and 0.014 mg/L (TQ=7), respectively. The pond was shallow (less than five feet deep) and potentially dries up during droughts. Tailings pond BP-2 contained comparatively low levels of metals meeting all site-specific criteria and may not contain fish because it has never been stocked. As mentioned previously, tailings pond BP-1 (Ballard Pond) is a commercial chat-wash pond. Beyond the physical limitations presented by the fluctuating water level in the pond, cadmium and zinc concentrations exceed the TRVs of 0.034 mg/L (TQ=10)and 6.33 mg/L (TQ=1), respectively.

The aquatic communities within the subsite ponds have been established as a result of the development of the ponds. Although the subsite ponds are artificial they are inhabited by species expected to occur within aquatic systems of southeast Kansas. Based on the 1991 RI sampling results, the green sunfish, bluegill, and black bullhead may be reproducing naturally within the subsite ponds. The lack of fish in age classes greater than Class II may indicate an adverse effect of the life cycle of fish inhabiting subsite ponds. Numbers of other species collected were insufficient to determine reproductive fitness, and fecundity assessments were not conducted during the RI program.

Table 10-6. Ecological Endpoint Assessment Summary for Subsite Ponds

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key aquatic populations possible?	1.	Diverse populations of fish were sampled from subsite ponds (excluding Ballard Ponds) during the 1991 RI sampling program. The presence of age class 0 and I may indicate that green sunfish, bluegill, and black bullhead are naturally reproducing within the ponds to some extent. On the other hand, the presence of only two age classes could also indicate a substantial reduction in the life cycle of fish inhabiting these ponds.
2.	Could intake of metals result in chronic toxic effects in aquatic populations?	2.	An evaluation of TQs for the metals of concern within the subsite ponds indicates that adverse acute and chronic effects on aquatic organisms are possible from exposure to elevated levels of cadmium, iron and zinc.
3.	Are gross signs of acute toxicity absent?	3.	During the 1991 RI sampling program, no signs of acute toxicity were observed. Fish collected were in good condition with no evidence of stress (lesions, abnormal accumulations of external mucous or parasites).
4.	Has community structure been obviously impacted?	4.	The aquatic communities within the subsite ponds have been established as a result of the development of the ponds. Since pre-mining community structures were non-existent, it is impossible to assess the impact of mining on aquatic community structure.
5.	Have significant community level transformations occurred in plant and animal systems?	5.	Although the subsite ponds are artificial they are inhabited by species expected to occur within aquatic systems of southeast Kansas.
6.	Is water quality sufficient to support a diverse natural aquatic community?	6.	As indicated by the sampling of numerous fish species within the subsite ponds (excluding the Ballard Ponds), water quality appears to be sufficient to support a diverse natural aquatic community.
7.	Have significant habitat modifications occurred?	7.	Pond habitat within the subsites has been created by mining activities and, as such, only beneficial aquatic habitat modifications have occurred as a result of mining activities.
8.	Are significant reductions in reproductive fitness of key aquatic species probable?	8.	Based on the 1991 RI sampling results, the green sunfish, bluegill, and black bullhead may be reproducing naturally within the subsite ponds. The lack of fish in age classes greater than Class II may indicate an adverse effect of the life cycle of fish inhabiting subsite ponds. Numbers of other species collected were insufficient to determine reproductive fitness. Fecundity assessments were not conducted as part of the RI program. Results of the toxicity assessment suggests that reproductive effects are possible from exposure to zinc.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	TQs calculated for T&E species indicate that adverse chronic effects are not expected.
10.	Are soils phytotoxic?	10.	N/A

10.2 TERRESTRIAL RECEPTORS

One objective of the wildlife investigations conducted during the RI was to determine whether adverse chronic effects to terrestrial receptors are possible. Integral to this assessment was the determination of discernible differences between control and study sites and the derivation of TQs for key higher trophic level species.

As described in Section 5.0, whole body mouse and fish tissue results were used to estimate dose levels for higher trophic level organisms that prey on small mammals and fish inhabiting the subsites. These data as well as the measured concentration of metals in on-site surface water, air, and near-pile soils were used to estimate intakes by key terrestrial predators: the barred owl, the red-tailed hawk, and the mink. The red-tailed hawk and the barred owl were used to estimate potential risks to higher-level predators, while the mink was used to estimate risks to higher-level, more omnivorous receptors. The focus on primary and higher-level consumers was based on the rationale that results of these investigations would provide sufficient data to assess the general condition of the terrestrial ecosystem.

Since the worst-case TQs (i.e., the TQ based on the worst-case exposure scenario) for cadmium, lead, and zinc were less than 1 for the raptors (Table 8-3), adverse chronic impacts from exposure to site-related metals are not expected to occur in higher trophic level species that have a similar prey base as raptors. Results of the toxicity assessment for the mink, however, indicate that chronic adverse effects from exposure to cadmium, lead and zinc are possible, since the worst-case and RME TQs are slightly above 1 (RME TQs range from 1 to 3). These data indicate that terrestrial species who consume fish may experience adverse chronic effects from exposure to cadmium, lead and zinc.

A summary of the ecological endpoints relevant to the terrestrial community evaluation is presented in Table 10-7. The results of field investigations indicate that existing exposures are not causing acute effects or mortality in exposed populations, since obvious signs of toxicity (such as those reported in the FWS *Contaminant Hazard Reviews*) associated with

Table 10-7. Ecological Endpoint Assessment Summary for the Terrestrial Community

	Assessment Endpoints		Measurement Endpoints
1.	Is a significant reduction in key terrestrial species possible?	1.	Based on the field investigations conducted during the Fall 1991 RI, other available information, and on professional judgement, the terrestrial wildlife community in the Baxter Springs/Treece subsites appears to be normal in comparison with the control area and in relation to what might be expected to exist within Cherokee County.
2.	Could intake of metals result in chronic toxic effects in key terrestrial populations?	2.	The primary focus of the environmental evaluation was on potential toxicity and associated impacts and food chain effects on higher trophic level organism (i.e., raptors and the mink). Toxicity quotients calculated for these higher trophic level organisms indicate that carnivorous receptors are not expected to experience adverse chronic effects while omnivorous receptors that consume fish might experience adverse effects. Furthermore, the arithmetic mean concentration of cadmium, lead, and zinc measured in whole-body mice samples taken from the subsites are 14, 28, and two times higher than arithmetic mean levels measured in mice taken from the Cherokee County Control site.
3.	Are gross signs of acute toxicity absent?	3.	During the 1991 RI field investigations, no signs of acute toxicity were observed. Wildlife observed were in good condition with no evidence of stress.
4.	Has community structure been obviously impacted?	4.	Based on field investigations and professional judgement (certified wildlife biologist), there was no evidence of obvious impacts to the wildlife community structure, with the exception of areas were habitat was precluded by actual mining activities (chat piles, mine workings, etc.)
5.	Have significant community level transformations occurred in plant and animal systems?	5.	The plant and animal systems within the subsites have been altered to a certain extent by both mining and agricultural activities. Agricultural activities have removed much of the natural habitat by plowing and tilling of fields while mining has, to a lesser extent, removed habitat associated with the tailings piles and mine workings. Additionally, both activities have also provided and enhanced habitat by creating and "edge effect" adjacent to fields relating to agricultural practices, while mining has in effect prevented the expansion of agriculture and has "preserved" those areas adjacent to the mine disturbances.
6.	Is water quality in on-site streams sufficient to support a diverse natural aquatic community?	6.	N/A
7.	Have significant habitat modifications occurred?	7.	Habitat modifications have occurred with respect to the mining activities disturbing existing habitat. However, as previously described, these disturbances have also preempted the expansion of agricultural practices.
8.	Are significant reductions in reproductive fitness of key terrestrial species probable?	8.	Based on the results of the toxicity assessment, reduced reproductive fitness of key terrestrial omnivorous species is probable. However, wildlife populations observed during field investigations were represented in numbers typical of southeast Kansas and appeared to be naturally reproducing.
9.	Could T&E species that may occur onsite be adversely impacted?	9.	Toxicity quotients calculated for T&E species show that adverse effects to this ecologic group are not expected.

Table 10-7. Ecological Endpoint Assessment Summary for the Terrestrial Community (Concluded)

	Assessment Endpoints		Measurement Endpoints
10.	Are soils phytotoxic?	10.	Upper-bound levels of cadmium, cobalt, lead, and manganese in near-pile soils and upper-bound levels of copper, lead, and manganese in Ag/A horizon soils did not exceed the median phytotoxic concentration (TQs based on the median phytotoxic concentration were less than 1). Therefore, these metals are not expected to produce phytotoxic effects in on-site vegetation. Conversely, upper-bound levels of zinc in near-pile soils did exceed the median phytotoxic concentration reported in the literature (TQs equal 4 and 3, respectively), which indicates that near-pile soils contaminated with elevated levels of zinc could be phytotoxic. Other factors that may affect the potential phytotoxicity of zinc include: (1) near-pile soils were typically covered with a well-defined organic layer (humus) that tends to reduce the plant-available concentration of most metals; (2) near-pile soils cover only 4 percent of the total subsite area; and (3) the available toxicity data were for agronomic species versus the species that are likely to inhabit near-pile soils.

chronic metals poisoning were not observed. Terrestrial species and communities appear to be normal in comparison to populations inhabiting the control area. No wildlife groups appeared to be missing, nor were poor or low population levels observed among the most highly-exposed species (e.g., primary and secondary consumers), although population numbers were not directly measured. The wildlife community in the vicinity of mine-related disturbances exhibited greater diversity and abundance than the community associated with intensively cultivated or more urban portions of the two subsites. Although approximately 1180 acres of wildlife habitat has been impacted by mining activities, these areas appear to provide high quality wildlife habitat relative to and in juxtaposition with surrounding agricultural tracts. This observation is supported by the fact that survey results obtained for predators, raptors, songbirds, and small mammals did not indicate significant differences in animal presence and appearance between disturbed versus control areas. For example, the number of barred owls observed in the control and test areas was similar.

The measured concentration of metals in whole body mice samples from the subsites and mice taken from various reference location within Cherokee County were compared. Results show that the arithmetic mean concentration of cadmium, lead, and zinc measured in subsite mice are, respectively, 14, 28, and two times higher than arithmetic mean levels measured in mice taken from the Cherokee County Control site (Table 5-10).

It is possible that the white-footed mouse could be more exposed than the raptors and the mink, since mice have closer and more frequent contact with contaminated soils. Accurate dose estimates for the white-footed mice were not possible, since: (1) the concentration of site-related metals was not measured in the vegetation and seeds on which mice primarily forage; (2) body burden data were measured with the pelt on; and (3) tissue levels for various organs were not obtained. These limitations did not allow a comprehensive evaluation of potential impacts to all terrestrial prey species (using the white-footed mice as a surrogate). Based on the results for higher-trophic level organisms, adverse chronic impacts to exposed terrestrial prey species are not expected.

10.3 PHYTOTOXICITY OF SOILS

In general, vegetation systems within the subsites have been altered to a certain extent by both mining and agricultural activities. Agricultural activities have removed much of the natural habitat by plowing an tilling of fields, while mining has, to a lesser extent, removed habitat by the placement of mill waste piles and mine workings.

The comparison of concentrations of metals in Ag/A soils to the median concentration reported to be phytotoxic in either laboratory or field studies indicated that metal concentrations were not likely to be phytotoxic. However, concentrations of zinc in near-pile soils may be phytotoxic (upper-bound and mean TQs were 2 and 1, respectively). The TQ assessment yielded numbers that are thought to be conservative since it was based on agronomic species rather than those more tolerant species established in near-pile soils. If effect, the concentration of zinc in near-pile soils could potentially result in reduced yield in plants, but little if any measurable impact on native species is expected.

10.4 T&E SPECIES

One state-listed species has critical habitat within areas impacted by mine and mill waste while several others have critical habitat that could be affected by the migration of site-related metals associated with streamflow in Willow Creek and/or Spring Branch. TQ's for surrogate amphibians based on the upper-bound concentration of metals in on-site surface water bodies (conservatively includes Tar Creek and subsite ponds) indicate that exposure to site-related metals is not expected to cause adverse impacts in exposed individuals or populations (all TQs were less than 1).

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CALCULATION OF FAVs

The Final Acute Values were calculated using the four selected GMAVs and cumulative probabilities for each metal of concern according to Equation (A-1). Specific calculations and toxicity data used to derive the site-specific FAVs for each metal are also provided.

$$FAV = e^{A} \qquad \text{(A-1)}$$

$$A = S(\sqrt{0.05}) + L$$

$$S^2 = \frac{\Sigma((\ln GMAV)^2) - ((\Sigma(\ln GMAV))^2/4)}{\Sigma(P) - ((\Sigma\sqrt{P}))^2/4)}$$

$$L = \frac{(\Sigma(\ln GMAV) - S(\Sigma(\sqrt{P}))}{4}$$

where:

	FAV	=	Final Acute Value
	A	=	Intermediate Step
	S	=	Intermediate Step
	L	=	Intermediate Step
	P	=	Cumulative Probability
	GMAV	=	Genus Mean Acute Value
	In	=	Natural Logarithm

FINAL ACUTE VALUE CALCULATIONS ZINC ($\mu g/l$)						
Rank	N	GMAV	lnGMAV	$(lnGMAV)^2$	P=R/(N+1)	$\sqrt{P}$
4	14	3830	8.2506	68.0727	0.26667	0.51640
3	14	3265	8.0910	65.4645	0.20000	0.44721
2	14	1578	7.3639	54.2272	0.13333	0.36515
1	14	1353	7.2101	51.9852	0.06667	0.25820
Sum			30.9156	239.7497	0.66667	1.58696
S^2	21.742					
L	5.8790					
A	6.9216					
FAV	1013.9536					

Genus Mean Acute Value	Species
88,960	Damselfly, *Argia*
19,800	Amphipod, *Crangonyx*
18,400	Worms, *Nais*
17,940	Banded Killifish, *Fundulus*
16,820	Snail, *Amnicola*
10,560	Pumpkinseed, *Lepomis*
10,250	Goldfish, *Carassius*
9,712	Worm, *Lumbriculus*
8,157	Isopod, *Asellus*
6,053	Guppy, *Poecilla*
3,830 Rank (4)	Fathead minnow, *Pimephales*
3,265 Rank (3)	Isopod, *Lirceus*
1,578 Rank (2)	Snail, *Hellsoma*
1,353 Rank (1)	Snail, *Physa*

$$S_2=[\Sigma((ln\ GMAV)^2) - ((\Sigma(ln\ GMAV))^2)/4] \ / \ [\Sigma(P) - ((\Sigma(\sqrt{P}))2/4)]$$

$$L = (\Sigma(ln\ GMAV) - (S(\Sigma(\sqrt{P}))))/4$$

$$A = S(\sqrt{0.05}) + L$$

$$FAV = e^A$$

FINAL ACUTE VALUE CALCULATIONS NICKEL (μg/l)						
Rank	N	GMAV	lnGMAV	$(\ln GMAV)^2$	P=R/(N+1)	$\sqrt{P}$
4	10	9661	9.1759	84.1963	0.36364	0.60302
3	10	9530	9.1622	83.9459	0.27273	0.52223
2	10	8027	8.9906	80.8303	0.18182	0.42640
1	10	4636	8.4416	71.2607	0.09091	0.30151
Sum			35.7702	320.2332	0.90909	1.85317
S^2	7.043					
L	7.7130					
A	8.3064					
FAV	4049.8789					

Genus Mean Acute Value	Species
43,250	Banded killifish, *Fundulus*
40,460	Stonefly, *Acroneuria*
21,320	Goldfish, *Carassius*
21,200	Damselfly, Unidentified sp.
14,100	Worm *Nais*
12,770	Snail, *Amnicola*
9,661 Rank (4)	Guppy, *Poecilla*
9,530 Rank (3)	Pumpkinseed, *Lepomis*
8,027 Rank (2)	Fathead minnow, *Pimephales*
4,636 Rank (1)	Mayfly, *Ephemerella*

$S_2=[\Sigma((ln\ GMAV)^2) - ((\Sigma(ln\ GMAV))^2)/4]\ /\ [\Sigma(P) - ((\Sigma(\sqrt{P}))2/4)]$

$L = (\Sigma(ln\ GMAV) - (S(\Sigma(\sqrt{P}))))/4$

$A = S(\sqrt{0.05}) + L$

$FAV = e^A$

FINAL ACUTE VALUE CALCULATIONS CADMIUM (µg/l)						
Rank	N	GMAV	lnGMAV	$(\ln GMAV)^2$	P=R/(N+1)	$\sqrt{P}$
4	25	104	4.6444	21.5704	0.15385	0.39223
3	25	98.79	4.5930	21.0956	0.11538	0.33968
2	25	42.8	3.7565	14.1116	0.07692	0.27735
1	25	30.5	3.4177	11.6809	0.03846	0.19612
Sum			16.4117	68.4584	0.38462	1.20538
S^2	52.520					
L	1.9190					
A	3.5395					
FAV	34.4509					

Genus Mean Acute Value	Species
8,325	Goldfish, *Carassius*
8,100	Damselfly, (Unidentified)
7,921	Tubificid worm, *Rhyacordrilus*
7,685	Mosquitofish, *Gambusia*
6,915	Tubificid worm, *Stylodrilus*
5,708	Channel catfish, *Ictalurus*
4,990	Tubificid worm, *Spirosperma*
4,778	Tubificid worm, *Varichaeta*
4,024	Tubificid worm, *Tubifex*
4,024	Tubificid worm, *Quistradilus*
3,800	Snail, *Amnicola*
3,641	Green sunfish, *Lepomis*
3,570	Guppy, *Poecilla*
3,018	Tubificid worm, *Branchiura*
2,310	Mayfly, *Ephemerella*
2,137	Tubificid worm, *Limnodrilus*
1,700	Worm, *Nais*
1,200	Midge, *Chironomus*

Genus Mean Acute Value	Species
400.5	Isopod, *Asellus*
322.8	Mayfly, *Paraleptophlebia*
156.9	Snail, *Physa*
104.0 Rank (4)	Snail, *Aplexa*
98.79 Rank (3)	Banded killifish, *Fundulus*
42.80 Rank (2)	Isopod, *Lirceus*
30.50 Rank (1)	Fathead minnow, *Pimephales*

$S_2=[\Sigma((ln\ GMAV)^2) - ((\Sigma(ln\ GMAV))^2)/4] \ / \ [\Sigma(P) - ((\Sigma(\sqrt{P}))2/4)]$

$L = (\Sigma(ln\ GMAV) - (S(\Sigma(\sqrt{P}))))/4$

$A = S(\sqrt{0.05}) + L$

$FAV = e^A$

FINAL ACUTE VALUE CALCULATIONS LEAD (µg/l)						
Rank	N	GMAV	lnGMAV	$(\ln GMAV)^2$	P=R/(N+1)	$\sqrt{P}$
4	6	66140	11.0995	123.1995	0.57143	0.75593
3	6	52310	10.8649	118.0470	0.42857	0.65465
2	6	25440	10.1441	102.9023	0.28571	0.53452
1	6	1040	6.9470	48.2605	0.14286	0.37796
Sum			39.0555	392.4093	1.42857	2.32307
S^2	139.479					
L	2.9049					
A	5.5458					
FAV	256.1522					

Genus Mean Acute Value	Species
235,900	Midge, *Tanytarsus*
101,100	Goldfish, *Carassius*
66,140 Rank (4)	Guppy, *Poecilia*
52,310 Rank (3)	Bluegill, *Lepomis*
25,440 Rank (2)	Fathead minnow, *Pimephales*
1,040 Rank (1)	Snail, *Aplexa*

$S_2=[\Sigma((ln\ GMAV)^2) - ((\Sigma(ln\ GMAV))^2)/4] / [\Sigma(P) - ((\Sigma(\sqrt{P}))2/4)]$

$L = (\Sigma(ln\ GMAV) - (S(\Sigma(\sqrt{P}))))/4$

$A = S(\sqrt{0.05}) + L$

$FAV = e^A$

FINAL ACUTE VALUE CALCULATIONS MERCURY (μg/l)						
Rank	N	GMAV	lnGMAV	$(lnGMAV)^2$	P=R/(N+1)	$\sqrt{P}$
4	21	50	3.9120	15.3039	0.18182	0.42640
3	21	30	3.4012	11.5681	0.13636	0.36927
2	21	20	2.9957	8.9744	0.09091	0.30151
1	21	20	2.9957	8.9744	0.04545	0.21320
Sum			13.3047	44.8209	0.45455	1.31039
S^2	22.450					
L	1.7740					
A	2.8334					
FAV	17.0040					

Genus Mean Acute Value	Species
2,000	Stonefly, *Acroneuria*
2,000	Mayfly, *Ephemerella*
1,200	Damselfly, (Unidentified)
1,000	Worms, *Nais*
406.2	Tubificid worm, *Spirosperma*
370	Snail, *Aplexa*
250	Tubificid worm, *Quistadrilus*
240	Tubificid worm, *Ryacodrilus*
180	Tubificid worm, *Limnodrilus*
180	Mosquitofish, *Gambusia*
160	Bluegill, *Lepomis*
158.7	Fathead minnow, *Pimephales*
140	Tubificid worm, *Tubifex*
140	Tubificid worm, *Stylodrilus*
100	Tubificid worm, *Varichaeta*
80	Tubificid worm, *Branchiura*
80	Snail, *Amnicola*
50 Rank (4)	Crayfish, *Orconectes*

Genus Mean Acute Value	Species
30 Rank (3)	Guppy, *Poecilla*
20 Rank (2)	Crayfish, *Faxonella*
20 Rank (1)	Midge, *Chlronomus*

$S_2=[\Sigma((ln\ GMAV)^2) - ((\Sigma(ln\ GMAV))^2)/4] / [\Sigma(P) - ((\Sigma(\sqrt{P}))2/4)]$

$L = (\Sigma(ln\ GMAV) - (S(\Sigma(\sqrt{P}))))/4$

$A = S(\sqrt{0.05}) + L$

$FAV = e^A$

ADJUSTING FAVs FOR HARDNESS-

FAVs were adjusted for a hardness of 190 mg $CaCO_3$/L using the following equation obtained from the AWQC documents for the metals of concern:

$$\text{Adjusted FAV} = e^{[(\text{Slope} \times \ln \text{Hardness}) + \ln \text{Intercept}]} \quad \text{(A-2)}$$

where:

where:	slope	=	pooled slope provided in the appropriate AWQC document;
	hardness	=	hardness for which LC_{50} or acute value is to be adjusted (in this case, 190 mg $CaCO_3$/L);
	intercept	=	Y-intercept from the regression equation used to adjust the individual species values to a hardness of 50 mg/L.

Since Y-intercept values are not reported in AWQC documents, they were calculated using the following equation obtained from the AWQC documents for the metals of concern:

$$\ln \textit{Intercept} = \ln (\textit{FAV}) - [\textit{Slope} \times \ln (\textit{Hardness})] \quad \text{(A-3)}$$

where:	intercept	=	the Y-intercept of the regression equation;
	FAV	=	the final acute value calculated for that metal for a hardness of 50 mg $CaCO_3$/L;
	slope	=	pooled slope provided in the AWQC document for the metal of interest;
	hardness	=	50 mg $CaCO_3$/L.

Example Calculation - Zinc

The species mean acute for zinc for the pumpkinseed sunfish adjusted for hardness of **50** mg $CaCO_3$/L is **18,700** μg/L, and the pooled slope is **0.847** (EPA, 1987). Therefore, using Equation A-3, the Y-intercept is estimated to be 6.52 as follows:

$$\ln(\text{Intercept}) = \ln(18{,}700) - [0.8473 \times \ln(50)]$$
$$\ln(\text{Intercept}) = 9.836 - [0.847 \times 3.91]$$
$$= 6.52$$

Using Equation A-2, the FAV (at 190 mg $CaCO_3$/L) for the pumpkinseed sunfish is calculated to be 57,884 μg/L as follows:

$$\text{Adjusted FAV} = e^{([\text{slope} \times \ln 190] + \ln \text{Intercept})}$$

$$\text{Adjusted FAV} = e^{[(0.847 \times 5.247) + 6.522]}$$

$$= 57{,}884\ \mu g/L.$$

The FAV for mercury was not adjusted, since water hardness has no apparent effect on the toxicity of mercury (EPA, 1984d). Site-specific FAVs adjusted for a hardness of 190 mg $CaCO_3$/L are shown in Table 7-4. An example calculation (for adjusting cadmium is provided below:

Example Calculation - Cadmium

$$\ln(\text{Intercept}) = \ln(34.4509) - [1.128 \times \ln(50)] \quad \text{(A-3)}$$

$$= 3.5395 - [1.128 \times 3.9120)$$

$$= -0.8733$$

Using equation 7-2, the FAV (at 190 mg$CaCO_3$/L) is calculated to be 155 μg/L as follows:

$$\text{Adjusted FAV} = e^{([\text{slope} \times \ln 190] + \ln \text{Intercept})}$$

$$\text{Adjusted FAV} = e^{([1.128 \times 5.2470) + (-0.8733)]}$$

$$= 155 \mu g/L$$

MAXIMUM ALLOWABLE TOXICANT CONCENTRATION CALCULATIONS-

The following regression equation developed by (Suter *et al.*, 1987) was used to predict the chronic Maximum Allowable Toxicant Concentrations (MATC) from an acute LC_{50}.

$$\log \text{MATC } (\mu g/L) = -0.70 + 0.73 \times \log LC_{50} \ (\mu g/L) \qquad \text{(A-4)}$$

In this case, the site-specific FAV for cadmium listed in Table 7-3 (155 μg/L) was inserted into Equation A-4 for the log LC_{50} as shown in the example calculation below.

Acute to Chronic Calculation - Aquatic Vertebrates

$$\begin{aligned} \log \text{MATC } (\mu g/L) &= -0.70 + 0.73 \text{ x Log } (155 \ \mu g/L) \qquad \text{(A-4)} \\ &= -0.70 + 0.73 \text{ x } (2.19) \\ &= 0.90 \\ \text{MATC } (\mu g/L) &= 7.92 \end{aligned}$$

The data set used to generate this regression equation was compiled from published results of life cycle, partial life cycle, and early life stage tests **performed** on freshwater fish **(vertebrates)**. It includes 25 tests on nine metals with 18 species (Suter *et al.*, 1987). Concentration-response data were averaged across duplicates within the same study. Data were eliminated if more than 30 percent mortality occurred in the control population. The LC_{50} and chronic data used for the acute-to-chronic extrapolations were taken from the same study so that consistent fish populations and water concentrations were used. Differences in hardness were not specified. Therefore, since the equation used to predict MATCs for aquatic organisms is derived from tests done on a variety of species and life stages with varying sensitivities, the resulting TRVs should also be protective of numerous species and life stages.

Suter (1986) reported a similar regression equation **(A-5)** designed to predict chronic MATCs from acute-to-chronic extrapolations done for *Daphnia spp.*, since invertebrate chronic data are limited to life-cycle tests with *Daphnia spp.* (i.e., there are little chronic data for any other freshwater invertebrate species). The data set used to generate the regression equation developed for aquatic invertebrates was compiled from published results of 27 life cycle test data on nine metals taken from the 1980 and 1984 AWQC support documents (Suter, 1986).

$$\log MATC \ (\mu g/L) = -1.08 + 0.96 \times \log LC_{50} \ (\mu g/L) \qquad \text{(A-5)}$$

Acute to Chronic Calculation - Aquatic Invertebrates

log *MATC* (μg/L)	=	-1.08 + 0.96 x Log (155 μg/L)	(A-5)
	=	-1.08 + 0.96 x (2.1903)	
	=	1.0227	
MATC	=	10.5363 μg/L	

DOSE ESTIMATES FOR THE RED-TAILED HAWK, BARRED OWL, AND THE MINK, ECOLOGICAL RISK ASSESSMENT, BAXTER SPRINGS/TREECE SUBSITES, CHEROKEE COUNTY

INTAKE = [(CM x QM x FI x BA) + (CA x QA x FI) + (CS x QS x FI) + (CW x QW x FI)] / BW

CM = concentration in mice (Table 5-6); CA = concentration in air (Table 5-4); CS = concentration in near-pile soils (Table 3-3).
CW = concentration in surface water (Table 5-5); BW = body weight (Table 5-8).
QM, QA, QS, and QW = quantity of mice, air, soil, and water, respectively, taken in by the receptor of concern (Table 5-8).

RED-TAILED HAWK - WORST-CASE

Metal	CM (mg/kg)	QM (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	1.5	0.135	1	5.2E-06	0.6	1	6.6	0.014	0.1	0.04	0.045	0.25	1.32	0.16
Lead	3.7	0.135	1	7.6E-05	0.6	1	113.9	0.014	0.1	0.04	0.045	0.25	1.32	0.50
Zinc	39.4	0.135	1	9.1E-04	0.6	1	995.6	0.014	0.1	7.1	0.045	0.25	1.32	5.15

RED-TAILED HAWK - RME

Metal	CM (mg/kg)	QM (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	0.8	0.135	0.75	5.2E-06	0.6	0.75	4.5	0.014	0.1	0.03	0.045	0.1	1.32	0.07
Lead	2.2	0.135	0.75	7.6E-05	0.6	0.75	88.1	0.014	0.1	0.02	0.045	0.1	1.32	0.26
Zinc	35.6	0.135	0.75	9.1E-04	0.6	0.75	710	0.014	0.1	5.00	0.045	0.1	1.32	3.50

BARRED OWL - WORST-CASE

Metal	CM (mg/kg)	QM (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	1.5	0.09	1	5.2E-06	0.3	1	6.6	0.009	0.1	0.04	0.03	0.25	0.74	0.19
Lead	3.7	0.09	1	7.6E-05	0.3	1	113.9	0.009	0.1	0.04	0.03	0.25	0.74	0.59
Zinc	39.4	0.09	1	9.1E-04	0.3	1	995.6	0.009	0.1	7.1	0.03	0.25	0.74	6.08

BARRED OWL - RME

Metal	CM (mg/kg)	QM (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	0.8	0.09	0.75	5.2E-06	0.3	0.75	4.5	0.009	0.1	0.03	0.03	0.1	0.74	0.08
Lead	2.2	0.09	0.75	7.6E-05	0.3	0.75	88.1	0.009	0.1	0.02	0.03	0.1	0.74	0.31
Zinc	35.6	0.09	0.75	9.1E-04	0.3	0.75	710	0.009	0.1	5.00	0.03	0.1	0.74	4.13

INTAKE = [(CM x QM x FI) + (CF x QF x FI) + (CA x QA x FI) + (CS x QS x FI) + (CW x QW x FI)] / BW

CM = concentration in mice (Table 5-6); CF = concentration in fish (Table 5-7); CA = concentration in air (Table 5-4)
CS = concentration in near-pile soils (Table 3-3); CW = concentration in surface water (Table 5-5); BW = body weight (Table 5-8).
QM, QF, QA, QS, and QW = quantity of mice, fish, air, soil, and water, respectively, taken in by the receptor of concern (Table 5-8).

Mink assumed to forage 50% on subsite mice and 50% on subsite fish.

MINK - WORST-CASE

Metal	CM (mg/kg)	QM (mg/kg)	FI	CF (mg/kg)	QF (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	1.5	0.23	0.5	0.3	0.23	0.5	5.2E-06	0.45	1	6.6	0.006	0.1	0.04	0.076	0.5	1.125	0.19
Lead	3.7	0.23	0.5	5.05	0.23	0.5	7.6E-05	0.45	1	113.9	0.006	0.1	0.04	0.076	0.5	1.125	0.96
Zinc	39.4	0.23	0.5	115.1	0.23	0.5	9.1E-04	0.45	1	995.6	0.006	0.1	7.1	0.076	0.5	1.125	16.6

MINK - RME

Metal	CM (mg/kg)	QM (mg/kg)	FI	CF (mg/kg)	QF (kg/d)	FI	CA (mg/m3)	QA (m3/d)	FI	CS (mg/kg)	QS (kg/d)	FI	CW (mg/L)	QW (L/day)	FI	BW (kg)	Intake (mg/kg-d)
Cadmium	0.8	0.23	0.5	0.2	0.23	0.5	5.2E-06	0.45	1	4.5	0.006	0.1	0.03	0.076	0.5	1.125	0.11
Lead	2.2	0.23	0.5	3.5	0.23	0.5	7.6E-05	0.45	1	88.1	0.006	0.1	0.02	0.076	0.5	1.125	0.63
Zinc	35.6	0.23	0.5	92.5	0.23	0.5	9.1E-04	0.45	1	710	0.006	0.1	5.00	0.076	0.5	1.125	13.6

APPENDIX B

SURFACE WATER DATA AND STATISTICS

BAXTER SPRINGS/TREECE REMEDIAL INVESTIGATION
SURFACE WATER DATA - PRELIMINARY DATA 22-Mar-93

Sampling Date	Station ID	Hardness as CACO3 mg/l
2/19/91	WC1	118
2/19/91	WC2	170
2/19/91	WC3	320
5/05/91	WC1	55
5/05/91	WC2	59
5/05/91	WC3	71
8/02/91	WC3	660
11/20/91	WC1	45
11/20/91	WC2	79
11/20/91	WC3	100
12/13/91	WC3	74
WILLOW CREEK	MAXIMUM	660
n=11	MINIMUM	45
	AVERAGE	159
	GEOMETRIC MEAN	109
2/21/91	SB1	534
2/21/91	SB2	578
5/05/91	SB1	140
5/05/91	SB2	160
8/02/91	SB2	760
11/18/91	SB1	389
11/18/91	SB2	359
12/13/91	SB2	270
SPRING BRANCH	MAXIMUM	760
n=8	MINIMUM	140
	AVERAGE	399
	GEOMETRIC MEAN	344
2/20/91	TC1	55
2/20/91	TC2	306
2/20/91	TC3	410
5/05/91	TC1	41
5/05/91	TC2	110
5/05/91	TC3	220
12/12/91	TC1	44
12/12/91	TC2	390
12/12/91	TC3	470
TAR CREEK	MAXIMUM	470
n=9	MINIMUM	41
	AVERAGE	227
	GEOMETRIC MEAN	156

Sampling Date	Station ID	Hardness as CACO3 mg/l
2/20/91	TT-1	287
5/05/91	TT-1	130
12/13/91	TT-1	320
Tar Creek Trib	MAXIMUM	320
n=3	MINIMUM	130
	AVERAGE	246
	GEOMETRIC MEAN	227
2/21/91	LC-1	94
5/05/91	LC-1	34
11/19/91	LC-1	55
Lytle Creek	MAXIMUM	94
n=3	MINIMUM	34
	AVERAGE	64
	GEOMETRIC MEAN	56
	BP1	1,110
	BP2	130
	BP3	840
	BP4	92
	BP5	470
BAXTER POND	MAXIMUM	1,110
n=5	MINIMUM	92
	AVERAGE	528
	GEOMETRIC MEAN	350
	TP1	560
	TP5	740
	TP6	680
	TP7	110
	TP9	320
TREECE PONDS	MAXIMUM	740
n=5	MINIMUM	110
	AVERAGE	482
	GEOMETRIC MEAN	397
Geometric Mean (All Ponds)		373
Geometric Mean (All Streams)		158
Geometric Mean (All Surface Water)		191
Arthimetic Mean (All Ponds)		504
Arthimetic Mean (All Streams)		233
Arthimetic Mean (All Surface Water)		294

Surface Water Data for Baxter Springs and Treece Subsites

Station ID	Date Sampled	Cadmium Tot.Rec. mg/l	Copper Tot.Rec. mg/l	Iron Tot.Rec. mg/l	Lead Tot.Rec. mg/l	Manganese Tot.Rec. mg/l	Nickel Tot.Rec. mg/l	Silver Tot.Rec. mg/l	Zinc Tot.Rec. mg/l
WC2	02/19/91	0.0021	(0.0025)	0.4	0.007	0.14	(0.02)	(0.0001)	0.900
WC2	05/05/91	0.0012	(0.0025)	3.9	0.019	0.19	(0.02)	(0.00025)	0.360
WC2	11/20/91	0.0023	(0.0025)	3.9	0.014	0.13	(0.02)	(0.0001)	0.670
Average		0.0019	0.0025	2.7333	0.0133	0.1533	0.0200	0.0002	0.6433
Stnd Dev		0.0006	0.0000	2.0207	0.0060	0.0321	0.0000	0.0001	0.2710
95% UCL		0.0029	0.0025	6.1399	0.0235	0.2075	0.0200	0.0003	1.1002
Geometric Mean		0.0018	0.0025	1.8245	0.0124	0.1515	0.0201	0.0001	0.6013
WC3	02/19/91	0.0009	(0.0025)	1.1	0.006	0.35	0.03	(0.0002)	1.000
WC3	05/05/91	0.0018	(0.0025)	5.4	0.019	0.20	0.02	(0.00025)	0.540
WC3	08/02/91	(0.00025)	(0.0025)	0.2	0.002	0.21	(0.02)	(0.0003)	0.220
WC3	11/20/91	0.0020	0.009	5.6	0.023	0.19	(0.02)	(0.0001)	0.810
WC3	12/13/91	0.0011	0.009	11.0	0.025	0.28	(0.02)	(0.0001)	0.530
Average		0.0011	0.0051	4.6600	0.0150	0.2460	0.0220	0.0002	0.6200
Stnd Dev		0.0009	0.0036	4.3067	0.0104	0.0680	0.0045	0.0001	0.2979
95% UCL		0.0020	0.0085	8.7662	0.0249	0.3109	0.0263	0.0003	0.9040
Geometric Mean		0.00098	0.0042	2.3598	0.0106	0.2391	0.0217	0.0002	0.5515

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Surface Water Data for Baxter Springs and Treece Subsites

Station ID	Date Sampled	Cadmium Tot.Rec. mg/l	Copper Tot.Rec. mg/l	Iron Tot.Rec. mg/l	Lead Tot.Rec. mg/l	Manganese Tot.Rec. mg/l	Nickel Tot.Rec. mg/l	Silver Tot.Rec. mg/l	Zinc Tot.Rec. mg/l
SB1	02/21/91	0.1600	(0.0025)	(0.15)	0.006	0.09	0.05	(0.00005)	20.000
SB1	05/05/91	0.0360	(0.0025)	(0.25)	0.022	0.07	(0.02)	(0.00025)	3.800
SB1	11/18/91	0.1600	(0.0025)	0.1	0.006	0.05	0.04	(0.0001)	18.000
Average		0.1187	0.0025	0.1667	0.0113	0.0700	0.0367	0.0001	13.9333
Stnd Dev		0.0716	0.0000	0.0764	0.0092	0.0200	0.0153	0.0001	8.8325
95% UCL		0.2394	0.0025	0.2954	0.0269	0.1037	0.0624	0.0003	28.8233
Geometric Mean		0.0975	0.0025	0.1557	0.0093	0.0682	0.0343	0.0001	11.0743
SB2	02/21/91	0.0270	(0.0025)	(0.1)	(0.0015)	0.16	(0.02)	(0.00005)	5.500
SB2	05/05/91	0.0300	(0.0025)	0.6	0.018	0.07	0.01	(0.00025)	3.200
SB2	08/02/91	0.0230	(0.0025)	(0.05)	0.001	0.23	0.04	(0.0005)	3.700
SB2	11/18/91	0.1100	(0.0025)	0.2	(0.002)	0.04	0.05	(0.000005)	14.000
SB2	12/13/91	0.0800	0.005	0.3	0.005	0.03	(0.02)	(0.0001)	11.000
Average		0.0540	0.0030	0.2500	0.0041	0.1060	0.0280	0.0002	7.4800
Stnd Dev		0.0390	0.0011	0.2179	0.0082	0.0862	0.0164	0.0002	4.7809
95% UCL		0.0912	0.0041	0.4578	0.0120	0.1882	0.0437	0.0004	12.0383
Geometric Mean		0.0439	0.0029	0.1783	0.0031	0.0791	0.0240	0.00008	6.3132

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Surface Water Data for Baxter Springs and Treece Subsites

Station ID	Date Sampled	Cadmium Tot.Rec. mg/l	Copper Tot.Rec. mg/l	Iron Tot.Rec. mg/l	Lead Tot.Rec. mg/l	Manganese Tot.Rec. mg/l	Nickel Tot.Rec. mg/l	Silver Tot.Rec. mg/l	Zinc Tot.Rec. mg/l
TC2	02/20/91	0.0036	(0.0025)	0.4	0.025	0.22	(0.02)	(0.00005)	4.700
TC2	05/05/91	0.0015	(0.0025)	2.5	0.025	0.24	(0.02)	(0.00025)	0.700
TC2	12/12/91	0.0300	0.007	2.3	0.043	1.10	0.04	(0.0001)	21.000
Average		0.0117	0.0040	1.7333	0.0310	0.5200	0.0267	0.0001	8.8000
Stnd Dev		0.0159	0.0026	1.1590	0.0104	0.5024	0.0346	0.0001	10.7531
95% UCL		0.0385	0.0084	3.6872	0.0485	1.3669	0.0851	0.0003	26.9278
Geometric Mean		0.0055	0.0035	1.3196	0.0301	0.3876	0.0253	0.0001	4.0976
TC3	02/20/91	0.0200	(0.0025)	0.6	0.075	0.44	0.02	(0.00005)	5.400
TC3	05/05/91	0.0070	(0.0025)	0.9	0.073	0.59	(0.02)	0.0021	1.500
TC3	12/12/91	0.0400	0.012	1.3	0.078	1.10	0.05	(0.0001)	22.000
Average		0.0223	0.0057	0.9333	0.0753	0.7100	0.0300	0.0008	9.6333
Stnd Dev		0.0166	0.0055	0.3512	0.0025	0.3460	0.0173	0.0012	10.8859
95% UCL		0.0504	0.0149	1.5254	0.0796	1.2933	0.0592	0.0027	27.9850
Geometric Mean		0.0178	0.0042	0.8889	0.0755	0.6588	0.0002	0.0002	5.6176

() - Value in Parenthesis is one-half the detection limit

CHEROKEE SURFACE WATER STATISTICS

Spring River Near Baxter Springs

Cadmium

Diss. (mg/l) 02/05/74 to 06/06/78	Total Rec. (mg/l) 03/14/79 to 04/14/92
0.0100	0.0040
0.0000	0.0020
0.0000	0.0020
0.0000	0.0010
0.0000	0.0060
0.0000	0.0010
0.0000	0.0020
0.0100	0.0160
0.0000	0.0020
0.0000	0.0010
0.0000	0.0010
0.0000	0.0010
0.0000	0.0030
	0.0040
	0.0030
	0.0030
	0.0020
	0.0020
	0.0020
	0.0020
	0.0020
	0.0020
	0.0020

	Diss.	Total Rec.
n=	13	23
max=	0.0100	0.0160
min=	0.0000	0.0010
Avg.=	0.0015	0.0029
Std. Dev.=	0.0038	0.0031
95% UCL=	0.0034	0.0040

Copper

Diss. (mg/l) 02/05/74 to 06/06/78	Total Rec. (mg/l) 03/14/79 to 04/14/92
0.1000	0.0300
0.1000	0.0100
0.1000	0.0100
0.0200	0.0000
0.0000	0.0000
0.1000	0.0200
0.0200	0.0100
0.1000	0.0100
0.0300	0.0100
0.0100	0.0100
0.0100	0.0200
0.0200	0.0200
0.0900	0.0090
0.0100	0.0090
	0.0060
	0.0140
	~~[illegible]~~ .0060
	0.0150
	0.0210
	0.0110
	0.0110
	0.0160
	0.0140

	Diss.	Total Rec.
n=	14	23
max=	0.1000	0.0300
min=	0.0000	0.0000
Avg.=	0.0507	0.0123
Std. Dev.=	0.0434	0.0069
95% UCL=	0.0713	0.0147

CHEROKEE SURFACE WATER STATISTICS

Spring River Near Baxter Springs

Lead

Diss. (mg/l) 06/04/74 to 06/06/78
0.0000
0.0000
0.0000
0.0000
0.0000
0.0000
0.1000
0.0000
0.0000
0.0400
0.5000
0.2000

n=	12
max=	0.5000
min=	0.0000
Avg.=	0.0700
Std. Dev.=	0.1486
95% UCL=	0.1470

Total Rec. (mg/l) 03/14/79 to 04/14/92
0.0300
0.0100
0.0100
0.0000
0.0100
0.0200
0.2000
0.0050
0.0100
0.0130
0.0090
0.0200
0.0090
0.0060
0.0240
0.0010
0.0200

n=	17
max=	0.2000
min=	0.0000
Avg.=	0.0234
Std. Dev.=	0.0462
95% UCL=	0.0429

Zinc

Diss. (mg/l) 02/05/74 to 06/06/78
0.3900
0.4200
0.4500
0.6300
0.5400
0.2400
0.4200
1.6000
0.5100
0.7200
1.1000
1.2000
0.3400
0.6700
14
1.6000
0.2400
0.6593
0.3844
0.8143

Total Rec. (mg/l) 03/14/79 to 04/14/92
0.5200
0.3900
0.2600
0.2000
0.4100
0.2200
0.3100
1.7100
0.1600
0.5200
0.1700
0.4900
0.3290
0.1780
0.5280
0.2850
0.1790
0.2300
0.1220
0.0900
0.4030
0.2520
0.3740

n=	23
max=	1.7100
min=	0.0900
Avg.=	0.3610
Std. Dev.=	0.3225
95% UCL=	0.4765

CHEROKEE SURFACE WATER STATISTICS

Neosho River 300 Yards Downstream of the confluence with Tar Creek (*)

Cadmium				Lead			
Diss. (mg/l)		Total Rec. (mg/l)		Diss. (mg/l)		Total Rec. (mg/l)	
8/82		8/82		8/82		8/82	
n=	---	n=	6	n=	---	n=	6
max=	---	max=	0.003	max=	---	max=	0.02
min=	---	min=	0.002	min=	---	min=	0.02
Avg.=	---	Avg.=	0.0021	Avg.=	---	Avg.=	0.02

Zinc			
Diss. (mg/l)		Total Rec. (mg/l)	
8/82		8/82	
n=	---	n=	6
max=	---	max=	0.72
min=	---	min=	0.058
Avg.=	---	Avg.=	0.325

(*) Data from: Oklahoma Water Resources Board, Effects of Acid Mine Discharge on the Surface Water Resources in the Tar Creek Area Ottowa County, Oklahoma, March 1983

TABLE 1.6-2

HISTORICAL WATER QUALITY DATA SUMMARY FOR WILLOW CREEK AND SPRING RIVER KANSAS

Station		Cadmium Dissolved (mg/l)	Cadmium Total (mg/l)	Copper Dissolved (mg/l)	Copper Total (mg/l)	Iron Dissolved (mg/l)	Iron Total (mg/l)	Lead Dissolved (mg/l)	Lead Total (mg/l)	Manganese Dissolved (mg/l)	Manganese Total (mg/l)	Zinc Dissolved (mg/l)	Zinc Total (mg/l)
WILLOW CREEK													
(1)Willow Creek at Baxter Springs, KS	Average	0.000	—	0.005	—	0.24	—	0.000	—	0.12	—	0.39	—
From 6/81 to 8/81	Maximum	0.000	—	0.010	—	0.30	—	0.000	—	0.14	—	0.63	—
	Minimum	0.000	—	0.000	—	0.17	—	0.000	—	0.10	—	0.14	—
	Number of Samples	2	—	2	—	2	—	2	—	2	—	2	—
(1)Willow Creek Tributary 3 near BS,KS	Sampled on 3-17-82	0.110	—	0.000	—	0.03	—	0.000	—	0.09	—	13.60	—
(1)Willow Creek 1 mile W. BS, KS	Sampled on 8-13-81	0.000	—	0.010	—	0.09	—	0.000	—	0.04	—	0.15	—
(1)Willow Creek 2 miles W. BS, KS	Sampled on 8-13-81	0.000	—	0.010	—	0.25	—	0.000	—	0.15	—	0.28	—
(1)Willow Creek 3 miles W. BS, KS	Sampled on 6-17-81	0.000	—	0.000	—	0.19	—	0.000	—	0.35	—	0.18	—
SPRING RIVER													
(2)Spring River near	Average	0.002	—	0.051	—	—	1.10	0.070	—	—	0.25	0.66	—
Baxter Springs, KS	Maximum	0.010	—	0.100	—	—	1.60	0.500	—	—	0.27	1.60	—
From 2/74 to 6/78	Minimum	0.000	—	0.000	—	—	0.19	0.000	—	—	0.22	0.24	—
	Number of Samples	13	—	14	—	—	3	12	—	—	2	14	—
(2)Spring River near	Average	—	0.003	—	0.012	0.762	—	—	0.024	0.225	—	—	0.37
Baxter Springs, KS	Maximum	—	0.016	—	0.030	10.08	—	—	0.200	0.974	—	—	1.71
From 3/79 to 11/91	Minimum	—	0.001	—	0.000	0.04	—	—	0.000	0.010	—	—	0.09
	Number of Samples	—	21	—	21	44	—	—	16	44	—	—	21
(1)Spring River near BS, KS	Sampled on 8-11-81	0.000	—	0.020	—	0.02	—	0.000	—	0.11	—	0.09	—

Sources: (1) U.S. Geological Survey Open-File Report 84-439: Assessment of Water Resources in Lead-Zinc Mined Areas in Cherokee County, and Adjacent Areas.

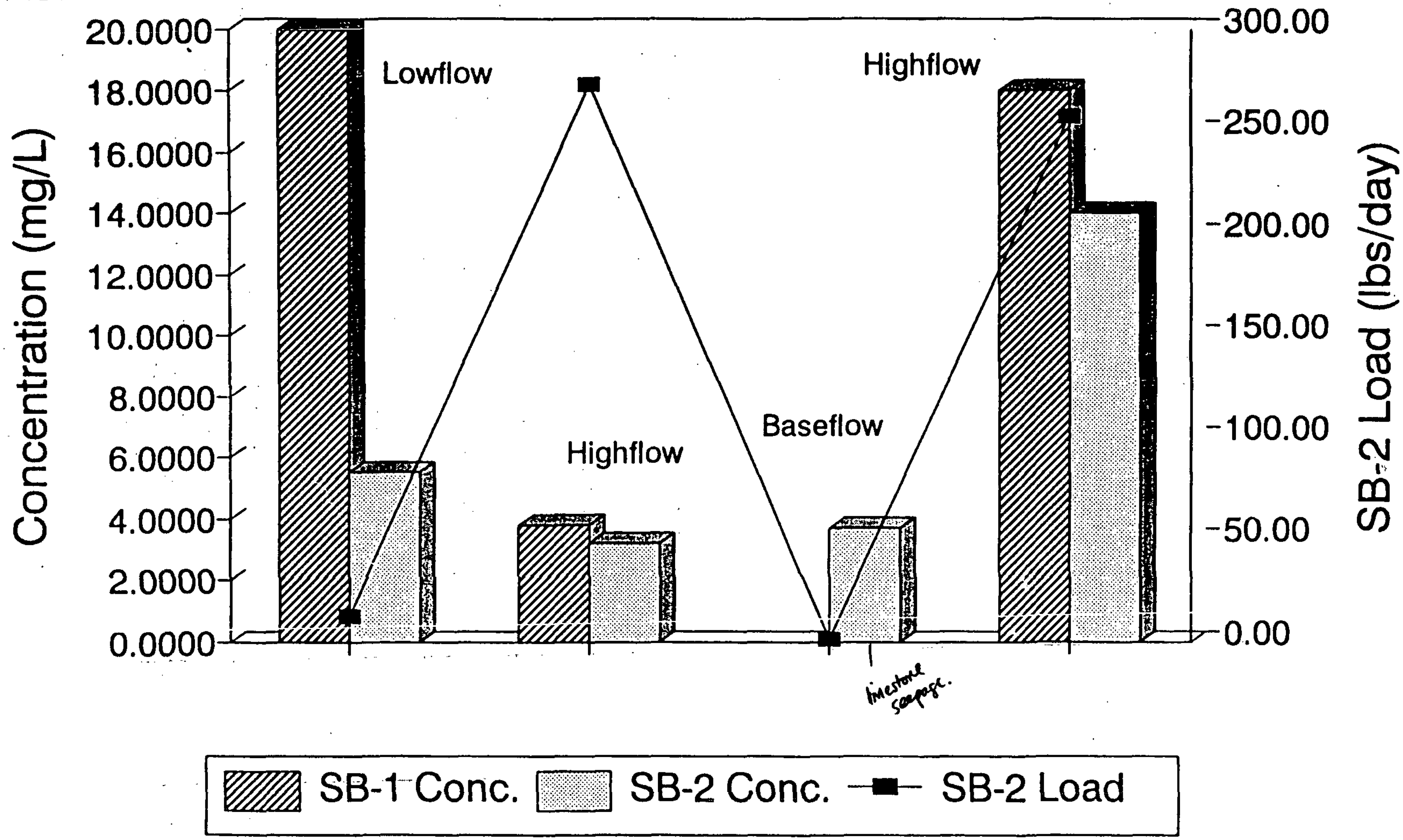
(2) Kansas Department of Health and Environment - Storet Data Summary 12/91. Dates of actual data may vary slightly among constituents.

TABLE 1.6-4
HISTORICAL WATER QUALITY DATA SUMMARY FOR TAR CREEK AND NEOSHO RIVER IN OKLAHOMA

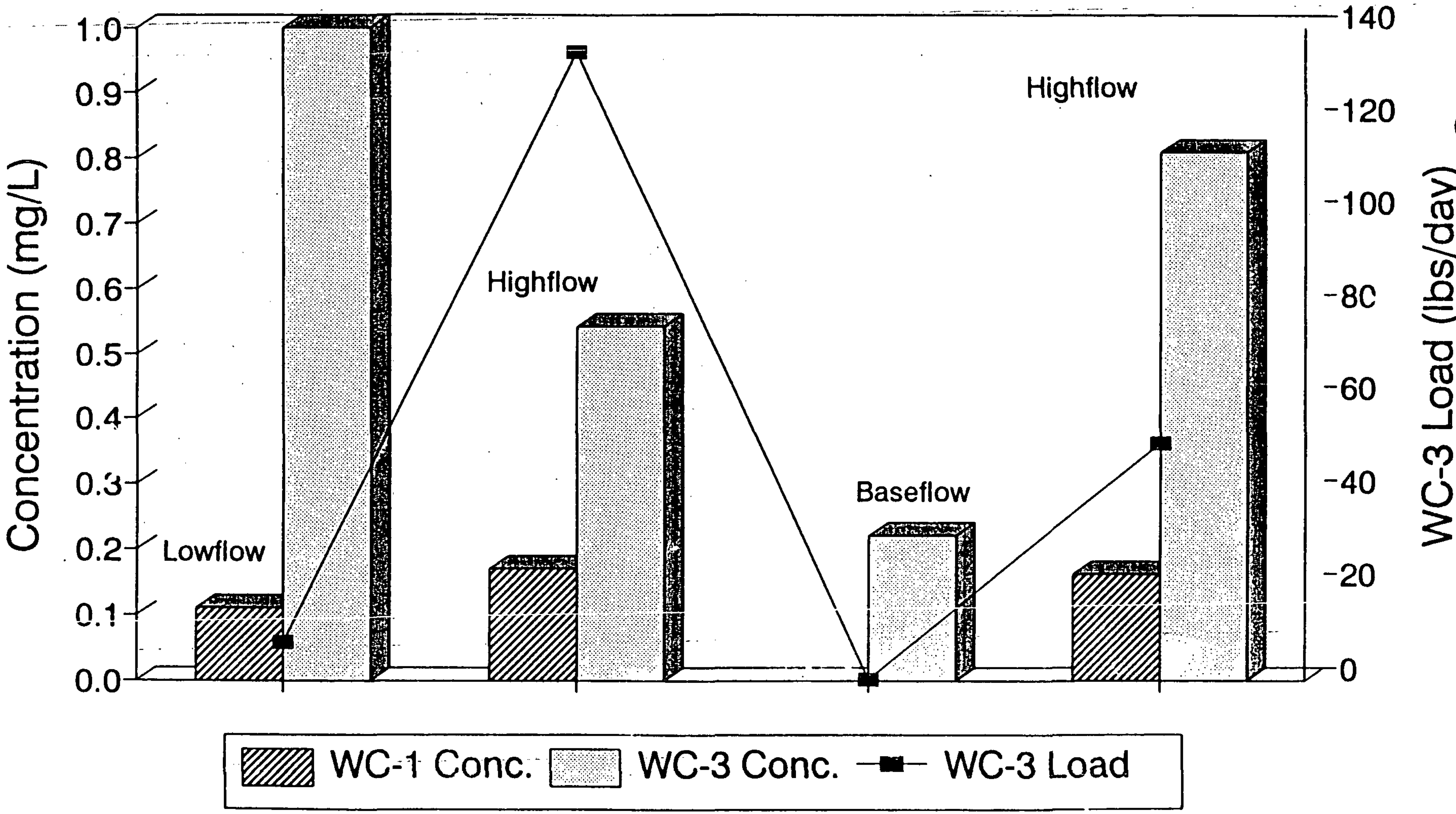
Station		Cadmium Dissolved (mg/l)	Cadmium Total (mg/l)	Copper Dissolved (mg/l)	Copper Total (mg/l)	Iron Dissolved (mg/l)	Iron Total (mg/l)	Lead Dissolved (mg/l)	Lead Total (mg/l)	Manganes Dissolved (mg/l)	Manganes Total (mg/l)	Zinc Dissolved (mg/l)	Zinc Total (mg/l)
TAR CREEK													
(1)Tar Creek on the Oklahoma-	Average	0.054	0.017	0.010	0.011	1.06	5.68	0.028	0.058	0.19	0.18	9.06	5.87
Kansas state line south of	Maximum	0.230	0.023	0.017	0.020	3.20	5.20	0.066	0.247	0.50	0.37	39.40	13.60
low water bridge.	Minimum	0.009	0.011	0.004	0.004	0.10	0.15	0.020	0.020	0.05	0.08	1.24	2.10
From 2/80 to 12/82	Number of Samples	8	10	3	3	6	10	6	8	5	5	8	10
(1)Tar Creek on the Oklahoma-	Average	0.016	0.017	—	—	8.18	0.80	0.038	0.107	—	—	3.24	3.52
Kansas state line south of	Maximum	0.034	0.035	—	—	64.30	1.70	0.045	0.469	—	—	7.21	7.34
low water bridge.	Minimum	0.006	0.006	—	—	0.05	0.18	0.030	0.030	—	—	1.50	1.86
From 1/87 to 2/89	Number of Samples	3	8	—	—	8	8	7	6	—	—	7	8
(4)Tar Creek on the Oklahoma-	Average	—	0.0176	—	—	—	7.871	—	0.0718	—	—	—	6.483
Kansas state line south of	Maximum	—	0.023	—	—	—	52	—	0.247	—	—	—	13.8
the low water bridge.	Minimum	—	0.011	—	—	—	0.15	—	0.02	—	—	—	2.1
6/82 to 8/82	Number of Samples	—	7	—	—	—	7	—	7	—	—	—	7
(2)Tar Creek at Treece, KS	Average	0.020	—	0.007	—	0.12	—	0.010	—	0.11	—	2.87	—
From 6/81 to 3/82	Maximum	0.040	—	0.010	—	0.18	—	0.030	—	0.16	—	5.80	—
	Minimum	0.010	—	0.000	—	0.02	—	0.000	—	0.04	—	1.30	—
	Number of Samples	3	—	3	—	3	—	3	—	3	—	3	—
(2)Tar Creek 1 mile NW Treece, KS	Sampled on 8-13-81	0.000	—	0.000	—	0.17	—	0.000	—	1.10	—	0.22	—
(2)Tar Creek near Cravensville, KS	Sampled on 6-17-81	0.000	—	0.000	—	0.22	—	0.000	—	0.11	—	0.07	—
(2)Chat Seepage near Treece, KS	Sampled on 3-18-82	0.057	—	0.010	—	0.00	—	0.010	—	0.14	—	5.80	—
(1)Tar Creek above Neosho River, OK	Average	0.003	0.005	0.004	0.004	0.20	1.26	0.020	0.020	0.165	0.335	1.803	6.083
From 6/81 to 8/82	Maximum	0.004	0.011	—	—	0.30	2.89	0.020	0.020	0.200	0.480	3.480	14.200
	Minimum	0.002	0.002	—	—	0.10	0.10	0.020	0.020	0.130	0.190	0.690	1.550
	Number of Samples	3	3	1	1	3	3	2	2	2	2	3	3
(4)Tar Creek 1 mile upstream from	Average	—	0.0186	—	—	—	8.853	—	0.033	—	—	—	21.333
the Neosho River.	Maximum	—	0.063	—	—	—	52	—	0.196	—	—	—	104
From 6/82 to 8/82	Minimum	—	0.002	—	—	—	0.55	—	0.02	—	—	—	0.281
	Number of Samples	—	20	—	—	—	20	—	19	—	—	—	20
NEOSHO RIVER													
(3)Neosho River near Commerce, OK	Average	0.0015	0.0015	0.0088	0.0111	0.0885	1.875	0.0032	0.0128	0.0173	0.26	0.0275	0.0614
and above Tar Creek Confluence	Maximum	0.004	0.002	0.06	0.048	0.46	55	0.02	0.025	0.048	2.2	0.15	0.33
From 10/66 - 5/89	Minimum	0.001	0.0	0.0	0.002	0.0	0.0	0.0	0.002	0.002	0.0	0.0	0.004
	Number of Samples	11	24	24	24	20	120	23	24	11	56	24	24
(4)Neosho River 300 yards upstream	Average	—	0.0027	—	—	—	1.703	—	0.02	—	—	—	0.485
of the confluence with Tar Creek	Maximum	—	0.003	—	—	—	2.86	—	0.02	—	—	—	1.19
During 8/82	Minimum	—	0.002	—	—	—	1.1	—	0.02	—	—	—	0.066
	Number of Samples	—	3	—	—	—	3	—	2	—	—	—	3
(4)Neosho River 300 yards downstream	Average	—	0.0021	—	—	—	1.083	—	0.02	—	—	—	0.325
of the confluence with Tar Creek	Maximum	—	0.003	—	—	—	2.59	—	0.02	—	—	—	0.72
During 8/82	Minimum	—	0.002	—	—	—	0.28	—	0.02	—	—	—	0.058
	Number of Samples	—	6	—	—	—	6	—	6	—	—	—	6

Sources: (1) Oklahoma State Health Department - Water Quality Data.
(2) U.S. Geological Survey Open-File Report 84-439: Assessment of Water Resources in Lead-Zinc Mined Areas in Cherokee County, and Adjacent Areas.
(3) Kansas Department of Health and Environment - Storet Data Summary 12/91. Dates of actual data may vary slightly among constituents.
(4) Oklahoma Water Resources Board, Effects of Acid Mine Discharge on the Surface Water Resources in the Tar Creek Area Ottawa County, Oklahoma, March 1983

Spring Branch
Total Rec. Zinc Analysis

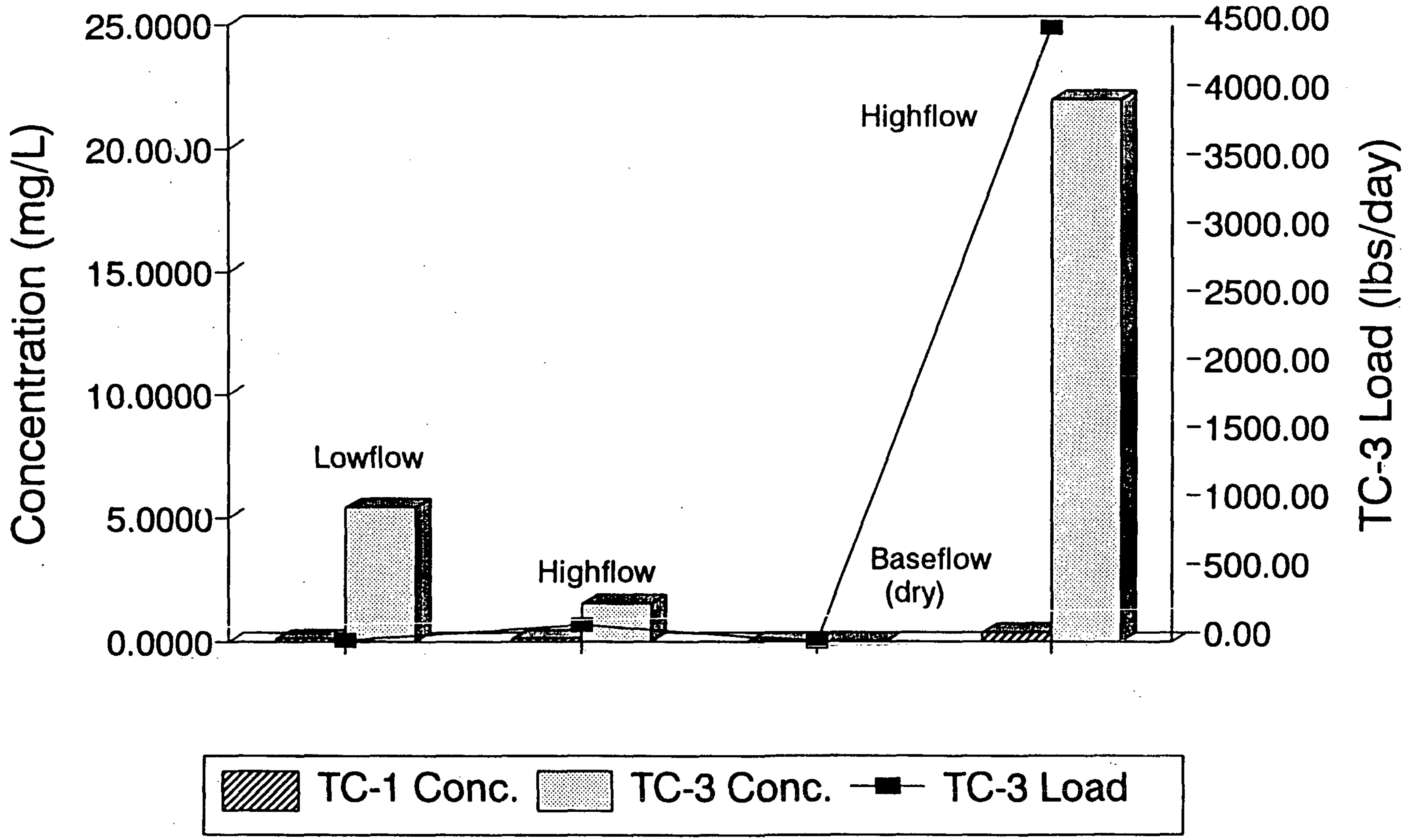


Willow Creek
Total Rec. Zinc Analysis

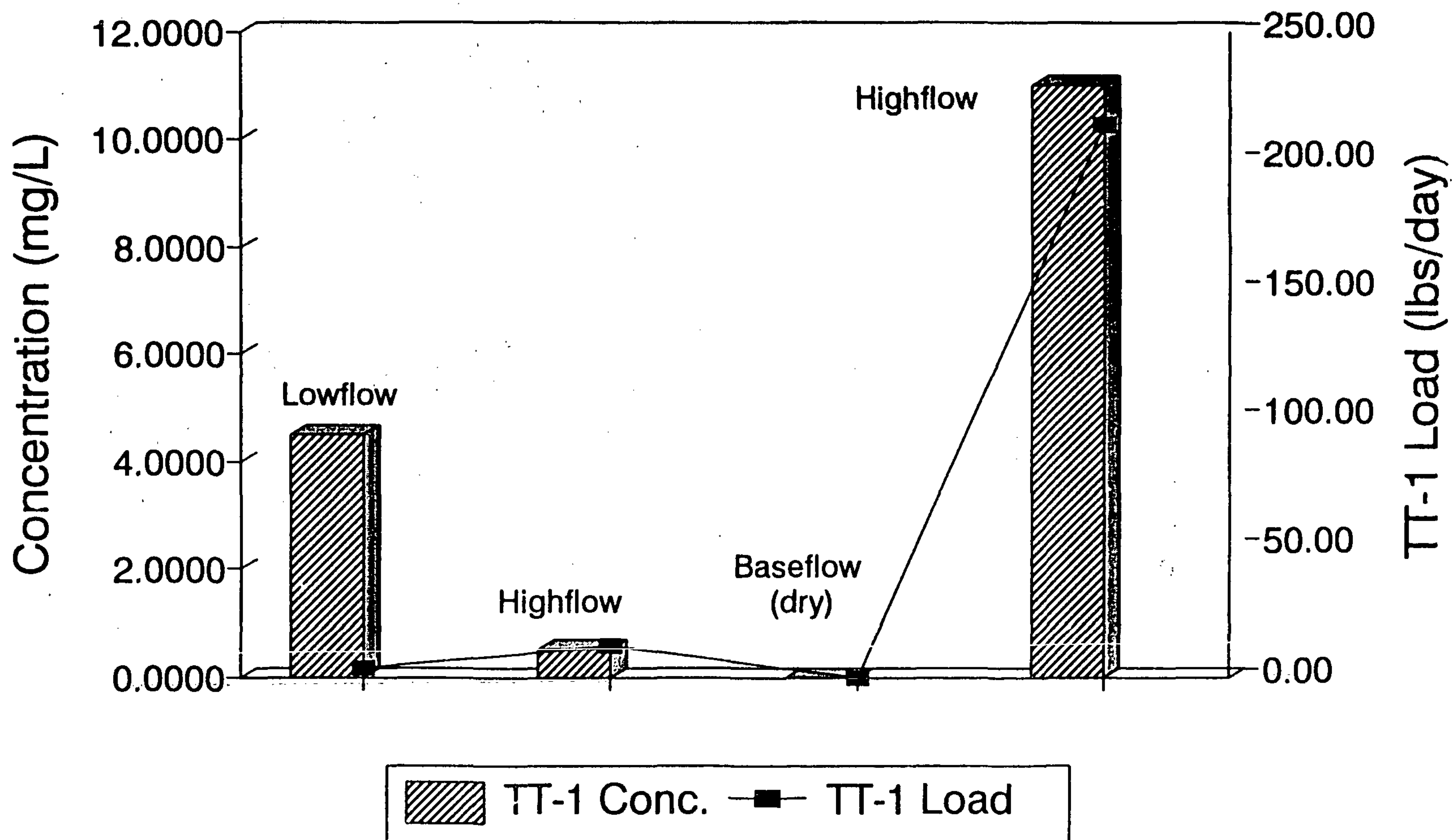


Tar Creek

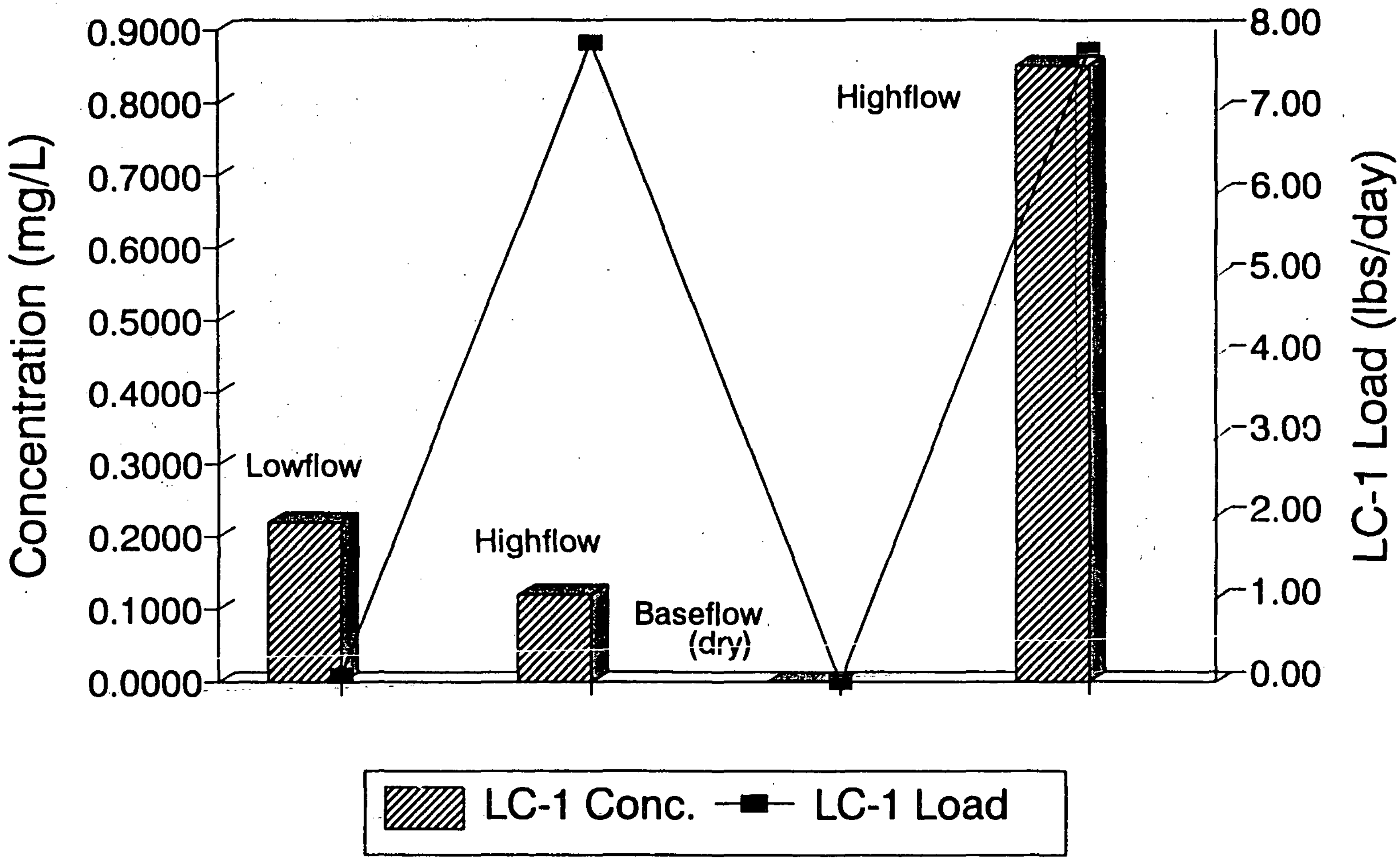
Total Rec. Zinc Analysis



Tar Creek Tributary
Total Rec. Zinc Analysis



Lytle Creek
Total Rec. Zinc Analysis



CHEROKEE STATS SW

1) NEOSHO RIVER NEAR COMMERCE

Hardness $CaCO_3$ mg/l [8/27/44 – 8/26/80]

MAX = 535

MIN = 51

AVG = 204.5900

n = 575

2.) SPRING RIVER NEAR BAXTER SPRIN[GS]

HARDNESS $CaCO_3$ (mg/l) [7/6/67 - 11/5/91]

MAX = 236

MIN = 60

AVG = 159.5600

n = 216

Cd, Cu, Pb, Zn, Mn [TOTAL REC.]

3.) SPRING RIVER NEAR WACO

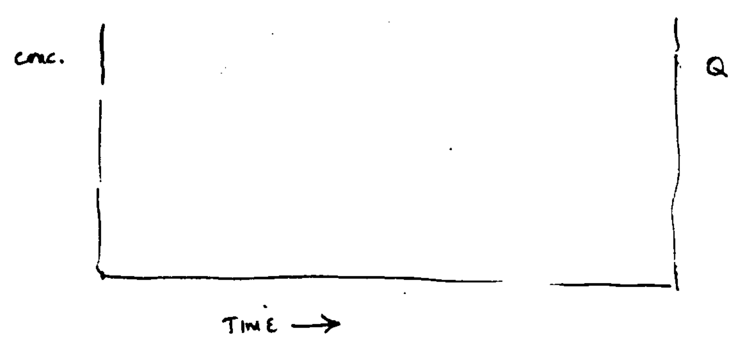
HARDNESS $CaCO_3$ (mg/l) [7/11/67-10/31/

MAX = 216

MIN = 41

AVG = 150.1300

n = 205



APPENDIX C

PHYTOTOXICITY DATA

Phytotoxicity of total lead in soils.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/Part	Hazard Response	Significance Level	Reference
Drummer Silt Loam	1400 (Calc)	5.9	Pb Acetate	Field	Corn/Stover-Grain	No Effect	NR	Baumhardt and Welch (1972)
Hjorth Silty Clay Loam	1000	3.8	$PbCl_2$	Greenhouse/Soil Pots	Lettuce/Leaf	35.5 % YR	0.05	John and Van Laerhoven (1972)
Hjorth Silty Clay Loam	1000	3.8	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Lettuce/Leaf	25.0 % YR	0.05	John and Van Laerhoven (1972)
Hjorth Silty Clay Loam	1000	3.8	$PbCO_3$	Greenhouse/Soil Pots	Lettuce/Leaf	17.1 % YR	0.05	John and Van Laerhoven (1972)
Hjorth Silty Clay Loam	1000	3.8	$PbCl_2$	Greenhouse/Soil Pots	Oats/Tops	No Effect	0.05	John and Van Laerhoven (1972)
Hjorth Silty Clay Loam	1000	3.8	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Oats/Tops	No Effect	0.05	John and Van Laerhoven (1972)
Hjorth Silty Clay Loam	1000	3.8	$PbCO_3$	Greenhouse/Soil Pots	Oats/Tops	No Effect	0.05	John and Van Laerhoven (1972)
Yolo Loam	1000	4.0	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Barley/Tops	33.3 % YR	0.05	Patel et al. (1977)
Yolo Loam	1000	6.0	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Barley/Tops	17.3 % YR	0.05	Patel et al. (1977)
Yolo Loam	1000	7.8	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Barley/Tops	1.9 % YR (N.S.)	0.05	Patel et al. (1977)
Yolo Loam	1000	8.5	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Barley/Tops	No Effect	0.05	Patel et al. (1977)
Dytchleys Brown Earth	1000	NR	$PbCl_2$	Greenhouse/Soil Pots	Oats/Roots	42.9 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	1000	NR	PbO	Greenhouse/Soil Pots	Wheat/Roots	6.7 % YR (N.S.)	0.05	Khan and Frankland (1984)
Weald Park Brown Earth	1000	NR	$PbCO_3$	Greenhouse/Soil Pots	Wheat/Roots	12.8 % YR	0.05	Khan and Frankland (1984)
Weald Park Brown Earth	1000	NR	$PbSO_4$	Greenhouse/Soil Pots	Wheat/Roots	7.4 % YR (N.S.)	0.05	Khan and Frankland (1984)
Weald Park Brown Earth	1000	NR	$PbCl_2$	Greenhouse/Soil Pots	Wheat Roots	33.7 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	1000	NR	$PbCl_2$/PbO	Greenhouse/Soil Pots	Radish/Roots	19.8 % YR	0.01	Khan and Frankland (1984)
Dytchleys Brown Earth	500	NR	$PbCl_2$	Greenhouse/Soil Pots	Oat/Roots	36.8 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	500	NR	$PbCl_2$	Greenhouse/Soil Pots	Wheat/Roots	14.8 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	500	NR	$PbCl_2$/PbO	Greenhouse/Soil Pots	Radish/Roots	4.6 % YR (N.S.)	0.05	Khan and Frankland (1984)
	400				Oats	No YR		Pruves (1977)
	400				Lettuce	No YR		Pruves (1977)
	400				Clover	No YR		Pruves (1977)
Paxton Fine Sandy Loam	250	4.5-6.4	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Ryegrass/Tops	No YR	0.01	Allinson and Dzlaco (1981)
Paxton Fine Sandy Loam	250	4.5-6.4	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Oats/Seed	No YR	0.01	Allinson and Dzlaco (1981)
Merrimac Fine Sandy Loam	250	6.9	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Alfalfa/Tops	17.9 % YR (N.S.)	0.01	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	250	6.9	$Pb(NO_3)_2$	Greenhouse/Soil Pots	Alfalfa/Tops	6.7 % YR (N.S.)	0.01	Taylor and Allinson (1981)
Bloomfield Loamy Sand	250	6.0	$PbCl_2$	Greenhouse/Soil Pots	Corn/Shoots	41.7 % YR	0.01	Miller et al. (1977)

Phytotoxicity of total lead in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Light Textured	214	5.-8.1	Sludge	Field	Spring Greens	Satisfactory Yields	NA	Shumbley and Unwin (1982)
Chester Silt Loam	212	5.2	$PbCl_2$	Greenhouse/Soil Pots	Corn/Tops	2.1 % YR (N.S.)	0.05	Lagerwerff et al. (1973)
Chester Silt Loam	212	7.2	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	12.1 % YR (N.S.)	0.05	Lagerwerff et al. (1973)
Chester Silt Loam	212	5.2	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	2.8 % YR (N.S.)	0.05	Lagerwerff et al. (1973)
Chester Silt Loam	212	7.2	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	17.5 % Yield Increase	0.05	Lagerwerff et al. (19[illegible]
Sango Silt	186	5.6	Sludge	Field	Corn/Grain	No YR	0.05	Giordano et al. (1975[illegible]
Light Textured	176	5.-8.1	Sludge	Field	Potato (Tuber)	Satisfactory Yields	NA	Chumbley and Unwin (1[illegible]
Light Textured	156	5.-8.1	Sludge	Field	Sweet Corn (Edible POR)	Satisfactory Yields	NA	Chumbley and Unwin (1982)
Light Textured	155	5.-8.1	Sludge	Field	Lettuce (Edible POR)	Satisfactory Yields	NA	Chumbley and Unwin (1982)
Bloomfield Loamy Sand	125	6.0	$PbCl_2$	Greenhouse/Soil Pots	Corn/Shoots	13.5 % YR (N.S.)	0.01	Miller et al. (1977)
Light Textured	117	5.-8.1	Sludge	Field	Cabbage	Satisfactory Yields	NA	Chumbley and Unwin (1982)
Chester Silt Loam	113	5.2	$PbCl_2$	Greenhouse/Soil Pots	Corn/Tops	7.8 % Yield Increase	0.05	Lagerwerff et al. (1973)
Chester Silt Loam	113	7.2	$PbCl_2$	Greenhouse/Soil Pots	Corn/Tops	13.8 % YR (N.S.)	0.05	Lagerwerff et al. (1973)
Chester Silt Loam	113	5.2-7.2	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	No Effect	0.05	Lagerwerff et al. (1973)
Oxbow Loam	109	7.7	$PbCl_2$	Greenhouse/Soil Pots	Bromegrass/Tops	7.9 % YR from 29 ppm (N.S.)	0.05	Karamanos et al. (1976)
Oxbow Loam	109	7.7	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	24.5% YR from 29 ppm	0.05	Karamanos et al. (1976)
Waltville Loam	108	6.3	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	0.09 % YR from 28 ppm (N.S.)	0.05	Karamanos et al. (1976)
Asquith Fine Sandy Loam	106	6.6	$PbCl_2$	Greenhouse/Soil Pots	Alfalfa/Tops	18.7 % YR from 26 ppm (N.S.)	0.05	Karamanos et al. (1976)
Asquith Fine Sandy Loam	106	6.6	$PbCl_2$	Greenhouse/Soil Pots	Bromegrass/Tops	17.8 % Yield Increase 26 ppm (N.S.)	0.05	Karamanos et al. (1976)
Dytchleys Brown Earth	100	NR	$PbCl_2$	Greenhouse/Soil Pots	Oats/Roots	15.9 % Yr (N.S.)	0.05	Khan and Frankland (1983)
Oxbow Loam	9	7.7	None	Field	NR	Background	NA	Karamanos et al. (1976)
Waltville Loam	8	6.3	None	Field	NR	Background	NA	Karamanos et al. (1976[illegible]
Asquith Fine Sandy Loam	6	6.6	None	Field	NR	Background	NA	Karamanos et al. (1976[illegible]

Phytotoxicity of total zinc in soils.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Hartsells Fine Sandy Loam	960	5.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	98.2 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	960	6.0	$ZnSO_4$	Greenhouse Soil Pots	Corn/Forage	96.7 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	960	6.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	96.7 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	960	7.0	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	86.7 % YR	NR	Mortvedt and Giordano (1975)
Domino Silt Loam	660	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	75 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	660	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	53 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	660	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	27 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	660	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	81 % YR	NR	Mitchell et al. (1978)
Blount Silt Loam	606	7.4	Sludge	Field	Corn/Stover	No YR	0.05	Hinesly et al. (1982)
Blount Silt Loam	606	7.4	Sludge	Field	Corn/Grain	No YR	0.05	Hinesly et al. (1982)
Redding Fine Sandy Loam	580	5.7	Sludge/$ZnSO_4$	Greenhouse/Soil Pots	Wheat/Grain	25 % YR	0.05	Mitchell et al. (1978)
Sassafras Silt Loam	524	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	72.4 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	524	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	26.2 % YR	NR	White and Chaney (1980)
Shano Silt Loam 15-30 cm	>500	7.0	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Pea/Tops	8 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	>500	7.0	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Clover/Tops	9 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	>500	7.0	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Potato/Tops	8 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	500	7.0	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Tomato/Tops	26 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	500	7.0	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Lettuce/Tops	31 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	400	7.1	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	17 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	400	7.1	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Field Corn/Tops	26 % YR	0.05	Boawn and Rasmussen (1971)
Sassafras Silt Loam	393	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	33.3 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	393	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	15.9 % YR	NR	White and Chaney (1980)
Domino Silt Loam	340	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	29 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	340	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	12 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	340	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	12 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	340	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	55 % YR	NR	Mitchell et al. (1978)
Lakeland Sand	300	NR	$ZnSO_4$	Greenhouse/Soil Pots	Slash Pine Seedling/ Shoots	59.6 % YR	NR	VanLear and Smith (1972)
Shano Silt Loam 15-30 cm	300	7.3	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Wheat/Tops	18 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	300	7.3	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Sweet Corn/Tops	32 % YR	0.05	Boawn and Rasmussen (1971)
Sassagras Silt Loam	262	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans	10.3 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	252	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans	22.1 % YR	NR	White and Chaney (1980)
Hartsells Fine Sandy Loam	240	5.9	Sludge	Greenhouse/Soil Pots	Corn/Forage	Yield Increase	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	240	5.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	49.1 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	240	6.0	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	35.0 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	240	6.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	8.3 % YR	NR	Mortvedt and Giordano (1975)
Hartsells Fine Sandy Loam	240	7.0	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	5.0 % YR	NR	Mortvedt and Giordano (1975)
Shano Silt Loam 15-30 cm	200	7.5	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Barley/Tops	16 % YR	0.05	Boawn and Rasmussen (1971)
Shano Silt Loam 15-30 cm	200	7.5	$Zn(NO_3)_2 \cdot 6H_2O$	Greenhouse/Soil Pots	Sorghum/Tops	30 % YR	0.05	Boawn and Rasmussen (1971)

Phytotoxicity of total zinc in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Sassafras Silt Loam	196	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	81.6 % YR	NR	White and Chaney (1980)
Sassafras Silt Loam	196	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	0.6 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	196	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	6.4 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	196	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	13.0 % YR	NR	White and Chaney (1980)
Domino Silt Loam	180	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	12 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	180	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	No YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	180	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	9 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	180	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	32 % YR	NR	Mitchell et al. (1978)
Sassafras Silt Loam	131	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	20.1 % YR	NR	White and Chaney (1980)
Sassafras Silt Loam	131	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	19.9 % Yield Increase	NR	White and Chaney (1980)
Pocomoke Silt Loam	131	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	10.1 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	131	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	0.7 % YR	NR	White and Chaney (1980)
Redding Fine Sandy Loam	130	5.7	Sludge/$ZnSO_4$	Greenhouse/Soil Pots	Lettuce/Shoots	25 % YR	0.05	Mitchell et al. (1978)
Domino Silt Loam	100	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	14 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	100	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	4 % Yield Increase	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	100	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	3 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	100	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	13 % YR	NR	Mitchell et al. (1978)
Sassafras Silt Loam	65	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	0.2 % Yield Increase	NR	White and Chaney (1980)
Sassafras Silt Loam	65	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	10.3 % Yield Increase	NR	White and Chaney (1980)
Pocomoke Silt Loam	65	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	0.6 % YR	NR	White and Chaney (1980)
Pocomoke Silt Loam	65	6.3	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	10.3 % YR	NR	White and Chaney (1980)
16 Minn. Surface Soils	60	5.3-8.2	None	Field	NR	Background	NA	Pierce et al. (1982)
Hartsells Fine Sandy Loam	60	5.5	Sludge	Greenhouse/Soil Pots	Corn/Forage	Yield Increase	NR	Mortvedt and Giordano (197
Hartsells Fine Sandy Loam	60	5.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	No YR	NR	Mortvedt and Giordano (197
Hartsells Fine Sandy Loam	60	6.0	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	5 % YR	NR	Mortvedt and Giordano (197
Hartsells Fine Sandy Loam	60	6.5	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	Yield Increase	NR	Mortvedt and Giordano (197
Hartsells Fine Sandy Loam	60	7.0	$ZnSO_4$	Greenhouse/Soil Pots	Corn/Forage	Yield Increase	NR	Mortvedt and Giordano (197
Lakeland Sand	60	NR	$ZnSO_4$	Greenhouse/Soil Pots	Slash Pine Seedlings/ Shoots	42.9 % YR	NR	VanLear and Smith (1972)
Domino Silt Loam	60	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	6 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	60	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	10 % Yield Increase	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	60	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	6 % Yield Increase	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	60	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	2 % YR	NR	Mitchell et al. (1978)
16 Minn. Soils Series - All Depths	54	5.3-8.2	None	Field	NR	Background	NR	Pierce et al. (1982)
16 Minn. Soils Parent Material	52	5.3-8.2	None	Field	NR	Background	NR	Pierce et al. (1982)
16 Minn. Subsoils	49	5.3-8.2	None	Field	NR	Background	NR	Pierce et al. (1982)

Phytotoxicity of total zinc in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
13 Laden Fine Sandy Loam	41.3	NR	None	Greenhouse/Soil Pots	Slash Pine Seedlings/ Shoots	Background	NR	VanLear and Smith (1972)
Domino Silt Loam	40	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	6 % YR	NR	Mitchell et al. (1978)
Domino Silt Loam	40	7.5	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	4 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	40	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Wheat/Grain	2 % YR	NR	Mitchell et al. (1978)
Redding Fine Sandy Loam	40	5.7	$ZnSO_4$/Sludge	Greenhouse/Soil Pots	Lettuce/Tops	No YR	NR	Mitchell et al. (1978)
Leon Fine Sand	37.5	NR	None	Greenhouse/Soil Pots	Slash Pine Seedlings/ Shoots	Background	NR	VanLear and Smith (1972)
Sassafras Silt Loam	33	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	9.7 % Yield Increase	NR	White and Chaney (1980)
Pocomoke Silt Loam	33	5.5	$ZnSO_4 \cdot 7H_2O$	Greenhouse/Soil Pots	Soybeans/Leaf	9.5 % YR	NR	White and Chaney (1980)
Lakeland Sand	30	NR	$ZnSO_4$	Greenhouse/Soil Pots	Slash Pine Seedlings/ Shoots	11.8 % YR	NR	VanLear and Smith* (1972)
Lakeland Sand	30	NR	None	Greenhouse/Soil Pots	Slash Pine Seedlings/ Shoots	Background	NR	VanLear and Smith (1972)

Phytotoxicity of total cadmium in soils.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Domino Silt Loam	5640	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Rice/Grain	25 % YR	NR	Bingham et al. (1975)
Merrimac Fine Sandy Loam	250	6.9	$Cd(NO_3)_2\ 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	46.5 % YR (N.S.)	0.01	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	250	6.9	$Cd(NO_3)_2\ 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	71.9 % YR	0.01	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	250	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	21 % YR	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	250	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	62.1 % YR	NR	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	250	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	29.0 % YR	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	250	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	67.4 % YR	NR	Taylor and Allinson (1981)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Grain	56.0 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Leaves	10.2 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Stalks	22.1 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Carrots/Tubers	96.4 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Radish/Tubers	93.2 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Peas/Pods	92.1 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Peas/Seed	99.2 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Cauliflower/Leaves	96.9 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Broccoli/Leaves	83.3 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Spinach/Leaves	98.5 % YR	0.05	John (1973)
Hazelwood Silt Loam	200	5.1	$CdCl_2$	Greenhouse/Soil Pots	Leaf Lettuce/Leaves	91.1 % YR	0.05	John (1973)
Domino Silt Loam	170	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Cabbage/Head	25 % YR	NR	Bingham et al. (1975)
Domino Silt Loam	160	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Bermuda Grass/Tops	25 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	160	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Tomato/Ripe Fruit	25 % YR	NR	Bingham et al. (1975)
Domino Silt Loam	160	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Zucchini/Fruit	25 % YR	NR	Bingham et al. (1975)
Domino Silt Loam	160	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Sudan Grass/Tops	98 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	160	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	White Clover/Tops	59 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	160	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	56 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	160	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Tall Fescue/Tops	30 % YR	NR	Bingham et al. (1976)
Redding Fine Sandy Loam	125	5.7	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Lettuce/Shoots	25 % YR	0.05	Mitchell et al. (1978)
Merrimac Fine Sandy Loam	125	6.9	$Cd(NO_3)_2\ 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	15.8 % YR (N.S.)	0.01	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	125	6.9	$Cd(NO_3)_2\ 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	56.2 % YR	0.01	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	125	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	0.7 % Yield Increase	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	125	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	23.6 % YR	NR	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	125	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	13.0 % YR	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	125	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	31.2 % YR	NR	Taylor and Allinson (1981)
Plainfield Sand	100.3	4.9	$CdCl_2$	Greenhouse/Soil Pots	7 Species Native Plants	Almost Total Mortality	NR	Miles and Parker (1979)
Marengo Silty Clay Loam	100	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	73.0 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	100	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops	85.6 % YR	NR	Haghiri (1973)
Weald Park Brown Earth	100	NR	CdO	Greenhouse/Soil Pots	Wheat/Roots	47.5 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	100	NR	$CdCO_3$	Greenhouse/Soil Pots	Wheat/Roots	13.4 % YR	0.05	Khan and Frankland (1984)
Weald Park Brown Earth	100	NR	$CdSO_4$	Greenhouse/Soil Pots	Wheat/Roots	67.7 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	100	NR	$CdCl_2$	Greenhouse/Soil Pots	Radish/Roots	42.6 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	100	NR	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Roots	67.7 % YR	0.01	Khan and Frankland (1984)
Dytchleys Brown Earth	100	NR	$CdCl_2$	Greenhouse/Soil Pots	Oats/Roots	76.7 % YR	0.01	Khan and Frankland (1984)
Domino Silt Loam	96	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Radish/Tuber	25 % YR	NR	Bingham et al. (1975)
Domino Silt Loam	80	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Sudan Grass/Tops	57 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	80	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	White Clover/Tops	43 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	80	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	40 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	80	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Tall Fescue/Tops	21 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	80	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Bermuda Grass/Tops	12 % YR	NR	Bingham et al. (1976)

Phytotoxicity of total cadmium in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Redding Fine Sandy Loam	60	5.7	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Wheat/Leaves	25 % YR	0.05	Mitchell et al. (1978)
Paxton Fine Sandy Loam	50	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	9.8 % Yield Increase	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	50	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfala/Tops	3.6 % YR	NR	Taylor and Allinson (1981)
Paxton Fine Sandy Loam	50	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	3.5 % Yield Increase	NR	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	50	6.9	$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	4.3 % Yield Increase	NR	Taylor and Allinson (1981)
Weald Park Brown Earth	50	NR	$CdCl_2$	Greenhouse/Soil Pots	Radish/Roots	31.9 % YR	0.01	Khan and Frankland (1984)
Weald Park Brown Earth	50	NR	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Roots	61.3 % YR	0.01	Khan and Frankland (1984)
Dytchleys Brown Earth	50	NR	$CdCl_2$	Greenhouse/Soil Pots	Oats/Roots	61.5 % YR	0.01	Khan and Frankland (1984)
Domino Silt Loam	50	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Wheat/Grain	25 % YR	NR	Bingham et al. (1975)
Merrimac Fine Sandy Loam	50	6.9	$Cd(NO_3)_2 \cdot 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	1.4 Yield Increase (N.S.)	0.01	Taylor and Allinson (1981)
Merrimac Fine Sandy Loam	50	6.9	$Cd(NO_3)_2 \cdot 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops - 2nd cutting	27.3 % YR	0.01	Taylor and Allinson (1981)
Flanagan Silt Loam	50	7.3	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Shoots	9.3 % YR	0.01	Boggess et al. (1978)
Marengo Silty Clay Loam	50	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	49.8 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	50	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops	85.3 % YR	NR	Haghiri (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Grain	36.3 % YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Leaves	No YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Oats/Stalks	No YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Carrots/Tubers	8.3 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Radish/Tubers	27.9 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Peas/Pods	29.7 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Peas/Seed	38.1 % YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Cauliflower/Leaves	2.7 % YR (N.S.)	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Broccoli/Leaves	No YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Spinach/Leaves	96 % YR	0.05	John (1973)
Hazelwood Silt Loam	40	5.1	$CdCl_2$	Greenhouse/Soil Pots	Leaf Lettuce/Leaves	No YR	0.05	John (1973)
Domino Silt Loam	40	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Field Bean/Dry Bean	25 % YR	NR	Bingham et al. (1975)
Domino Silt Loam	40	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Sudan Grass/Tops	43 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	40	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	29 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	40	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	White Clover/Tops	21 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	40	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Tall Fescue/Tops	19 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	40	7.5	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Bermuda Grass/Tops	12 % YR	NR	Bingham et al. (1976)
Marengo Silty Clay Loam	40	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	49.8 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	40	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops	84.8 % YR	NR	Haghiri (1973)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Kentucky Bluegrass/ Shoots	90.1 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Little Bluestem/ Shoots	18.1 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Rough Blazing Star/ Shoots	88.5 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Poison Ivy/Shoots	61.3 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Black-eyed Susan/ Shoots	98.5 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Wild Bergamot/Shoots	67.9 % YR	NR	Miles and Parker (1979)
Plainfield Sand	30.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Long-Fruited Thimble Weed/Shoots	30.4 % YR	NR	Miles and Parker (1979)
Marengo Silty Clay Loam	30	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops		NR	Haghiri (1973)
Marengo Silty Clay Loam	30	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops		NR	Haghiri (1973)
Domino Silt Loam	28	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Turnip/Tuber	25 % YR	NR	Bingham et al. (1975)
Flanagan Silt Loam	25	7.3	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Shoots	9.8 % YR	0.01	Boggess et al. (1978)
Domino Silt Loam	20	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Carrots/Tuber	25 % YR	NR	Bingham et al. (1975)
Dytchleys Brown Earth	20	NR	$CdCl_2$	Greenhouse/Soil Pots	Oats/Roots	56.7 % YR	0.01	Khan and Frankland (1984)
Marengo Silty Clay Loam	20	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops		NR	Haghiri (1973)
Marengo Silty Clay Loam	20	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybean/Tops		NR	Haghiri (1973)
Domino Silt Loam	18	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Corn/Kernel	25 % YR	NR	Bingham et al. (1975)
Marengo Silty Clay Loam	15	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	34.0 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	15	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybean/Tops	65.3 % YR	NR	Haghiri (1973)
Domino Silt Loam	[illegible]	7.5-7.8	Sludge/$CdSO_4$	Greenhouse/Soil Pots	Lettuce/Head	[illegible] % YR	NR	Bingham et al. (1975)
Loams	10.8	5-8.1		Field	Potato/Tuber	"Satisfactory Yields"	NR	Chumbley and Unwin (1982)

Phytotoxicity of total cadmium in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Kentucy Bluegrass/ Shoots	18.7 % YR	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Little Bluestem/ Shoots	21.1 % YR	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Rough Blazing Star/ Shoots	29.6 % YR	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Poison Ivy/Shoots	28.9 % Yield Increase	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Black-Eyed Susan/ Shoots	78.5 % YR	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Wild Bergamot/Shoots	23.3 % YR	NR	Miles and Parker (1979)
Plainfield Sand	10.3	4.8	$CdCl_2$	Greenhouse/Soil Pots	Long-Fruited Thimble Weed/Shoots	8.7 % YR	NR	Miles and Parker (1979)
Dytchleys Brown Earth	10	NR	$CdCl_2$	Greenhouse/Soil Pots	Oats/Roots	24.5 % YR	0.01	Khan and Frankland (1984)
Domino Silt Loam	10	7.5	$Sludge/CdSO_4$	Greenhouse/Soil Pots	White Clover/Tops	33 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	10	7.5	$Sludge/CdSO_4$	Greenhouse/Soil Pots	Sudan Grass/Tops	20 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	10	7.5	$Sludge/CdSO_4$	Greenhouse/Soil Pots	Alfalfa/Tops	17 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	10	7.5	$Sludge/CdSO_4$	Greenhouse/Soil Pots	Bermuda Grass/Tops	6 % YR	NR	Bingham et al. (1976)
Domino Silt Loam	10	7.5	$Sludge/CdSO_4$	Greenhouse/Soil Pots	Tall Fescue/Tops	2 % YR	NR	Bingham et al. (1976)
Flanagan Silt Loam	10	7.3	$CdCl_2$	Greenhouse/Soil Pots	Soybean/Shoots	4.3 % YR	0.01	Boggess et al. (1978)
Marengo Silty Clay Loam	10	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	28.4 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	10	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops	49.2 % YR	NR	Haghiri (1973)
Loams	9.3	5.-8.1	Sludge	Field	Spring Greens/Leaves	"Satisfactory Yields"	NR	Chumbley and Unwin (1982)
Loams	7.8	5.-8.1	Sludge	Field	Lettuce/Leaf	"Satisfactory Yields"	NR	Chumbley and Unwin (1982)
Loams	7.8	5.-8.1	Sludge	Field	Sweet Corn/Grain	"Satisfactory Yields"	NR	Chumbley and Unwin (1982)
Loams	6.5	5.-8.1	Sludge	Field	Beet Root/Tuber	"Satisfactory Yields"	NR	Chumbley and Unwin (1982)
Grenville Loam 0-15 cm	5.6	6.6	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	7.5 % YR (N.S.)	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.5	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	13.9 % YR (N.S.)	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.5	Fe Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	8.9 % YR (N.S.)	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.5	Fe Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	21.9 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.7	Al Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	12.7 % YR (N.S.)	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.6	Al Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	15.2 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.6	Mn Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	5.7 % YR (N.S.)	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.7	Mn Precip $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	18.5 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	7.1	$CaCO_3$ + $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	16.6 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	7.1	$CaCO_3$ + $CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	27.2 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	7.0	$CdCl_2$ + $CaCO_3$	Greenhouse/Soil Pots	Lettuce/Tops	14.6 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.9	$CdCl_2$ + $CaCO_3$	Greenhouse/Soil Pots	Lettuce/Tops	23.2 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.8	Sludge	Greenhouse/Soil Pots	Lettuce/Tops	29.3 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	6.7	Sludge	Greenhouse/Soil Pots	Lettuce/Tops	52.3 % Yield Increase	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	7.0	Sludge	Greenhouse/Soil Pots	Lettuce/Tops	19.1 % YR	0.05	Singh (1981)
Grenville Loam 0-15 cm	5.6	7.0	Sludge	Greenhouse/Soil Pots	Lettuce/Tops	55.0 % Yield Increase	0.05	Singh (1981)
Grenville Sandy Loam 0-15 cm	5.60	7.4	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	9.7 % YR	NR	MacLean (1976)
Romona Sandy Loam	5.57	6.0	Sludge	Greenhouse/Soil Pots	Barley-Barsoy/Tops	15 % YR (N.S.)	0.01	Chang et al. (1982)
Romona Sandy Loam	5.57	6.0	Sludge	Greenhouse/Soil Pots	Barley-Briggs/Tops	27 % YR (N.S.)	0.01	Chang et al. (1982)
Romona Sandy Loam	5.57	6.0	Sludge	Greenhouse/Soil Pots	Barley-Florida 103/ Tops	14 % Yield Increase	0.01	Chang et al. (1982)
Romona Sandy Loam	5.57	6.0	Sludge	Greenhouse/Soil Pots	Barley-Larker/Tops	11 % Yield Increase	0.01	Chang et al. (1982)
Uplands Sand 0-15 cm	5.50	5.5	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	5.0 % YR	NR	MacLean (1976)
Uplands Sand 0-15 cm	5.50	7.0	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	9.9 % YR	NR	MacLean (1976)
Rideau Clay 0-15 cm	5.50	6.1	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	0.4 % Yield Increase	NR	MacLean (1976)
Rideau Clay 0-15 cm	5.50	6.8	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	7.6 % YR	NR	MacLean (1976)
Granby Sandy Loam 0-15 cm	5.45	6.7	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	4.2 % YR	NR	MacLean (1976)
Uplands Sand 15-30 cm	5.38	5.2	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	31.3 % YR	NR	MacLean (1976)
Uplands Sand 15-30 cm	5.30	5.8	$CdCl_2$	Greenhouse/Soil Pots	Lettuce/Tops	26.4 % YR	NR	MacLean (1976)
Marengo Silty Clay Loam	5	6.7	$CdCl_2$	Greenhouse/Soil Pots	Wheat/Tops	27.9 % YR	NR	Haghiri (1973)
Marengo Silty Clay Loam	5	6.7	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Tops	10.3 % YR	NR	Haghiri (1973)
Merrimac Fine Sandy Loam	5	6.3	$Cd(NO_3)_2 \cdot 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	25.7 % YR (N.S.)	0.01	Taylor and Allinson
Merrimac Fine Sandy Loam	5	6.9	$Cd(NO_3)_2 \cdot 4H_2O$	Greenhouse/Soil Pots	Alfalfa/Tops	16.5 % YR	0.01	Taylor and Allinson

Phytotoxicity of total cadmium in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Bloomfield Loamy Sand	1.0	5.5	$CdCl_2$	Greenhouse/Soil Pots	Soybeans/Shoots	10.6 % YR from 0.5 ppm Soil Level	0.01	Boggess et al. (1978)
33 Fraser Valley Ag. Soils	0.88		None	Field	Farmland	Background	NR	John et al. (1972)
Grenville Loam 0-15 cm	0.60	6.7	None	Greenhouse/Soil Pots	Lettuce	Background	NR	Singh (1981)
U.S. Soils	0.1-0.8	NR	None	Field	NA	Background	NA	Meyer et al. (1982)
16 Minn. Surface Soils	0.39	5.3-8.2	None	Field	NR	Background	NA	Pierce et al. (1982)
Plainfield Sand	0.33	4.8	None	Field	"Uncontaminated Site"	Background	NA	Miles and Parker (1982)
Domino Silt Loam	0.3	7.8	None	Field	Crop Land	Background	NR	Chang et al. (1982)
16 Minn. Subsoils	0.23	5.3-8.2	None	Field	NR	Background	NA	Pierce et al. (1982)
Greenfield Sandy Loam	0.1	7.1	None	Field	Crop Land	Background	NR	Chang et al. (1982)
Romona Sandy Loam	0.1	6.0	None	Field	Crop Land	Background	NR	Chang et al. (1982)

Phytotoxicity of total arsenic in soils.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Hagerstown Silty Clay Loam	1000	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	100 % YR	0.05	Woolson et al. (1973)
Hagerstown Silty Clay Loam	1000	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	90 % YR	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	1000	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	100 % YR	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	1000	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	100 % YR	0.05	Woolson et al. (1973)
Burnt Fork Cobbly Loam	315	6.1	Smelter Contamination	Field	Corn/Shoots	28 % YR	NR	Woolson et al. (1971)
Hagerstown Silty Clay Loam	100	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	4 % YR (N.S.)	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	100	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	45 % YR	0.05	Woolson et al. (1973)
Hagerstown Silty Clay Loam	100	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	81 % YR	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	100	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	98 % YR	0.05	Woolson et al. (1973)
Plainfield Sand	100	5.5	$NaAsO_2$	Field	Peas/Seeds	94.9 % YR	0.01	Steevens et al. (1972)
Plainfield Sand	100	5.5	$NaAsO_2$	Field	Potatoes/Tubers	75.2 % YR	0.01	Steevens et al. (1972)
Houston Black Clay	90	7.6	As_2O_3	Field Pots	Bermuda Grass/Leaves	Sig. Growth Reduction (50 %)	NR	Weaver et al. (1984)
Wesvood Black Clay	90	7.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	Growth Prevented	NR	Weaver et al. (1984)
Arenosa Fine Sand	90	4.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	Growth Prevented	NR	Weaver et al. (1984)
Avg. 13 Soils	85	NR	NR	NR	Corn	Level of Sig YR	NR	Walsh et al. (1977)
Plainfield Loamy Sand	68	NR	NR	NR	Potato	Level of Sig YR	NR	Walsh et al. (1977)
Plainfield Loamy Sand	68	NR	NR	NR	Sweet Corn	Level of Sig YR	NR	Walsh et al. (1977)
Plainfield Sand	45.0	5.5	$NaAsO_2$	Field	Peas/Seed	39.9 % YR	0.10	Steevens et al. (1972)
Plainfield Sand	45.0	5.5	$NaAsO_2$	Field	Potatoes/Tubers	17.1 % YR	0.10	Steevens et al. (1972)
Houston Black Clay	45	7.6	As_2O_3	Field Pots	Bermuda Grass/Leaves	Slight YR (10 %)	NR	Weaver et al. (1984)
Wesvood Silt Loam	45	7.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	83 % YR	NR	Weaver et al. (1984)
Arenosa Fine Sand	45	4.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	No YR	NR	Weaver et al. (1984)
Colton Loamy Sand	44	NR	NR	NR	Blueberry	Level of Sig YR	NR	Walsh et al. (1977)
Plainfield Sand	27	5.5	$NaAsO_2$	Field	Peas/Seed	2.0 % Yield Increase (N.S.)	0.10	Steevens et al. (1972)
Plainfield Sand	27	5.5	$NaAsO_2$	Field	Potatoes/Tuber	0.6 % YR (N.S.)	0.10	Steevens et al. (1972)
Plainfield Loamy Sand	25	NR	NR	NR	Snap Beans and Peas	Level of Sig YR	NR	Walsh et al. (1977)
Plainfield Sand	14.1	5.5	$NaAsO_2$	Field	Peas/Seed	15.0 % Yield Increase (N.S.)	0.10	Steevens et al. (1972)
Plainfield Sand	14.1	5.5	$NaAsO_2$	Field	Potatoes/Tubers	1.7 % YR (N.S.)	0.10	Steevens et al. (1972)
Hagerstown Silty Clay Loam	10	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	Yield Increase (N.S.)	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	10	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Corn/Shoots	3 % YR (N.S.)	0.05	Woolson et al. (1973)
Hagerstown Silty Clay Loam	10	5.5	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	22 % YR	0.05	Woolson et al. (1973)
Lakeland Loamy Sand	10	6.2	Na_2HAsO_4	Greenhouse/Soil Pots	Oats/Shoots	6 % YR	0.05	Woolson et al. (1973)

Phytotoxicity of total arsenic in soils, continued.

Soil Type	Soil Concentration (ppm)	Soil pH	Chemical Form Applied	Type of Experiment	Plant Species/ Part	Hazard Response	Significance Level	Reference
Houston Black Clay	10	7.6	As_2O_3	Field Pots	Bermuda Grass/Leaves	No YR	NR	Weaver et al. (1984)
Weswood Silt Loam	10	7.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	No YR	NR	Weaver et al. (1984)
Arenosa Fine Sand	10	4.7	As_2O_3	Field Pots	Bermuda Grass/Leaves	No YR	NR	Weaver et al. (1984)
Helena Valley	6	NR	None	Field	NA	Background	NA	Miesch and Huffman (1972)
NA	5.8	NR	NR	Field	NA	Background	NA	Shacklette and Boerngen (1984)
Weswood Silt Loam	5.6	7.7	None	Field	NA	Background	NA	Weaver et al. (1984)
Houston Black Clay	4.8	7.6	None	Field	NA	Background	NA	Weaver et al. (1984)
Plainfield Sand	3.6	5.5	None	Field	NA	Background	NA	Steevens et al. (1972)
Arenosa Fine Sand	1.2	4.7	None	Field	NA	Background	NA	Weaver et al. (1984)
NR	1.02 ± 0.5 Wet Weight	NR	None	Field	Vegetables	Background	NA	Anderson et al. (1978)