CHINO MINES ADMINISTRATIVE ORDER ON CONSENT

SITEWIDE ECOLOGICAL RISK ASSESSMENT TECHNICAL MEMORANDUM NO. 1: ERA WORKPLAN

CMC Agreement No. C59938

May 14, 1999

Prepared for:

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1.0 INTRODUCTION

1.1 Background

This document represents the initial phase of the sitewide ecological risk assessment (ERA) for the Chino Mine Investigation Area (IA), Grant County, New Mexico. The Chino Mine site, located approximately 12 miles southeast of Silver City, includes open pit copper mining facilities, rock stockpiles, leach stockpiles, mineral processing facilities, and tailings impoundments. Chino Mines Company (CMC) controls approximately 116,000 acres around the mining and mineral processing facilities.

In December 1994, CMC and the New Mexico Environment Department (NMED) entered into an Administrative Order on Consent (AOC) to conduct environmental investigations at the Chino Mine site and surrounding area as appropriate. As a result of the AOC, the site was divided into six investigation units (IUs)(Figure 1-1):

- Lampbright Draw
- Hanover Creek Channel
- Whitewater Creek Channel
- Smelter
- Hurley Soils
- Tailings Impacted Soils

The AOC requires that a Remedial Investigation/Feasibility Study (RI/FS), including human and ecological risk assessments, be completed for each IU. In December 1995, CMC and NMED agreed to combine the ERAs for the IUs into a single study based on suggestions that an ERA could be more effectively conducted on a sitewide basis. An Ecological IU was designated for this purpose and added to the AOC in December 1995 (NMED 1995). The Ecological IU encompasses areas of the other IUs that may contain ecological resources and may be affected by contaminant release (NMED 1995).

The sitewide ERA will focus on areas of the site that may have been affected by historical release of contaminants from mining and milling operations. Potential contaminant sources and transport mechanisms are identified in Section 2.2 below. Certain aspects of the site operation require state or federal permits for normal operations. For example, emissions from the Hurley Smelter are subject to requirements of the New Mexico Air Quality Control Regulations and permitted under Air Quality Permit No. 376-M-1 (CMC 1995). In addition, CMC maintains multiple state groundwater discharge plans in the AOC area and a National Pollutant Discharge Elimination System (NPDES) permit for two zero-discharge outfalls in the operations area (CMC 1995). The permits and licenses specify procedures and/or chemical concentrations in released materials that comply with state or federal guidelines for protecting human health or the environment. However, the permits do not address exposure or risks resulting from releases that may have occurred prior to permitting and that may affect the same areas of the site. For purposes of the ERA, current potential sources that are operated under state or federal permits will not be considered, but areas affected by historical releases occurring from the sources prior to permitting will be addressed if data from RIs indicate contamination..

The sitewide ERA will be conducted in accordance with US Environmental Protection Agency (EPA) guidance on conducting ERAs at Superfund (Comprehensive Environmental Response, Compensation, and Liability Act of 1980 [CERCLA]) sites (EPA 1992a, 1997). While the Chino site is not a Superfund site, the intent of the AOC is to produce CERCLA-like investigations and remedies. In addition, the EPA guidance is most appropriate for the Chino site because its focus is on evaluating the risk and impacts associated with ecotoxicity of chemicals. More recent general guidance conducting ERAs (EPA 1998) was used in developing problem formulation, risk terminology, and risk characterization approach for the Chino Mines ERA.

The EPA (1997) guidance includes eight steps in performing an ERA:

1. Screening-level Problem Formulation and Ecological Effects Evaluation
2. Screening-level Exposure Estimate and Risk Calculation
3. Baseline Risk Assessment Problem Formulation
4. Study Design and Data Quality Objective Process
5. Field Verification of Sampling Design
6. Site Investigation
7. Risk Characterization
8. Risk Management

Steps 1 and 2 comprise the screening-level risk assessment, the results of which are used to determine whether further data collection and/or risk analysis is necessary. The screening-level analysis may consist of quantitative or qualitative analyses and professional judgement of the risk assessors and risk managers. At the end of Step 2, risk managers make the decision whether to proceed with further data collection or analyses to support additional risk assessment or remediation planning.

If the need for further risk analyses is indicated at the end of Step 2, planning for analyses and any additional data collection are conducted in Step 3, and a sampling and analysis plan is prepared in Step 4. The field investigation is implemented in Step 5 and Step 6, and the analysis and risk characterization is conducted in Step 7. Risk management actions taken as a result of the analyses are planned in Step 8. Risk management activities are separate from risk assessment, but the risk assessors provide technical support to site managers in decisions. The data collection and analyses conducted in the ERA should be based on the data needed to best support risk management decisions for the site.

1.2 Purpose and Scope of Technical Memorandum 1 (TM-1)

The purpose of TM-1 is to present information associated with the screening-level risk assessment (Steps 1 and 2), and preliminary information for Step 3. Some preliminary risk evaluation for copper has been conducted for the site (WCC 1997), and risk managers at CMC and NMED have determined that additional characterization of risk at the site is needed. Therefore, the Scientific Management Decision Point (SMDP) that is normally addressed at the end of Step 2 was largely addressed prior to development of TM-1. However, documentation of the screening-level analysis associated with Steps 1 and 2 for copper and other potential contaminants is needed and is presented in TM-1. In addition, TM-1 also includes initiation of the Problem Formulation for the Baseline ERA (Step 3).

TM-1 includes the following elements identified in the EPA guidance (1997):

(Step 1)

- A description of the site setting
- Identification of the potential constituents of concern (PCOCs)
- The preliminary conceptual site model (CSM) identifying incomplete and potentially complete exposure pathways
- Initiation of the Data Quality Objective (DQO) process (i.e., identification of site management goals and general project objectives)

(Step 2)

- Conservative Screening-level exposure and risk calculations
- Identification of constituents of concern (COCs) to be evaluated in the Baseline Problem Formulation

(Step 3)

- Identification of assessment endpoints based on the management goals
- General approach including risk questions and measurement endpoints for evaluating risk to assessment endpoints

Once the above points are agreed upon, the balance of Step 3 can be conducted, including identification of specific data needs and the sampling strategies to be employed. The process described in the EPA guidance is essentially equivalent to the DQO process (EPA 1997). However, the DQO process will be formally implemented to identify the types and quantities of samples needed. A detailed sampling and analysis plan (Step 4) will then be developed. These items will be documented in TM-2. If additional analyses or data needs are identified during the course of the project, TM-2 will be revised to address the data needs through preparation of addenda.

Results of the analyses described in TM-2 will be presented in the ERA Report. The report will include a summary of TM-1 and TM-2 and present analysis results and the risk characterization for the site.

2.0 SCREENING-LEVEL PROBLEM FORMULATION

2.1 Environmental Setting

2.1.1 Site Overview

Copper deposits in the Chino area were identified in Spanish records as early as 1804. Historic mining operations established for the extraction of silver, gold, iron, and copper are common in the area. CMC currently operates the Santa Rita pit, an open pit copper mine approximately 2 miles northeast of Bayard. Mineral processing facilities located near the Santa Rita pit include rock stockpiles, leach stockpiles, water supply reservoirs, a precipitation plant, mill and concentrator, and solvent extraction/electrowinning plant (CMC 1995). Other facilities are located approximately six miles south of the mine area near the town of Hurley. These include a power plant, the Hurley Smelter, and tailings impoundments (Figure 1-1).

CMC currently controls approximately 116,000 acres around the facilities described above. The AOC identified the IA, encompassing approximately 31,700 acres, which includes the IUs identified previously (Figure 1-1). Historic mining operations resulted in release of metals and other chemicals from mineral processing and storage facilities into the surrounding area (CMC 1995). The following sub-sections describe the physical and biological characteristics of the area potentially affected by the mine and processing facilities.

2.1.2 Physical Setting

The IA at the Chino Mine site is located in southwestern New Mexico, near the eastern limits of the Basin and Range physiographic province in the northern end of the Mexican Highlands section (Thornbury 1965, Hunt 1967). The Mexican Highlands are bounded on the east by the Sacramento Mountains and on the west by the Sonoran Desert, two other sections of the Basin and Range province. The northern edge of the Mexican Highlands grade into the Datil section of the Colorado Plateau physiographic province.

The IA is in a semi-arid region, with average annual precipitation in Hurley of about 14 inches; most of the precipitation falls during the summer "monsoon" period (mid-July through August). Average monthly temperatures are 38°F in January and 72°F in July.

2.1.2.1 Topography

Major topographic features in the AOC investigation area include the Cobre Mountains and the San Vicente Basin. The following discussion is summarized from previous site investigations (Trauger 1972). Erosion of the plateau surface in the Cobre Mountains southeast of Bayard has resulted in a series of even-crested, southward-sloping ridges that gradually become low hills. The topographic high within the AOC investigation area is approximately 7,700 feet.

The San Vicente basin is a broad lowland that extends northward from the Mimbres Valley. The basin terminates against the Big Burro and Little Burro mountains on the west, Silver City and Pinos Altos ranges on the north, and the Cobre Mountains on the east. The slope of the terrain is from these mountains toward the San Vicente Arroyo. The San Vicente Basin is characterized by several dry, sandy washes and gullies.

2.1.2.2 Regional Geology

The Chino Mine is located within a complex geologic region that constitutes a transition zone between Colorado Plateau structures to the north and Basin and Range structures to the south. This transition zone is a northwest-trending structural belt that is 50 to 75 miles wide and characterized by features related to Late Cretaceous and Miocene intrusive activity.

Prominent features include abundant normal faulting and fracturing of the rock mass, intrusive igneous rocks, and extrusive volcanic rocks. The Santa Rita Pit area and upland areas are characterized by a variety of igneous and sedimentary rocks. The rock types include limestone, sandstone, shale, intrusive diorite, volcanic breccias, tuffs, and rhyolite. Structural features characteristic of the bedrock units include faults, fractures, joints, and intrusive dikes and sills. Geologic units of Paleozoic to Quaternary age are exposed in the region.

2.1.2.3 Surface Water Hydrology

The primary surface water drainages are shown on Figure 1-2. In the vicinity of the CMC site, they include Hanover Creek, Whitewater Creek, and Lampbright Draw. Man-made surface water features associated with mining operations are primarily located in the Santa Rita Pit area and the tailings area south of Hurley. General discussions of each of the surface water drainages and their tributaries are presented below.

Hanover Creek

Hanover Creek originates in the higher terrain north of the Santa Rita Mine and flows in a southwesterly direction. Hanover Creek is a tributary of Whitewater Creek and they join southwest of the Ivanhoe Concentrator. Hanover Creek and its tributaries, including Buckhorn Gulch and other unidentified tributaries north and south of the town of Hanover, are ephemeral. Flow occurs in Hanover Creek during periods when no precipitation has been recorded at locations further to the south. This is due to the fact that Hanover Creek originates in the higher elevations where it is more likely to receive rain or snow.

Whitewater Creek (including Santa Rita and Bolton Drainages)

The Whitewater Creek drainage basin covers an area of approximately 57 square miles and ranges in elevation from 5,300 to 7,600 feet above mean sea level (msl). To prevent surface water from entering the mine area, most of the runoff north of the Santa Rita Pit is intercepted by a reservoir system. Runoff from upper Whitewater Creek (Santa Rita drainage) to the south of the pit and rock stockpiles is contained by Reservoir 3A and Reservoir 9. The creek bed commences immediately west of Last Chance Reservoir, flows southwest toward Bayard, then flows to the south-southwest toward Hurley. As with Hanover Creek, Whitewater Creek is an ephemeral stream flowing only in response to runoff events. Ephemeral flow is also received from Gold Gulch, a tributary of Whitewater Creek originating west of the Santa Rita Mine near the town of Vanadium.

Bayard Canyon and Lucky Bill Canyon are tributaries of Whitewater Creek north of Hurley. South of Hurley, Whitewater Creek flows in a south-southeasterly direction along the east side of the tailings area. In 1988, Whitewater Creek was diverted to the east to allow for the construction of tailings impoundments, which are now located over the original creek bed. South of the tailings area, the creek resumes its original course and heads south-southeast toward its confluence with San Vicente Arroyo and Lampbright Draw.

Lampbright Draw

Lampbright Draw is located approximately 4 miles east of the Santa Rita Pit. Lampbright Draw is an ephemeral stream that is dry most of the year and flows only during storm events. Intermittent tributaries to Lampbright Draw include Rustler Canyon, located approximately 5 miles southeast of Santa Rita and Martin Canyon, located approximately 5 miles east of Hurley. Lampbright Draw flows southwest where it joins Whitewater Creek near the town of Faywood.

2.1.2.4 Soils

This section includes a general discussion of soil types in the region. The information is provided by the Soil Survey of Grant County, New Mexico (SCS 1983)(U.S. Department of Agriculture, Soil Conservation Service [now the National Resources Conservation Service]).

Soils in the region generally originate from two distinct parent materials:

- Valley-fill sediments that form broad upland plains between mountain ridges
- Igneous rocks that crop out on hills and mountains and in deep canyons and entrenched streams

Soil characteristics within the IA are controlled largely by the type of parent material, but are influenced also by climate, topography, time, flora, and fauna. General soil types include:

- Luzena – Rock Outcrop – Muzzler: moderately sloping to extremely steep, well-drained shallow soils and rock outcrop, on mountains and hills;
- Lonti – Manzano – Ustorthents: Nearly level to extremely steep, well-drained deep and moderately deep soils, on hills and terraces;
- Santana – Rock Outcrop – Lithic Ustorthents: Gently sloping to very steep, well-drained, shallow soils and rock outcrop; on hills and mountains;
- Santa Fe – Rock Outcrop – Encierro: Nearly level to very steep, well drained, shallow and deep soils and rock outcrop, on hills; and
- Plack – Lonti – Pits: Nearly level to very steep, well-drained shallow and deep soils and pits, on broad terraces and hills.

The predominant soil type in the Tailing IU and Hurley IU areas is the Plack-Lonti Pits. Tailings impoundments are generally classified as pits. The Santana-Rock Outcrop-Lithic Ustorthents and the Santa Fe-Rock Outcrop-Encierro predominate in the Lampbright IU. Within the IA boundary, the Hanover Creek IU contains Santana-Rock Outcrop-Lithic soils and the Whitewater Creek IU contains Santana, Lonti, and Plack soils. The Lonti and Plack soils are predominantly loam-type soils, while the other soils have significant rock outcrops.

2.1.3 Ecology

The structure and function of an ecosystem are important considerations in performing an ecological risk assessment because of the influence that biotic and abiotic factors can have on exposure points, movement of contaminants through a food web, and potential adverse effects at the individual, population, or community levels. Two general types of landscapes (i.e., mountain and plain), each composed of different habitat types, occur in the immediate project vicinity. The site lies near the northern edge of the Deming Plain, a broad depositional environment (a bolson or interior-draining basin) cloaked with alluvial and eolian materials. To the south are fault-block and extrusive mountains (e.g., the Animas and Peloncillo mountains), while to the north are sedimentary mountains that form the lower edge of the Colorado Plateau (e.g., the Mogollon Mountains)(Findley et al. 1975).

2.1.3.1 Vegetation

Major Habitat Types and Controlling Factors

Shelford (1963) described most of the region around CMC as being at the margins of the mesquite grassland association of the northern temperate grassland and southern temperate grassland biomes. Dick-Peddie (1993) divided this general association into two subcategories based on the presence or absence of mesquite. Rocky highlands within the site and more mountainous terrain on the northern and southern boundaries of the Deming Plain are dominated by associations of woody plants. Shelford described these as the oak-juniper-agave woodland, pinyon pine woodland, and mountain forest (oak-ponderosa pine) associations. Dick-Peddie (1993) followed basically the same classification in describing the oak/juniper, juniper/pinyon pine/oak, and ponderosa pine/juniper habitats of the coniferous and mixed woodland vegetation type. It is worth noting that more than one species of oak and juniper are contained within the names of these units.

Differences in the major habitat types are controlled by differences in moisture related to elevation, slope, aspect, and substrate. Higher-elevation areas have cooler temperatures, slightly greater precipitation, and differences in exposure that affect the moisture regime. They also tend to be somewhat rockier. Rocky soils tend to be more moist because the incident rainfall is concentrated into the portion of the soil without rocks. Additionally, rocky soils tend to have a somewhat coarse matrix, further facilitating percolation of precipitation into the root zone. Flatter areas are often underlain by fine soils that retard infiltration of moisture (as well as development of plant roots), which in turn may lead to development of saline or alkaline conditions that are hostile to all but specifically adapted species.

Because much of the rainfall occurs during the summer “monsoon,” warm-season grasses dominate most of the area. Prevalent species include various grama grasses, dropseeds, three-awn, galleta, curly-mesquite, Indian ricegrass, and muhly. Bluestems (beardgrasses) have perhaps been harder hit than the other grasses by cattle, both because of their palatability and height, and because they are more moisture-dependent. Cool-season grasses are present and gain in importance at higher elevations or other relatively cool and moist sites, as do the bluestems. Examples include bluebunch wheatgrass, junegrass, muttongrass, and Arizona fescue. Higher elevation areas also support succulents such as beargrass and sotol and shrubs such as mountain-mahogany, mountain spray, apache plume, and buckbrush in addition to the oaks, junipers, pines, and agaves for which these communities are named.

Along drainages with intermittent surface flow and more protracted subsurface flow, the availability of water beyond the meager rainfall far outweighs most other factors. These areas generally support riparian communities consisting of mesophytic or hydrophytic shrubs and discontinuous ribbons of phreatophyes such as plains (Fremont) cottonwood, Arizona walnut, and Arizona sycamore.

Nearby areas were classified by Shelford (1963) as components of the hot desert biome. These include extensions of the Chihuahuan desert along the Rio Grande and of the Arizona-Sonoran desert along the Gila. In both of these biomes, creosote bush is dominant along with yuccas, cacti, and sparse grasses. At a landscape level, the shift from "southern temperate" to "hot desert" biomes is related to a combination of precipitation and temperature. At a smaller scale, similar but more subtle shifts can accompany topographic or edaphic variability within the larger biomes (e.g., rock outcrops, areas historically subjected to less grazing, dune fields).

Identification of vegetation communities/wildlife habitat in and around the Chino Mine site was conducted by CMC based on examination of aerial photographs from 1991 and site visits by wildlife biologists in 1994 and 1995 (CMC 1995). Field work for a more detailed site survey was conducted in 1997 in association with an Environmental Impact Statement (EIS) being prepared for expansion of the Santa Rita Pit. The survey was also conducted for areas identified in the AOC to be addressed under the ERA. However, data and comprehensive mapping from the surveys were not available at the time this document was prepared. Therefore, vegetation mapping used for preliminary site descriptions is based on the information from the earlier survey of aerial photographs.

The Background Report identified seven vegetation communities outside of the current actively used areas of the mine:

- honey mesquite/cactus
- mixed grasses/yucca-cactus
- oak/juniper
- juniper/pinyon pine/oak
- ponderosa pine/juniper
- riparian/mixed deciduous shrub
- sparsely vegetated cliffs

Habitat designations were based on regional information described above from Dick-Peddie (1993). Two additional designations were described for actively used areas:

- disturbed—no revegetation
- disturbed—revegetated

The AOC Background Report (CMC 1995) includes a map of habitat designations for the site that was based on previous qualitative surveys of the site. However, vegetation communities are currently being mapped in greater detail as a result of data collected in 1997, but mapping is not complete for the IA.

Influence of Grazing

Impacts from grazing are often a dominant factor that influences shifts in native vegetation. Throughout much of the region, the current landscape is less grassy than prior to settlement by

Europeans and subsequent introduction of cattle. Grasses and palatable forbs, quickly removed by cattle, were replaced by "desert scrub," especially low-growing mesquite. Bison were not present during the shaping of the Deming Plain ecosystem, perhaps because of high temperatures and scarce water. Consequently, native grass communities did not evolve in the presence of heavy herbivory and trampling, as was the case with prairie grasslands of the Great Plains (including eastern New Mexico).

Another related factor in the relatively rapid spread of mesquite is the fact that cattle eat the beans, which they then spread, along with a favorable environment for germination and early growth, in their manure (Shelford 1963). Findley (1987) suggested that yet another factor is the shade that the mesquites provide for black-tailed jackrabbits. The jackrabbits may consume large quantities of grassy vegetation, reducing the plant cover and creating conditions more favorable for establishment of additional mesquite seedlings.

Low rainfall and poor soils combine to slow the recovery rate from disturbance, including non-sustainable grazing rates. Even areas where cattle are no longer part of the landscape (e.g., national parks) may show the effects of habitat modification for decades or perhaps centuries.

Substrate and topography can have an indirect effect on vegetation through preferential use or avoidance by cattle. For example, dune fields and steep or rocky terrain tend to receive less use by cattle and thus are generally less altered than other areas. Impacts of cattle can be especially dramatic along drainages, which provide shade, relatively lush forage, and water. Riparian habitats are often the most severely impacted of native habitats.

The impact of cattle on vegetation near areas of surface water is exacerbated when water diversions reduce the already limited amount of surface flows in ephemeral streams. Effects from cattle can be especially severe because of their large size, their tendency to trample areas of concentrated use (e.g., beneath trees and near water), and their artificially large numbers (i.e., at greater than sustainable populations).

2.1.3.2 Wildlife

As described above, vegetation is influenced by the abiotic factors of water, soil, and topography. In turn, the abiotic factors and vegetation combine to control spatial and temporal patterns of wildlife occurrence and relative abundance. The process is brought full circle when wildlife modify the habitat through differential removal of biomass and seed-caching.

Representative species groups expected to occur in the project area are described briefly below. As with vegetation, more detailed information on community composition, species relationships, and habitat/landscape affinities will be developed as part of the final ERA Work Plan.

Mammals

Although habitat-specific communities of small mammals (and other taxa) are normal in most regions, the pattern is amplified in arid or semi-arid regions because of the habitat extremes. Two groups of small mammals—the *Peromyscus* species and the Heteromyids—exemplify the patchiness of semidesert landscapes. The deer mouse group (*Peromyscus*) may consist of several "microallopatric" species within the same general area in desert mountain ranges. For example, the deer mouse or white-footed mouse may occur in grasslands, with the latter more common on sandy soils and arroyo banks. Where mesquite is common, the cactus mouse may be common in bajada grasslands but otherwise prefers rocky jumbles at the base of hot foothills slopes. Throughout most of the oak zone, the brush mouse is the prevalent *Peromyscus*, while the pinyon mouse dominates in pinyon-juniper woodland.

Heteromyid rodents, including kangaroo rats and pocket mice, are both interesting and important in arid environments such as the flat terrain of the Deming Plain. These species are highly adapted in terms of not requiring free water since they fulfill water requirement from the moisture content of their diets and production of metabolic water from consumption of seeds. The granivorous habit is particularly well suited to arid regions because one of the reproductive strategies for vegetation is to produce large quantities of seed. This strategy increases the likelihood that some seeds will fall into suitable microsites and that not all seeds in a particular site will germinate in any given moisture event (which may be insufficient for seedling establishment). Co-evolution of granivores probably affects the strategy as well: Plants need to produce copious quantities of seeds to ensure that some will escape the eyes or noses of hungry rodents.

Heteromyids have responded to this "feast or famine" seed abundance by developing external, fur-lined cheek pouches that enable them to collect large quantities of seeds when they find them. The distribution of seeds tends to be patchy because of differential accumulation on the ground surface as a result of small-scale wind effects and the patchy distribution of plants. They also tend to cache the seeds when they are available. Extreme "antisocialism" of some heteromyids (including intolerance for members of the opposite sex except when mating) may be related for the need to aggressively defend a favorite feeding area or seed cache. Seed-storing behavior also enables heteromyids to attain surprisingly large numbers because they can extend a "time of plenty" instead of being limited to food materials available at any one time.

Heteromyids occurring in the general region of southern Grant County show some marked habitat preferences. For example, the silky pocket mouse prefers soils that are loose and friable, particularly sands, while the rock pocket mouse is limited to rocky areas. The desert pocket mouse occurs primarily in mesquite grasslands. Among kangaroo rats, Ord's prefers sandy soils, while Merriam's is generally found on finer soils where the two species occur together (Findley 1987). The banner-tailed kangaroo rat is found on a variety of soils but prefers areas of well-developed grass, while the other two species may also be found in desert scrub as well as grasslands.

Landscape influences on small mammal communities in the region are also reflected by the grasshopper mice and woodrats. The northern grasshopper mouse occurs on sandy soils, while the southern grasshopper mouse is found on finer soils. Both of these species are largely predaceous, feeding on insects, other invertebrates, and even other mice. Among woodrats, which are mostly herbivorous, the southern plains woodrat occurs primarily in grasslands while the white-throated woodrat occurs in rocky sites. A third species, Stephens' woodrat, could also occur at the Chino Mine site. If so, it would probably be found among junipers, in which it may nest.

Taken together, these patterns of distribution suggest the presence of several, somewhat distinct small-mammal communities dominated by combinations of deer mice *(Peromyscus* sp.*)*, wood rats *(Neotoma* sp.*)*, kangaroo rats *(Dipodomys* sp.*)*, and pocket mice *(Perognathus* sp.*)* species.

dominance of different taxa, with their varying dietary habits, could be an important component of both the food web and risk evaluation for specific landscape units.

Other potentially common small mammals in the area include the western harvest mouse, hispid cotton rat, tawny-bellied cotton rat, Botta's pocket gopher, desert cottontail, and black-tailed jackrabbit. The harvest mouse and cotton rats would be expected in well-developed grasslands. The cottontail and jackrabbit could occur in either grassland or desert scrub habitats, with the cottontail extending farther into rocky or wooded uplands. Sciurids potentially present in the area include the cliff chipmunk, Harris antelope squirrel, and rock squirrel in rocky uplands and the spotted ground squirrel in sandy grasslands.

Ungulates are probably less important herbivores than the rodents or lagomorphs in terms of ecosystem function. However, they can be important from a risk assessment perspective because of their longer lives and potential for being consumed by humans (not an issue for the ERA). Both mule deer and white-tailed deer occur in the area. Both would be expected primarily in areas of broken country or drainages that provide hiding cover, thermal cover, water, and adequate forage. Pronghorn, if present, would be expected in open grasslands.

Predators potentially present include omnivores such as the spotted, striped, and hooded skunks, as well as the ringtail in broken terrain and the raccoon along water courses. Larger predators include the gray fox in wooded or rocky country, kit fox in open grasslands, ubiquitous coyote, and (potentially) both the bobcat and mountain lion in rugged uplands.

Birds

In general, avian species are influenced by the same types of landscape components as mammals, although vegetation is by far the most important factor. Other factors are important primarily as they affect vegetation, although rocky areas, cliffs, and surface water can be critical for some birds. From a risk assessment standpoint, birds are generally less important than mammals because they live in less intimate contact with the soil, are highly mobile, and in many cases are present only seasonally. The major exception to this generalization is the category of avian predators, many of which feed on small mammals and, being secondary, tertiary, or even quaternary consumers, are particularly vulnerable to food web effects.

Representative avian predators (raptors) for the study area include the golden eagle and red-tailed hawk, both of which are large, diurnal species; the great horned owl, a large, nocturnal species; and smaller species such as the American kestrel, western screech-owl, and greater roadrunner. The kestrel and screech-owl consume both insects and small mammals, while the roadrunner eats a wide variety of small prey. Other raptors potentially present in the area could include two federally listed threatened or endangered species, the peregrine falcon and bald eagle. Both of these species tend to be tied to some extent to water, where they specialize in feeding on waterfowl and fish, respectively. However, peregrines take other avian prey, and bald eagles consume waterfowl, rabbits, and carrion, particularly during the winter. Both of these species would be expected to occur primarily during winter, as would the ferruginous hawk and northern harrier. The prairie falcon and Swainson's hawk could be expected during the summer breeding season.

Two gallinaceous birds occur in the area: the scaled quail in open country and the Gambel's quail in areas of greater shrub cover or closer to water. Both quails feed on a wide variety of insect and plant food.

Common songbirds include the pinyon jay, plain titmouse, bushtit, blue-gray gnatcatcher, gray vireo, black-throated gray warbler, and spotted towhee in pinyon-juniper; the western kingbird, ash-throated flycatcher, western meadowlark, horned lark, curve-billed thrasher, Scott's oriole, lark sparrow, black-throated sparrow, and Cassin's sparrow in mesquite grassland and desert scrub; and the scrub jay, canyon towhee, black-chinned sparrow, and rufous-crowned sparrow in woody canyons or hillsides. White-throated swifts, rock wrens, canyon wrens, and various raptors may nest on cliffs or rock ledges. Portions of the drainages with cottonwoods or other trees also support a distinct avian community. Potentially present "riparian" species include various woodpeckers, Cassins's kingbird, Say's phoebe, black phoebe, vermillion flycatcher, crissal thrasher, northern oriole, blue grosbeak, and indigo bunting.

Most of the birds listed above have flexible diets that emphasize specific types of plant or animal material during certain seasons. For example, many species consume flower buds, leaf buds, fruit, or seeds during most of the year but shift to insect prey as a protein source for forming eggs and feeding young. Most species are somewhat opportunistic, feeding on whatever food source is most abundant or particularly nutritious/palatable at a given time. The warblers and vireos approach being totally insectivorous, while flycatcher (including kingbirds and phoebes), swifts, and swallows are true insectivores.

Reptiles and Amphibians

These species, collectively called "herptiles," are generally of secondary importance in the movement of contaminants and energy through the food web because of the relatively small proportion of total biomass that they represent. However, these groups may contain species that are either considered rare in a specific area or especially sensitive to contaminants. The latter is of special concern for amphibians, which lay their eggs in water and have aquatic larvae. Thus, waterborne contaminants represent the same type of potential impact as to fish. Amphibians are also important because they are included in both aquatic and terrestrial food webs.

Numerous species of lizards and snakes are potentially present in the study area. Most either remain insectivorous throughout their lives (smaller species such as the western hognose snake and most lizards) or are insectivorous when young but shift to eating small mammals, ground-nesting birds, and other reptiles as they grow (larger species such as the coachwip and gopher snake). One species, the western terrestrial garter snake, prefers habitats near water, and adults may even consume fish. Large lizards such as the collared lizard readily consume other lizards and may take small birds or mammals. Vipers, including the western and black-tailed rattlesnakes, feed primarily on vertebrates throughout most of their lives.

Reptiles likely to occur in the area, in addition to those listed above, include the desert striped whipsnake, Big Bend and mountain patch-nosed snakes, glossy snake, western hook-nosed) snake, greater and lesser earless lizards, Texas horned lizard, short-horned lizard, Clark's spiny lizard, eastern fence lizard, tree lizard, side-blotched lizard, Great Plains skink, and several species of whiptails.

Amphibians potentially present are mostly limited to species that require water only for breeding and are either terrestrial as adults or can burrow into the mud as their breeding pool begins to dry. Species potentially present in the project vicinity include the red-spotted toad, Great Plains toad, southwestern toad, Woodhouse's toad, Couch's spadefoot toad, New Mexico spadefoot toad, and plains spadefoot toad (Williamson et al. 1994). The tiger salamander and canyon treefrog are also potentially present.

Threatened/Endangered Species and Sensitive Habitats

Table 2-1 lists federal- and state-listed threatened or endangered (T/E) animals and plants that potentially occur in the IA and were previously identified based on consultation with state and federal resource management agencies (Appendix A in CMC 1995). No T/E species or their critical habitats have been observed within the AOC IA. However, peregrine falcons are federally protected and nest on a formation known as the Kneeling Nun near the Santa Rita Pit. The New Mexico figwort (*Scrophularia macrantha)* is also known to occur near the Kneeling Nun. This plant species is a candidate for federal listing and is listed as endangered in the state. Additional wildlife surveys are currently planned for 1998 as part of a baseline Environmental Impact Statement (EIS) for expansion of the mine, and will provide additional information on the occurrence of T/E species in the area.

The AOC Background Report (CMC 1995) summarized the T/E wildlife and plant species that could occur in habitat types found in the IA:

Wildlife

Sparsely Vegetated Cliffs: Gila monster, spotted bat, peregrine falcon, and bald eagle

Oak/Juniper: Gila monster, peregrine falcon, Lucifer hummingbird, Costa's hummingbird, Gila wood pecker, gray vireo, Mexican spotted owl, and spotted bat

Juniper/Pinyon Pine/Oak: Gila monster, peregrine falcon, Lucifer hummingbird, Costa's hummingbird, gray vireo, and spotted bat

Ponderosa Pine/Juniper: Gila monster, peregrine falcon, Mexican spotted owl, and spotted bat

Riparian Mixed Deciduous Shrub: Gila monster, lowland leopard frog, green rat snake, narrowhead garter snake, Mexican garter snake, Gila woodpecker, willow flycatcher, Bell's vireo, Abert's towhee, and the common black hawk

Honey Mesquite/Cactus-Yucca/Mixed Grasses: Aplomado falcon, Lucifer hummingbird, and Gila woodpecker

Mixed Grasses/Yucca/Cactus: Aplomado falcon, Lucifer hummingbird, Baird's sparrow, and Gila woodpecker

Active Use Areas: None

Disturbed-Revegetation: Aplomado falcon and Lucifer hummingbird

In addition to the bat species listed in Table 2-1 are several other bat species that may be present (although rare) in the study area. Habitat associations for bats tend to be more varied than for terrestrial mammals as feeding and roosting areas are often distinct. In the CMC study area, rocky bluffs and steep cliffs are likely to provide roosting habitat for a variety of species, including those listed in Table 2-1. Feeding areas for insectivorous species may also vary, however, riparian areas or areas with high insect abundance are of primary importance.

Plants

Five endangered plant species may occur in the region of the study area (Table 2-1). The New Mexico figwort has been documented to occur on the eastern slopes of Kneeling Nun Ridge. During the 1994 site investigation, one colony observed during the site investigation was found with young plants (i.e., 1994 growth) and the previous year's growth was evident. It is possible that other colonies exist along the eastern slope of Kneeling Nun Ridge. The other four species of plants are not known to occur within the study area. Investigations into the habitat preference of the two species of cactus (Wright's fishhook cactus and grama grass cactus) are ongoing, but they are not known to occur in the study area. Parish's alkali grass occurs within CMC property but south of the study area. An occurrence of the Pinos Altos Mountains flameflower has not been recorded within the study area or within CMC property boundary.

2.2 Nature and Extent of Contamination

Data on the nature and extent of contamination at the Chino site used in TM-1 were available from the AOC Remedial Investigation Background Report, Chino Mine Investigation Area ("Background Report") (CMC 1995). Additional data collection is planned for individual IUs by their respective RIs. To date, only sampling at the Hurley Soils IU has been conducted. The following subsection summarizes the information and data available from the Background Report and the RI Proposals for the individual IUs.

2.2.1 Primary Sources and Transport Mechanisms

The primary sources of contamination at the Chino site are mine waste and mine process materials. Historical releases of waste and process materials has occurred in the form of spills, leaks, surface water runoff from rock stockpiles and tailings deposits, wet and dry deposition of smelter emissions, and deposition of windblown concentrator dust and tailings (CMC 1995). The Background Report provides a more detailed summary of historic mining operations at the site and the approximate dates of known/suspected waste releases.

Since the 1970s, recent releases of mine waste/process materials have been controlled and monitored through the implementation of engineered structures and monitoring programs (CMC 1995). Control structures include stormwater intercept systems, upgraded smelting facilities, and surface water diversion systems. Since control structures and monitoring systems were installed, the release of contaminants has been confined to discreet events, which are summarized in the Background Report and RI proposals. Thus, the only source of existing uncontrolled contaminant releases to the environment are from "secondary" sources, i.e., environmental media that were impacted prior to the installation of engineered control structures. These secondary sources are releasing contaminants to the environment by processes such as desorption, leaching, erosional transport, and redistribution by wind.

Historical releases (pre-1970) of contaminants as the result of erosional runoff from rock stockpiles and tailings and from spills and leaks have primarily impacted drainage channels that receive surface water runoff from the source areas. Areas impacted by runoff include:

- Lampbright Draw and its tributaries near the Lampbright leach stockpile

- Hanover Creek channel downgradient from CMC's West Leach Stockpile and other non-CMC mines facilities
- Whitewater Creek channel downgradient from the West and South Leach Stockpiles and other (non-CMC) mining facilities north of Hurley, and downstream of the tailings impoundments east and south of Hurley

Because the major drainages at the Chino Mine site are ephemeral, the most significant impact from the release of waste materials is contamination of drainage sediments and adjacent riparian soils. Metals and other chemicals deposited in sediments are susceptible to re-suspension and further transport during surface water runoff events, resulting in potential exposure to biota in areas downgradient of the site. Accumulation of metals in sediments and adjacent soils can inhibit the establishment of vegetation along drainage courses and can act as a secondary source of contaminants to underlying groundwater via leaching by rainfall and snowmelt. Deposition of metals onto plant foliage may significantly impair the vegetative cover, thereby accelerating soil and sediment erosion and transporting metals to downgradient receptors.

The additional accumulation of metals and salts in surficial sediments is enhanced by evapoconcentration processes in which alternating periods of wet and dry conditions causes the upward transport of metals, sulfates, and other inorganic constituents where they accumulate as "evaporites" in surficial sediments. The existence of metal-rich evaporite deposits along sections of Whitewater Creek is evidence that this phenomena is occurring. Biota may be further exposed to metals through contact or ingestion of these evaporitic salt deposits.

Deposition of smelter emissions and windblown concentrator dust and tailings are mainly confined to areas east of the smelter complex and tailings impoundments, in the direction of the prevailing winds. However, elevated concentrations of copper and other metals have also been detected in soils to the west of these facilities, in the towns of Hurley and North Hurley, and in the areas between them (CMC 1995, Golder 1996, SRK 1998). Deposition of windblown tailings have also affected the mesquite grasslands to the east and southeast of the site. The full extent of areas impacted by smelter and windblown wastes have not yet been defined, but will be investigated by upcoming RI activities. Current releases from smelter operations is regulated under the New Mexico Air Quality Control Act.

Following airborne deposition onto soils, metals and other inorganic constituents may be further redistributed by a combination of physical (air and water erosion) and/or chemical (leaching) processes. Infiltration of soil by rainfall and snowmelt may solubilize the metals, causing them to migrate downward in the soil profile by leaching processes. Metals vary significantly in their ability to be dissolved and transported by water in soils. At neutral pHs for example, most metals occur as relatively insoluble oxides or hydroxides and remain confined to surficial soil layers (e.g., iron). Other metals such as copper and lead are strongly adsorbed by organic matter where they become concentrated in the upper soil o-horizon. Conversely, in saturated soils, dissolved organic carbon may complex with copper and lead and transport them to deeper soil horizons. This is atypical of New Mexico soils which are typically dry, but may occur in sediments that are periodically saturated. In the presence of sufficient soil alkalinity (usually as calcium carbonate) typical of New Mexico soils, metals such as cadmium, lead, and zinc can be removed from solution as carbonate minerals, such as otavite ($CdCO_3$), cerussite ($PbCO_3$), or smithsonite ($ZnCO_3$). Other inorganic constituents such as the metaloids arsenic, selenium, and molybdenum tend to form negatively charged oxyanions in soil solutions (e.g., AsO_4^{2-}, SeO_4^{2-} and MoO_4^{2-}) that are relatively immobile when pHs are less than 7, but become mobile under slightly alkaline pH ($pH>7$). Most of the metal COCs at the site (including arsenic, selenium, and molybdenum) are very susceptible to adsorption to aluminum, iron, and manganese oxy-hydroxide solids ("sesquioxides") in the soil zone. This is an extremely important removal mechanism because sesquioxides are abundant in New Mexico soils, and adsorption to these solids occurs even when COC levels are below that required for metal precipitation.

2.2.2 Contaminant Form and Bioavailability

In general, as metals and inorganic constituents are re-transported and re-distributed over time, a decrease in concentration and bioavailability is often observed. This is due to a combination of chemical and physical interactions that bind the COC to solids particles (sorption), result in the precipitation of or inclusion into (e.g., solid solution partitioning) a solid phase of low solubility, or result in the formation of stable chemical complexes that reduce the proportion of free metal ions available to biota.

Chemical exposure to biota is greatest from dissolved metals that occur as free ions or unstable (labile) complexes, and to a lesser extent by metals that occur as stable complexes or sorbed to particulate matter (Suavé et al. 1998, also see discussion in Appendix A). The proportion of free metal ion that exists in solution is a function of the pH and the concentration of complexing ligands (e.g., sulfate, carbonate, humic and fulvic acids). Copper and other metals released by smelter emissions are expected to occur in an oxidized form, probably encapsulated in a refractory silicate or oxide matrix that results from the high heat of combustion. Depending on the degree of encapsulation, smelter emissions may exhibit low solubility and therefore low dissolved metal content. In contrast, residual copper and other metals in tailings wastes are expected to occur in an altered form derived from the parent ore material, and may or may not be encapsulated by a recalcitrant matrix. Copper ores typically exist as sulfide, oxide, or carbonate minerals. Oxide and carbonate minerals tend to be relatively stable under surficial conditions, but can be solubilized under certain conditions (e.g., low pH). Sulfide minerals in particular are very susceptible to chemical alteration (oxidation) and can release metals, sulfate, and hydrogen ions (acidic pHs) to the surrounding environment. The presence of pyrite (FeS_2) or chalcopyrite ($FeCuS_2$) in tailings often act as catalysts to increase acid production and the subsequent release of metals and sulfate from other minerals.

The geochemical behavior of metals and inorganics following deposition onto soils and sediments greatly affects their mobility, speciation, and bioavailability. At low levels, some of the target metals are micronutrients required by biota for survival. However, excessive uptake of metals can result in toxic effects. The rate at which metals are taken up depends on a number of physical and chemical factors.

Important geochemical reactions occur in soils that strongly affect the speciation of metals and the ease with which they are assimilated by plants. Most important is the pH of the immediate environment, and secondarily is the concentration of dissolved ligands (Appendix A). At acidic pHs, most metals occur in solution as the free metal ion (e.g., Cu^{2+} or Pb^{2+}). As pH increases, the free metal ion bonds with dissolved ligands to form charged and uncharged dissolved complexes of varying stability and bioavailability (e.g., $CuSO_4^{o}$, $CuHCO_3^{+}$, $CuCO_3^{o}$, Cu-organic). Stable complexes exhibit substantially lower bioavailability, and hence toxicity, than weak complexes or the free metal ion. Depending on the pH, the proportion of metal complexes may comprise a significant portion of the total metal load in a system. Consequently, the total content of metals in soil and water is less important than the abundance of the speciation and bioavailable fraction present.

Other factors that affect speciation and mobility include the presence of iron, aluminum, and manganese oxyhydroxides, organic carbon content, and clay content. These phases act as strong sorbents that remove metals from solution and render them unavailable to biota. Metals exhibit variable propensity to partition to these solid phases. For example, copper forms strong complexes with organic carbon compounds and forms relatively insoluble carbonate or oxide compounds above a pH of 5.5. As such, copper may be largely bioavailable in acidic soils that are low in organic carbon, and unavailable in neutral pH, clayey soils rich in carbonate and organic matter.

The characteristics of soils in the landscape play a strong role in determining the bioavailability of copper deposited from natural or anthropogenic sources. Area soils were mapped by the Soil Conservation Service in 1973 through 1978 (SCS 1983). Soils east of the smelter and tailings area include a mosaic of series that differ in:

- depth—including soils with depth less than20 inches to greater than 40 inches or to bedrock
- organic matter content—including soils high in organic matter (mollisols), and low in organic matter (aridisols)
- clay content—with texture in the 10 to 40 control horizon varying from "fine" to "coarse-loamy", with and without argillic subsoil horizons
- carbonate levels—including noncalcareous soils developed from volcanics or Gila Conglomerate and soils with carbonate-cemented caliche layers

The soils perceived to have the highest risk of affecting vegetation would be noncalcareous, shallow, coarse-textured (i.e., low clay) soils. These include the Santana, Oro Grande, and rock outcrop areas. Metals deposited in these areas are less likely to be complexed with organic or calcareous materials and therefore may be more bioavailable. However, deposition in these areas would be limited to accumulation in cracks or depressions in the surface.

2.3 Potentially Complete Exposure Pathways

In general, biota can be exposed to chemical stressors through direct exposure to abiotic media, or through ingestion of forage or prey that themselves have accumulated contaminants. Exposure routes are the mechanisms by which a chemical may enter an individual receptor's body. Possible exposure routes include (EPA 1989):

- *Absorption* across external body surfaces such as cell membrane, skin, integument, or cuticle from air, soil, or water (microbes, plants, animals)
- *Ingestion* including direct ingestion of food and incidental ingestion of soils, sediments, or water along with food (animals only)
- *Inhalation* of gaseous or particulate forms of contaminants with respired air (animals only)

Potential chemicals of concern at the Chino Mine site primarily consist of metals, associate inorganics (e.g., sulfate), and acidic pH.

Absorption across external surfaces is the only exposure route of concern for plants. Chemicals may be absorbed by roots from the soil matrix or water (aquatic plants), or by foliage from the gaseous phase and airborne materials that have been deposited on leaves and stems. Potential exposure pathways at Chino Mines do not currently include gaseous or vapor phase materials. However, historic smelter emissions probably contained sulfur dioxide, which can be phytotoxic and may have resulted in adverse effects during emissions. Similarly, windblown tailings and/or concentrator dust deposited on foliage may have resulted in increased exposure to plants. Tailings impoundments are currently managed to minimize wind dispersion, thus reducing the potential for exposure via this pathway. Phytotoxicity from historic processes such as these can result in alterations of the vegetation community that persist after the attenuation of release (Galbraith et al. 1995).

Currently, the primary chemical stresses of concern for plants are metals and acidic pH in soils. The primary pathways of concern are absorption across root surfaces, and deposition of tailings materials from windblown soils. Acidic pH alone can be phytotoxic and increases the solubility of most metals, making them more bio-available.

Invertebrates in soils and (dry) sediments may be exposed through ingestion of soils during feeding, or absorption across external body surfaces particularly when exposed to aqueous phase solutions in soils or sediments.

For terrestrial animals, the possible exposure routes include absorption, ingestion, and inhalation. Ingestion is the most important potential route for exposure of terrestrial organisms to metals and other inorganic contaminants. Direct ingestion of soils or sediments could occur as animals accidentally ingest soil from the ground during grazing, soil particles adhering to food, or during grooming. Animals may also ingest prey or forage items that have taken up contaminants from the environment.

Dermal contact is usually not considered an important pathway for uptake of metals in terrestrial animals because skin, integument, and other biological barriers are not permeable to most forms of metals (EPA 1989). Inhalation is a potentially important pathway for burrowing animals in areas where subsurface soils are contaminated with contamination. Data on metal concentrations for the IA indicate that contamination in the Tailing and Smelter IUs is primarily in surface soils, with decreasing concentrations with depth. Soils/sediments of fluvially environments may contain elevated concentrations at greater depths due to the downward leaching of metals from infiltrating runoff. Methods for assessing exposure of burrowing animals to airborne particulates is not well developed and this investigation will focus on ingestion as the primary exposure pathway.

In general, most metals do not tend to biomagnify between trophic levels in aquatic or terrestrial food webs. Exceptions include mercury vapor (non-ionic) and organomercury which are more lipid-soluble than inorganic forms and, therefore, tend to pass through biological membranes and be retained in hydrophobic environments (Eisler 1987, Hrudely et al. 1996). As a result, upper-level consumers in terrestrial environments are unlikely to experience exposures greater than primary consumers. However, some terrestrial plants and animals can accumulate metals to concentrations that are potentially toxic to upper-level consumers. A well-known example of this phenomenon is the accumulation of selenium by some plants of the genus *Astragalus* (among others), with resultant toxic effects on herbivores.

The most common mechanism of metal accumulation is bioconcentration of metals in aquatic animals and plants. Metals dissolved in surface water or in the interstitial water of sediments tend to cross biological barriers, and accumulate in tissues where they may induce toxicity. As a result, aquatic organisms at all trophic levels could accumulate metals to potentially toxic levels, including the ingestion of forage or prey that contain elevated metal concentrations. For vertebrate organisms feeding in the aquatic habitats, bioconcentration of metals in aquatic prey could lead to increased exposure.

Wildlife receptors predicted to experience the greatest exposure to affected soils are small herbivorous, granivorous, or omnivorous birds and mammals. These receptors are in frequent close contact with soils and have relatively small feeding ranges. For the Chino Mine ERA, exposure to metals in the terrestrial-based food web of the mesquite grasslands and riparian communities in the ephemeral drainages is likely to be most important for organisms that ingest ground-dwelling invertebrates and vegetation. Insects and other invertebrates can accumulate metals from soils and vegetation, resulting in increased exposure to secondary consumers. Small mammals and other small vertebrates may accumulate metals to a lesser extent, primarily from soils adhering to body surfaces contained in the gut lumen. A model of the food web for the site was developed based on available information (Figure 2-2).

Temporary water bodies are critical breeding habitat for amphibians in semi-arid lands. In the Chino area, temporary flow in ephemeral drainages results in creation of isolated pockets and pools following runoff after rainstorms. Runoff that collects from contaminated areas can contain elevated levels of mine waste constituents. Aquatic larval stages of amphibians are more sensitive than adult stages to water quality and could be adversely affected by mine waste in runoff.

Identification of potentially complete exposure pathways is used to evaluated the risk of direct effects on ecosystem components. Indirect effects on components that are not exposed to contaminants can result if an important component of its habitat, such as nesting sites or prey base, is adversely affected by toxicity.

3.0 SCREENING-LEVEL EXPOSURE ESTIMATE AND RISK CALCULATION

The screening-level exposure and risk calculation corresponds to Step 2 of the EPA guidance. Step 2 includes an assessment of potential ecotoxicity of chemical stressors based on information available prior to performing the ERA. The result of Step 2 is a decision on whether additional ecological risk assessment is needed. Specifically, Step 2 is intended to support the following decisions (EPA 1997):

1. Available information is adequate to conclude there is no need for remediation within the Chino Mines IA on the basis of ecological risk.

OR

2. Data are adequate to indicate that risks may not be negligible and further assessment is warranted.

OR

3. Available information is not adequate to determine that risks are negligible and more evaluation is necessary to determine the need for further action.

Information available prior to the initiation of the Chino Mines ERA indicated that risks may not be negligible and assessment was needed because:

- Potentially complete pathways exist at the site for the exposure of wildlife and vegetation to mine-related contaminants in soil, surface water, and sediments, including metals and depressed pH
- Analyses presented in the Phase I ERI Proposal (WCC 1997) concluded that copper is present in soils at concentrations that exceed natural concentrations and literature-based toxicity thresholds
- Qualitative inspection of the site indicates areas where deposition of mine waste has apparently resulted in adverse effects on the grassland community
- Direct measures of biological effects such as tissue residues, mortality, population and community measures were not available to directly assess impacts to biotic components of the ecosystem.

This information indicates that risks may not be negligible, but further risk characterization is needed to determine the significance of ecological risks and support decisions regarding the need for remedial action.

Available information has not included formal presentation of an ecotoxicity screen which includes all potential contaminants. Such a screen is an important aspect of the ERA process as described by EPA (1997), and was requested by regulatory agencies with oversight of the Chino Mines ERA. While additional screening is not needed to satisfy the basic decisions listed above, results of the preliminary ecotoxicity screen are needed to define the scope of the baseline ERA.

As suggested in the EPA guidance, screening-level assessment endpoints were adopted that describe the ecological resources for which risk is being assessed. The assessment endpoint(s) for the screening-level exposure and risk estimate is:

- *Identify chemicals that are present at concentrations in abiotic media that could result in adverse effects on populations or communities, as represented by the following ecological receptors:*

Feeding Guild or General Category	Representative Receptor	Media
Large ground-feeding birds	*Gambel's quail*	*soil/sediment*
Small ground-feeding birds	*dark-eyed junco*	*soil/sediment*
Avian predator	*red-tailed hawk*	*soil/sediment*
Omnivorous small mammals	*deer mouse*	*soil/sediment*
Mammalian predator	*Coyote*	*soil/sediment*
Ruminant	*mule deer*	*soil/sediment*
Vegetation		*soil/sediment*
Aquatic life and amphibians		*surface water/sediment*

Wildlife receptor types were identified based on ecological functional groups since soil and dietary ingestion are the primary pathways of concern and potential sensitivity of receptor types to toxicity from metals expected at the site. (See Appendix D for brief description of toxicity from metals.) Measurement endpoints used in the screen were chemical concentrations of potential contaminants in abiotic environmental media at the site (see Section 3.1.10. Species representing each of the wildlife feeding groups and that are likely to be present at the site are included. The objective of the screening-level exposure analysis is to conservatively evaluate potential risk to vegetation and wildlife species that may use areas within the IA.

Large and small ground-feeding birds were included because differences in feeding and soil ingestion rates result in an apparent 10-fold difference in sensitivity to metal concentrations in soil, with smaller birds potentially experiencing toxicity at lower concentrations. Both groups are potentially present at the site. The screen included the most sensitive group, the smaller birds, to represent the most conservative scenario. Larger ground-feeding birds were included to provide a broader scope of information to support design of baseline risk assessment.

Vegetation was included as an assessment endpoint based on the potential for direct contact with contaminants in the soil matrix and its importance in local ecosystems. Vegetation was selected because the toxicological literature indicates that vascular plants tend to be more sensitive to metals contamination than other soil biota. In addition, more complete information on potential toxicity is available for plants. Limited information on the potential toxicity to soil invertebrates and microbes is also presented where appropriate.

Aquatic habitat at the site is limited to ephemeral flows in drainages and temporary pools that develop during wet seasons. However, such areas provide habitat for breeding amphibians and invertebrates that require water for reproduction. Therefore, the screening-level analysis included assessment of the potential toxicity of surface water collected as runoff.

The screening-level analysis was conducted for each of the receptor types by comparing maximum estimates of chemical concentration in environmental media (i.e., soil, sediment, water) to benchmark concentrations that represent non-toxic conditions. Selection and/or development of benchmark concentrations used in the screening process is described in the following section. Screening-level risk estimates are based on the potential for toxic effects directly on the receptor types under evaluation. Indirect effects, such as degradation of habitat quality, were not explicitly considered since toxic effects on individual system components is a more conservative estimate of ecological effects.

3.1 Technical Approach

The initial ecotoxicological screening was conducted by comparing maximum concentrations from site data to risk-based screening criteria. A chemical was included as a COC if (1) maximum concentrations exceeded risk-based criteria for one or more receptors, (2) detection limits from site analyses exceeded benchmarks, or (3) toxicity information was inadequate to set a benchmark. Data available for use in the screen are discussed in Section 3.1.1.

The risk-based criteria developed for the screen are concentrations of chemicals in environmental media that are associated with non-toxic exposures. Criteria were developed for each of the groups represented in the assessment endpoints. The criteria were estimated from toxicity reference values (TRVs) which were developed for each chemical to represent non-toxic dose rates. TRVs were then used to estimate concentrations in environmental media that would result in non-toxic exposures. These criteria were then compared to data available chemical concentrations in environmental media at the Chino site. Development of TRVs is discussed in Section 3.1.2. Use of TRVs in developing risk-based criteria used in the screen is discussed in Section 3.1.3.

3.1.1 Data Used in Screening-level Exposure and Risk Estimates

Environmental data available for use in the screening-level assessment were limited to metal concentrations in soil, dry sediment, and stormwater monitoring data. Data on metal concentrations in biota were not available prior to the ERA. Therefore, concentrations of potential contaminants in forage and prey items were estimated where necessary. Available analytical data also did not include concentrations of anthropogenically generated organic contaminants.

The ecotoxicity screen was conducted for metals and metalloids which are expected to be the primary types of contaminants at the Chino Mines site. No data were available on other potentially ecotoxic chemicals such as chlorinated solvents, polychlorinated biphenyls, or pesticides. Available information does not suggest that these chemicals were used in large quantities at the site or reason to consider them contaminants of concern. However, the lack of data precluded specific evaluation of these chemicals in the ecotoxicity screen.

Soils data available for use in the screening-level assessment were from the Baseline Remedial Investigation and reported in the AOC Background Report (CMC 1995). In general, data were available for metals included in the EPA Target Analyte List (TAL), plus boron and molybdenum. However, the target analytes for each IU were different, and data were not available for all metals in all IUs (Table 3-1). A sitewide summary of soil data is shown in Table 3-2. A summary of data for individual IUs is shown in Table 3-3.

Data were available from the Hurley Soils RI conducted during the summer of 1997. These data are for areas entirely within the developed residential and commercial areas of the town of Hurley. Metal concentrations in this area tend to be higher than in soils from other areas of the site due to the proximity of the smelter and other processing facilities. Therefore, chemical concentrations in these soils may not be representative of concentrations in other IUs or the entire ecological IU. The soils in Hurley were evaluated in the Hurley Soils IU Screening-level Ecological Risk Assessment (Schafer 1998) and are not included in this screen.

Based on guidance from EPA Region 6, risk screening was conducted for all metals for which data were available, including those for which site concentrations did not exceed reference conditions (natural background) according to the AOC Background Report (CMC 1995). However, the risk screen excluded calcium, magnesium, potassium, and sodium which are considered macronutrients and are toxic only at concentrations much greater than those needed to meet minimum nutritional requirements (EPA 1989). Other metals such as chromium and zinc are also essential nutrients, but can be toxic at doses that are more similar to nutritional requirements.

Concentrations of magnesium, potassium, and sodium reported in the AOC Background Report were generally within the range of those detected in background samples (CMC 1995). Calcium concentrations exceeded reference concentrations (CMC 1995), but were generally within the concentration range reported for soils in the western U.S. (Conner and Shacklette 1975, Ebens and Shacklette 1982). Concentrations of these elements seemed adequate to meet minimum nutritional requirements for receptors.

Data on metal concentrations in stormwater were available from ongoing monitoring programs at the site. Stormwater data collected for the sitewide monitoring program were used to assess the potential effect of runoff on aquatic biota in ephemeral drainages and temporary pools. Stormwater data are summarized for sampling sites in Table 3-4. Stormwater sampling locations are shown in Figure 3-1.

3.1.2 Toxicity Reference Values

TRVs used in initial risk estimations for vegetation are presented in Table 3-5, for wildlife in Table 3-6, and for aquatic life in Table 3-7. TRVs were developed from various sources and based on original toxicological research. If possible, primary literature sources were used. If secondary sources were used, both the primary and secondary sources are cited. In all cases, TRVs were developed from results of the original study. TRVs taken from information provided in the Oak Ridge National Laboratory (ORNL) ecotoxicity database were derived from the primary information on original studies, and do not include any adjustments by ORNL to account for population effects.

For vegetation, TRVs are expressed as a total concentration in soil (e.g., mg/kg). In some cases, secondary information from studies described by Kabata-Pendias and Pendias (1992) was used for some metals (Table 3-5). The values presented in this publication represent "maximum acceptable concentrations" and are developed from several investigations. The TRVs developed from this information are one-half of the median concentrations presented. This approach has been used in risk assessments at other sites (Greystone 1997, Schafer & Associates 1997).

Wildlife TRVs were developed to approximate the no-observed-adverse-effects level (NOAEL). Direct measurement of NOAELs or equivalent values were used when available. If a NOAEL was not available, uncertainty factors were applied to original research results using an approach recommended by EPA (Region 6) in which the NOAEL is assumed to be one tenth (0.1) of the lowest-observed-adverse-effects level (LOAEL). The resulting TRV is applied to all potential receptor species in the taxonomic class.

State water quality standards for livestock watering were used to provide a preliminary screen for risk to wildlife from contaminants in drinking water (Table 3-7). These standards are most applicable to ungulates that may drink from stormwater and other temporary water sources. The state standards may not be legally applicable to all water sources. However, they provide an effective tool for screening available surface water data.

For aquatic life, TRVs are based on New Mexico State Water Quality Standards (NMWQS) for protection of aquatic life, or EPA Ambient Water Quality Criteria (AWQC)(Table 3-7). Most of the NMSWQS for metals are based on the AWQC, which in turn are based on protecting approximately 95 percent of taxa for which toxicity test data are available (EPA 1992b). Therefore, both sets of criteria are toxicity- and risk-based. Comparison of stormwater quality to aquatic life standards is intended to provide a preliminary assessment of risk from contaminated runoff.

3.1.3 Development of Risk-based Screening Criteria

As noted in Section 3.1, toxicological information was used to estimate concentration based criteria which were then compared to environmental data to determine which chemicals were present at potentially ecotoxic concentrations. For aquatic organisms, available TRVs are based on federal water quality guidelines or state water quality standards, both of which are already expressed as concentrations and can be compared directly to site data.

No federal or state standards have been promulgated for assessing chemical concentrations in soils. Therefore, available information was used to estimate soil screening criteria (SSCs) for use in evaluating site soils. Vegetation TRVs are expressed as concentrations which can be compared directly to site soil data (Table 3-5). For terrestrial animals, TRVs are based on rates of chemical intake (e.g., mg ingested/kg body wt./day)(Table 3-6). Therefore, SSCs for birds and mammals were calculated from available information and are expressed as chemical concentrations in soils that would result in intake rates equal to the TRV:

[Equation 1]

$$\text{Daily Intake (mg/kg bw/day)} = [((FIR*(C_{soil}*BAF)) + (SIR*C_{soil})]*AUF$$

where:

FIR = *daily food ingestion rate (kg/kg body weight/day)*

SIR = *daily rate for incidental ingestion of soil or sediment (kg/kg body weight/day)*

C_{soil} = *concentration of COC in soil and/or sediment (mg/kg)*

BAF = bioaccumulation factor (ratio of concentrations in food and soil, based on empirical data)(unitless)

AUF = area use factor; the proportion of the daily intake obtained from the study area (unitless)

Where possible, maximum daily rates for intake of forage and prey and incidental ingestion of soils were obtained for representative species from EPA *Wildlife Exposure Factors Handbook* (EPA 1993). Alternative sources for information were used if data were more representative of western species, or more accurate data were available. Intake parameters used in estimating SSCs are shown in Appendix B. For screening, the assimilation efficiency of all metals in ingested soils or biota was assumed to be 100 percent. This is a conservative estimate since the bioavailability of most metals is less, especially directly from incidentally ingested soils or gut content of prey items. Calculation of screening-level SSCs also assumed that animals were obtaining 100 percent of exposure from areas under evaluation (i.e., area use factor = 100 percent).

Because data on chemical concentrations in biota were not available, the concentration of chemicals in food items (i.e., forage or prey) was estimated using bioaccumulation factors (BAFs). Estimates of BAFs for transfer of metals from soil to vegetation (above-ground biomass), invertebrates, and small mammals were obtained from data collected at other mine or mineral processing sites (Table 3-8)(Sample et al. 1998) and other sources. These estimates are based on sites where soils and biota were collected from co-located sampling sites and represent conservative (higher) estimates of uptake. Concentrations in biota are from unwashed samples and, therefore, include metals adhering to the surface as well as internalized forms of the metals. If BAFs were not available for a given metal, a BAF of 1.0 was assumed. This conservative assumption implies that the metal concentration in food items is equal to that of soil.

Details of SSC calculation are shown in Appendix C. The SSCs used in screening are presented in Table 3-9. Neither the process for development of SSCs or selection of COCs included comparison to background or reference area conditions. Consideration of site-specific reference conditions will be included in the initial stages of the baseline ERA.

SSCs were compared to the minimum detection limits (DL) for metals reported in the data base associated with the AOC Background Report (Table 3-10). Detection limits for boron, molybdenum, thallium, and vanadium exceeded SSCs for at least one receptor type, indicating risk estimates for these receptor and chemical combinations may not be reliable. As a result, these elements were retained for further analysis in the baseline risk assessment.

3.2 Screening-Level Exposure and Risk Estimates

3.2.1 Soils/Sediments Compared to SSCs

SSCs were compared to sitewide maximum metal concentrations to determine the metal concentrations that exceed SSCs for the entire AOC Investigation Area, including all potential contaminant sources (Table 3-11). Comparison to soil data for individual IUs is shown in Tables 3-12 through 3-15, and includes maximum and average concentrations. Selection of COCs was based on comparison of SSCs to maximum concentrations. Results were expressed using the hazard quotient (HQ) approach (EPA 1997), which is the ratio of the estimated exposure to the SSC (i.e., [maximum concentration at site] ÷ SSC). An HQ < 1 indicates negligible risk. An HQ $\geq$1 indicates that further analyses are necessary to characterize the extent and magnitude of risk.

Results for comparison of sitewide maxima to SSCs are summarized in Table 3-16. Concentrations of beryllium and silver did not exceed the SSC for any receptor. However, neither beryllium nor silver was analyzed in samples from the Hanover-Whitewater IU and, therefore, were included as COCs for this IU. For other metals, either the maximum concentration for each of the other metals exceeded at least one SSC, or information was insufficient to develop an SSC.

Evaluation of results for individual IUs reveals that the Hanover-Whitewater Creek IU contains the highest concentrations and is potentially affected by the broadest range of chemicals (Table 3-17). Data from the Hanover-Whitewater IU includes predominantly sediments collected from the dry beds of the ephemeral drainages. The drainages may be vegetated and used by wildlife during periods of low or no flow. Potentially affected media in the Lampbright IU is also predominantly composed of dry sediment. Data for the Smelter and Tailing IUs is predominantly from upland soils which support various vegetation communities and wildlife habitats.

As noted earlier, the comparisons shown in Tables 3-11 to 3-15 do not consider background/reference area concentrations. The AOC Background report presented comparisons of site concentrations to reference concentrations for the IUs. However, analyses were not available for all analytes in all IUs (Table 3-1) and suitable reference area data have been established only for the soils in the HSIU. Therefore, information is currently insufficient to determine whether metal concentrations in site soils exceed concentrations in reference area. Establishment of reference concentrations and statistical comparison of site data to reference conditions will be conducted as part of the remedial investigations for each of the IUs.

3.2.2 Stormwater Compared to Aquatic TRVs

Evaluation of stormwater indicated that concentrations of some chemicals exceeded aquatic TRVs (Table 3-18). Table 3-18 shows only analytes for which concentrations exceeded the benchmarks. The list of analytes for which stormwater is analyzed is shown in Table 3-4. Data were not available for runoff from reference area soils or sediments. Therefore, the potential for natural concentrations of chemicals to exceed TRVs could not be assessed.

3.3 Summary of Screening-level Problem Formulation and Ecotoxicity Assessment

The screening-level problem formulation and risk calculations were intended to support decisions regarding the need for further ecological risk characterization and the scope of the additional analysis (EPA 1997). The following summarizes the results of the screening-level assessment:

- Data from the AOC Background Report (CMC 1995) indicate that release of mine waste may resulted in increased concentrations of some metals and reduction of soil pH in affected areas of the site
- Current data are insufficient to quantitatively determine which chemicals occur at concentrations that exceed natural reference concentrations for specific soils and other geologic materials
- Potentially complete pathways exist at the site for the exposure of wildlife and vegetation to mine-related contaminants including metals and depressed pH
- Analyses presented in the Phase I ERI Proposal (WCC 1997) concluded that copper is present at concentrations that exceed natural and literature-based toxicity thresholds
- The screening-level assessment in the preceding section identify metals other than copper that occur at potentially ecotoxic concentrations
- Qualitative inspection of the site indicates areas where deposition of mine waste has apparently resulted in adverse effects on the grassland community
- Data available for risk characterization are not sufficient to support decisions regarding need for remediation to reduce ecological risk. Direct measures of biological effects such as tissue residues, mortality, population and community measures were not available to directly assess impacts to biotic components of the ecosystem.

Tables 3-15 and 3-18 summarize chemicals for which available data indicate that sitewide maximum concentrations exceeded risk-based benchmarks for terrestrial and aquatic organisms. Based on available data, concentrations of silver do not exceed SSCs and may represent negligible risk. However, data on silver concentrations were not available from all areas of the site (Table 3-1 and 3-16) and further sampling or analysis of mine waste streams is needed to eliminate this chemical from further consideration.

Concentrations of other metals exceeded vegetation, wildlife, or aquatic biota screening criteria. The Hanover-Whitewater IU sediments contained the greatest number of constituents in this category (Table 3-16).

4.0 BASELINE RISK ASSESSMENT PROBLEM FORMULATION

The Baseline Risk Assessment Problem Formulation is Step 3 in the EPA Superfund ERA process. The objective of this step is to plan further risk analyses based on results of the screening-level risk assessment. The process was begun by NMED and CMC in preparation of the Phase I ERI Proposal and the statement of work for the ERA.

According to EPA (1997) guidance, the issues to be addressed in the Problem Formulation are:

- Identification of COCs
- Literature search on the ecotoxicity of COCs
- Description of COC fate and transport as it relates to the ERA
- Identification of ecosystems (or ecosystem components) potentially at risk
- Identification of potentially complete exposure pathways
- Determination of appropriate assessment endpoints and risk questions

Identification of COCs to be included in the Baseline ERA is presented in Section 4.2. A summary of toxicity of potential COCs is presented in Appendix D. A summary of fate and transport mechanisms and potentially complete exposure pathways were presented in Section 2 as part of the initial site description. The portions of the Chino site and ecosystem components potentially at risk are presented in Section 4.3. A summary of the goals and objectives of the ERA, including identification of assessment endpoints, is presented in Section 4.4.

Section 4.5 presents the proposed approach for analysis and risk characterization. This includes activities described for Step 4 of the EPA guidance (Study Design and Data Quality Objectives) includes (EPA 1997):

- Establishment of measures to be used in the analysis risk characterization
- Completion of the site conceptual model
- Establishment of study design and data quality objectives

Once a study design has been approved, a detailed sampling and analysis plan will be developed including statistical considerations. The detailed study design will be provided in TM-2.

4.1 Summary of Screening-level Problem Formulation

The rationale for proceeding with the Baseline ERA is based on several lines of evidence. Identification of information used in determining the need for the ERA is essentially equivalent to the initial step in the DQO process, which is to "state the problem" (EPA 1994), or characterize the site conditions that have lead risk managers to conduct an RI/FS and ERA. For the Chino Mine site, these conditions include:

- Knowledge of historical releases from CMC mining and mineral processing facilities
- Knowledge of historical release from mining and mineral processing facilities not owned or controlled by CMC in the Hanover Creek and Whitewater Creek watersheds
- Elevated concentrations of copper and other metals in site media (WCC 1997)

- Results of screening-level assessment indicate that some metals are present at concentrations that exceed risk-based estimates of ecotoxic threshold concentrations (Section 3 and WCC 1997)
- Evidence of ecological stress (apparent phytotoxicity and resulting soil erosion)

Potentially affected areas of the site can be grouped into two categories, (1) areas subject to fluvial transport of contaminants in the ephemeral drainages (Hanover-Whitewater and Lampbright IUs) and (2) more upland areas (primarily soils) affected by aerial deposition of contaminants from historic smelter emissions and windblown tailings (Hurley, Smelter, and Tailing IUs)(Figure 1-2). Fluvial areas may also be subject to runoff from soils affected by windblown materials. Metal concentrations, especially copper, lead, and zinc, tend to be higher in the fluvial environments than in areas affected by windblown transport of smelter emissions and tailings.

Within the Hanover-Whitewater IU, releases from Chino sources and other historic mining sources are mixed with instream sources such as the tin-can plants. Tin-can plants are areas of the drainage where local citizens had historically placed "tin" cans and other iron-containing materials for the purpose of scavenging dissolved copper from process waters released into the streams by the mining company. Tin can plants are no longer in use because the mine no longer releases water.

Some areas of the IU may have higher metal concentrations due to the effect of depositional processes on contaminant distribution. Contaminants and low pH in sediments and soils along the sides of the drainage may adversely affect vegetation and wildlife that occur in these areas. Currently, several sections of the bed in the Hanover-Whitewater drainage lacks vegetation. The lack of vegetation could be due to natural processes such as shifting sediment or to effects of metals released from the mine. It currently is unclear to what extent release of mine waste from the Chino facilities has resulted in adverse effects in the ephemeral drainages of the Hanover-Whitewater and Lampbright IUs.

Soils and other surface materials in the Smelter and Tailing IUs have varying metal concentrations depending on the source of contamination and the distance from the source (Figure 4-1). Ecological stress due to the deposition of mine waste and/or smelter emissions is observable in some sections of the site. Specifically, the area immediately east of the tailings impoundments lacks the herbaceous plant component that typically occurs in the mesquite grasslands in other areas of the site. The cause of the stress has not been confirmed, but deposition of smelter emissions and/or windblown tailings may have played a role.

The phytotoxicity of several metals, including cadmium, copper, lead, and zinc, increases with decreasing pH due to the increased solubility and bioavailability of metals. In addition, pH below ~5 alone can inhibit seed germination and plant growth. For copper, solubility and free ion activity increases steadily with decreasing pH below pH 7 (Kabata-Pendias and Pendias 1992, Schafer & Associates 1993, Sauvé et al. 1998). The weathering of residual sulfides in tailings and smelter emissions has apparently led to the development of acidic pH in soils east of the source areas (Figure 4-1)(CMC 1995). Areas affected by tailings and smelter emissions appear to have similar pH, although copper concentrations are highest in areas affected primarily by smelter emissions (Figure 4-2). Copper concentrations in smelter-impacted soils exhibit a logarithmic decrease with distance, consistent with airborne dispersion processes. The effects of elevated metal concentrations may be exacerbated by the depressed pH in some areas affected by the sources.

The toxicity of metals to aquatic insects and other aquatic organisms is well known, and TRVs and state water quality standards have been developed that are protective of most species. However, metal toxicity to amphibians is less well known, especially to species that are native to the arid and semi-arid regions of the southwestern US. Runoff from affected soils and sediments may accumulate in drainages, forming temporary or permanent water bodies that are used by amphibians for breeding. These water bodies are very important to breeding amphibians and other wildlife.

4.2 Constituents of Concern

Identification of COCs for the baseline ERA was based primarily on exceedence of risk-based criteria in the screening-level assessment by maximum site concentrations. The screen was completed for potentially complete exposure pathways. Metals for which the maximum concentration exceeded criteria for at least one receptor type are shown for soils in Tables 3-16 and 3-17, and for stormwater in Table 3-18. Most of the available analytes exceeded at least one criterion. The COCs proposed for inclusion in the terrestrial and aquatic components of the Baseline ERA are:

Terrestrial

aluminum—(vegetation and wildlife)
antimony—(vegetation and wildlife)
arsenic—(vegetation and wildlife)
barium—(wildlife)
boron—(vegetation and wildlife)
cadmium—(vegetation and wildlife)
chromium—(vegetation and wildlife)
copper—(vegetation and wildlife)
iron—(vegetation and wildlife)
lead—(vegetation and wildlife)
manganese—(vegetation and wildlife)
mercury—(vegetation and wildlife)
nickel—(vegetation)
selenium—(vegetation and wildlife)
thallium—(vegetation and wildlife)
vanadium—(vegetation and wildlife)
zinc—(vegetation and wildlife)
hydrogen ion activity (pH)—(vegetation)

Aquatic

arsenic
cadmium
chromium
copper
nickel
lead
zinc

Silver was excluded from further consideration because maximum concentrations did not exceed risk-based benchmarks at any of the IUs (Table 3-16).

Molybdenum was excluded from this list because the primary toxicological effect of molybdenum is copper-deficient molybdenosis. This is typically expressed in domestic cattle and sheep when the copper-to-molybdenum ratio in forage materials is less than about 3 (Eisler 1989). Ruminant wildlife such as deer and elk appear to be resistant to elevated molybdenum concentrations, regardless of copper concentrations (Eisler 1989). Moose (*Alces alces*) in some areas of Sweden have been reported to be susceptible to molybdenosis in areas where the pH of soils has been artificially elevated due to liming (Frank 1998). Copper concentrations in soils and sediments of the Chino Mine area are naturally high due to natural mineralization and effects of mine waste. As a result, the probability of molybdenosis at the site appears to be small. However, molybdenum will be included in analysis of samples collected for the ERA because of its tendency to reduce copper toxicity in mammals (Eisler 1989) and some plants (Kabata-Pendias and Pendias 1992).

Concentrations of several metals soil samples from the Tailing IU did not exceed SSCs. However, many of these metals were not included in analyses for the Smelter IU, so their concentrations could not be established for these areas of the site (see Section 3.3). These metals will be included in the ERA analysis until further data or rationale support their removal from the COC list. Boron, thallium, molybdenum, and vanadium were also included because detection limits in the RI data set exceeded the SSCs for soils and sediments.

As discussed previously, the screening-level assessment did not consider reference area concentrations. Metals included in the assessment occur naturally in soils and geologic materials of the area. Given the highly mineralized nature of the area, many metals occur at naturally elevated levels. In some cases, natural concentrations may exceed literature-based toxicity benchmarks (e.g., see Table 3-9). In many cases, biotic components of the ecosystem are adapted to the naturally elevated concentrations and the forms of metals. However, since mining and mineral processing activities tend to alter the concentration and the form of metals in geologic materials, evaluation of ecological risk from exposure to metals should include some consideration of the natural background conditions.

It is the opinion of NMED that data available prior to preparation of this TM were not adequate to characterize reference concentrations of metals and, therefore, to determine whether metal concentrations in site soils and sediments are elevated due to mine operation. The AOC Background Report included a preliminary comparison of site data for each IU to available reference area data. However, the method and metals included in the analyses varied among IUs. Therefore, metals were not excluded from COCs based on available reference comparisons. Characterization of reference and comparison to site data will be an important function of the RIs.

The evaluations included in the baseline ERA will include each of the primary COCs listed above. However, an initial step of the ERA will be to consider results of reference comparisons conducted in the RIs when they become available, or to include reference comparisons using RI data if they are not conducted during the RI. Methods to be used in such comparisons will be consistent with those agreed upon for use in RIs. Constituents that are not enriched in the waste stream elements (i.e., tailings, smelter emissions, concentrate) and are not present at concentrations that exceed reference conditions, but that exceed screening level benchmarks will be retained and evaluated in the baseline risk assessment.

Comparison to reference (background) conditions is particularly important for aluminum, barium, chromium, thallium, and vanadium. These constituents were associated with HQs >1 for one or more receptors in the screening assessment, but preliminary analysis conducted for the AOC Background Report indicated that they may not be present in soils at concentrations that exceed reference conditions (CMC 1995). Aluminum, for example, is a primary component of clay minerals, which act as natural sorbents for metals such as chromium and vanadium.

4.3 Ecosystems Potentially at Risk

Ecosystems that are potentially at risk occur in areas of the site where concentrations are elevated to potentially toxic concentrations, as defined in the screening-level ERA. The ecosystem components and geographic areas potentially at risk were identified based on information on the nature and extent of contamination in the AOC Background Report (CMC 1995), remedial investigation proposals, NPDES stormwater sampling data, other data sources, and results of the screening-level ERA. In general, the areas coincide with IU descriptions. The areas include:

1. Areas in the Smelter and Tailing IUs east and west of the smelter facilities and tailings impoundments. The most extensive habitat types in these areas include the Honey Mesquite/Cactus-Yucca/Mixed Grasses and Mixed Grasses/Yucca-Cactus (CMC 1995). In addition, the more upland areas northeast of the smelter facility include Oak/Juniper and Juniper/Pinyon Pine/Oak habitat types. Sections of Whitewater Creek, Lampbright Draw, and Martin Canyon contain the Riparian/Mixed Deciduous Shrub vegetation community type and could also be affected by runoff from affected upland areas.
2. Areas affected by fluvial transport of COCs in sediments and stormwater runoff include the Riparian/Mixed Deciduous Shrub vegetation community type in Hanover-Whitewater Creek drainages and Tributaries 1 and 2 of Lampbright Draw, and possibly the main channel of Lampbright Draw. Terrestrial, semiaquatic (e.g., amphibian), and aquatic organisms are potentially affected. Aquatic habitats are primarily restricted to temporary pools in drainages of the site. However, temporary pools in more upland areas may also provide ephemeral habitat for amphibians and insects that require standing water for reproduction.

Each of the areas contains inclusions of other habitat types that may be affected by mine waste. A conceptual model for contaminant flow and exposure of ecological receptors in the habitat types is presented in Section 2 (Figure 2-1).

4.4 ERA Goals and Objectives

The ERA goals and objectives were developed according to EPA guidance on conducting ERAs (EPA 1997, 1998) and the Data Quality Objectives Process (EPA 1994). EPA develops goals, objectives, and data needs for ERAs through identification of management goals, assessment endpoints, risk questions, and measures to be used in the risk analysis. Management goals define the broad objective of ecological resource management on which the ERA is based.

Assessment endpoints are explicit expressions of the ecological resource to be protected (EPA 1992a, 1997, 1998) and provide the focus for the ERA. Identification of assessment endpoints is necessary to focus the ERA on more sensitive and ecologically relevant receptors, rather than attempt to evaluate risks to all potentially affected ecological receptors. Assessment endpoints should be consistent with management policy goals and ecological values for the site.

"Risk questions" as described by EPA (1997) are the questions the ERA will attempt to answer regarding whether or not assessment endpoints have been adversely affected by exposure to COCs. They form the basis for identifying the specific analyses to be conducted and the data needed to perform the analysis. In some cases, risk questions may be stated as "risk hypotheses" (EPA 1998) which form the basis for identifying the data collection and analysis to be performed. Evaluation of risk hypotheses is not equivalent to formal statistical tests of null hypotheses (EPA 1998).

Assessment endpoints and risk questions/hypotheses are used to identify the types of measures needed to perform ERA. Three types of measures will be used in the ERA (EPA 1998):

- **Measures of effects**—measurement of changes in an attribute of the assessment endpoint in response to exposure
- **Measures of exposure**—measures that describe the location and concentrations of COCs in abiotic and biotic media that can be used to estimate exposure of receptors
- **Measures of ecosystem and receptor characteristics**—measures of factors such as receptor behavior, life history characteristics, and transport of COCs that may affect intensity of exposure or manifestation of effects

As noted previously, the Problem Formulation process described for ERAs (EPA 1997) is similar to the DQO process (EPA 1994). However, the components of the DQO process that require *a priori* identification of decision rules and statistically based decision criteria in the form of SSCs are not always applicable to risk hypotheses used in the ERA (EPA 1998). Decision criteria were used in the screening-level analysis described in Section 3. Such binary decisions are not applicable to many aspects of the baseline ERA because of the need to describe impacts, risk, and respective sources prior to developing decision criteria for remedial actions, if any (EPA 1998). The following section describes the initial stages of the Problem Formulation process for the Chino ERA.

4.4.1 Identification of Management Goals and Decisions

Management Goals

Management goals are used to identify the inputs to the decisions. The (proposed) ecological risk management goal on which the ERA design is based is:

- *Prevent or remediate adverse direct or indirect effects on populations or ecological communities due to toxic exposure to chemicals in mine waste.*

Adverse effects are defined as those that result in reduced capacity for sustainable populations or communities. The level of evaluation is affected by the rarity of the species under question. Where threatened or endangered species may be exposed to mine waste, the management goal is to protect individual organisms from effects that result in reduced survivorship or capacity for reproduction.

Evaluation of community-level impacts and risks should consider the current and projected land uses of the site (Harwell et al. 1994). Cattle are grazed in a large portion of the site; grazing can affect the vegetation community composition and structure. Altered vegetation communities may in turn affect use of the site by wildlife, including vertebrates and invertebrates. A goal of the risk assessment will be to determine whether adverse effects are due to the presence of mine waste or to other stresses such as grazing.

Decisions

NMED has ultimate responsibility for remediation decisions at the Chino Mine site, but will consider input EPA and CMC. Remediation decisions will be based on the results of the ERA and other environmental investigations performed at the site. Risk assessors will provide risk managers with the information to make remediation decisions.

Two levels of decisions are to be based on the ERA (EPA 1993):

- *Early Assessment Decisions* in which the goal is to determine whether a contaminant release represents a threat to the environment, and
- *Advance Assessment Decision, Phase I* in which the goal is to determine whether contaminant concentrations exceed ARARs or risk-based criteria developed based on site-specific information.

The fundamental decisions that the ERA is designed to support are:

1. Determine whether COCs released from CMC operations have resulted, or are likely to result, in adverse effects to assessment endpoints
2. If adverse effects have occurred or are likely to occur, determine which COCs, exposure pathways, and fate and transport mechanisms are most important in causing the effects
3. Determine whether adverse impacts or risk of adverse effects warrant remediation
4. Determine the impacts to the environment caused by potential remedial actions

4.4.2 Assessment Endpoints, Approach Objectives, and Risk Questions

Assessment Endpoints

Assessment endpoints were identified based on ecological relevance, potentially complete exposure pathways, taxonomic groups that may be sensitive to chemical stressors and are potentially exposed, and site management goals (EPA 1998). Terrestrial pathways include exposure of plant and wildlife to contaminated soils. “Aquatic” pathways refer to the ephemeral drainages and include exposure of riparian vegetation and wildlife and aquatic or semi-aquatic (e.g., amphibians) species to runoff from mine waste and contaminated sediments. Amphibian populations could be especially sensitive since they require aquatic habitat for breeding and are typically very sensitive to aquatic contamination. The proposed assessment endpoints and rationale for their inclusion are summarized below.

Terrestrial vegetation community (growth, community composition, reproduction, recruitment):

> *Ecological relevance:* Vegetation is critical as a food source and physical habitat for wildlife. Changes in plant communities can affect overall habitat quality. Loss of vegetative cover can also result in erosion of surface soils, which can inhibit revegetation.

Susceptibility to COCs: Various plant species have been shown to exhibit toxic responses to elevated concentrations of metals including copper, and acidic soil pH. Adverse effects of metals and acidity on plants include failure to germinate, stunted growth, failure to produce viable seeds or propagules, and death.

Elevated metal concentrations, acidic pH, and deposition of historic smelter emissions has been shown to adversely affect vegetation communities at other mining sites. Factors such as grazing and past land use can also affect plant community structure.

Relevance to Management Goals: Metal toxicity to vegetation can alter the plant community composition and structure, which can result in decreased wildlife habitat and range quality.

Ephemeral Drainage plant community (growth, community composition, reproduction, recruitment):

Ecological relevance: Vegetation in ephemeral drainages is a critical food source and provides physical habitat for wildlife. Although relatively small in area compared to more extensive grassland habitats, ephemeral drainages contain much of the vegetation and wildlife diversity of the hot desert grasslands such as those found at the Chino Mine. As a result, alteration or loss of plant communities in ephemeral drainages can affect wildlife populations.

Susceptibility to COCs: Susceptibility is similar to that described for terrestrial vegetation. The primary route of exposure is through uptake of metals across root and leaf surfaces.

Relevance to Management Goals: Metal toxicity to vegetation can alter the plant community composition and structure, which can result in decreased wildlife habitat and range quality. Loss of vegetative cover on stream banks can also lead to degradation of water quality.

Terrestrial invertebrate community (growth, reproduction, community composition, biomass):

Ecological relevance: Terrestrial invertebrates are critical components of terrestrial food webs and play important roles in energy transfer among trophic levels. Invertebrates are also important in other ecosystem processes such as pollination of plants, decomposition of plant and animal matter, and nutrient cycling. Alteration in plant community or soil condition can affect the abundance and composition of invertebrates in terrestrial communities.

Susceptibility to COCs: Elevated metal concentrations in soils can result in toxic effects on invertebrates of various taxa (Newman and McIntosh 1991). Toxic effects include reduced reproductive capacity, decreased lifespan, and acute mortality.

Relevance to Management Goals: Terrestrial invertebrates are important components of energy transfer in local food webs and importance in nutrient cycling. Adverse effects to terrestrial invertebrate abundance could result in reduced food supplies for insectivorous species.

Herbivorous, insectivorous, and omnivorous birds (community composition, survival, growth, reproduction):

Ecological relevance: Migratory and resident birds provide a food source for avian and mammalian predators, can be important in dispersal of seeds, and help control insect populations. Ephemeral drainages are important migration corridors for several species of warblers and other small birds that may breed in more northerly and upland areas.

Susceptibility to COCs: Several metals can cause sublethal and lethal effects in birds. The primary exposure routes are through ingestion of contaminated soil, food, or water.

Relevance to Management Goals: The potential exists for direct effects on birds inhabiting affected portions of the site, and indirect effects due to loss of habitat due to phytotoxicity, especially along the ephemeral drainages.

Raptors (survival, growth, reproduction):

Ecological relevance: Raptors are important predators in the hot desert grassland habitats. Red-tailed hawks and peregrine falcons are known to inhabit cliffs of the Kneeling Nun formation approximately 5 miles northeast of Hurley.

Susceptibility to COCs: Several metals can cause sublethal and lethal effects in birds. The primary exposure routes are through ingestion of contaminated soil, food, or water.

Relevance to Management Goals: The potential exists for direct effects on raptors ingesting prey from potentially contaminated areas of the site, and indirect effects if food supplies are adversely affected at the site. Because of typically large home ranges, few individuals would be affected at a given time. However, assessment of risk to raptors will provide an evaluation of risk on a large spatial scale.

Herbivorous, insectivorous, and omnivorous small mammals (community composition, survival, growth, reproduction):

Ecological relevance: Small mammals are critical components of local food webs in many habitat types. Significant long-term reductions in local small mammal populations could affect predator populations. Indirect adverse effects to small mammals could result from loss of habitat or food source associated with effects on vegetation.

Susceptibility to COCs: Several metals can cause sublethal and lethal effects in rodents and lagomorphs (rabbits and hares). The primary exposure route is through ingestion of contaminated soil, food, or water. Many species of small mammals are frequently in close contact with soils in burrows and feed on vegetation and invertebrates.

Relevance to Management Goals: Due to their relatively small home ranges, small mammals may represent the most highly exposed vertebrates. In addition, small mammals can be sampled and evaluated for adverse effects more efficiently than large mammal or bird species.

Ruminants (survival, growth, reproduction):

Ecological relevance: Ruminants such as mule deer are important primary consumers in many ecosystems. Mule deer also provide an important food source for mountain lion, and to a lesser extent, coyotes. Individuals and groups of mule deer typically occupy large areas.

Susceptibility to COCs: Several metals can cause sublethal and lethal effects in mammals including ruminants. The primary exposure routes are through ingestion of contaminated soil, food, or water.

Relevance to Management Goals: Assessment of mule deer provides a representative of a primary consumer with a large home range which may include affected and unaffected areas of the site and adjacent regions.

Mammalian Predators (survival, growth, reproduction):

Ecological relevance: As with raptors, mammalian predators such as coyotes and mountain lions represent the top consumers of local food webs.

Susceptibility to COCs: Several metals can cause sublethal and lethal effects in mammals. The primary exposure routes of exposure for mammalian predators are through ingestion of contaminated soil, food, or water.

Relevance to Management Goals: Large mammalian predators typically have large home ranges and would require a large affected area, or impacts to a critical resource, to result in impacts to local populations. However, their density is lower so loss of a few individuals is more important than with mice. Inclusion of such receptors in the evaluation will provide assessment of ecological risk on a large spatial scale.

Amphibians (survival, growth, reproduction):

Ecological relevance: Most amphibians depend on availability of standing water for successful breeding. In arid areas, species are adapted to using isolated temporary pools that form during wet seasons. Contamination of important pools could result in significant adverse effects on local populations. The drainages where amphibians breed are potential sinks for the contaminants.

Susceptibility to COCs: Amphibians are known to be sensitive to toxicity from several metals and depressed pH of water. The primary route of exposure is direct contact with surface water, sediments, and soils.

Relevance to Management Goals: The Chino site largely lacks surface water features and aquatic habitat. Therefore, adverse effects on water quality in important breeding pools could affect a disproportionately large percentage of local populations.

The Phase I ERI Proposal included "ecosystem integrity" as the overall initial assessment endpoint and data on range condition were collected to assess impacts at landscape level of organization (WCC 1997). However, assessment of risk to individual ecosystem components as a measure of ecosystem integrity is necessary to generate a predictive risk assessment. Measurements at the organism (or suborganismal) level are often the most reliable measurements. This will ultimately increase the utility of the ERA to the risk managers because more lines of evidence will be available for review.

Objectives of the ERA

As noted above, ecological stress on vegetation is visually apparent in some areas of the site. However, for other receptor types and assessment endpoints, impacts or risk of impact are not visually apparent and therefore, additional data and analyses are necessary to characterize impact and risk. Therefore, the overall approach to the ERA has three main objectives:

1. *Characterize soil/sediment conditions (e.g., metal concentration, pH) associated with adverse impacts on vegetation and invertebrate assessment endpoints at specific sampling sites in affected areas of the IUs.* The objective is to develop a set of criteria that can be used to evaluate the potential for ecological risk in areas of the site that have not been sampled or characterized. This approach is intended to fit with the staggered schedule for performing the RIs for different IUs. The criteria would be used to evaluated uncharacterized areas of the as data on nature and extent of contamination become available from RI field studies.

2. *Identify areas of the site with soil conditions associated with those identified as causing adverse impacts as defined under objective No. 1.* The criteria developed under No. 1 will be applied to other portions of the site as data become available from RIs.

3. *Evaluate the potential that adverse impacts have occurred or are likely to occur for wildlife receptors occupying areas of the site that are potentially affected.* This includes potential effects on wildlife and vegetation communities for which adequate reference conditions are not available. For wildlife, accurate assessment of population-level effects is highly uncertain without long-term monitoring. Therefore, risk characterization for wildlife will focus on comparison of estimated exposures of receptors to ecotoxicological benchmarks. However, the baseline analysis will more fully characterize the magnitudes and spatial distribution of exposures at the Chino site. This assessment will be used to more accurately reflect exposure and risks to the local population. Where practical, the assessment will also include effects such as accumulation of COCs in tissues.

Risk hypotheses and measurements are listed in Table 4-1. The risk hypotheses can be classed into two basic categories (EPA 1998):

1. *Exposure Assessment.* Available data indicate that ecological receptors in areas with elevated metal concentrations in soils have the potential for exposure to COCs. However, the bioavailability of metals in soils can be highly variable. Exposure estimation is based on site-specific estimation of bioavailable metal.

2. *Effects Assessment.* The effects assessment consists of measurements of endpoints that can be indicative of ecological stress, if present. This includes measurements at the individual level (e.g., histopathology, chemical residues, toxicity tests) and higher levels of organization including population (density, biomass), community (taxonomic composition and abundance), and landscape (range condition).

4.4.3 Identification of Data Gaps

Data available at the time this TM was prepared included:

- metal concentrations (total) and pH from a limited number of surface soils/sediment samples from each of the IUs (CMC 1995)
- metal concentrations (total) and pH of subsurface soils/sediments from a subset of sites sampled for surface soils (CMC 1995)
- data on stormwater quality from the site stormwater discharge monitoring program (personal communication, R. Quintana)

Note: Analytes for which data were available varied among IUs.

Additional data collection is planned for each of the IUs, and will be implemented according to Remedial Investigation Proposals (RIPs) currently being prepared by CMC. The data needs addressed in the RIPs focus primarily on defining the nature and extent of contamination and needs of human health risk assessments. Ecological risk assessors will also review the RIPs to determine whether additional sampling may be required to characterize areas of each IU for ecological risk.

Data were collected in 1997 to analyze range condition for soil and vegetation community types within the AOC IA. Data from this sampling effort are currently being entered into an electronic data base for subsequent analysis of range condition. These data were not available for detailed review during preparation of this TM. However, the data may be used in the ERA when they become available.

The primary data needs for the ERA will be addressed in a specific sampling program designed to collect data from co-located sites. General data gaps were identified based on review of available data from the AOC Background Report, sampling proposed in available RIPs, and the types of data collected for the range condition analysis. As a result of this analysis, the following general data gaps were identified:

Data Gap Type	Data Use
Definitive comparison of metal concentrations in site samples to reference (background) samples	Determine which chemicals are elevated in site media due to release of mine waste and should be evaluated in the ERA versus naturally elevated levels of mine waste
Adverse effects of exposure to COCs on plant communities that are not apparent upon casual observation	Evaluate adverse effects on plant community as related to soil contamination
COC concentrations, pH, and other conditions with associated phytotoxicity	Evaluate the area over which adverse effects to vegetation recruitment may be expected
COC content of biological tissues in potentially affected areas and reference (unaffected) areas	Characterize uptake of COCs by plants and exposure to consumer species
COC content of biological tissues in reference (unaffected) areas	Characterize natural conditions to which ecosystem components are exposed
Locations and water quality of temporary and permanent water bodies in potentially affected areas	Assess potential effects on receptors that require standing water

Data Gap Type	Data Use
Water quality of temporary water bodies in reference areas	Characterize natural conditions to which ecosystem components are exposed

4.5 Technical Approach

4.5.1 RI Comparisons of Site Data to Reference Concentrations

Formal comparison of site soil and sediment data to reference area concentrations will be performed as a component of the RIs. Results and statistical methods for comparisons will be presented in the RI reports. Results of the comparisons will be used in the ERA to determine the extent to which COCs will be evaluated in the risk characterization—the analysis and risk characterization will focus on COCs found at concentrations that exceed reference area concentrations and SSCs. COCs for which site concentrations are below reference concentrations will be evaluated qualitatively.

Data from some RIs may not be available prior to field data collection activities proposed for the ERA. Therefore, the initial planning for ERA field sampling will include collection of data needed to address risk from all COCs identified in Section 4.1. COCs for which site concentrations in soils or sediments are shown not to exceed reference area concentration but that exceed SSCs will be evaluated further in the baseline risk assessment. Specific sampling or analyses for such chemicals could be eliminated from ERA sampling plans as appropriate.

4.5.2 Assessment of Risk to Terrestrial Components of the Ecosystem

Sampling to collect data identified in Table 4-1 will be performed using a modified "gradient" approach (EPA 1997) in which a suite of analyses will be performed at sites selected to represent the range of observed COC concentrations and pH. The general objective is to identify a combination of COC concentrations, pH, and other environmental factors protective of assessment endpoints. The conditions can then be used to identify areas of the site that are associated with potential adverse effects.

The analyses described below constitute a "weight of evidence" approach that requires professional judgement of risk assessors. However, biological responses are often associated with a large amount of variability. Comprehensive quantitative characterization of the relationship between physical parameters and adverse effects could require an exhaustive "research" level analysis that may not be required to support risk management decisions for the Chino ERA.

As noted above, fluvial deposition and aerial transport are the two primary modes of contaminant transport at the site. Analysis of risk will be stratified to evaluate areas affected by each mode separately. A suite of sampling and analysis identified in Table 4-2 will be conducted at each sampling site.

4.5.2.1 Sampling Site Selection

Sampling sites will be selected in the two main types of affected areas: (1) areas affected by airborne deposition or mechanical placement of waste materials and (2) ephemeral drainages. Within each of these main categories, sites will be selected based on the following criteria:

1. Sites will be selected to include major soil types/vegetation types. Information on vegetation and soil types will be derived from range condition analysis conducted for the AOC IA during 1997. Data were not available during preparation of TM-1.
2. Sites will include the range of copper concentrations observed in site samples or other important constituents (other metals, sulfate, or pH) in soil/sediment for each unit.
3. Sites will also include the range of pH in soils/sediments observed in site samples. Where possible, sites will include a range of pH at approximately equal copper concentrations.
4. Sites will be selected to allow general assessment of the spatial extent of exposures/effects by including sites throughout the IA.

Sampling sites will be comprised of rectangular plots with the long axis oriented in a north-south direction, perpendicular to the primary direction of transport in the Smelter and Tailing IUs (west to east) where topography permits. Plots in ephemeral drainages will be perpendicular to the thalweg of the drainage. TM-2 will contain a detailed sampling and analysis plan, including numbers of sampling sites, numbers of samples, sampling methods, DQOs, required analytical detection limits and method requirements, and other factors. TM-2 will also include final identification of sampling sites. The following section describes a process for identification of sampling sites. However, results of soil characterization, vegetation surveys and associated range condition analysis, and a site visit by risk assessors are needed to finalize sampling site selection.

Sampling efforts proposed for gradient locations in upland areas and ephemeral drainages are listed in Table 4-2. The gradient approach is applicable to components of the site where soil profiles are intact. Areas of the site where soil profiles may have been lost due to erosion cannot be included in the gradient evaluation because copper or other metals in surface soils may have been removed with lost soils.

For upland soils, a preliminary evaluation of candidate sampling sites was conducted using surface soil data from sites in the Smelter and Tailing IUs (Figure 4-3). The primary factors used in site classification were copper concentration and pH. The objective was to identify soils that best represent the range of conditions for each parameter. Copper concentrations were used as an indicator of affected soils based on background copper levels established in the AOC Background Report (CMC 1995). Historically, copper is a major constituent of mine wastes, and copper concentrations are elevated in soils and sediments to greater degree than other potential metal contaminants (CMC 1995, WCC 1997). Copper was also associated with the highest HQs for vegetation and other receptors (Tables 3-11 to 3-15). Copper is therefore likely to heavily influence risk estimates. Soil pH was also used because the bioavailability, and consequently toxicity, of many metals increases with decreasing pH (Kabata-Pendias and Pendias 1992). Low soil pH may be the result of sulfide oxidation or SO_2 deposition.

Concentrations of copper are positively correlated with concentrations of cadmium, molybdenum, and lead (Figure 4-4) and, therefore, ranges of these metals will be adequately represented if sampling site selection is based on copper concentrations. Copper concentrations are not well correlated with concentrations of other potential contaminants such as arsenic, chromium, cobalt, manganese, selenium, and zinc. The apparent lack of correlation may be due to different sources, differential transport of the constituents from sites of deposition and/or differential enrichment in the mine waste. Preliminary analysis presented in the AOC Background Report suggest that cadmium, lead, and molybdenum are constituents of mine waste and occur at concentrations that exceed reference concentrations in soil (CMC 1995). Other constituents may not be important or exceed reference concentrations. However, results of RI investigations are needed for further definition of reference concentrations and identification of contaminants for each IU. Site selection will include locations where concentrations of other potential COCs appear to be elevated and exceed SSCs.

Copper concentrations from site data were divided into quartiles, and pH was classified as ≥ 7 or <7. This approach to classifying soils by pH classification was based on two factors: (1) the mean pH for reference areas is approximately 7 (CMC 1995)and (2) solubility of copper in soils generally increases rapidly below pH 7 (Kabata-Pendias and Pendias 1992, Schafer & Associates 1993, Sauvé et al. 1998).

The matrix in Table 4-3 shows how the existing sampling sites are distributed among these classifications. Sampling sites will be selected from candidate sites to represent the range of chemical conditions. To the extent practicable, sampling sites will also be dispersed within the study area, in order to allow assessment of spatial distribution of effects (if any) within the study area. For example, the sampling sites will include locations near Lampbright Draw, at the eastern edge of the current Smelter/Tailing IUs. Soils in this area appear to represent minimal levels of contamination and may represent the eastern extent of the area affected by smelter emissions and/or windblown tailing.

In addition to sites within the IUs, the sampling design will include multiple sites in the reference area(s) The candidate sites will be used only to identify general areas for sampling and so the approach does not depend upon identifying the exact location of the previous sampling. This design was selected over sitewide comprehensive sampling because it combines the gradient approach, which can be used to identify remediation goals, with site-specific sampling at the margins of contaminated areas to help identify the spatial extent of effects.

The relative concentrations of potential COCs in sediments sampled from the Hanover-Whitewater IU differs from that of soils in other IUs. Figure 4-5 shows copper concentration in sediments along the Hanover-Whitewater drainage. Copper concentrations range from approximately 200 to 1,200 mg/kg in channel sediments, and up to 93,000 mg/kg in sediments from tin-can plants. Preliminary analysis presented in the AOC Background Report indicates that concentrations of several other metals are elevated with respect to reference samples. The correlation among metal concentrations with copper in samples is not strong (Figure 4-6). The complex distribution of metals along the drainage may result from the mixed influence from CMC and other historical non-CMC sources along the drainage in addition to sites where two drainages converge, thereby mixing sources. Additional characterization of sediments is planned for the RI for the Hanover-Whitewater IU. This sampling will characterize sediments of different geomorphological classifications (i.e., depositional areas, erosion areas, etc.). Sampling is planned for summer and fall 1998. When available, data from the RI will be used to select sampling sites for the ERA.

4.5.2.2 Data Collection and Uses

Uses of the data presented in Table 4-2 are summarized below. Statistical analyses are identified where applicable. In some cases statistical analysis may not be applicable or required. A detailed sampling and analysis plan which identifies sampling methodology, sample handling protocols, laboratory analyses, will be developed for TM-2.

Samples of soils, vegetation foliage, seeds, invertebrates, and small mammals will be collected from a subset of candidate sites such as those shown in Table 4-3 that represent (1) reference (low metal concentration and normal [high] pH), (2) high metal concentrations and low pH; and (3) high metal concentrations and high pH. These data will be collected primarily for comparison to reference concentrations and assessment of metal uptake and exposure of upper level consumers. If results indicate concentrations in biota that are elevated with respect to reference areas and are associated with risk to consumers, additional sampling may be necessary to further characterize sitewide exposure and risk.

Comparison to reference:

COC concentrations in soil, sediment, and biota samples will be compared to site reference concentrations to determine if site concentrations are elevated. Since ERA sampling sites are selected to represent the range of concentrations, the resulting data sets will not be unbiased and independent. Therefore, strict comparison of site mean concentrations to the reference mean is not applicable and alternative methods of comparing site data to reference area data is required.

Data for individual sampling sites within the study area will be compared to the reference data set using interval and equivalence tests as described by Kilgour et al. for using "normal range" of the reference data (McBride et al. 1993, Kilgour and Somers 1998, Kilgour et al. 1998). This approach avoids the difficulties of interpretation resulting from simple testing for a difference in means (McBride et al. 1993, Kilgour et al. 1998). The approach requires *a priori* definition of a practical significant difference (psd), then testing the probability that the difference between site and reference data exceed the psd. Kilgour et al. (1998) recommend using the normal range to define the psd. The normal range includes the portion of the distribution that includes 95 percent of the reference area population. For a normally distributed population, the normal range is approximately equal to the mean $\pm$ 1.96σ, where σ is the standard deviation. The psd is then set equal to 1.96σ for two-tailed tests. For a one-sided test aimed at determining exceedence of the mean, such as when determining whether a concentration is greater than the reference concentration, the mean + 1.67σ is used. Use of the "95 percent rule" implies decision criteria based on a 5 percent Type I error rate. The power of the test depends primarily on the sample size of the reference area population. Equations associated with the test are presented in Appendix E.

Data from reference area samples will be used to determine acceptable levels for plant community parameters. Species richness, biomass, and range condition are proposed as the parameters to be included as response variables in the analysis. The acceptable levels will be defined as the normal range for the parameter in the reference data, as described in the preceding paragraph (Kilgour et al. 1998). If data are not distributed normally, appropriate transformation will be performed prior to analysis.

Plant community analysis:

Vegetation community data will be used in conjunction with data on soil contamination to determine the metal concentrations and pH that are correlated with areas of adversely affected communities. Of primary importance are factors that affect recruitment of vegetation in areas impacted by mine waste. The initial release of mine waste occurred in the relatively distant past, and has been attenuated for much of the site. However, the altered chemistry of soils has apparently affected the ability of vegetation to recolonize the area. As a result, the plant community analysis will concentrate on factors that may affect recruitment of native species. Information on interaction of metal concentrations and pH is important in estimating risk.

Plant community data will be collected using the Whittaker belt sampling approach (Mueller-Dombois and Ellenberg 1974). Trees and shrubs of each species within the belt will be counted. Canopy cover of herbaceous species and bare ground will be evaluated in 10 one-meter square cells randomly located within the belt. Total species richness will be assessed by surveying the belt for all species present. Herbaceous cover and woody plant stem density will be analyzed using analysis of variance and linear regression procedures.

The range condition analysis conducted by CMC for the soil types in the AOC IA will also be conducted for the ERA sampling sites. Conducting the analysis at each sampling site is necessary because the sampling locations for the AOC IA were based on soil types within the IA, not on specific metal concentration ranges, as is proposed for the ERA. The resulting data will be used to correlate the ERA sampling with the larger site-wide effort.

The objective of the vegetation community analyses is to describe the relationship between effects of mine waste on soil chemistry and plant community structure. The potential for change in plant community structure will be evaluated by identifying the range of metal concentrations and pH associated with community structure that is degraded with respect to reference areas. The range condition analysis for each site will be used, in conjunction with similar data collected by CMC for soil classification units throughout the site, to assess the extent to which a degraded condition may be due to grazing.

Laboratory phytotoxicity testing:

Laboratory phytotoxicity testing is proposed to support field investigations by providing an additional line of evidence for assessing the potential for revegetation in affected areas. Soils collected from the sampling sites will be used in the laboratory tests. Two native plant species will be used to assess the potential interaction of pH, metal concentration, and metal bioavailability. Species will be selected at a later date. A phased approach to phytotoxicity testing is proposed. Samples from the areas of highest metal concentrations and low pH will be tested first. If phytotoxicity is not observed, no further testing is planned. If toxicity is observed, tests will be conducted on each of the samples collected for phytotoxicity testing.

Tests will be conducted by a qualified laboratory according to ASTM protocols (ASTM 1994). For herbaceous species, the tests will evaluate seed germination and early seedling growth. Seeds will be planted in soils from the site and reference sampling areas, as well as laboratory control soils. Tests will be conducted for a duration of 14 days past the time when 50 percent of plants in control soils have germinated (ASTM 1994). Positive control tests will also be conducted using boric acid added to water. Standard endpoints measured during tests include germination rate, survival, shoot length, root length, and dry weight.

The seed germination tests will be conducted to assess ability of seeds to germinate and establish in site soils. Data from the phytotoxicity tests will be used to help determine metal concentrations and pH under which recruitment and growth may be inhibited. Adverse effects will be defined as significant reduction in germination and/or growth relative to reference soils from unaffected parts of the site.

If mesquite or other woody shrub species are used, either plant cuttings or seedlings started in control soils will be transplanted to test soils and grown for 28 days. Endpoints measured for shrub growth will include maximum branch and root length, stem diameter, and wet and dry weights of primary stem branches and leaves without primary stem, and roots.

Metal concentrations in test soils will be used to estimate the NOAEL and LOAEL. The laboratory analyses will include all COCs for vegetation, but analyses will focus on the metals that are present at concentrations that exceed reference values. Toxicological endpoints will be estimated based on estimates of bioavailable metal concentrations (see discussion in the following section).

Plant Effects Levels:

Background

Plant Effects Levels (PELs) will be developed from laboratory phytotoxicity testing and field data to describe the combination of environmental factors that are associated with adverse effects on plant communities. The objective of this task is to develop a set of criteria based on analysis of data from a relatively few sample sites, then use this information to evaluation the potential for phytotoxic risk in other areas of the site.

PELs will be based on common measurements, such as soil pH, total recoverable or water soluble metal concentrations, that will be available or easily obtainable for areas of the site that have not been characterized during the ERA. Metal concentration data that are commonly collected for RIs are based on total recoverable metals (i.e., EPA Method 3050 digestion). Total recoverable metal concentrations are the most abundant type of information available for the Chino Mine site.

However, total recoverable metal concentrations are typically not well correlated with degree of phytotoxicity because this measurement does not reflect the bioavailable fraction of metals in soils that contribute to phytotoxicity (Sauvé et al. 1998). The bioavailable fraction typically is a small fraction of the total metals in soils. Soluble metal and free metal ion activity are typically better correlated with phytotoxicity, but these parameters are controlled by physical and chemical factors, such as pH and total organic carbon, and other physico-chemical factors that influence the balance between adsorption-desorption and dissolution-precipitation reactions in the soil matrix (Sauvé et al. 1996, 1997, 1998; McBride et al. 1997; Jopony and Young 1994). Such factors are naturally variable and may vary independently of total metal concentration. Therefore, the PELs should include a mechanism for adjustment for soil pH, TOC or other factors so the criteria can be applied to differing soil conditions.

Sauvé et al. (1998) used empirical equations that described the relationship between free ion activity of cupric ions (pCu^{2+}) and lead (pPb^{2+}) to evaluate the relationship between free-ion activity and phytotoxic response of various plant species. For example, the following equations were used to estimate pCu^{2+} and pPb^{2+}:

$$pCu^{2+} = (1.4*pH) - (1.7*\log_{10}[Cu_{total}]) + 3.42$$

$$pPb^{2+} = (0.62*pH) - (0.84*\log_{10}[Pb_{total}]) + 6.78$$

The relationships were developed from regression analysis of data from several published investigations for a variety of soil types. Results from Sauvé et al. (1998) indicated that free ion activity was better correlated with degree of phytotoxicity than total metal content. Similar relationships between bioavailable and total metal concentrations have been developed for cadmium and zinc (McBride et al. 1997):

$$\log[Cd]_{sol} = 3.62 - (0.5*pH) + (0.96*\log[Cd]_{total}) - (0.45*\log[TOC])$$

$$\log[Zn]_{sol} = 4.44 - (0.71*pH) + (0.68*\log[Zn]_{total})$$

The toxicity values developed by Sauvé et al. (1998) and shown in Figure 4-7 were developed for soils of higher organic carbon content and different mineralogy than those at Chino. Expression of phytotoxicity is site- and soil-specific and dependent upon natural levels of metals in soils. Concentrations required for phytotoxicity at Chino may be higher than those indicated in Sauvé et al. (1998) because of the natural mineralization of geologic materials. However, the results of Sauvé et al. indicate the feasibility of developing such relationships on a site-specific basis. This approach will be used in some form to evaluate the effect of soils on plants at the site.

The results also show that the copper concentrations required for phytotoxicity must be increased by over 40-fold when pH is increased from pH 6 to 8 (Figure 4-7). The pH of reference area soils for the Hurley IU range from 5.5 to 8 (CMC 1995). Therefore, it is important that any risk or remediation criteria developed for the site include consideration of pH. The importance of pH can also be exploited in developing remediation technologies for site soils.

The approach will be based on empirical analysis of data collected from the Chino Mine site and is not based on assumption regarding mechanism of toxicity or indirect effects. If results of the ERA indicate the need for site remediation, the results are intended to support development of criteria for identifying cleanup options. Development of such criteria may include consideration of other factors such as the total area affected and the relative importance of the affected plant communities. Development of remediation criteria must also consider of factors such as socioeconomic impacts (EPA 1988). The approach for the Chino Mines site has been adapted from similar investigations at other mining sites (ARCO 1994).

Proposed approach

Development of the risk criteria will be based on describing a relationship among soil chemistry parameters and biological response variables that will allow estimation of the metal concentration and pH that result in adverse biological effects. An initial step of the analysis will focus on estimating bioavailable metal concentrations by calibrating the above equations, or describing other empirical relationships for soils at the Chino site. The analyses proposed for soil samples is described in Appendix A. These metals and other factors are the focus of the analysis because of the abundant scientific information that suggest they are most important in influencing toxicity in a given soil type (Chang et al. 1992, Jopony and Young 1994, McBride et al. 1997, Sauvé et al. 1996, Sauvé et al. 1998). Other important environmental factors which may affect plant communities, including soil nutrient concentrations, soil type, and grazing history. These factors will also be included in the analysis.

The initial phase of the evaluation will be an attempt to quantify a relationship using stepwise multiple linear regression (MLR)(McBride et al. 1997). Biological response variables proposed for inclusion in the analyses are phytotoxicity (germination rate and seedling growth), species richness, and biomass. Range condition will also be assessed to control for effects of grazing on plant communities. MLR will be performed for each biological response variable using the environmental factors identified above. The resulting equations will be used to determine the levels of environmental factors that result in values for the response variables that are equal to the acceptable levels calculated from reference data. These values will then be used as PELs. The resulting PELs may be composed of a "sliding scale" in which the value of one parameter changes with another. For example, the metal concentrations that result in plant community effects may decrease with decreasing pH.

Success of the gradient approach does not depend upon obtaining significant results from the MLR analysis. It is possible that MLR will not result in statistically significant correlation based on the modest number of sampling sites proposed. If satisfactory results are not obtained, the same data can be used with other approaches to estimate phytotoxic concentrations. Comparison of results from each site within the IA to the reference data set will be used to determine the sites (if any) that exhibit adverse effects. Methods for these comparisons were described in a previous section. The resulting information can be combined with non-metric cluster analysis and graphic analysis to estimate the conditions that result in adverse effects. For example, the NOAELs or LOAELs resulting from phytotoxicity tests could be used in conjunction with plant community analyses to estimate the conditions resulting in adverse effects.

Areas of Eroded Soils

As noted earlier, soil and vegetation along the eastern edge of Lake 1 and Tailings Pond 1 show signs of phytotoxicity and erosion. Eroded areas could have resulted from loss of vegetative cover due to phytotoxicity from smelter emissions and/or tailings deposited in the area. If so, elevated metal concentrations and other chemical conditions causing phytotoxic effects may no longer exist due to erosion of the surficial materials that may now exist in downstream areas. If this is the case, adverse effects on vegetation may not be correlated with elevated COC concentrations in soils and an approach other than that described for the gradient sampling is necessary to characterize impact and risk.

Sampling at these sites will be aimed at characterizing current conditions that may have inhibited the natural revegetation of the soils. Soil type and structure will be evaluated to determine the extent to which A and O horizons have been lost. Soil samples will be collected from the area and tested for COC concentration, nutrient concentrations, pH, acidity, alkalinity, and total organic carbon. Data will be evaluated to determine if COC concentrations are elevated with respect to reference, and whether nutrients concentrations, pH, and TOC are adequate to support vegetation. This information can then be used to determine whether soils can be amended to encourage vegetative growth, or whether removal and replacement may be necessary for revegetation of the slopes.

Relationship between soil chemistry and COC residues in biota:

This analysis will be conducted to determine the relationship between COC content of biota and concentrations in soils. The analysis will be conducted using two relationships (1) COC concentrations in biota and soils at given sites and (2) the ratio of COC concentration in biota vs. soil (i.e., the BAF), compared to the total concentration in soil. Evaluation of the second relationship is necessary because the ratio of COC concentrations in biota vs. soil often changes with the total concentration in soil. Typically, the BAFs decrease with increasing soil concentration. Relationships will be quantified using model II regression analysis (Sokal and Rholf 1968). The extent to which measures of soil metal concentration are predictive of tissue residues or BAF will be assessed based on correlation coefficients that are "significant" at a Type I error rate of 10 percent or less ($p \leq 0.1$). This information will be used primarily to support exposure estimation for wildlife.

Exposure and risk assessment for wildlife:

Risk of impacts to local wildlife populations from exposure to mine-affected media will be assessed using standard methods for estimating exposure as intake of COCs from environmental media (EPA 1997). Screening-level exposure estimates that were presented in Section 3 were conducted using site (or IU) maximum concentrations of COCs in each abiotic or biotic medium to which a receptor type is exposed. Further analysis will be conducted to determine the likelihood that exposures within the area will exceed the ecotoxicological benchmarks.

Exposures will estimated by calculating intake of COCs by representative receptors using concentrations of COCs in soil/sediment, food, and water as appropriate for a given receptor. Receptors to be evaluated will include the birds and mammals evaluated in Section 3. Exposures will be estimated for populations within the IA. For receptors with individual home ranges that are larger than an individual IU or other area for which exposure is being estimated, data from adjacent areas will be used in estimates. If adjacent areas are outside of potentially affected areas, reference area concentrations will be used to represent that portion of the exposure area, unless other data specific to the site are available. Intakes will be calculated using Equation 1. Intake parameters to be used for representative receptors are presented in Appendix B.

Risk estimates will be provided for individual sampling locations and for areas encompassing multiple sampling locations and IUs. For example, risk estimates will be provided for the combined Tailing and Smelter IUs to assess risk to populations of receptors on the site. Hazard quotients will be used to express risk estimates for individual locations. For larger areas, point estimates of exposure will be developed for the median, 75th, and 95th percentile concentrations in available soil/sediment data to provide risk estimate for a wider range of conditions than just the maximum exposure as was done in the screening level assessment. Available data are expected to include existing data from the AOC Background Report, data from the proposed ERA sampling locations, and additional data to be collected during remedial investigations. Existing sampling locations are evenly distributed throughout the site.

COC concentrations in biota will be calculated from soil/sediment data using site-specific BAFs estimated from co-located soil and biota samples. If the correlation coefficient between BAF and soil concentrations exceeds 0.5, SSCs will be calculated using the regression equation to predict uptake of COCs by biota. If the correlation is lower, the mean BAF will be used and a sensitivity analysis performed to determine the effect of BAF variability on exposure estimates. This approach will result in more representative data on distribution of metals in biota because: (1) data on COC content in soils will be available from more locations than biota and (2) biota samples will be collected from ERA sampling that are selected to represent the range of metal concentrations in soils and, therefore, represent a biased and non-random subsample of the total.

Probabilistic simulation modeling will be used to characterize uncertainty associated with point estimates of exposure (Bartell et al. 1992). The simulation modeling will involve [pseudo-] random sampling of soil metal concentrations 1,000 times from the data distributions using a stratified random, or Latin hypercube, procedure (Iman and Conover 1980, Bartell et al. 1992). Simulations will be conducted using commercially available software (@ Risk, Palisade Corp.). Each randomly sampled soil or biota datum will then used to calculate an estimate of intake (i.e., exposure). The process will result in 1,000 estimates of intake, which will be used to construct a probability density function describing the statistical distribution of intake. The cumulative form of the pdf will then be used to estimate the probability of exceeding the TRV.

In theory, all inputs to the exposure equation could be modeled simultaneously. However, in practice increasing the number of modeled parameters can make the results difficult to interpret. Therefore, only data on chemical concentrations in environmental media will be modeled.

Probabilistic site estimates provide the risk manager with additional information on the relative likelihood of exceeding various benchmarks. This information may be useful in prioritizing areas for remediation or choosing remedial actions.

Statistical distributions for input soil data will be estimated from visual inspection of frequency histograms constructed from original soil data. The histograms will be assigned common distributions (i.e., normal, lognormal) based on their shape and common knowledge of the statistical behavior of specific types of environmental data (Gilbert 1987).

Additional analysis will be conducted for COCs that are associated with exposures that exceed TRVs. The analysis will include calculation of COC concentration in soils that would result in exposures equal to the TRV. These calculated concentrations will be similar to the SSCs, except that they will be calculated for a range of AUFs from 0.1 to 1.0. Examples of such calculations are presented in Figure 4-5. The spatially weighted average (or other descriptor such as percentile) can then be compared to the criteria for varying assumptions about site use. If remediation of a site is deemed necessary, the criteria can also be used to prioritize areas within the site for removal or other action.

4.5.3 Assessment of Risk to Aquatic Ecosystem Components

In addition to characterization of dry sediments in ephemeral stream channels, sampling will be conducted to assess the potential effect of contaminated surface water in stream channels and temporary pools within the IA. Temporary water bodies provide breeding habitat for amphibians and some insects and other invertebrates that require standing water to complete their life cycles. Habitat for breeding amphibians may be a more critical consideration because insects may be capable of breeding in a broader range of water bodies. However, insects may be more sensitive to metal contamination in water or sediments. Contaminants entering temporary pools or stream channels in runoff or leachate could result in toxicity to aquatic organisms.

Results of surveys will be used to determine the locations of temporary and permanent water bodies that may serve as habitat for aquatic life. Identification of suitable water bodies will be based on professional judgement of biologists involved in site surveys and risk assessments. If no such areas are identified, no further analysis is necessary. If suitable sites are identified in the area of potentially contaminated soils and sediments, further analysis will be conducted as follows.

Two lines of evidence will be assessed: (1) evaluation of metal concentrations and water quality of surface waters to determine whether they may cause toxicity to aquatic stages of insects or amphibians and (2) evaluation of presence or absence of aquatic life.

Data needed to assess potential risk in temporary or permanent surface water bodies include:

1. Metal concentrations (total and dissolved) and water quality parameter measurements (i.e., pH, hardness, DO)
2. Metal concentrations in sediments
3. Metal concentrations and pH in sediment and/or water that result in toxicity to aquatic organisms or amphibians
4. Information on the location and size of water bodies within the IA and reference areas that are suitable for amphibian breeding
5. The proportion of the above water bodies that show evidence of use by amphibians (egg masses, tadpoles, presence of adults during breeding season

Metal concentrations in water and sediment will be compared to reference metal concentrations to determine if COC concentrations in site waters exceed those in natural conditions. Concentrations will also be compared to risk-based criteria such as state water quality standards and data from EPA AWQC documents. Information from AWQC documents includes toxicological data for multiple species. This information will be used to estimate risk to the types of organisms that are expected to inhabit temporary or permanent aquatic habitats in the Chino Mine area. Data in AWQC documents typically do not include amphibian species. Therefore, additional information from the scientific literature will be used to determine potentially toxic concentrations.

If COC concentrations do not exceed TRVs, risk will be assumed to be negligible. If aquatic life is absent, and COC concentrations exceed TRVs and reference concentrations, further evaluation may be required. The type and extent of further evaluation will be determined by risk managers and risk assessors. Potential evaluation could include identification of potential sources of elevated COC concentrations, or toxicity testing of water or sediment to determine potential toxicity.

If aquatic organisms are present and COC concentrations exceed TRVs and reference concentrations, the types of aquatic life present will be evaluated to determine if taxa present represent more metal-tolerant species than in reference areas. If amphibian adults, larvae, or eggs are present, development will be monitored to determine the extent of successful metamorphosis. The risk assessors and risk managers will determine by professional judgement if additional analyses are necessary.

The above information may be augmented by use of Frog Embryo Teratogenesis Assay—*Xenopus* (FETAX) which is designed to assess effects on developing amphibians (ASTM 1991). The FETAX bioassay involves exposure of embryos and larvae of the South African clawed toad (*Xenopus laevis*) or other anuran species, to test solutions. The developing organisms are then examined to determine whether the test solutions have induced abnormal development. This species does not occur in the Chino Mines area and is not adapted to arid environments. The assay has been adapted for use with other anuran species (ASTM 1991). If implemented, the teratogenicity assays would also test water from nearby unaffected sites as a reference condition.

4.6 Schedule

The proposed schedule is based on commencement of sampling in 1999. Commencement of sampling in early 1999 is also important to allow time for additional data collection in 2000 if analysis of initial data indicates the need. The schedule is also based on identification of COCs from data available in May 1998. Data from remedial investigations to be conducted in 1998 through 2000 may provide additional information on the types and distribution of contaminants at the site.

5.0 REFERENCES

ARCO. 1994. Regional Ecorisk Field Investigation: Upper Clark Fork River Basin. November.

ASTM. 1991. Standard guide for conducting the frog embryo teratogenesis assay-*Xenopus* (Fetax). In: ATSM Standards on Aquatic Toxicology and Hazard Evaluation. Philadelphia, PA, p. 457-467.

ASTM. 1994. Standard Practice for Conducting Early Seedling Growth Tests. Prepared by the American Society of Testing and Materials. Designation: E1598-94. Philadelphia, PA.

Bartelle, S.M., R.H. Gardner, and R.V. O'Neill. 1992. Ecological Risk Estimations. Lewis Publishers, Boca Raton, FL.

Chang, A.C., T.C. Granato, and A.L Page. 1992. A methodology for establishing phytotoxicity criteria for chromium, copper, nickel, and zinc in agricultural land application of municipal sewage sludges. J. Environ. Contam. 21:521-536.

Conner, J.J. and H.T. Shacklette. 1975. Reference geochemistry of some rocks, soils, plants, and vegetables in the conterminous United States. Geological Survey Professional Paper 574-F.

CMC. 1995. Administrative Order on Consent, Investigation Area Remedial Investigation Background Report, Chino Mine Investigation Area. October.

Dick-Peddie, W.A. 1993. New Mexico Vegetation: Past, Present, and Future. University of New Mexico Press, Albuquerque, NM.

Ebens, R.J. and H.T. Schacklette. 1982. Geochemistry of some rocks, mine spoils, stream sediments, soils, plants, and waters in the Western Energy Region of the Conterminous United States. Geological Survey Professional Paper 1237.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: A Synoptic Review. Contaminant Hazard Review Report USFWS, Vol. 10.

Eisler, R. 1989. Molybdenum hazards to fish, wildlife, and invertebrates: A Synoptic Review. Contaminant Hazard Review Report USFWS, Vol. 19.

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

EPA (U.S. Environmental Protection Agency). 1992a. EPA. Framework for Ecological Risk Assessment. EPA/630/R-92/001; February 1992.

EPA (U.S. Environmental Protection Agency). 1992b. Interim Guidance on Interpretation and Implementation of Aquatic Life Criteria for Metals. May.

EPA (U.S. Environmental Protection Agency). 1993. Handbook of Wildlife Exposure Factors.

EPA (U.S. Environmental Protection Agency). 1994. Data Quality Objectives Process for Superfund. Interim Final Guidance. EPA 540/R/93/01. September.

EPA (U.S. Environmental Protection Agency). 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. June.

EPA (U.S. Environmental Protection Agency). 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. April.

Findley, J.S. 1987. The Natural History of New Mexican Mammals. University of New Mexico Press, Albuquerque, NM.

Findley, J.S., Harris, A.H., Jones, C., Wilson, D.E. 1975. Mammals of New Mexico. Van Nostrand Reinhold, New York, NY.

Frank, A. 1998. Mysterious moose disease in Sweden. Similarities to copper deficiency and/or molybdenosis in cattle and sheep. Biochemical background of clinical signs and organ lesions. Science of the Total Environment. 209:70-26.

Galbraith H., K. LeJeune, and J. Lipton. 1995. Metal and arsenic impacts to soils, vegetation communities, and wildlife habitat in southwest Montana uplands contaminated by smelter emissions: I. Field Evaluation. Environ. Toxicol. Chem. 14(11):1895-1903.. 1995.

Gilbert R.O. 1987. Statistical Methods for Environmental Pollution Monitoring. Van Nostrand Reinhold, New York, NY.

Golder & Associates, Inc.. 1996. Administrative Order on Consent, Remedial Investigation Proposal, Hurley Soil Investigation Unit. April.

Greystone. 1997. Environmental Assessment: Land Application of Wastewater at Closure, Black Pine Mine Project, Cassia County, Idaho. Prepared for USDI BLM Upper Snake River District, Malad Resource Area and USDA Forest Service Sawtooth National Forest, Burley Ranger District. May

Harwell, M., J. Gentile, B. Norton, and W. Cooper. 1994. Issue Paper on Ecological Significance. Prepared for Risk Assessment Forum, U.S. EPA. EPA/630/R-94/009 (November).

Hrudely, S.E., W. Chen, and C.G. Rousseaux. 1996. Bioavailability in Environmental Risk Assessment. CRC/Lewis Press. 294 pp.

Hunt, C.B. 1967. Physiography of the United States. W.H. Freeman and Company, San Francisco.

Iman, R.L. and W.J. Conover. 1980. Risk Methodology for Geologic Disposal of Radioactive Waste: A Distribution-free Approach to Inducting Correlations among Input Variables for Simulation Studies. Technical Report SAND80-0157. Sandia Laboratories, Albuquerque.

Jopony, M. and S.D. Young. 1994. The solid-solution equilibria of lead and cadmium in polluted soils. Eur. J. Soil Sci. 45:59-70.

Kabata-Pendias, A. and H. Pendias. 1992. Trace Elements in Soils and Plants. 2nd ed. Ann Arbor, MI: CRC Press. 365 pp.

Kilgour, B.W., K.M. Somers. 1998. The statistics of testing generic biological criteria. Setac News. July.

Kilgour, B.W., K.M. Somers, and D.W. Matthews. (In press). Using the normal range as a criterion for ecological significance in environmental monitoring and assessment. Ecoscience.

McBride, M., S. Sauvé and W. Hendershot. 1987. Solubility control of Cu, Zn, Cd, and Pb in contaminated soils. Eur. J. Soil Sci. 48:337-346.

McBride, G.B., J.C. Loftis, N.C. Adkins. 1993. What do significance tests really tell us about the environment? Environ. Mgt. 17(4)423-432.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and Methods of Vegetation and Ecology. John Wiley & Sons, New York. 547 pp.

Newmont, M.C. and A.W. McIntosh. 1991. Metal Ecotoxicology: Concepts & Applications. Lewis Publishers, Boca Raton, FL.

NMED. 1995. Personal communication to Tom L. Shelley (Chino Mines Company) from James McDonald (NMED). December 27.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, and G.W. Suter II. 1998. Development and Validation of Bioaccumulation Models for Small Mammals. Prepared for U.S. DOE. ES/ER/TM-219.

Sauvé S., N. Cook, W.H. Hendershot, and M.B. McBride. 1996. Linking plant tissue concentrations and soil copper pools in urban contaminated soils. Environ. Poll. 94(2):153-157.

Sauvé, S., M.B. McBride, W.A. Norvell, and W.H. Hendershot. 1997. Copper solubility and speciation of in situ contaminated soils: effects of copper level, pH and organic matter. Water, Air, and Soil Pollution 100:133-149.

Sauvé S., A. Dumestre, M. McBride, and W. Hendershot. 1998. Derivation of soil quality criteria using predicted chemical speciation of Pb^{2+} and Cu^{2+}. Environ. Tox. Chem. (19)1481-1489.

Schafer & Associates and Reclamation Research Unit, MSU. 1993. Streambank Tailings and Revegetation Studies, Stars Phase III, Final Report, Volume 1. Prepared for Montana Department of Health and Environmental Sciences. November.

Schafer & Associates. 1993. Streambank tailings and Revegetation Studies, Stars Phase III, Final Report, Vol. 1. Prepared for Montana Dept. of Health and Environmental Sciences. November 10.

Schafer 1997. Kinross-DeLamar Mine soil assessment for land application treatment system. Prepared for Kinross-DeLamar Mining Company, Jordan Valley, Oregon. Prepared by Schafer & Associates, Inc., Bozeman, Montana. March 11.

Schafer & Associates. 1998. Chino Mines Administrative Order on Consent, Hurley Soils Investigation Unit Screening-Level Ecological Risk Assessment. Prepared for the New Mexico Environment Department. March.

SCS (Soil Conservation Service). 1983. Soil Survey, Grant County, New Mexico. U.S. Government Printing Office, Washington, D.C.

Shelford, V.E. 1963. The Ecology of North America. University of Illinois Press, Urbana, IL.

Sokal, R.R. and F.J. Rohlf. 1968. Biometry. W.H. Freeman and Company, San Francisco, CA. 776 pp.

SRK. 1998. Administrative Order on Consent, Draft Remedial Investigation Proposal, Smelter Investigation Unit. February.

Thornbury, W.D. 1965. Regional Geomorphology of the United States. John Wiley and Sons, Inc., New York.

Trauger, F.S. 1972. Water Resources and General Geology of Grant County, New Mexico. New Mexico State Bureau of Mines and Mineral Resources, Hydrologic Report No. 2.

WCC. 1997. Administrative Order on Consent, Phase I Ecological Remedial Investigation Proposal, Chino Mine Investigation Area. November.

Williamson, M.A., J.S. Applegarth, and P.W. Hyder. 1994. Snakes, Lizards, Turtles, Frogs, Toads & Salamanders of New Mexico: A Field Guide. Sunstone Press, Santa Fe, NM.

TABLES

Table 2-1

Threatened and Endangered Vertebrate and Plant Species that may Occur in the Vicinity of Chino Mines Operations Area in Habitats Similar to Those Found in the Operations Area[1]

Group/Species	Federal Status	State Status			
		Group 1[2]		Group 2[2]	
		Grant Co.	Luna Co.	Grant Co.	Luna Co.
Amphibians					
Lowland leopard frog (*Rana yavapaiensis*)	Rev. E/T[3]	***			
Reptiles					
Gila monster (*Heloderma suspectum*)	Rev. E/T	***	*		
Green rat snake (*Senticolis triaspis*)	-			**	
Narrowhead garter snake (*Thamnophis rufipunctatus*)	Rev. E/T			***	
Mexican garter snake (*Thamnophis egues*)	Rev. E/T			***	
Birds					
Bald eagle (Haliaeetus *leucocephalus*)	E			***	*
Aplomado falcon (*Falco femoralis*)	E	*	*		
Peregrine falcon (*Falco peregrinus*)	E&T	***	**		
Mexican spotted owl (*Strix occidentalis lucida*)	T				
Lucifer hummingbird (*Calothorax lucifer*)	-			*	
Costa's hummingbird (*Calypte costae*)	-			*	
Gila woodpecker (*Melanerpes uropygialis*)	-			***	
Willow flycatcher (*Empidonax traillii*)	Rev. E/T			***	**
Bell's vireo (*Vireo bellii*)	-			***	**
Gray vireo (*Vireo vicinior*)	-			***	
Abert's towhee (*Pipilo aberti*)	-			***	
Baird's sparrow (*Ammodramus bairdii*)	-			**	**
Common black hawk (*Buteo gallus anthracinus*)	-			***	**
Mammals					
Spotted bat (*Euderma maculatum*)	Rev. E/T			**	

	Federal Status	State Status
Plants		
Parish's alkali grass (*Puccinellia parishii*)	Candidate (C1)[4]	Endangered (E)[5]
New Mexico figwort (*Scrophularia macrantha*)	Candidate (C1)	Endangered (E)
Pinos Altos Mountains flameflower (*Talinum humile*)	Candidate (C2[6]	Endangered (E)
Wright's fishhook cactus (*Mammalaria wrightii*)		Endangered (E)
Grama grass cactus (*Pediocactus papyranthus*)	Candidate (C2)	Endangered (E)

(from Table 9 and Table 10 of Appendix A "Vegetation and Wildlife Report," CMC (1995).

[1] Vertebrate data obtained from the Handbook of Species Endangered in New Mexico, New Mexico Department of Game and Fish (no publication date indicated). Plant data obtained from information supplied in a May 4, 1994, letter and a June 5, 1995, fax from the New Mexico Natural Heritage Program, Univ. of New Mexico.

[2] Group 1 = Endangered taxa whose prospects of survival or recruitment within the state are in jeopardy.
Group 2 = Endangered taxa whose prospects of survival or recruitment within the state are likely to become jeopardized in the foreseeable future. *** = indicated a county where the species is known or is highly likely to occur regularly in recent time (i.e., 1960 or later).
** = indicates a county where the species is known to occur less than regularly, but where regular occurrence is likely in recent time (i.e., 1960 or later). * = indicates a county where the species is known to occur less than regularly and where regular occurrence is unlikely in recent time (i.e., 1960 or later).

[3] E = endangered, T = threatened, Rev. E/T = under review as endangered or threatened.

[4] Taxa for which the USFWS has on file enough substantial information on biological vulnerability and threat(s) to support proposals to list them as endangered or threatened species.

[5] "Endangered plant species," any plant species whose prospects of survival within the state are in jeopardy or are likely, within the foreseeable future, to become jeopardized.

[6] Taxa for which there is some evidence of vulnerability, but for which there are not enough data to support listing proposals at this time.

Table 3-1
Metals Analyzed in Soil/Sediment Samples Included in AOC Background Report

	Investigation Unit			
Analyte	Hanover-Whitewater	Lampbright	Smelter	Tailing
Aluminum	X	X		X
Antimony			X	X
Arsenic	X	X	X	X
Barium	X	X	X	X
Beryllium	X	X		X
Boron	X	X		X
Cadmium	X	X	X	X
Calcium	X	X		X
Chromium	X	X	X	X
Cobalt		X		X
Copper	X	X	X	X
Iron	X	X		X
Lead	X	X	X	X
Magnesium	X	X		X
Manganese	X	X		X
Mercury	X	X	X	X
Molybdenum	X	X	X	X
Nickel	X	X		X
Potassium	X	X		X
Selenium	X	X	X	X
Silver	X	X	X	X
Sodium	X	X		X
Sulfate	X	X	X	
Thallium				X
Vanadium	X	X		X
Zinc	X	X		X

Source: Electronic database provided by Woodward-Clyde Consultants, December 1997 (file = bri_data.csv)

Table 3-2
Summary of Analyte Concentrations in Surface Soil and Sediment of Chino Mines Investigation Area[1]
Baseline Remedial Investigation Samples (CMC 1995)
(units = mg/kg)

Analyte	Mean	Standard Deviation	Minimum Detected Concentration	Maximum	95th Percentile	# Samples with Detectable Concentration
Aluminum	11,915	6,392	4,200	29,400	23,100	73
Antimony	3.3	1.0	2.0	5.9	5.0	39
Arsenic	5.5	7.0	1.1	37.4	18.3	82
Barium	114	49	39	342	198	105
Beryllium	0.5	0.3	0.1	1.8	1.1	61
Boron	4.5	3.6	1.2	28.0	8.6	67
Cadmium	2.2	6.3	0.2	53.3	4.8	95
Chromium	17	14	3	90	47	105
Cobalt	10	5	4	24	21	58
Copper	1,187	7,592	12	93,300	2,185	151
Iron	43,467	26,391	9,660	159,000	99,900	101
Lead	169	425	4	2,730	924	105
Manganese	772	612	134	2,910	2,092	73
Mercury	0.1	0.0	0.1	0.2	0.1	21
Molybdenum	10	13	1	76	33	90
Nickel	12	15	2	128	22	70
Selenium	1.2	1.3	0.1	5.6	4.2	60
Silver	0.2	1.0	-1.7	4.1	2.3	45
Thallium	7.6	1.3	5.6	10.4	10.3	19
Vanadium	34	27	9	147	103	73
Zinc	505	889	21	6,000	1,640	73

[1] Includes data from all IUs except Hurley Soils

Table 3-3

Mean and Maximum Concentrations of Metals Analyzed in Soil/Sediment Samples Included in AOC Background Report

	Mean				Maximum			
	Investigation Unit				*Investigation Unit*			
Analyte	Hanover-Whitewater	Lampbright	Smelter	Tailing	Hanover-Whitewater	Lampbright	Smelter	Tailing
Aluminum	9,238	12,141	na	16,775	23,400	29,400	na	28,400
Antimony	na	na	3	3	na	na	6	5
Arsenic	8	5	3	2	37	14	7	3
Barium	81	128	137	122	153	342	240	199
Beryllium	0.4	1	na	1	2	1	na	1
Boron	4	2	na	6	28	8	na	9
Cadmium	4	1	2	1	53	2	29	1
Calcium	11,741	8,452	na	14,434	26,100	43,800	na	59,000
Chromium	18	10	15	23	90	51	27	48
Cobalt	na	9	na	12	na	22	na	24
Copper	3,434	129	1,150	339	93,300	721	5,240	1,330
Iron	58,700	18,753	na	44,103	159,000	30,300	na	120,410
Lead	436	36	34	18	2,730	171	143	32
Magnesium	3,076	3,493	na	3,620	5,360	8,900	na	6,920
Manganese	1,091	529	na	396	2,910	1,050	na	596
Mercury	0.0	0.1	nd	nd	0.2	0.1	nd	nd
Molybdenum	11	3	12	15	57	12	76	34
Nickel	11	11	na	13	128	36	na	19
Potassium	1,301	1,865	na	3,037	2,220	3,880	na	4,770
Selenium	1	0	nd	2	5	2	nd	6
Silver	0.1	0.2	0.3	nd	4	0.3	0.3	nd
Sodium	282	181	na	140	728	701	na	273
Sulfate	nd	1,950	449	na	99,000	40,300	7,150	na
Thallium	na	na	na	8	na	na	na	10
Vanadium	28	20	na	57	53	43	na	147
Zinc	927	131	na	58	6,000	427	na	82

Source: Electronic database provided by Woodward-Clyde Consultants, December 1997 (file = bri_data.csv)

na = not analyzed

nd = analyzed, but not detected

Table 3-4
Dissolved Concentrations (mg/L) of Analytes in Stormwater Discharge Chino Mines Investigation Area

Average Concentrations

	Lampbright				Sampling Site								
Chemical	LBS1CS	LBS1FM	LBS2CS	LBS2FM	HANMB	HAN0	HAN1	HAN2	HAN12	HAN12A	GRUNER	BRANCH	HWY180
Arsenic	na	na	na	na		na	0.2	0.4	0.2	0.31	na	na	0.040
Cadmium	0.001	0.002	0.001	0.003	0.0	na	0.006	0.02	0.008	0.006	na	na	0.010
Chromium	0.004	0.015	0.004	0.003	0.0	na	0.009	0.01	0.024	0.016	0.011	0.014	0.037
Cobalt	na	na	0.02	na	0.01	na	0.01	0.02	0.02	0.02	0.01	0.034	0.05
Copper	0.1	0.2	0.03	0.1	0.1	0.1	0.04	0.04	0.016	0.02	0.4	0.6	4.6
Iron	0.5	7.0	0.4	1.0	2.0	1.1	0.4	0.6	0.1	0.5	10.6	18.0	45.6
Lead	na	0.2	na	na	0.1	na	0.1	0.1	0.2	0.2	0.2	0.1	0.3
Manganese	0.8	0.9	0.1	0.1	0.4	0.5	0.2	3.0	0.5	0.3	0.9	1.6	2.7
Nickel	0.01	0.04	0.01	0.02	0.03	na	0.02	0.06	0.06	0.03	na	na	0.04
pH	6.9	7.0	7.1	7.6	7.3	7.2	7.4	7.5	7.2	7.7	7.2	6.9	7.0
Sulfate	243.7	159.3	184.4	131.1	453.2	1,230.0	1,459.4	1,596.8	1,500.0	1,533.3	310.0	118.0	183.0
Total Diss Solids	481.8	358.7	442.4	321.3	816.8	2,050.0	2,334.7	2,536.8	2,290.0	2,450.0	523.0	263.0	310.0
Zinc	0.1	0.2	0.0	0.0	0.9	1.6	1.8	4.8	2.8	1.3	0.8	0.9	2.3

Maximum Concentrations

	Lampbright				Sampling Site								
Chemical	LBS1CS	LBS1FM	LBS2CS	LBS2FM	HANMB	HAN0	HAN1	HAN2	HAN12	HAN12A	GRUNER	BRANCH	HWY180
Arsenic	na	na	na	na		na	0.2	0.4	0.4	0.3	na	na	0.040
Cadmium	0.002	0.003	0.001	0.003	0.01	na	0.011	0.2	0.010	0.013	na	na	0.017
Chromium	0.007	0.050	0.005	0.003	0.01	na	0.015	0.02	0.038	0.029	0.011	0.014	0.046
Cobalt	na	na	0.02	na	0.01	na	0.02	0.06	0.04	0.03	0.01	0.034	0.05
Copper	0.3	1.5	0.1	0.3	0.2	0.1	0.2	0.2	0.030	0.1	0.4	0.6	6.4
Iron	5.0	84.0	3.9	8.2	4.5	1.1	2.5	2.6	0.1	2.0	10.6	18.0	53.4
Lead	na	0.2	na	na	0.1	na	0.1	0.2	0.2	0.2	0.2	0.1	0.3
Manganese	3.3	8.3	0.6	0.7	1.3	0.5	0.9	35.0	2.5	1.3	0.9	1.6	2.8
Nickel	0.02	0.08	0.02	0.03	0.03	na	0.02	0.16	0.06	0.04	na	na	0.04
pH	7.6	8.7	8.0	8.7	7.9	7.2	7.9	8.0	7.4	8.2	7.2	6.9	7.1
Sulfate	534	824	361	355	972	1,230	1,820	2,090	1,710	1,690	310	118	220
Total Diss Solids	977	1,260	851	739	1,700	2,050	2,940	4,090	2,490	3,000	523	263	364
Zinc	0.4	0.7	0.1	0.1	2.3	1.6	3.0	38.6	4.8	3.2	0.8	0.9	3.1

Table 3-5

Screening-level Toxicity Reference Values (TRVs) for Plants and Soil Fauna
Chino Mines Investigation Area Ecological Risk Assessment

Chemical	TRV for plants (mg/kg)	Reference	TRV for Soil Organisms (mg/kg)	Reference
Aluminum	730	Will and Suter 1995	2,800	ICF 1979
Antimony	5	Will and Suter 1995	na	
Arsenic	16	Kabata-Pendias & Pendias 1992[1]	60	Will and Suter 1995
Barium[2]	500	Will and Suter 1995	na	
Beryllium	10	Will and Suter 1995	na	
Boron	na		na	
Cadmium	3	Kabata-Pendias & Pendias 1992[1]	9	Donker et al. 1993
Chromium	150	CEPA 1994	0.4	Will and Suter 1995
Cobalt[2]	20	Will and Suter 1995	na	
Copper	46	Kabata-Pendias & Pendias 1992[1]	84	Donker et al. 1993
Iron	na		1,000	ICF 1979
Lead	250	Kabata-Pendias & Pendias 1992[1]	606	Donker et al. 1993
Manganese[2]	500	Will and Suter 1995	na	
Mercury	0.3	Will and Suter 1995	0.1	Will and Suter 1995
Molybdenum	2,200	Kabata-Pendias & Pendias 1992[1]	na	
Nickel	250	CEPA 1994	200	Will and Suter 1995
Selenium	4	Kabata-Pendias & Pendias 1992[1]	70	Will and Suter 1995
Silver	10	Kabata-Pendias & Pendias 1992[1]	na	Will and Suter 1995
Thallium	1	Will and Suter 1995	na	
Vanadium	2	Will and Suter 1995	20	Will and Suter 1995
Zinc[3]	380	Kapustka et al. 1995	864	Donker et al. 1993

[1] TRV is one half of median phytotoxic values from Table 5 of Kabata-Pendias and Pendias (1992); values are concentration in soil.

na = not available

Table 3-6
Toxicity Reference Values used in Calculating Soil Screening Criteria for Wildlife
Chino Mine Investigation Area

	Omnivorous Rodent (Deer Mouse)		Carnivorous Mammal (Coyote)		Ruminant (Mule Deer)	
Chemical	TRV_{NOAEL}	Reference	TRV_{NOAEL}	Reference	TRV_{NOAEL}	Reference
Aluminum	34	Ondreicka et al. 1966	34	Ondreicka et al. 1966	34	Ondreicka et al. 1966
Antimony	0.2	Friberg et al. 1979	0.2	Friberg et al. 1979	0.2	Friberg et al. 1979
Arsenic	0.6	Pershagen & Vahter 1979	0.6	Pershagen & Vahter 1979	0.5	Pershagen & Vahter 1979
Barium	5.1	Perry et al. 1983, Borzelleca et al. 1988	5.1	Perry et al. 1983, Borzelleca et al. 1988	5.1	Perry et al. 1983, Borzelleca et al. 1988
Beryllium	4.3	WHO 1990	4.3	WHO 1990	4.3	WHO 1990
Boron	na	--	na	--	na	--
Cadmium	2.5	Wilson et al. 1941	2.5	Wilson et al. 1941	0.4	Wilson et al. 1941
Chromium	1.8	NAS 1974	20	NAS 1974	20	NAS 1974
Cobalt	1.2	Domingo 1994	1.2	Domingo 1994	1.2	Domingo 1994
Copper	168	Hebert et al. 1993	18	Aulerich et al. 1982	1	Aulerich et al. 1982
Iron	260	NAS 1974	260	NAS 1974	260	NAS 1974
Lead	80	Stowe & Goyer 1971	1.3	Horwitt & Cowgill 1971	5	Horwitt & Cowgill 1971
Manganese	200	NTP 1993	200	NTP 1993	200	NTP 1993
Mercury (inorg)	13	Revis et al. 1989 (cited in ORNL 1996)	1	Aulerich et al. 1974 (cited in ORNL 1996)	13	Aulerich et al. 1974 (cited in ORNL 1996)
Molybdenum	0.3	Schroeder & Mitchner 1971	0.3	Schroeder & Mitchner 1971	0.3	Schroeder & Mitchner 1971
Nickel	40	Ambrose et al. 1976	40	Ambrose et al. 1976	40	Ambrose et al. 1976
Selenium	0.03	Rosenfeld & Beath 1954	0.03	Rosenfeld and Beath 1954	0.1	Rosenfeld and Beath 1954
Silver	65	Walker 1971, Van Vleet 1976	65	Walker 1971, Van Vleet 1976	65	Walker 1971, Van Vleet 1976
Thallium	0.3	Roll & Matthiaschk 1981	0.3	Roll & Matthiaschk 1981	0.3	Roll & Matthiaschk 1981
Vanadium	0.01	Friberg et al. 1979	0.01	Friberg et al. 1979	0.01	Friberg et al. 1979
Zinc	120	Schlicker & Cox 1968	314	Aulerich et al. 1991	30	Aulerich et al. 1991

Table 3-6
Toxicity Reference Values used in Calculating Soil Screening Criteria for Wildlife Chino Mine Investigation Area

Continued

	Ground-feeding Birds (Quail and Junco)		Avian Predator (Harrier)	
Chemical	TRV_{NOAEL}	Reference	TRV_{NOAEL}	Reference
Aluminum	963	Carriere et al. 1986	963	Carriere et al. 1986
Antimony	na	–	na	–
Arsenic	17.5	Stanley et al. 1994	17.5	Stanley et al. 1994
Barium	194	Johnson et al. 1960	208	Johnson et al. 1960
Beryllium	49	Friberg 1979	49	Friberg 1979
Boron	na	–	na	–
Cadmium	1.7	White & Finley 1978	1.7	White & Finley 1978
Chromium (III)	1.3	CEPA 1994	1.3	CEPA 1994
Cobalt	2.5	Friberg 1979	2.5	Friberg 1979
Copper	28	Jackson & Stevenson 1981	28	Jackson & Stevenson 1981
Iron	390	Wiseman 1987	390	Wiseman 1987
Lead	4	Edens & Garlich 1983	4	Edens & Garlich 1983
Manganese	977	Laskey & Edens (1985) (cited in ORNL 1996)	977	Laskey & Edens (1985) (cited in ORNL 1996)
Mercury (inorg)	0.5	Hill & Schaffner 1976 (cited in ORNL 1996)	0.5	Hill & Schaffner 1976 (cited in ORNL 1996)
Molybdenum	3.5	Lepore & Miller 1965	3.5	Lepore & Miller 1965
Nickel	77	Cain & Pafford 1981	77	Cain & Pafford 1981
Selenium	0.3	Heinz et al. 1989	0.3	Heinz et al. 1989
Silver	9	Friberg 1979	9	Friberg 1979
Thallium	2	Hudson et al. 1984	2	Hudson et al. 1984
Vanadium	2.4	Kubena et al. 1986	2.4	Kubena et al. 1986
Zinc	10	Gasaway & Bus 1972	10	Gasaway & Bus 1972

[1] TRVs developed as described in Section 3.1.2. As recommended by EPA, if no NOAEL was available, the LOAEL was divided by 10 to obtain an estimate of the NOAEL.

Table 3-7

Water Quality Criteria Used to Assess Risk to Aquatic Life and Larval Amphibians

All values refer to dissolved (filtered) concentrations; units = µg/L

	NMWQS for Aquatic Life			Livestock
Analyte	Acute	Chronic	Amphibian[3]	Watering[4]
Aluminum	750	87	100	5,000
Antimony	88[1]	30[1]	30	na
Arsenic	360[1]	190[1]	4	200
Barium	na	na	na	na
Beryllium	130	5	na	na
Boron	na	na	na	5,000
Cadmium[2]	14	3	4	50
Chromium[2]	4270	509	3	1,000
Cobalt	na	na	5	na
Copper[2]	50	30	20	500
Iron	na	na	20000	na
Lead[2]	331	13	20000	100
Manganese	na	na	142	na
Mercury	2.4	0.012	0.1	10
Molybdenum	na	na	na	na
Nickel[2]	3593	399	5	na
Selenium	20	2	9	na
Silver[2]	0.92[1]	0.12[1]	4.1	na
Thallium	1400[1]	40[1]	11	na
Vanadium	na	na	na	100
Zinc[2]	297	269	200	25,000

[1] EPA Ambient Water Quality Criteria

[2] Hardness dependent acute and chronic standards
Values shown for hardness = 300 mg/L ($CaCO_3$) based on site data

[3] Source: Harfenist et al. 1989

[4] Livestock watering intended for screening-level assessment of risk to wildlife

Table 3-8
Summary of Bioaccumulation Factors Used in Calculating Soil Screening Criteria

Chemical	Soil-Vegetation		Soil-Terrestrial Invertebrates		Soil-Small Mammals	
Aluminum	0.02	c	na		na	
Antimony	na		na		na	
Arsenic	0.06	a	0.4	a	0.003	d
Barium	na		na		0.06	d
Beryllium	na		na		na	
Boron	na		na		na	
Cadmium	0.14	a	0.6	a	0.3	d
Chromium	0.03	c	na		0.08	d
Cobalt	0.03	c	na		0.02	d
Copper	0.2	a	0.3	a	0.2	d
Lead	0.03	a	0.04	a	0.05	a
Manganese	0.1	c	na		na	
Mercury	na		na		0.05	d
Molybdenum	6	b	1.8	b	0.2	b
Nickel	0.1	c	na		na	
Selenium	0.8	a	0.93	a	0.2	d
Silver	na		na		na	
Thallium	na		na		0.1	d
Vanadium	0.01	c	na		na	
Zinc	0.1	b	1.6	a	0.8	d

na = not available

Sources:
a. Ecological Risk Assessment, North Oquirrh Mountains
b. Lincoln Park Ecological Risk Assessment
c. Toele Army Depot Ecological Risk Assessment

Table 3-9
Literature-based Soil Screening Criteria (SSCs) for Chino Mines Investigation Area (units = mg/kg)

Based on Site Soil Data and Bioaccumulation Factors[1]

	Reference Area Concentrations		Soil Screening Criterion						
Analyte	Mean	UCL_{95}	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Predatory Bird	Omnivorous Rodent	Mammalian Predator	Ruminant	Vegetation
Aluminum	21,500	24,186	90,678	9,068	3,210	164	309	21,573	730
Antimony	3	4	na	na	na	0.5	2	10	5
Arsenic	2.3	3	1,263	126	2,089	7	176	215	16
Barium	118	141	2,156	216	8,303	8	537	255	500
Beryllium	0.7	24	544	54	163	12	39	215	10
Boron	6.6	8	na	na	na			na1	--
Cadmium	0.6	1	84	8	18	16	71	105	3
Chromium	32	40	114	11	42	9	1,719	11,338	150
Cobalt	12.0	14	219	22	187	6	230	680	20
Copper	136	179	1,111	111	424	1,585	736	202	46
Lead	22	25	350	35	181	3,683	154	2,835	250
Manganese	537	596	57,135	5,713	3,257	894	1,818	64,935	500
Mercury	nd (0.1)	--	6	1	23	33	119	650	0.3
Molybdenum	nd (0.6)	--	7	1	53	0.2	12	3	2200
Nickel	16	18	4,503	450	257	100	364	12,987	250
Selenium	1.1	1.3	4	0.4	5	0.1	1	6	4
Silver	nd (0.2)	--	100	10	30	162	591	3,250	10
Thallium	7	8	22	2.2	54	0.5	22	15	1
Vanadium	101	121	245	24	8	0.05	0.1	7	2
Zinc	69	75	573	57	41	352	3,543	9,741	380

[1] Calculated using bioaccumulation factors from other sites.

Shaded cells indicate literature-based soil screening-criterion is less than UCL_{95} of mean for site-specific reference concentration for Hurley Soils IU.

Table 3-10
Comparison of Soil Screening Criteria to Sitewide Minimum Detection Limits for Soils and Sediments
Chino Mines Investigation Area
(units = mg/kg)

Analyte	Minumum DL for Site[1]	Lg. Ground-feeding Bird		Sm. Ground-feeding Bird		Avian Predator		Omnivorous Sm. Mammal		Mammalian Predator		Ruminant		Vegetation	
		SSC	HQ[2]	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ
Aluminum	0.8	90,678	0.0	9,068	0.0	803	0.0	164	0.0	309	0.0	21,573	0.0	730	0.0
Antimony	1.8		--		--		--	1	3.6	2	1.0	10	0.2	5	0.4
Arsenic	0.1	1,263	0.0	126	0.0	418	0.0	7	0.0	176	0.0	215	0.0	16	0.0
Barium	0.1	2,156	0.0	216	0.0	878	0.0	8	0.0	537	0.0	255	0.0	500	0.0
Beryllium	0.1	544	0.0	54	0.0	107	0.0	12	0.0	39	0.0	215	0.0	10	0.0
Boron	1.2		--		--		--		--		--	1	1.2	--	--
Cadmium	0.2	84	0.0	8	0.0	3	0.1	16	0.0	71	0.0	105	0.0	3	0.1
Chromium	0.3	114	0.0	11	0.0	10	0.0	9	0.0	1,719	0.0	11,338	0.0	150	0.0
Cobalt	0.4	219	0.0	22	0.0	300	0.0	6	0.1	230	0.0	680	0.0	20	0.0
Copper	0.3	1,111	0.0	111	0.0	91	0.0	1,585	0.0	736	0.0	202	0.0	46	0.0
Iron	1.7		--		--		--		--		--		--	--	--
Lead	0.1	350	0.0	35	0.0	45	0.0	3,683	0.0	154	0.0	2,835	0.0	250	0.0
Manganese	0.1	57,135	0.0	5,713	0.0	650	0.0	894	0.0	1,818	0.0	64,935	0.0	500	0.0
Mercury	0.1		--		--		--		--		--	650	--	0.3	0.3
Molybdenum	0.6	7.1	0.1	1	0.8	30	0.0	0.2	3.1	12	0.0	3	0.2	2200	0.0
Nickel	2.1	4,503	0.0	450	0.0	50	0.0	100	0.0	364	0.0	12,987	0.0	250	0.0
Selenium	0.1	4	0.0	0.4	0.3	5.0	0.0	0.1	1.2	1.2	0.1	6	0.0	4	0.0
Silver	0.2	100.0	0.0	10	0.0	20	0.0	162	0.0	591	0.0	3,250	0.0	10	0.0
Thallium	4.0	22	0.2	2	1.8	8	0.5	0	8.0	22	0.2	15	0.3	1	4.0
Vanadium	0.3	245	0.0	24	0.0	1	0.5	0.1	5.8	0.1	3.1	7	0.0	2	0.2
Zinc	0.2	573	0.0	57	0.0	41	0.0	352	0.0	3,543	0.0	9,741	0.0	380	0.0

[1] Maximum of concentrations from Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

Table 3-11
Comparison of Soil Screening Criteria to Sitewide Maximum Concentrations in Surface Soils
Chino Mines Investigation Area
(units = mg/kg)

Analyte	Maximum for AOC Area[1]	Reference Area UCL$_{95}$[2]	Lg. Ground-feeding Bird		Sm. Ground-feeding Bird		Avian Predator		Omnivorous Rodent		Mammalian Predator		Ruminant		Vegetation	
			SSC	HQ[3,4]	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ
Aluminum	29,400.0	24,186	90,678	0.3	9,068	3.2 X	3210	9.2 X	164	179.8 X	309	95.1 X	21,573	1.4 X	730	40.3 X
Antimony	5.9	4	na	--	na	--	na	--	1	11.7 X	2	3.3 X	10	0.6	5	1.2
Arsenic	37.4	3	1,263	0.0	126	0.3	2089	0.0	7	5.4	176	0.2	215	0.2	16	2.3
Barium	342.0	141	2,156	0.2	216	1.6	8303	0.0	8	45.6 X	537	0.6	255	1.3	500	0.7
Beryllium	1.8	24	544	0.0	54	0.0	163	0.0	12	0.1 X	39	0.0	215	0.0	10	0.2 X
Boron	28.0	8	na	--	na	--	na	--		-- X		-- X	1	28.0 X	--	--
Cadmium	53.3	1	84	0.6	8	6.4	18	3.0	16	3.3	71	0.7	105	0.5	3	17.8
Chromium	90.4	40	114	0.8	11	7.9 X	42	2.1	9	10.5 X	1,719	0.1	11,338	0.0	150	0.6
Cobalt	24.1	14	219	0.1	22	1.1	187	0.1	6	4.2 X	230	0.1	680	0.0	20	1.2
Copper	93,300.0	179	1,111	84.0	111	839.6 X	424	219.9	1,585	58.9	736	126.8	202	462.9	46	2,028.3 X
Iron	159,000.0	163,842	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Lead	2,730.0	25	350	7.8	35	78.0	181	15.1	3,683	0.7	154	17.7	2,835	1.0	250	10.9
Manganese	2,910.0	596	57,135	0.1	5,713	0.5	3257	0.9	894	3.3	1,818	1.6	64,935	0.0	500	5.8 X
Mercury	0.2	--	6.0	0.0 X	1	0.2 X	23	0.0 X	33	0.0 X	119	0.0 X	650	0.0 X	0.3	0.7 X
Molybdenum	75.9	--	7.1	10.7 X	1	107.4 X	53.0	1.4 X	0	390.0 X	12	6.2 X	3	28.9 X	2200	0.0 X
Nickel	128.0	18	4,503	0.0	450	0.3	257	0.5	100	1.3	364	0.4	12,987	0.0	250	0.5
Selenium	5.6	1.3	4	1.4	0.4	14.0 X	5	1.2	0.1	68.3 X	1.2	4.7 X	6	0.9	4	1.4
Silver	4.1	--	100.0	0.0 X	10	0.4 X	30	0.1 X	162	0.0 X	591	0.0 X	3,250	0.0 X	10	0.4 X
Thallium	10.4	8	22	0.5	2	4.7 X	54	0.2	0	20.9 X	22	0.5	15	0.7	1	10.4 X
Vanadium	147.0	121	245	0.6	24	6.0 X	8	18.4 X	0	2,820.9 X	0	1,539.5 X	7	20.4 X	2	73.5 X
Zinc	6,000.0	75	573	10.5	57	105.3 X	41	144.9 X	352	17.1	3,543	1.7	9,741	0.6	380	15.8

[1] Maximum of concentrations from Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] From Hurley Soils IU (see Table 3-9)

[3] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

[4] An "X" next to the HQ indicates that the UCL$_{95}$ of the mean for reference area soils exceeds the SSC for this receptor group. Reference data from the Hurley Soils IU.

Shaded cells indicate that HQ is ≥ 1.0

Table 3-12
Comparison of Soil Screening Criteria to Concentrations in Soils/Sediment
Hanover-Whitewater IU

Maximum Concentrations (units = mg/kg)

Analyte	Maximum for IU[1]	Reference Area UCL_{95}[2]	Lg. Ground-feeding Bird			Sm. Ground-feeding Bird			Avian Predator			Omnivorous Sm Mammal			Mammalian Predator			Ruminant			Vegetation		
			SSC	HQ[2]		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ	
Aluminum	23,400	24,186	90,678	0.3		9,068	2.6	X	3210	7.3	X	164	143.1	X	309	75.7	X	21,573	1.1	X	730	32.1	X
Antimony	na	4		–	X		–	X	na	–		1	–	X	2	–	X	10	–		5	–	
Arsenic	37	3	1,263	0.0		126	0.3		2089	0.0		7	5.4		176	0.2		215	0.2		16	2.3	
Barium	153	141	2,156	0.1		216	0.7		8303	0.0		8	20.4	X	537	0.3		255	0.6		500	0.3	
Beryllium	2	24	544	0.0		54	0.0		163	0.0		12	0.1	X	39	0.0		215	0.0		10	0.2	X
Boron	28	8	–	–		–	–		na	–		–	–		–	–		1	–	X	–	–	
Cadmium	53	1	84	0.6		8	6.4		18	3.0		16	3.3		71	0.7		105	0.5		3	17.8	
Chromium	90	40	114	0.8		11	7.9	X	42	2.1		9	10.5	X	1,719	0.1		11,338	0.0		150	0.6	
Cobalt	na	14	219	–		22	–		187	–		6	–	X	230	–		680	–		20	–	
Copper	93,300	179	1,111	84.0		111	839.6	X	424	219.9		1,585	58.9		736	126.8		202	462.9		46	2,028.3	X
Iron	159,000	163,842	–	–		–	–		–	–		–	–		–	–		–	–		–	–	
Lead	2,730	25	350	7.8		35	78.0		181	15.1		3,683	0.7		154	17.7		2,835	1.0		250	10.9	
Manganese	2,910	596	57,135	0.1		5,713	0.5		3257	0.9		894	3.3		1,818	1.6		64,935	0.0		500	5.8	X
Mercury	0.2	–	6.0	0.0	X	1	0.2	X	23	0.0	X	33	0.0	X	119	0.0	X	650	0.0	X	0.3	0.7	X
Molybdenum	57	–	7	8.0	X	1	80.2	X	53.0	1.1	X	0	291.4	X	12	4.6	X	3	21.6	X	2,200	0.0	X
Nickel	128	18	4,503	0.0		450	0.3		257	0.5		100	1.3		364	0.4		12,987	0.0		250	0.5	
Selenium	5	1.3	4	1.2		0.4	12.0	X	5	1.1		0.1	58.6	X	1.2	4.0	X	6	0.8		4	1.2	
Silver	4	–	100	0.0	X	10	0.4	X	30	0.1	X	162	0.0	X	591	0.0	X	3,250	0.0	X	10	0.4	X
Thallium	na	8	22	–		2	–	X	54	–		0.5	–	X	22	–		15	–		1	–	X
Vanadium	53	121	245	0.2		24	2.1	X	8	6.6	X	0.1	1,007.5	X	0.1	549.8	X	7	7.3	X	2	26.3	X
Zinc	6,000	75	573	10.5		57	106.3	X	41	144.9	X	352	17.1		3,543	1.7		9,741	0.6		380	15.8	

[1] From Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] From Hurley Soils IU (see Table 3-9)

[3] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

[4] An "X" next to the HQ indicates that the UCL_{95} of the mean for reference area soils exceeds the SSC for this receptor group. Reference data from the Hurley Soils IU.

Shaded cells indicate that HQ is $\geq$ 1.0

Table 3-13
Comparison of Soil Screening Criteria to Concentrations in Soils/Sediment
Lampbright IU

Maximum Concentrations (units = mg/kg)

Analyte	Maximum for IU[1]	Reference Area UCL$_{95}$[2]	Lg. Ground-feeding Bird			Sm. Ground-feeding Bird			Avian Predator			Omnivorous Sm Mammal			Mammalian Predator			Ruminant			Vegetation		
			SSC	HQ[3,4]		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ		SSC	HQ	
Aluminum	29400	24,186	90,678	0.3		9,068	3.2	X	3210	9.2	X	164	179.8	X	309	95.1	X	21,573	1.4	X	730	40.3	X
Antimony	na	4	--	--		--	--		na	--		1	--	X	2	--	X	10	--		5	--	
Arsenic	14.1	3	1,263	0.0		126	0.1		2089	0.0		7	2.0		176	0.1		215	0.1		16	0.9	
Barium	342	141	2,156	0.2		216	1.6		8303	0.0		8	45.6	X	537	0.6		255	1.3		500	0.7	
Beryllium	1.12	24	544	0.0		54	0.0		163	0.0		12	0.1	X	39	0.0		215	0.0		10	0.1	X
Boron	8.4	8		--	X		--	X	na	--			--	X		--	X		--	X		--	X
Cadmium	2.03	1	84	0.0		8	0.2		18	0.1		16	0.1		71	0.0		105	0.0		3	0.7	
Chromium	51.4	40	114	0.5		11	4.5	X	42	1.2		9	6.0	X	1,719	0.0		11,338	0.0		150	0.3	
Cobalt	22.4	14	219	0.1		22	1.0		187	0.1		6	3.9	X	230	0.1		680	0.0		20	1.1	
Copper	721	179	1,111	0.6		111	6.5	X	424	1.7		1,585	0.5		736	1.0		202	3.6		46	15.7	X
Iron	30300	163,842	--	--		--	--		--	--			--	X	--	--		--	--			--	X
Lead	171	25	350	0.5		35	4.9		181	0.9		3,683	0.0		154	1.1		2,835	0.1		250	0.7	
Manganese	1050	596	57,135	0.0		5,713	0.2		3257	0.3		894	1.2		1,818	0.6		64,935	0.0		500	2.1	X
Mercury	0.1	--	6.0	0.0	X	1	0.1	X	23	0.0	X	33	0.0	X	119	0.0	X	650	0.0	X	0.3	0.3	X
Molybdenum	12.4	--	7	1.8	X	1	17.5	X	53.0	0.2	X	0	63.7	X	12	1.0	X	3	4.7	X	2,200	0.0	X
Nickel	35.5	18	4,503	0.0		450	0.1		257	0.1		100	0.4		364	0.1		12,987	0.0		250	0.1	
Selenium	1.7	1.3	4	0.4		0.4	4.3	X	5	0.4		0.1	20.7	X	1.2	1.4	X	6	0.3		4	0.4	
Silver	0.31	--	100	0.0	X	10	0.0	X	30	0.0	X	162	0.0	X	591	0.0	X	3,250	0.0	X	10	0.0	X
Thallium	na	8	22	--		2	--	X	54	--		0	--	X	22	--		15	--		1	--	X
Vanadium	42.8	121	245	0.2		24	1.7	X	8	5.4	X	0	821.3	X	0	448.2	X	7	5.9	X	2	21.4	X
Zinc	427	75	573	0.7		57	7.5	X	41	10.3	X	352	1.2		3,543	0.1		9,741	0.0		380	1.1	

[1] From Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] From Hurley Soils IU (see Table 3-9)

[3] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

[4] An "X" next to the HQ indicates that the UCL$_{95}$ of the mean for reference area soils exceeds the SSC for this receptor group. Reference data from the Hurley Soils IU.

Shaded cells indicate that HQ is $\geq$ 1.0

Table 3-14
Comparison of Soil Screening Criteria to Concentrations in Soils/Sediment
Smelter IU

Maximum Concentrations (units = mg/kg)

Analyte	Maximum for IU[1]	Reference Area UCL_{95}[2]	Lg. Ground-feeding Bird		Sm. Ground-feeding Bird		Avian Predator		Omnivorous Sm Mammal		Mammalian Predator		Ruminant		Vegetation	
			SSC	HQ[2]	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ
Aluminum	na	24,186	90,678	–	9,068	– X	3210	– X	164	– X	309	– X	21,573	– X	730	– X
Antimony	5.9	4		– X		– X	na	–	1	11.7 X	2	3.3 X	10	0.6	5	1.2
Arsenic	6.7	3	1,263	0.0	126	0.1	2089	0.0	7	1.0	176	0.0	215	0.0	16	0.4
Barium	240.0	141	2,156	0.1	216	1.1	8303	0.0	8	32.0 X	537	0.4	255	0.9	500	0.5
Beryllium	na	24	544	–	54	–	163	–	12	– X	39	–	215	–	10	– X
Boron	na	8	–	–	–	–	–	–	–	–	–	–	1	– X	–	–
Cadmium	29.1	1	84	0.3	8	3.5	18	1.6	16	1.8	71	0.4	105	0.3	3	9.7
Chromium	27.1	40	114	0.2	11	2.4 X	42	0.6	9	3.2 X	1,719	0.0	11,338	0.0	150	0.2
Cobalt	na	14	219	–	22	–	187	–	6	– X	230	–	680	–	20	–
Copper	5,240.0	179	1,111	4.7	111	47.2 X	424	12.4	1,585	3.3	736	7.1	202	26.0	46	113.9 X
Iron	na	163,842	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Lead	143.0	25	350	0.4	35	4.1	181	0.8	3,683	0.0	154	0.9	2,835	0.1	250	0.6
Manganese	na	596	57,135	–	5,713	–	3257	–	894	–	1,818	–	64,935	–	500	– X
Mercury[3]	0.1	–	6.0	0.0 X	1	0.1 X	23	0.0 X	33	0.0 X	119	0.0 X	650	0.0 X	0.3	0.3 X
Molybdenum	75.9	–	7	10.7 X	1	107.4 X	53.0	1.4 X	0	390.0 X	12	6.2 X	3	28.9 X	2,200	0.0 X
Nickel	na	18	4,503	–	450	–	257	–	100	–	364	–	12,987	–	250	–
Selenium[3]	0.1	1.3	4	0.0	0.4	0.3 X	5	0.0	0.1	1.2 X	1.2	0.1 X	6	0.0	4	0.0
Silver	0.3	–	100	0.0 X	10	0.0 X	30	0.0 X	162	0.0 X	591	0.0 X	3,250	0.0 X	10	0.0 X
Thallium	na	8	22	–	2	– X	54	–	0	– X	22	–	15	–	1	– X
Vanadium	na	121	245	–	24	– X	8	– X	0	– X	0	– X	7	– X	2	– X
Zinc	na	75	573	–	57	– X	41	– X	352	–	3,543	–	9,741	–	380	–

[1] From Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] From Hurley Soils IU (see Table 3-9)

[3] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

[4] An "X" next to the HQ indicates that the UCL_{95} of the mean for reference area soils exceeds the SSC for this receptor group. Reference data from the Hurley Soils IU.

Shaded cells indicate that HQ is ≥ 1.0

Table 3-15
Comparison of Soil Screening Criteria to Concentrations in Soils/Sediment
Tailing IU

Maximum Concentrations (units = mg/kg)

Analyte	Maximum for IU[1]	Reference Area UCL_{95}[2]	Lg. Ground-feeding Bird		Sm. Ground-feeding Bird		Avian Predator		Omnivorous Sm Mammal		Mammalian Predator		Ruminant		Vegetation	
			SSC	HQ[2]	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ	SSC	HQ
Aluminum	28,400	24,186	90,678	0.3	9,068	**3.1** X	3210	**8.8** X	164	**173.7** X	309	**91.9** X	21,573	**1.3** X	730	**38.9** X
Antimony	5	4		-- X		-- X	na	--	1	**10.6** X	2	**2.9** X	10	0.5	5	**1.1**
Arsenic	3	3	1,263	0.0	126	0.0	2089	0.0	7	0.5	176	0.0	215	0.0	16	0.2
Barium	199	141	2,156	0.1	216	0.9	8303	0.0	8	**26.5** X	537	0.4	255	0.8	500	0.4
Beryllium	1	24	544	0.0	54	0.0	163	0.0	12	0.1 X	39	0.0	215	0.0	10	0.1 X
Boron	9	8	--	--	--	--	na	--	--	--	--	--	1	**9.0** X	--	--
Cadmium	1	1	84	0.0	8	0.1	18	0.1	16	0.1	71	0.0	105	0.0	3	0.4
Chromium	48	40	114	0.4	11	**4.3** X	42	**1.1**	9	**5.6** X	1,719	0.0	11,338	0.0	150	0.3
Cobalt	24	14	219	0.1	22	**1.1**	187	0.1	6	**4.2** X	230	0.1	680	0.0	20	**1.2**
Copper	1,330	179	1,111	**1.2**	111	**12.0** X	424	**3.1**	1,585	0.8	736	**1.8**	202	**6.6**	46	**28.9** X
Iron	120,410	163,842	--	--	--	--	--	--	--	**--**	--	--	--	--	--	--
Lead	32	25	350	0.1	35	0.9	181	0.2	3,683	0.0	154	0.2	2,835	0.0	250	0.1
Manganese	596	596	57,135	0.0	5,713	0.1	3257	0.2	894	0.7	1,818	0.3	64,935	0.0	500	**1.2** X
Mercury[3]	0.1	--	6.0	0.0 X	1	0.1 X	23	0.0 X	33	0.0 X	119	0.0 X	650	0.0 X	0.3	0.3 X
Molybdenum	34	--	7	**4.8** X	1	**48.4** X	53.0	0.6 X	0	**175.8** X	12	**2.8** X	3	**13.0** X	2,200	0.0 X
Nickel	19	18	4,503	0.0	450	0.0	257	0.1	100	0.2	364	0.1	12,987	0.0	250	0.1
Selenium	6	1.3	4	**1.4**	0.4	**14.0** X	5	**1.2**	0.1	**68.3** X	1.2	**4.7** X	6	0.9	4	**1.4**
Silver[3]	0.2	--	100	0.0 X	10	0.0 X	30	0.0 X	162	0.0 X	591	0.0 X	3,250	0.0 X	10	0.0 X
Thallium	10	8	22	0.5	2	**4.7** X	54	0.2	0	**20.9** X	22	0.5	15	0.7	1	**10.4** X
Vanadium	147	121	245	0.6	24	**6.0** X	8	**18.4** X	0	**2,820.9** X	0	**1,539.5** X	7	**20.4** X	2	**73.5** X
Zinc	82	75	573	**0.1**	57	**1.4** X	41	**2.0** X	352	0.2	3,543	0.0	9,741	0.0	380	0.2

[1] From Baseline Remedial Investigation data presented in AOC Background Report (CMC 1995)

[2] From Hurley Soils IU (see Table 3-9)

[3] HQ = hazard quotient. HQs rounded to nearest 0.1 unit

[4] An "X" next to the HQ indicates that the UCL_{95} of the mean for reference area soils exceeds the SSC for this receptor group. Reference data from the Hurley Soils IU.

Shaded cells indicate that HQ is ≥ 1.0

Table 3-16
Summary of Screening-level Exposure and Risk Assessment Sitewide Maximum Concentrations in Surface Soils and Sediment Chino Mines Investigation Area

"X" indicates that maximum concentration in soil exceeds literature-based soil screening criterion

Analyte	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Avian Predator	Omnivorous Sm. Mammal	Mammalian Predator	Ruminant	Vegetation
Aluminum		X	X	X	X	X	X
Antimony	X	X	X	X	X		X
Arsenic				X			X
Barium		X		X		X	
Beryllium							
Boron	X	X	X	X	X	X	X
Cadmium		X	X	X			X
Chromium		X	X	X			
Cobalt		X		X			X
Copper	X	X	X	X	X	X	X
Iron	X	X	X	X	X	X	X
Lead	X	X	X		X		X
Manganese			X	X	X		X
Mercury	X	X	X	X	X	X	
Molybdenum	X	X	X	X	X	X	
Nickel			X	X			
Selenium	X	X	X	X	X		X
Silver							
Thallium		X	X	X			X
Vanadium		X	X	X	X	X	X
Zinc	X	X	X	X	X		X

Table 3-17
Chemicals for which Maximum Concentrations Exceed SSCs in Soils/Sediment

Hanover-Whitewater IU

Analyte	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Avian Predator	Omnivorous Sm Mammal	Mammalian Predator	Ruminant	Vegetation
Aluminum		X	X	X	X	X	X
Antimony	NA	NA	NA	NA	NA	NA	NA
Arsenic				X			X
Barium				X			
Beryllium							
Boron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Cadmium		X	X	X			X
Chromium		X	X	X			
Cobalt	NA	NA	NA	NA	NA	NA	NA
Copper	X	X	X	X	X	X	X
Iron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Lead	X	X	X		X		X
Manganese				X	X		X
Mercury	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Molybdenum	X	X	X	X	X	X	
Nickel				X			
Selenium	X	X	X	X	X		X
Silver							
Thallium	NA	NA	NA	NA	NA	NA	NA
Vanadium		X	X	X	X	X	X
Zinc	X	X	X	X	X		X

Lampbright IU

Analyte	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Avian Predator	Omnivorous Sm Mammal	Mammalian Predator	Ruminant	Vegetation
Aluminum		X	X	X	X	X	X
Antimony	NA	NA	NA	NA	NA	NA	NA
Arsenic				X			
Barium		X		X		X	
Beryllium							
Boron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Cadmium							
Chromium		X	X	X			
Cobalt		X		X			X
Copper		X	X			X	X
Iron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Lead		X			X		
Manganese				X			X
Mercury	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Molybdenum	X	X		X	X	X	
Nickel							
Selenium		X		X	X		
Silver							
Thallium	NA	NA	NA	NA	NA	NA	NA
Vanadium		X	X	X	X	X	X
Zinc		X	X	X			X

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Table 3-17
Chemicals for which Maximum Concentrations Exceed SSCs in Soils/Sediment

(continued)

Smelter IU

Analyte	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Avian Predator	Omnivorous Sm Mammal	Mammalian Predator	Ruminant	Vegetation
Aluminum	NA	NA	NA	NA	NA	NA	NA
Antimony	NSC	NSC	NSC	X	X		X
Arsenic							
Barium		X		X			
Beryllium	NA	NA	NA	NA	NA	NA	NA
Boron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Cadmium		X	X	X			X
Chromium		X		X			
Cobalt	NA	NA	NA	NA	NA	NA	NA
Copper	X	X	X	X	X	X	X
Iron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Lead		X					
Manganese	NA	NA	NA	NA	NA	NA	NA
Mercury	ND	ND	ND	ND	ND	ND	ND
Molybdenum	X	X	X	X	X	X	
Nickel	NA	NA	NA	NA	NA	NA	NA
Selenium	ND	ND	ND	ND	ND	ND	ND
Silver							
Thallium	NA	NA	NA	NA	NA	NA	NA
Vanadium	NA	NA	NA	NA	NA	NA	NA
Zinc	NA	NA	NA	NA	NA	NA	NA

Tailing IU

Analyte	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Avian Predator	Omnivorous Sm Mammal	Mammalian Predator	Ruminant	Vegetation
Aluminum		X	X	X	X	X	X
Antimony	NSC	NSC	NSC	X	X		X
Arsenic							
Barium				X			
Beryllium							
Boron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Cadmium							
Chromium		X	X	X			
Cobalt		X		X			X
Copper	X	X	X		X	X	X
Iron	NSC	NSC	NSC	NSC	NSC	NSC	NSC
Lead							
Manganese							X
Mercury	ND	ND	ND	ND	ND	ND	ND
Molybdenum	X	X		X	X	X	
Nickel							
Selenium	X	X	X	X	X		X
Silver	ND	ND	ND	ND	ND	ND	ND
Thallium		X		X			X
Vanadium		X	X	X	X	X	X
Zinc		X	X				

X = maximum concentration exceeded SSC
NA = not analyzed in samples
NSC = no soil screening criterion
ND = not detected in samples

Table 3-18

Chemicals for which the Maximum Concentrations in Stormwater Samples Exceeded New Mexico State Standard for Ambient Water Quality Criteria[1]

Aquatic life standards are not ARARs for stormwater, but are used here to provide preliminary assessment of potential toxicity to aquatic life inhabiting ephemeral drainages, including amphibians

	Lampbright				Hanover-Whitewater								
	LBS1CS	LBS1FM	LBS2CS	LBS2FM	HANMB	HAN0	HAN1	HAN2	HAN12	HAN12A	GRUNER	BRANCH	HWY180
Arsenic[2]	na	na	na	na	na	na	C-L	C-L	C-L-Amp	A-C-L-Amp	na	na	Amp
Cadmium	na	A-C	na	C	C-Amp	na	C	A-C-Amp	C-Amp	C-Amp	na	na	A-C-Amp
Chromium	Amp	Amp	Amp		Amp	na	Amp	Amp	Amp	Amp	Amp	Amp	Amp
Copper	A-C-L-Amp	A-C-L-Amp	A-C-Amp	A-C-Amp	A-C-Amp	A-C-Amp	A-C	A-C-Amp	Amp	A-C-Amp	A-C-Amp	A-C-L-Amp	A-C-L-Amp
Nickel	Amp	Amp	Amp	Amp	Amp	na	Amp	Amp	Amp	Amp	na	na	Amp
Lead	na	C-L	na	na	C	na	C-L	C-L	C-L	C-L	C-L	C-L	A-C-L
Zinc	C-Amp	A-C-Amp	Amp	Amp	A-C-Amp	A-C-Amp	A-C	A-C-L-Amp	A-C-Amp	A-C-Amp	A-C-Amp	A-C-Amp	A-C-Amp

[1] Only analytes for which maximum concentrations exceeded benchmarks are shown

[2] No NM State standard; maximum concentrations compared to AWQC

A = indicates surface water concentration greater than Acute Aquatic life standard

C = indicates surface water concentration greater than Chronic Aquatic Life standard

L = indicates surface water concentration greater than New Mexico Livestock watering standard

Amp = exceeds amphibian TRV

Data were available for for dissolved (filtered samples) concentrations only. Aquatic life standards are dissolved.

na = not analyzed

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
	<u>Exposure Assessment</u>	
1. **Vegetation Community of Upland Sites**	1. COC concentrations in soils or vegetation do not exceed reference	Distribution of metals in soils and vegetation from site and reference areas
	2. COC concentrations in site soils do not exceed screening level TRVs	Metal concentrations in soils, TRVs for vegetation
	3. Nutrient levels are sufficient to support normal vegetation growth	K, P, NO_2+NO_3 TOC, pH in soils of site and background
	4. What proportion of landscape unit with [metals] in soils exceeding TRV or site-specific risk-based criterion	Distribution of elevated metal concentrations in soils or sediments
	<u>Effects Assessment</u>	
	5. Existing vegetation community at site is not degraded with respect to reference	Vegetation community structure in site and background areas; results of range quality assessment; sites located along gradient of conditions if possible
	6. Are COC concentrations or altered physical conditions in soils inhibiting recruitment?	Vegetation community and phytotoxicity test results for germination, root elongation, seedling growth from gradient of soil conditions
	7. Dose-response relationship exists between toxicity and soil contamination	" "
	8. What proportion of landscape unit(s) with adverse effects?	Spatial distribution of areas exhibiting adverse effects; elevated concentrations
	9. Are habitats in landscape unit fractionated by physical disturbance or chemical contamination?	Mapped distribution of vegetation types, wildlife species that may be restricted to habitat types against metal concentrations

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
	Exposure Assessment	
2 **Vegetation Community of Ephemeral Drainages**	1. COC concentrations in soils/sediments or vegetation exceed reference	Distribution of metals in soils and vegetation from site and reference areas
	2. COC concentrations in site soils exceed screening level TRVs	Metal concentrations in soils, TRVs for vegetation
	3. Dose-response relationship exists between residues and soil contamination	Metal concentrations in soils and plant tissues from co-located sites along gradient of conditions
	4. Nutrient levels are sufficient to support normal vegetation growth	K, P, NO_2+NO_3 TOC, pH in soils of site and background
	5. What proportion of landscape unit has [metals] in soils exceeding TRV or site-specific risk-based criterion?	Distribution of elevated metal concentrations in soils or sediments
	Effects Assessment	
	6. Existing vegetation community at site is not degraded with respect to reference area	Qualitative comparison of species present to unaffected or less affected sites (reference condition may not be available)
	7. COC concentrations are not accumulating in plant tissues	Metal concentrations in soils and plant tissues from gradient of conditions
	8. Are COC concentrations or altered physical conditions in soils inhibiting recruitment?	Phytotoxicity test results for germination, root elongation, seedling growth from gradient of soil conditions
	9. Dose-response relationship exists between toxicity and soil contamination	" "
	10. What proportion of landscape unit(s) with adverse effects?	Distribution of areas exhibiting adverse effects; elevated concentrations
	11. Habitats in landscape unit fractionated by physical disturbance or chemical contamination?	Mapped distribution of vegetation types, wildlife species that may be restricted to habitat types against metal concentrations

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
	Exposure Assessment	
3 **Herbivorous, Insectivorous, and Omnivorous Birds**	1. COC exposure do not exceed TRVs (estimate by habitat type [i.e., upland, ephemeral drainage] and location on site)	COC concentrations in soils, seeds, foliage, invertebrates; TRVs for small and large granivorous, omnivorous, and insectivorous birds; Intake calculations
	2. COC in exposure media do not exceed reference levels	COC concentrations in soils, seeds, foliage from site units and reference area
	3. What soil concentrations are associated with exposures that exceed TRVs?	Correlation between COC concentrations in soils and either (a) concentrations in forage or prey or (b) bioaccumulation factors
	Effects Assessment	
	4. Habitat quality is not degraded in potentially affected areas	Habitat quality (vegetation community structure) in site vs. reference
	5. What portion of landscape unit with [metals] in soils and vegetation exceed risk-based criterion?	Spatial distribution of elevated metal concentrations in sediments, soils, and vegetation in landscape unit(s)

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
	Exposure Assessment	
4 **Raptors**	1. COC exposure do not exceed TRVs (estimate by habitat type [i.e., upland, ephemeral drainage] and location on site)	COC concentrations in soils, invertebrates, small mammals TRVs for raptors; Intake calculations
	2. COC in exposure media do not exceed reference levels	COC concentrations in soils, prey
	3. What soil concentrations are associated with exposures that exceed TRVs?	Correlation between COC concentrations in soils and either (a) concentrations in forage or prey or (b) bioaccumulation factors
	Effects Assessment	
	4. Habitat quality is not degraded in potentially affected areas	Habitat quality (vegetation community structure) in site vs. reference
	5. What portion of landscape unit with [metals] in soils and vegetation exceed risk-based criterion?	Spatial distribution of elevated metal concentrations in sediments, soils, and vegetation in landscape unit(s)
	Exposure Assessment	
5 **Herbivorous, Granivorous, and Omnivorous Small Mammals**	1. COC exposure do not exceed TRVs (estimate by habitat type [i.e., upland, ephemeral drainage] and location on site)	COC concentrations in soils, seeds, foliage, invertebrates; TRVs for small and large granivorous, omnivorous, and insectivorous birds; Intake calculations
	2. COC in exposure media do not exceed reference levels	COC concentrations in soils, seeds, foliage from site units and reference area
	3. What soil concentrations are associated with exposures that exceed TRVs?	Correlation between COC concentrations in soils and either (a) concentrations in forage or prey or (b) bioaccumulation factors

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
	<u>Effects Assessment</u>	
	4 Histopathology is associated with elevated concentrations in tissues	COC concentrations in liver, kidney; Histopathological assessment of tissues
	5 Habitat quality is not degraded on site	Habitat quality (vegetation community structure) in site vs. reference
	6 What portion of landscape unit with [metals] in soils and vegetation exceed risk-based criterion?	Spatial distribution of elevated metal concentrations in sediments, soils, and vegetation in landscape unit(s)
	<u>Exposure Assessment</u>	
6 **Ruminant Wildlife**	1. COC exposure do not exceed TRVs (estimate by habitat type [i.e., upland, ephemeral drainage] and location on site)	COC concentrations in soils, foliage of palatable species; TRVs for ruminants; Intake calculations
	2. COC in exposure media do not exceed reference levels	COC concentrations in soils, seeds, foliage from site units and reference area
	3. What soil concentrations are associated with exposures that exceed TRVs?	Correlation between COC concentrations in soils and either (a) concentrations in forage (b) bioaccumulation factors for uptake soil-forage
	<u>Effects Assessment</u>	
	4. Habitat quality is not degraded on site	Habitat quality (vegetation community structure) in site vs. reference
	5. What portion of landscape unit with [metals] in soils and vegetation exceed risk-based criterion?	Spatial distribution of elevated metal concentrations in sediments, soils, and vegetation in landscape unit(s)

Table 4-1
Summary of Assessment Endpoints for Chino Mine Ecological Risk Assessment

Management Goal:

Prevent or remediate adverse direct or indirect effects on ecological communities or populations of ecological receptors from toxic exposure to chemicals in mine waste

Assessment Endpoint	Risk Hypotheses or Question	Measures
7 **Mammalian Predators**	Exposure Assessment	
	1. COC exposure do not exceed TRVs (estimate by habitat type [i.e., upland, ephemeral drainage] and location on site)	COC concentrations in soils, small mammals; TRVs for mammals; Intake calculations
	2. COC in exposure media do not exceed reference levels	COC concentrations in soils, seeds, foliage from site units and reference area
	3. What soil concentrations are associated with exposures that exceed TRVs?	Correlation between COC concentrations in soils and either (a) concentrations in forage (b) bioaccumulation factors for uptake soil-forage
	Effects Assessment	
	4. Habitat quality is not degraded on site	Habitat quality (vegetation community structure) in site vs. reference
	5. What portion of landscape unit with [metals] in soils and vegetation exceed risk-based criterion?	Spatial distribution of elevated metal concentrations in sediments, soils, and vegetation in landscape unit(s)
9 **Amphibians**	Exposure Assessment	
	1. Metal concentrations in water of breeding areas do not exceed toxicity thresholds for amphibians or aquatic life	Exposure Assessment Data on water quality from temporary and permanent aquatic habitat
	2. COC in exposure media do not exceed reference levels	Data on water quality from temporary and permanent aquatic habitat in reference area
	Effects Assessment	
	3. Determine whether amphibians occur in aquatic habitats to the extent expected	Presence/absence of breeding amphibians in aquatic habitats; site and reference (if available)
	4. Sediment are not toxic to aquatic stages of amphibians	Data on metal content of sediment in temporary and aquatic habitats; sediment toxicity testing if necessary

Table 4-2
Proposed Sampling at Gradient Locations

Tailing and Smelter Investigation Units

Sample Type	Endpoint(s)	Purpose
	Vegetation	
Foliage	COC concentrations	Estimate uptake and exposure herbivores; calculated BAF based on soil concentrations
Seeds	" "	" "
Phytotoxicity test for soils	germination rate, shoot length, root length, biomass	Determine whether soils toxic to native species; use two native species (grass, forb, shrub) typical of mesquite-grassland-cactus association
Range condition	range condition	repeat measure used in landscape study; for each new location--allows for extrapolation to available landscape data
Modified Whittaker community quantification	tree/shrub density, herbaceous cover, taxa richness, taxa similarity.	Compare among sites with varying metal concentrations and pH
	Invertebrates	
Taxa richness	number of species (or taxa)	Comparison among sites in areas of various copper concentrations
Biomass	total biomass of invertebrates	" "
Metal concentrations	COC concentration in whole -body composite samples	Uptake and estimation for risk to insectivorous fauna
	Small Mammals	
Tissue residue analysis (liver, kidney, carcass)	COC concentrations in tissue types	Uptake and estimation of risk to predators; used to determine whether exposure to media results in varying uptake; calculate BAF based on soil concentrations
Histopathology (liver, kidney)	Histopathology of internal organs	Determine whether sublethal effects are occurring

Table 4-2
Proposed Sampling at Gradient Locations

(continued)

Sample Type	Endpoint(s)	Purpose
Soils		
Total metals Surface (0-2") and Subsurface (2-12") Soils	total recoverable COC concentrations in soils	For comparison to background and potential development of remediation criteria; total metals (ICP or GFAA) represents most cost effective for delineating areas for potential clean up
Water soluble metals Surface (0-2") and Subsurface (2-12") Soils	COC concentrations in extractant	Evaluating potential bioavailability of metals; soluble metals may be better predictor of toxicity but not well correlated with total metals among sites
Soil nutrients, pH, and Total Organic Carbon Surface (0-2") and Subsurface (2-12")	Nutrient and TOC concentrations and pH in soil samples	Needed to help explain patterns of vegetation community
Soil/profiles type, texture	Soil characterization	Characterize soil structure. Texture important in assessing relative bioavailability and fate of contaminants (e.g., sorption *vs.* leaching)

Table 4-3

Classification of Sites[1] within the Tailing and Smelter Investigation Units with Regard to Copper Concentrations (mg/kg) and pH in Surface Soil

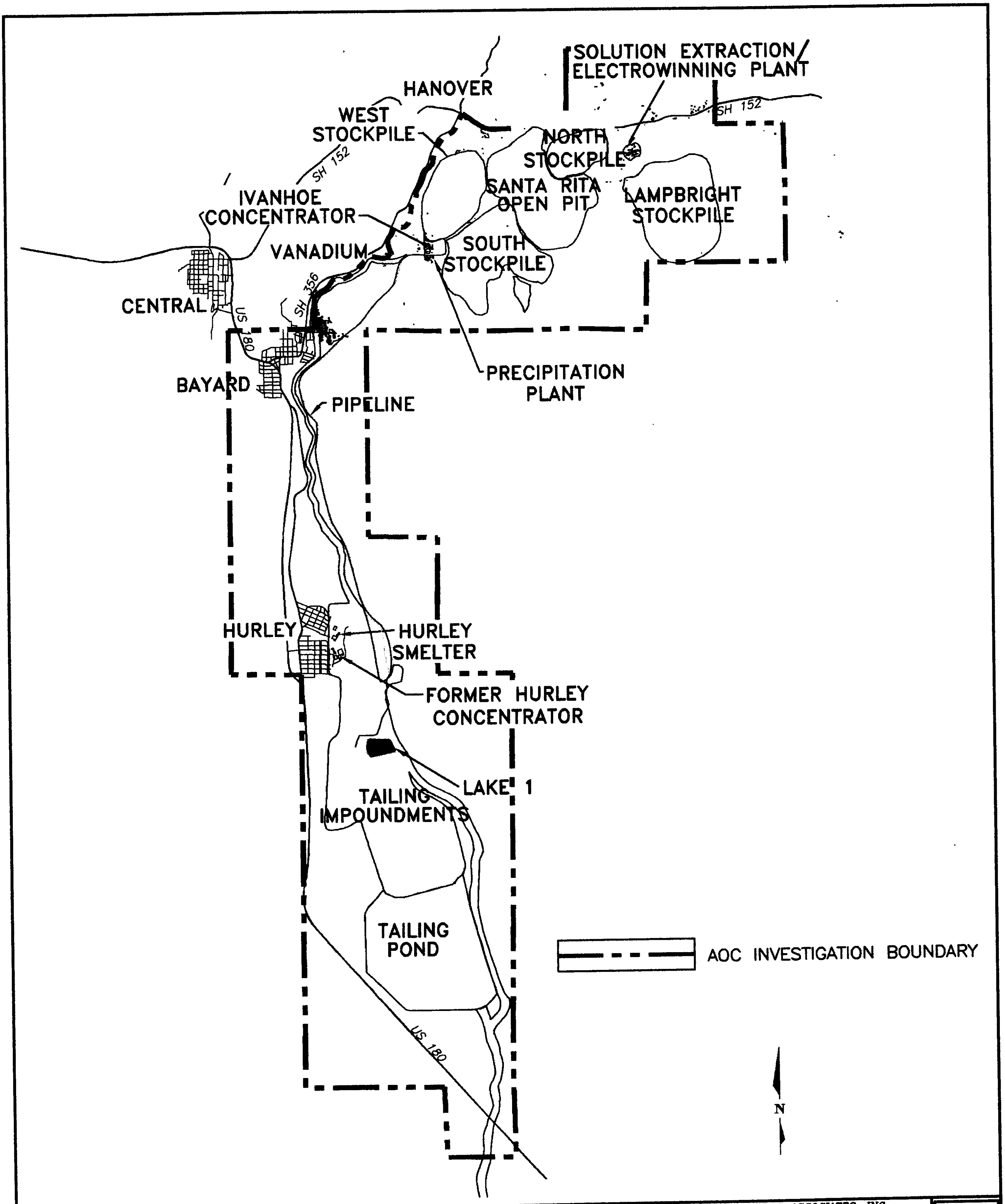
Sample Site ID	pH (<7)				pH (>7)			
	Bottom quartile (43-430)	2nd quartile (431-647)	3rd quartile (648-1280)	4th quartile (>1280)	Bottom quartile (43-430)	2nd quartile (431-647)	3rd quartile (648-1280)	4th quartile (>1280)
U04-1001				X				
U04-1002				X				
U04-1003								X
U04-1004			X					
U04-1007			X					
U04-1008		X						
U04-1009			X					
U04-1010			X					
U04-1011			X					
U04-1012	X							
U04-1013		X						
U04-1014		X						
U04-1015	X							
U04-1016			X					
U04-1017	X							
U04-1018	X							
U04-1019	X							
U04-1020		X						
U04-1021	X							
U04-1022				X				
U04-1023								X
U04-1024								X
U04-1025				X				
U04-1028				X				
U04-1029					X			
U04-1030							X	
U04-1031				X				
U04-1032							X	
U04-1033						X		
U04-1034					X			
U06-3001							X	
U06-3003					X			
U06-3007				X				
U06-3008		X						
U06-3012		X						
U06-3013		X						
U06-3020		X						
U06-3022		X						
U06-3025					X			

Reference Sites

U06-3015	X							
U06-3016	X							
U06-3024					X			
U06-3026					X			
U06-3028	X							
U06-3030					X			
U06-3031					X			

[1] includes only surface soil sites with data on both copper and pH

FIGURES



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Checked:	M.L.
Date:	10-6-98
AOC Area-schafer-1-	

FIGURE 1-1

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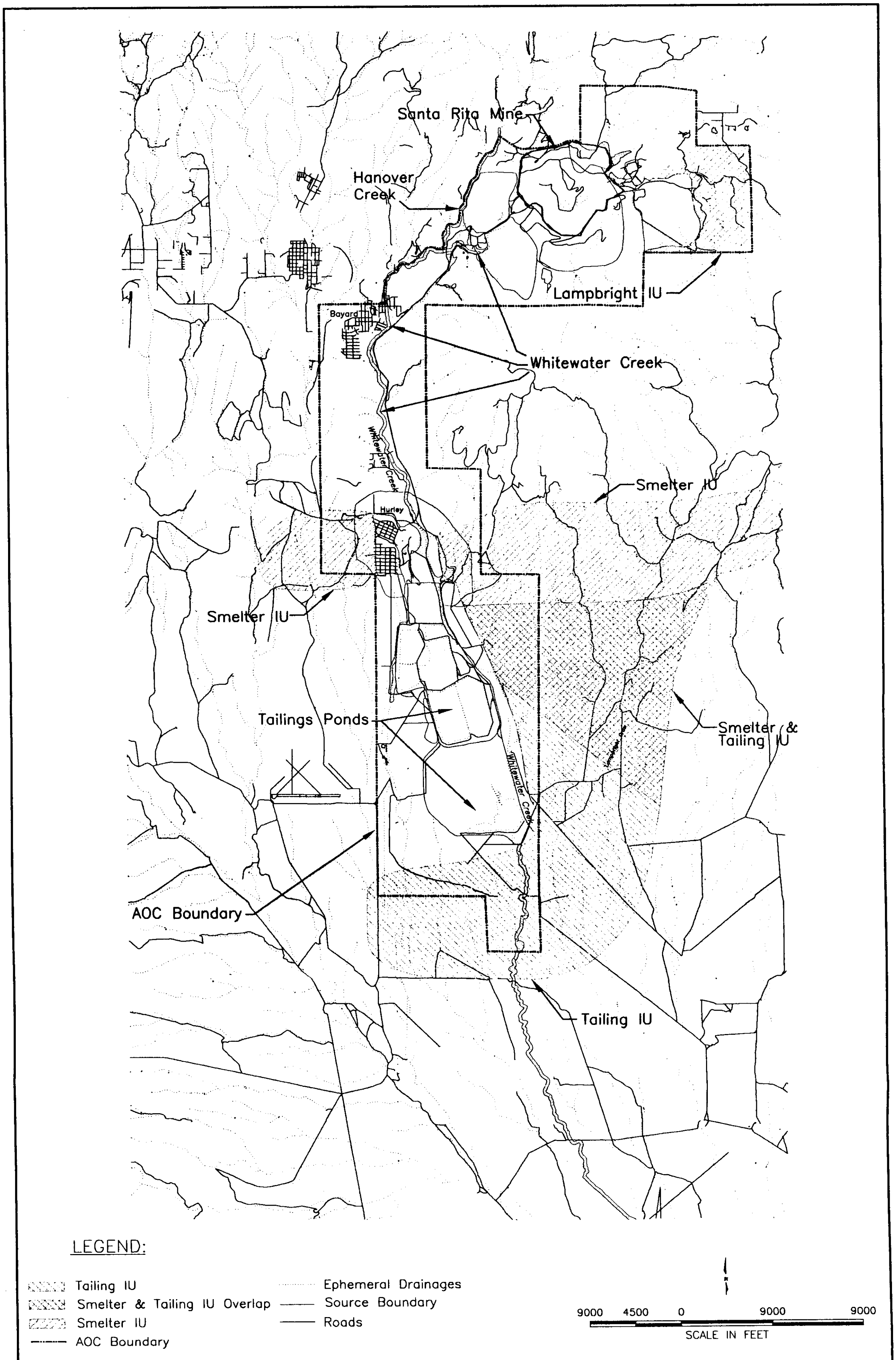
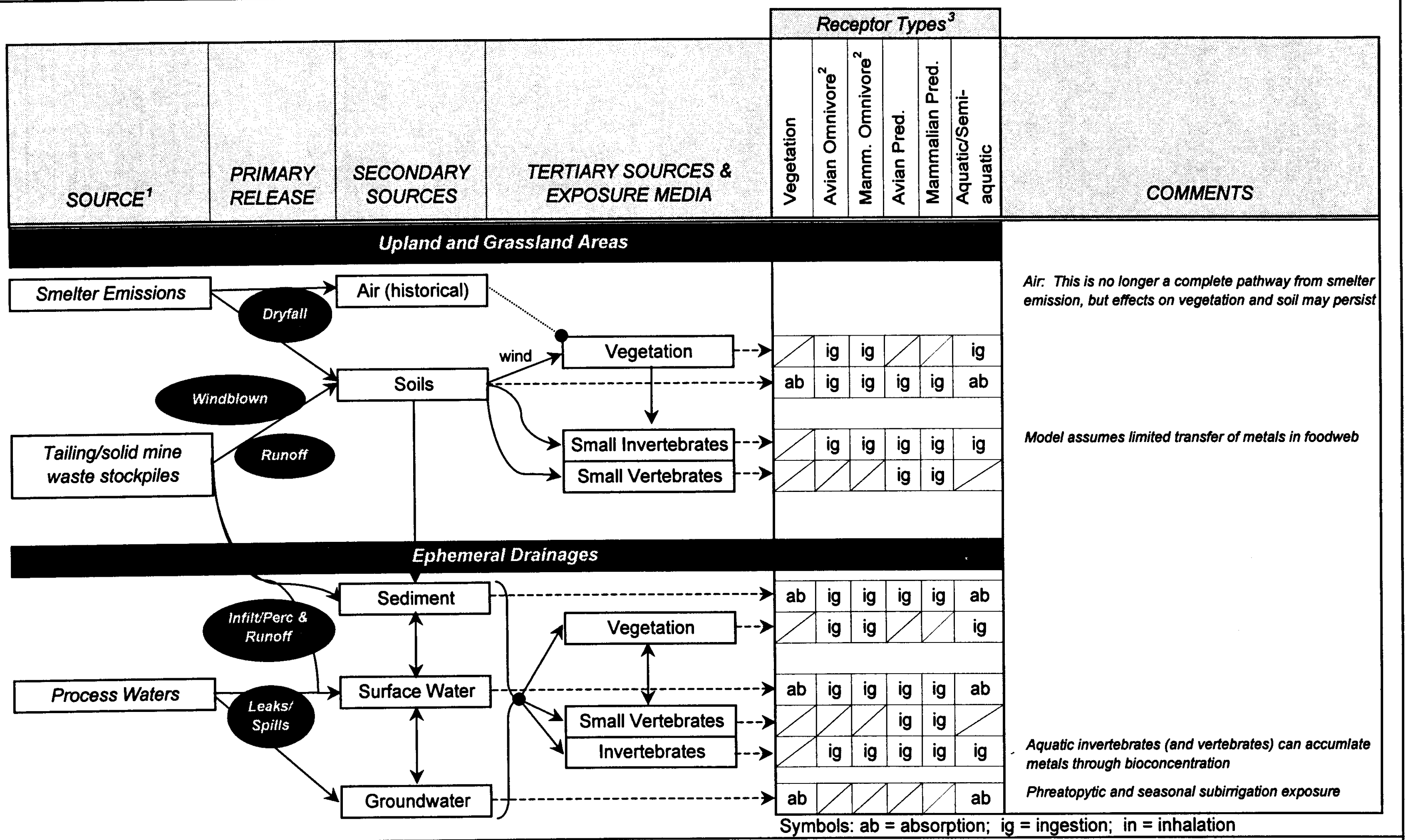
GENERAL SITE CONFIGURATION
CHINO MINES
INVESTIGATION AREA

SCHAFER & ASSOCIATES, INC.
ENVIRONMENTAL SOLUTIONS FOR WATER, WASTE AND LAND
801 14th STREET
GOLDEN, CO 80401
Phone (303) 216-1600 Fax (303) 216-1316

Schafer



Drawn: T.M.G. Designed: M.C.L. Checked: M.C.L. Date: 10-6-98 AOCsources-schafer1-2	**FIGURE 1-2** ISSUED FOR CLIENT REVIEW. Revision: REV 1	**APPROXIMATE** **FIELD INVESTIGATION AREAS** **CHINO MINES INVESTIGATION AREA**	**SCHAFER & ASSOCIATES, INC.** ENVIRONMENTAL SOLUTIONS FOR WATER, WASTE AND LAND 801 14th STREET GOLDEN, CO 80401 Phone (303) 216-1600 Fax (303) 216-1316 Schafer



Exposure media	Vegetation	Avian Omnivore[2]	Mamm. Omnivore[2]	Avian Pred.	Mammalian Pred.	Aquatic/Semi-aquatic	COMMENTS
Upland and Grassland Areas							
Air (historical)							*Air: This is no longer a complete pathway from smelter emission, but effects on vegetation and soil may persist*
Vegetation		ig	ig			ig	
Soils	ab	ig	ig	ig	ig	ab	
Small Invertebrates		ig	ig	ig	ig	ig	*Model assumes limited transfer of metals in foodweb*
Small Vertebrates				ig	ig		
Ephemeral Drainages							
Sediment	ab	ig	ig	ig	ig	ab	
Vegetation		ig	ig			ig	
Surface Water	ab	ig	ig	ig	ig	ab	
Small Vertebrates				ig	ig		
Invertebrates		ig	ig	ig	ig	ig	*Aquatic invertebrates (and vertebrates) can accumlate metals through bioconcentration*
Groundwater	ab					ab	*Phreatopytic and seasonal subirrigation exposure*

Symbols: ab = absorption; ig = ingestion; in = inhalation

Schafer

Figure 2-1. Conceptual Site Model for Exposure of Ecological Receptors Chino Mines ERA

[1] Includes CMC and non-CMC historical sources as identified in AOC Background Report and RI Proposals

[2] Includes herbivores and assumes most omnivores do not ingest vertebrate prey

[3] Characterisitic food web for the site is shown in Figure 2-2

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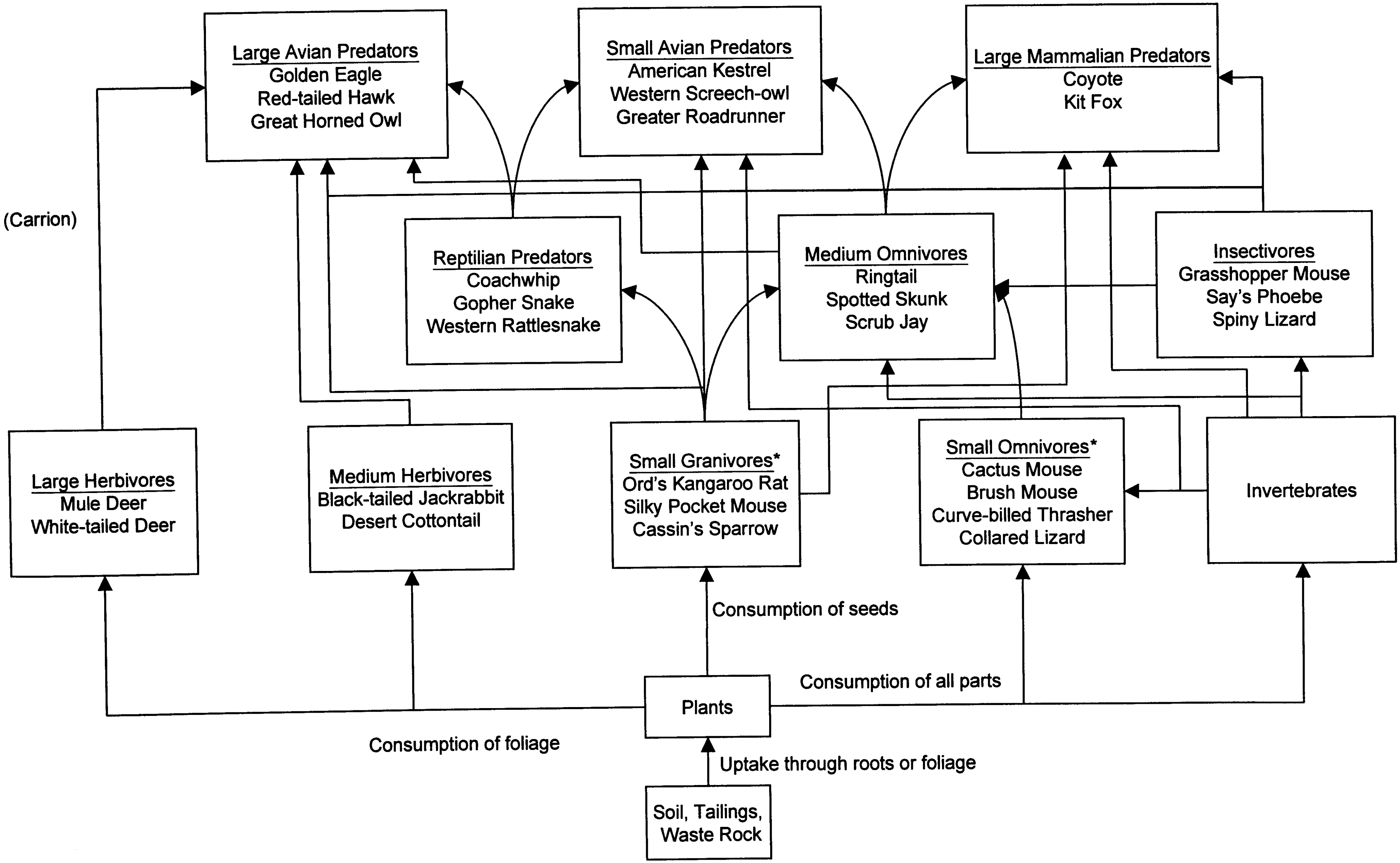
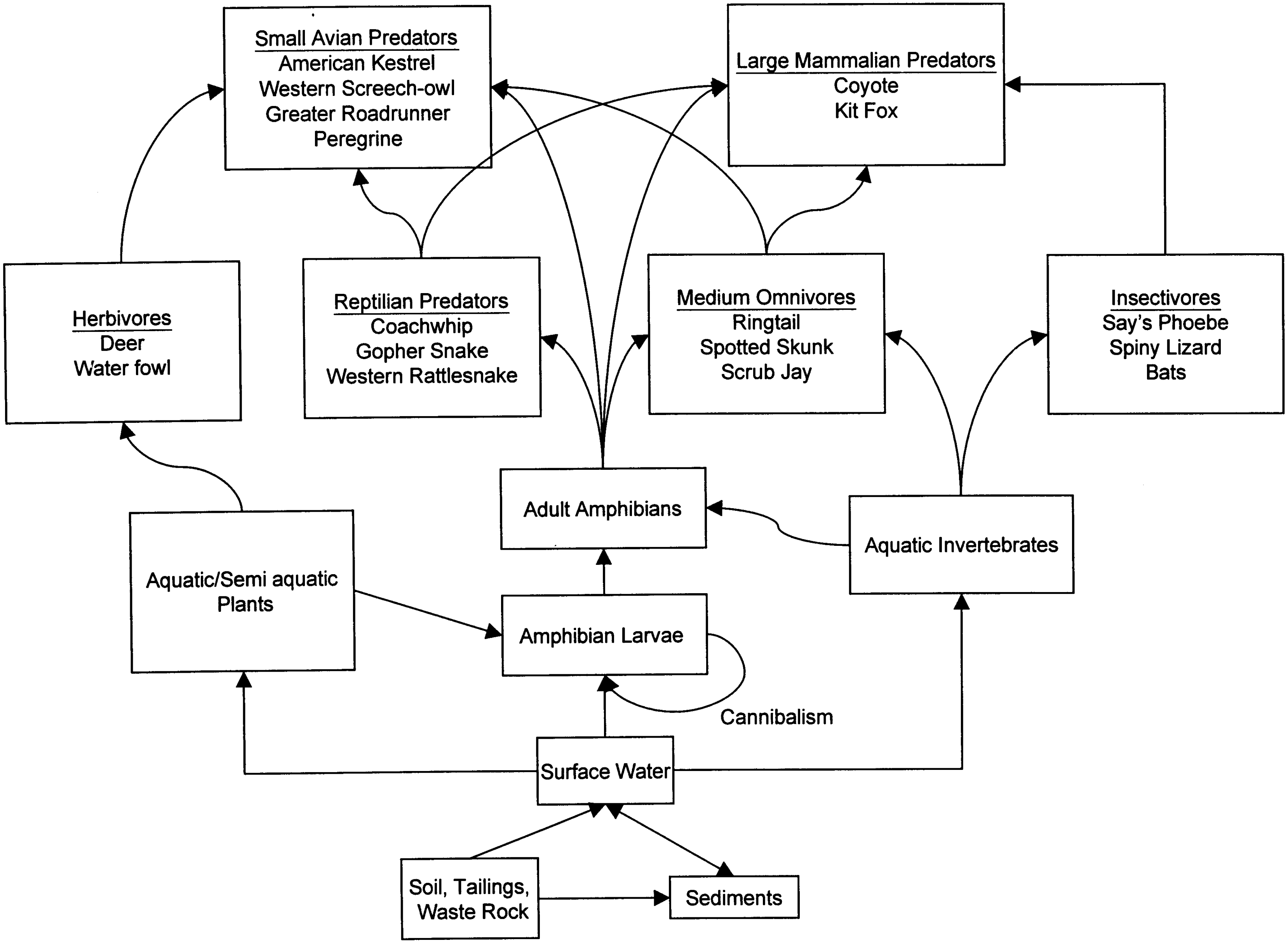
Figure 2-2
Terrestrial Conceptual Food Web



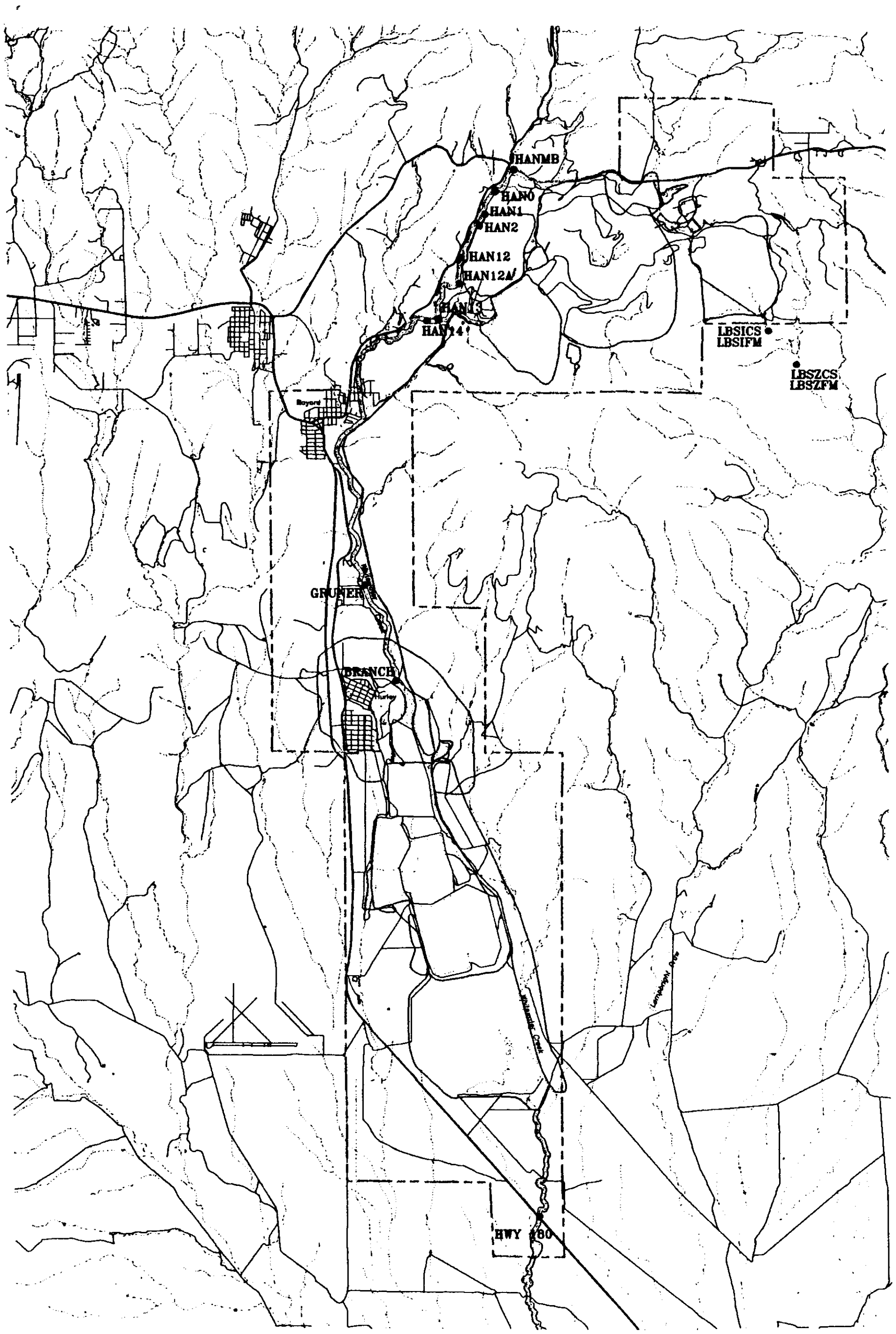
*Species assemblage determined by substrate and vegetation

ChinoERA\Foodweb.ppt\8/26/97

Figure 2-2 (cont'd)
Aquatic Conceptual Food Web



*Species assemblage determined by substrate and vegetation



LEGEND:

- ● **HAN1** Rainstorm Flow Sampler Location and Designation
- ——— Source Boundary
- – – – AOC Boundary
- ········ Ephemeral Drainages
- ——— Roads

N

9000 4500 0 9000 18000

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FIGURE 3-1

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RAINSTORM FLOW SAMPLER LOCATIONS
HANOVER-WHITEWATER & LAMPBRIGHT IUs

SCHAFER & ASSOCIATES, INC.
ENVIRONMENTAL SOLUTIONS FOR WATER, WASTE AND LAND
801 14th STREET
GOLDEN, CO 80401
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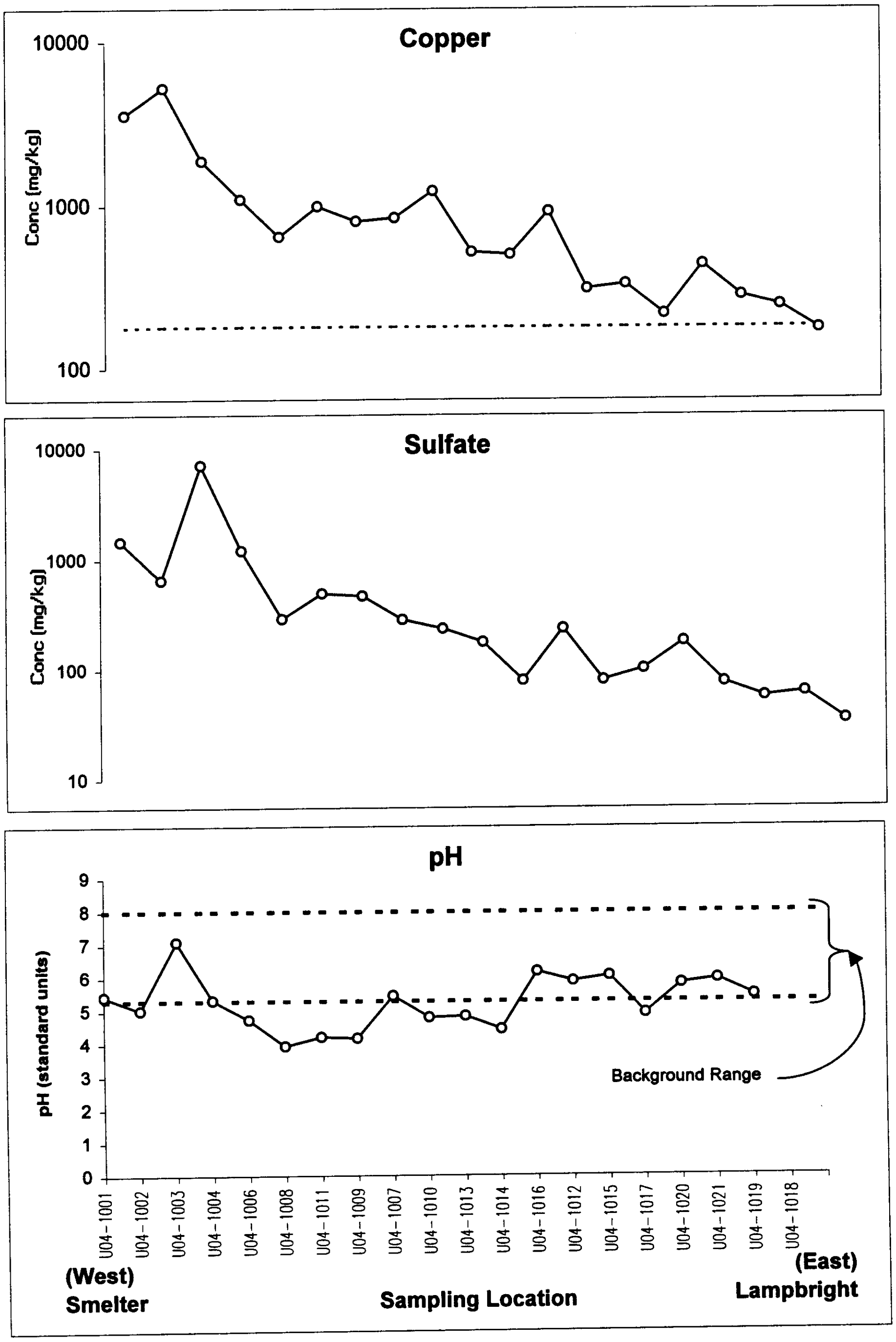
Figure 4-1

Copper, Sulfate, and pH in Surface Soils at Locations from Smelter East to Lampbright Draw

Horizontal lines indicate UCL95 for mean of background concentrations[1]



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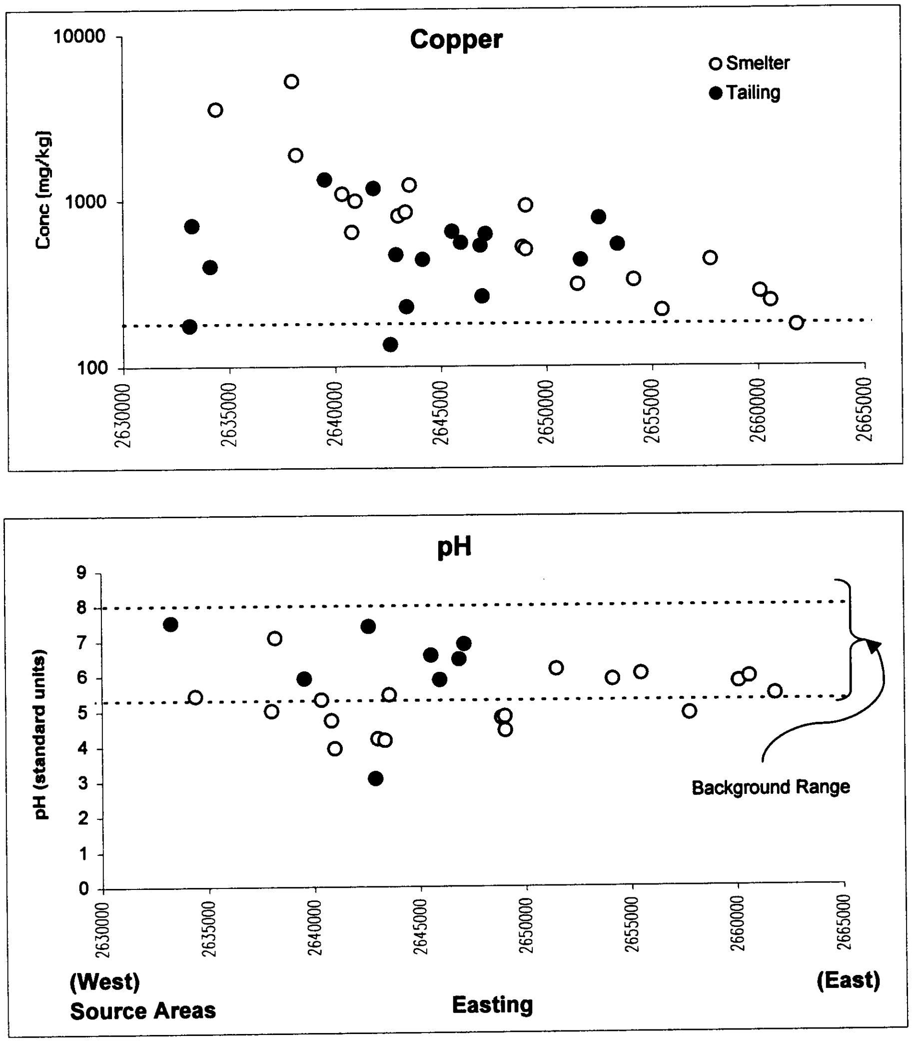
[1] Background data from reference samples reported for Hurley Soils Investigation Unit, Table 6.4-6 of AOC Background Report (CMC 1995)

Figure 4-2
Copper and pH in Surface Soil Samples from Smelter and Tailing IUs

Horizontal lines indicate UCL95 for mean of background concentrations[1]



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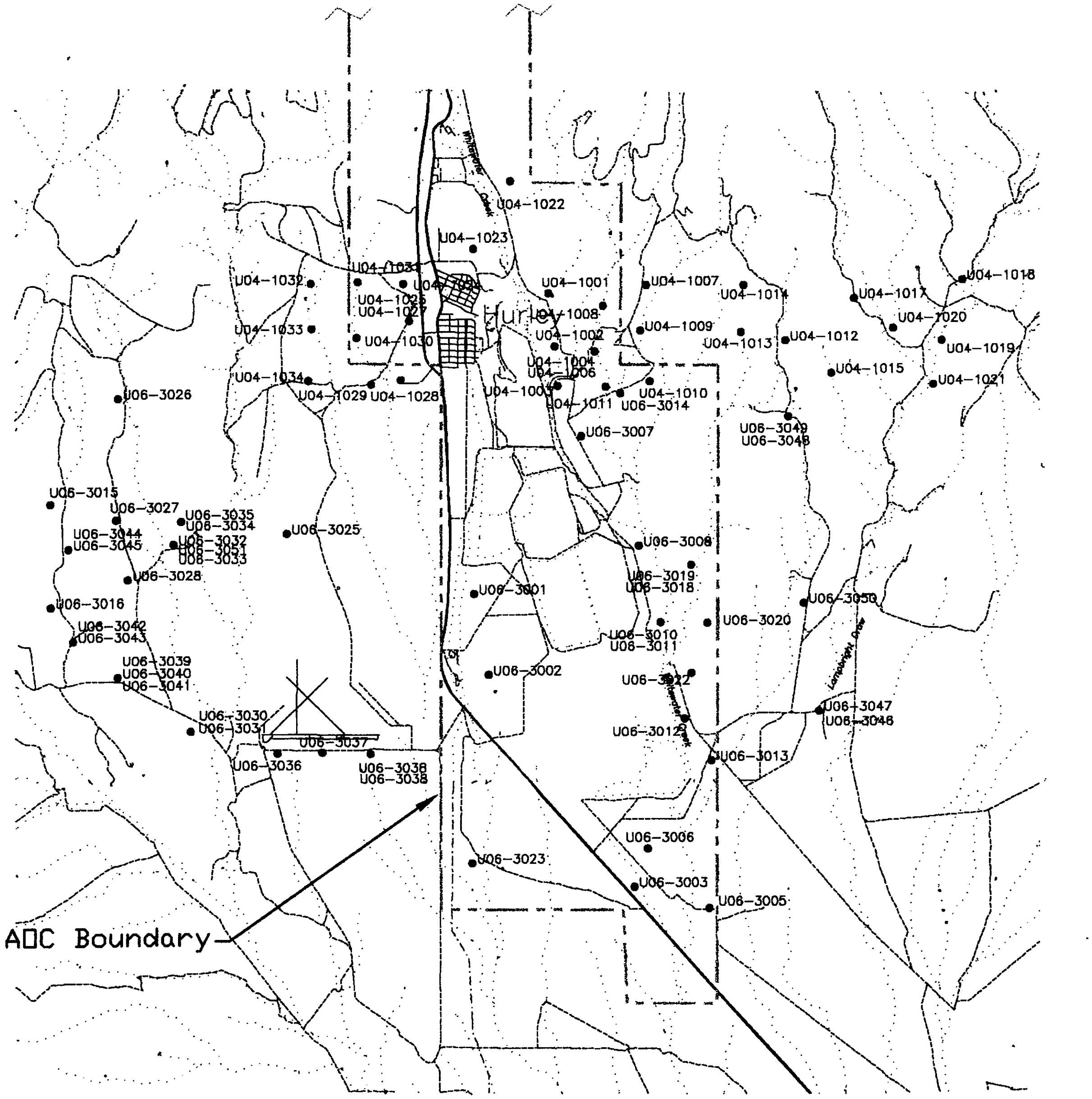
[1] Background data from reference samples reported for Hurley Soils Investigation Unit, Table 6.4-6 of AOC Background Report (CMC 1995)



LEGEND:

Source: AOC Background Report (CMC 1995)

● U06-3022 Soil Sampling Location and Designation

– – – – AOC Boundary

· · · · · · · Ephemeral Drainages

——— Roads

N

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Date: 10-6-98
AOC Area-schafer4-3

FIGURE 4-3

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Revision: REV 1

SOIL SAMPLING LOCATIONS SMELTER AND TAILING INVESTIGATION UNIT

SCHAFER & ASSOCIATES, INC.
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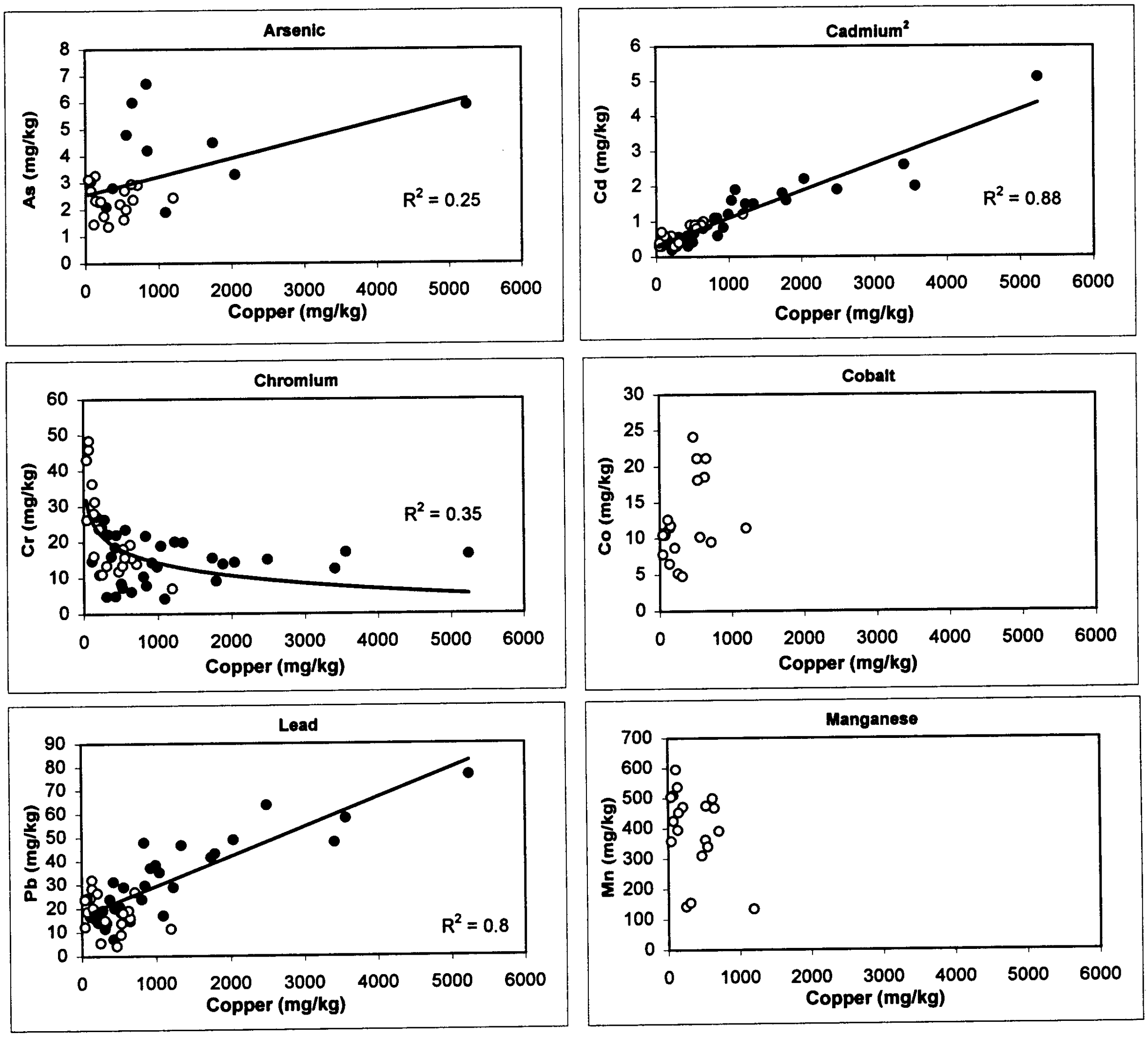
Figure 4-4

Relationship Between Copper and Concentrations of other Metals in Surface Soils from Tailing and Smelter Investigation Units

Open symbols for Tailing IU, Closed symbols for Smelter IU; trendlines for combined Tailing and Smelter data [1]



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[1] Data from Smelter IU does not include all analytes

[2] Does not include data from site 1003 in the Smelter IU. Sample from this site had anomolously high concentrations of cadmium, lead, and sulfate. Site and vicinity of 1003 requires further investigation

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Figure 4-4 (continued)
Relationship Between Copper and Concentrations of other Metals in Surface Soils from Tailing and Smelter Investigation Units

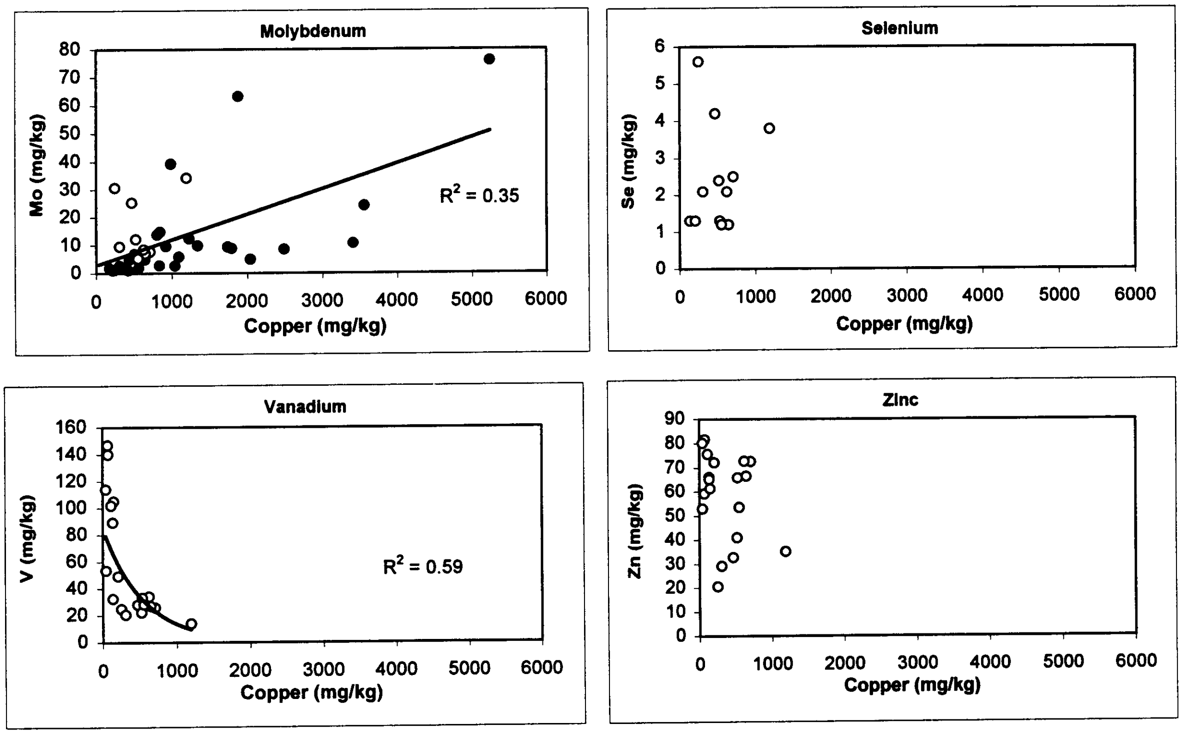
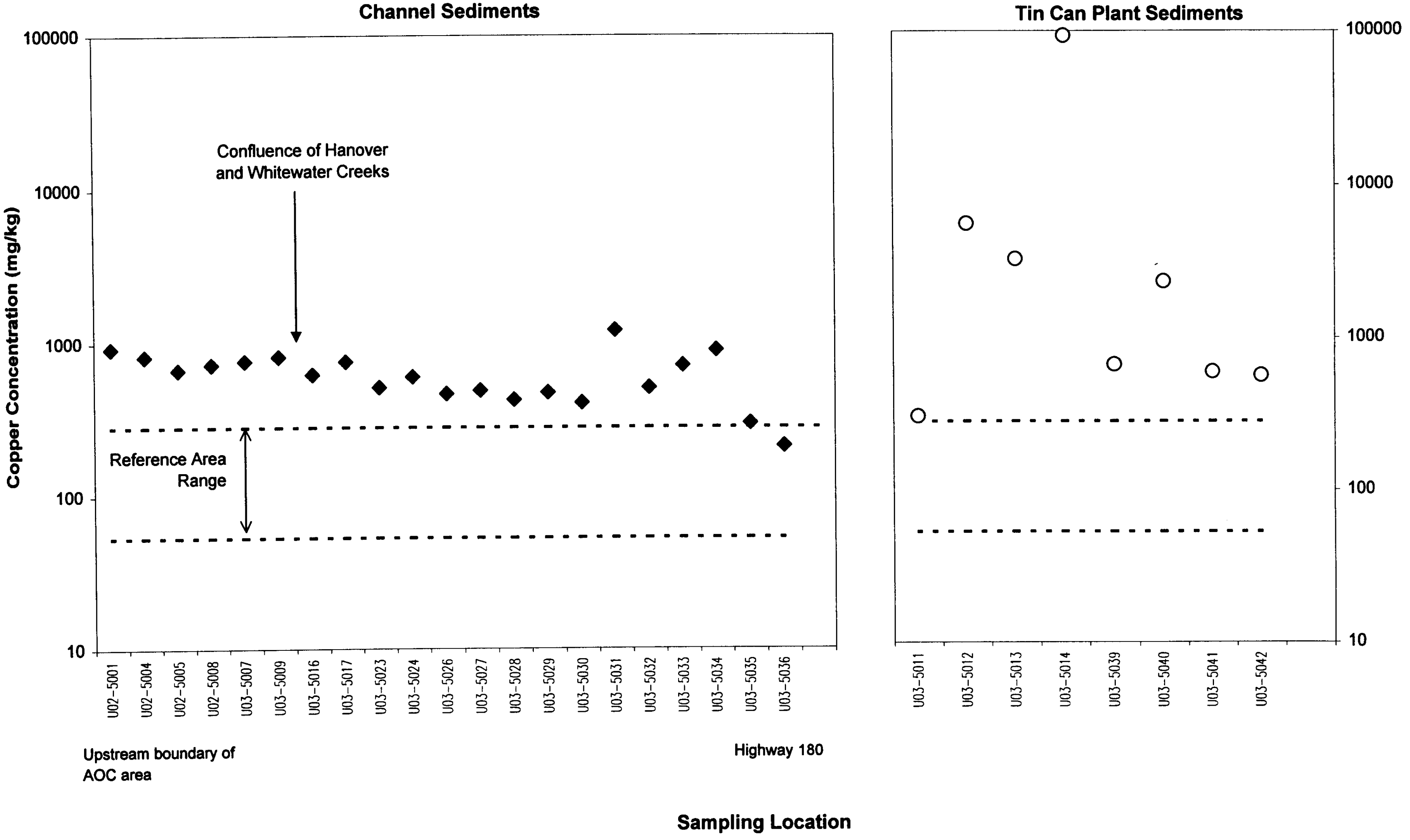


Figure 4-5

Summary of Copper Concentrations in Hanover-Whitewater Creek IU Sediments[1]



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[1] Sediment sampling locations from Figure 6.2-5 in AOC Background Report (CMC 1995)

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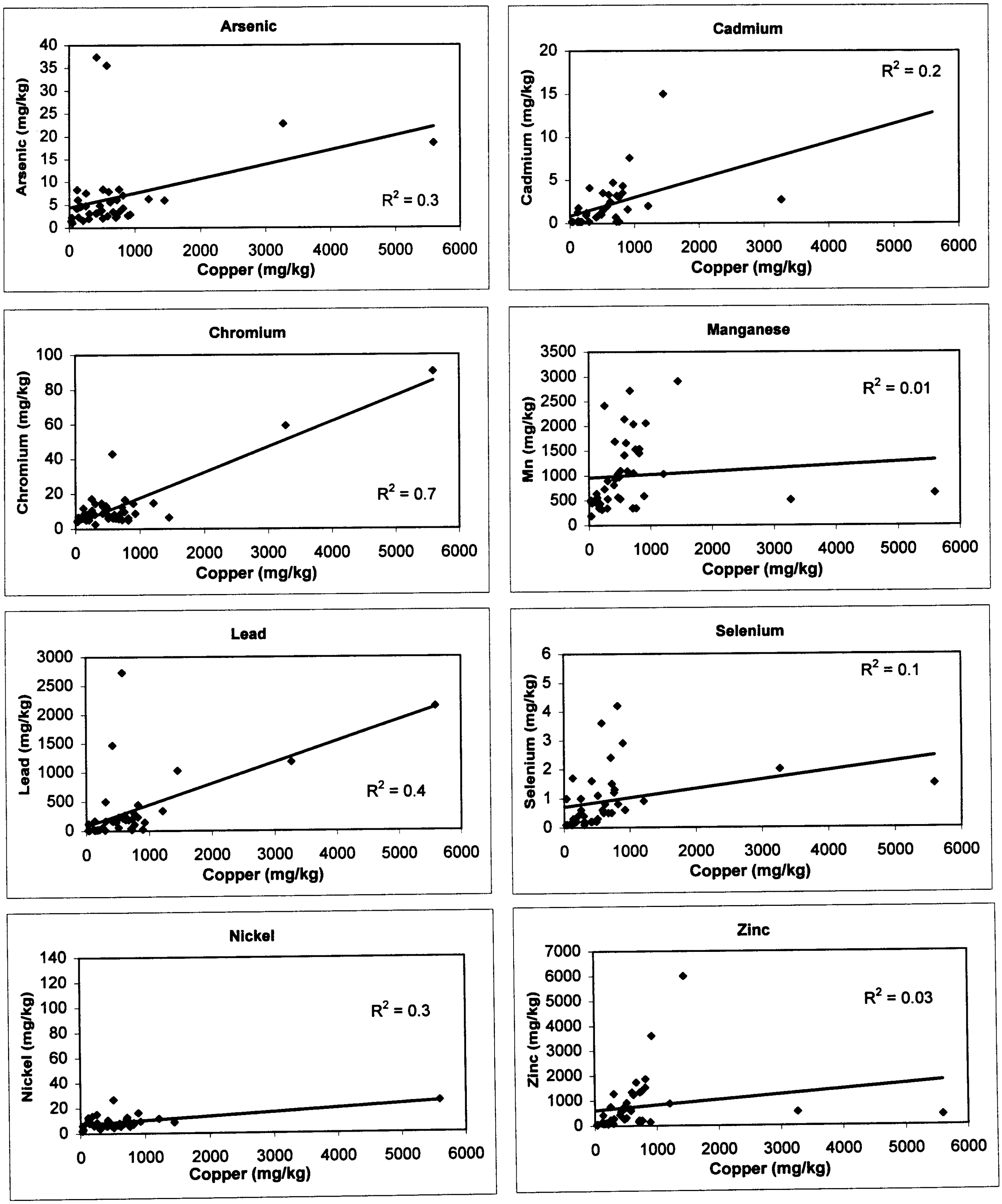
Figure 4-6
Relationship Between Copper and Concentrations of other Metals in Sediment from Hanover-Whitewater and Lampbright IUs



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APPENDIX A

SOIL AND SEDIMENT ANALYSIS PLAN

Appendix A

Soil and Sediment Analysis Plan

1.0 INTRODUCTION

This appendix outlines the proposed conceptual laboratory analytical protocol to evaluate potential bioavailability of constituents of concern (COCs) in Chino Mine Investigation Area (Chino Site) soil and sediment. Preliminary COCs at the Chino Site consist of 17 metals for the terrestrial environment and 7 metals for the aquatic environment (see Section 4.2, Constituents of Concern). The final COC list will be established using additional data obtained from ongoing remedial investigations (RIs) and by further risk screening using soil screening criteria (SSCs). For the purposes of this appendix, discussion of COCs will be limited to cadmium, copper, lead, and zinc.

Traditionally, ecological risk assessments have used the total metal content in soil and sediment to calculate risk to biota. However, this approach has resulted in poor statistical correlation between observed toxicological effects and the metal concentrations used in the risk calculations (cf. Sauvé et al. 1998a). This lack of correlation has been attributed to the existence of several metal "pools" that exhibit differing levels of metal solubility and differing degrees of bioavailability and toxicity (Sauvé et al. 1997). Recent studies have shown that more accurate predictions of metal toxicity and risk are attained using the concentration of "bioavailable" metals in soil (Sauvé et al. 1998). The most bioavailable metal fraction consists of the water-soluble metals, especially the free metal ion content (Sauvé et al. 1997, McGrath 1994, Campbell 1995, Allen and Hansen 1996). Unstable (labile) metal complexes are part of the water-soluble fraction and also impart toxic responses, but usually are secondary in importance to the free metal ion concentration in terms of bioavailability (Sauvé et al. 1998a).

2.0 SOIL METAL POOL AND METAL SPECIATION

In contaminated and pristine soil and sediment, the total metal pool is distributed among numerous solid phases that represent varying degrees of metal cycling in response to pedogenic and depositional processes. These metal-bearing phases consist of relatively insoluble primary minerals (e.g., quartz, pyroxenes, sulfides, etc.), sparingly soluble secondary minerals and low-temperature precipitates (e.g., carbonates, sulfates, and oxides), exchangeable and reactive surface phases (e.g., clays and sesquioxides), and organic phases (Figure C-1) (Sauvé et al. 1998, Tessler et al. 1979, McBride et al. 1997). These phases exhibit widely differing metal solubilities when placed in contact with aqueous solutions and therefore contribute substantially different amounts of soluble and free metal loads to the interstitial soil/sediment solution.

For example, in pure water at pH 7 and oxidizing conditions (pe = 8), the copper oxide mineral "cuprite" (Cu_2O) has a solubility of about 0.001 mg/L. Under the same conditions, the secondary precipitate mineral "malachite" ($Cu_2[OH]_2CO_3$) has a solubility of about 1 mg/L. Therefore, if two different soil/sediment samples have equal concentrations of total copper, but one contains copper in the form of cuprite and the other as malachite, the concentration of soluble copper in the interstitial solution will differ by three orders of magnitude. However, if exchangeable substrates (e.g., clays) or reactive sorptive surfaces (e.g., iron oxyhydroxides) are present, the concentration of soluble copper in the interstitial solution will be lower than that predicted by solubility calculations alone (Dzombak and Morel 1990). In contrast, the presence of soluble inorganic ligands and reactive forms of dissolved organic carbon (e.g., humic and fulvic acids), will increase copper solubility above that predicted by solubility calculations.

Sauvé et al. (1997) evaluated lead toxicity in four different soils (two pristine soils and two contaminated soils) having similar concentrations of lead and pH by quantifying the concentrations of total dissolved lead and free lead ion (Pb^{2+}) relative to the total soil lead content (Figure C-2). They used hot nitric acid digestion and graphite furnace atomic absorption spectrometry (GFAA) to determine total soil lead; a 0.01 *M* $CaCl_2$ aqueous extract and GFAA to determine total dissolved lead; and differential pulse anodic stripping voltammetry to measure free Pb^{2+} ion. Their results showed that the proportions of the potentially bioavailable lead fractions (i.e., total dissolved lead and free Pb^{2+} ion) were several orders of magnitude less than the total soil lead, and that the proportion of free Pb^{2+} ion represents a fraction of the total dissolved lead pool. They concluded that most of the soil Pb content is not extractable by water or dilute salt solutions and is therefore not bioavailable, and only a small portion of the total dissolved Pb content is actually present as the free Pb ion (Sauvé et al. 1997).

In an aqueous solution, the total dissolved lead pool is defined as the sum total of all dissolved lead species, including free Pb^{2+} ion and complexed lead (e.g., $PbSO_4^o$, $PbCO_3^o$, Pb-organic). The distribution of dissolved *inorganic* lead species in a dilute aqueous solution is shown on Figure C-3. The diagram shows that the proportion of free Pb^{2+} ion and complexed lead species is primarily a function of pH. Free Pb^{2+} ion, the most bioavailable form of Pb, is the dominant lead species only under acidic conditions ($pH < 6$). At $pH > 6$, lead bicarbonate ($PbHCO_3^+$) and lead carbonate ($PbCO_3^o$) complexes become the dominant species, with Pb^{2+} concentrations decreasing to insignificant levels above pH 7.

Similar "species-distribution" relationships exist as well for other cationic (positively-charged) metals such as copper (Cu), cadmium (Cd), and zinc (Zn) (Sauvé et al. 1997, Sauvé et al. 1998a). Distribution diagrams for inorganic copper, cadmium, and zinc is shown on figures C-3 through C-5. In the case of copper, free copper ("cupric") ion (Cu^{2+}) comprises about 90% of the total dissolved copper pool at $pH < 5$, with approximately 10% present as the uncharged copper sulfate ion, $CuSO_4^o$. At $pH > 6$, the proportion of copper bicarbonate ($CuHCO_3^+$), copper carbonate ($CuCO_3^o$), and copper hydroxide ($Cu[OH]_3^o$) increase dramatically at the expense of free Cu^{2+} ion. At pH 7.5, the proportion of free Cu^{2+} ion (most bioavailable form) is negligible relative to copper hydroxide (Figure C-3). At very alkaline pHs, only negatively charged copper hydroxide species are present. Both cadmium and zinc form weaker complexes that either copper or lead as is evident by the dominance of the free Cd^{2+} and Zn^{2+} ions up to about pH 8. Above pH 8, these metals exist primarily as complexed ions.

The complexation of metals with dissolved ligands is a function of the solution chemistry and the stability constant (K_{stab}) of the resulting complex ion (Thurman 1985, Drever 1988). For example, the stability constant of $CuSO_4^o$ is $K_{stab} = 10^{2.31}$ compared to the labile species $CuCl_2^o$ ($K_{stab} = 10^{0.16}$), indicating that copper in the presence of sulfate ion will form a stronger bond than with chloride (Allison et al. 1993). The implication is that the bioavailability of copper is lower in the presence of sulfate than chloride, even if concentrations of copper are identical in the two solutions.

Sauvé et al. (1998a) developed empirical regression equations to predict the concentration of free metal ion of copper (Cu^{2+}) and lead (Pb^{2+}) in soil using the total soil metal concentration and soil pH as input parameters. The regressions used to predict free Cu^{2+} were reported with and without soil organic carbon as a predictive parameter. In the case of free Pb^{2+}, soil organic matter was not a significant parameter for predicting free metal activity. Similar equations are presented for free cadmium (Cd^{2+}) and free zinc (Zn^{2+}) (McBride et al. 1997). The equation for cadmium included soil organic carbon as a predictive parameter. The 5 equations to predict free metal activities from total soil metal content, pH, and soil organic carbon (TOC) are:

$$_{P}Cu^{2+} = 1.4pH - 1.7 \log_{10} (\text{Total Cu}) + 3.42 \qquad \text{(eqn. 1)}$$

$$_{P}Cu^{2+} = 1.37pH - 1.95 \log10 (\text{Total Cu}) + 1.95 \log10 (\text{TOC}) \qquad \text{(eqn. 2)}$$

$$_{P}Pb^{2+} = 0.62pH - 0.84 \log_{10} (\text{Total Pb}) + 6.78 \qquad \text{(eqn. 3)}$$

$$_{P}Cd^{2+} = 0.5pH - 0.96 \log_{10} (\text{Total Cd}) + 0.45 \log_{10} (\text{TOC}) - 3.62 \qquad \text{(eqn. 4)}$$

$$_{P}Zn^{2+} = 0.71pH - 0.68\log_{10} (\text{Total Zn}) - 4.44 \qquad \text{(eqn. 5)}$$

Where, pMetal is the negative logarithm (base 10) of the free metal ion activity in micromoles (µM), the total metal content is in milligrams of metal per kilogram dry soil (mg/kg), soil pH is in standard units, and soil organic carbon is in percent carbon (w/w) measured using Walkey-Black potassium dichromate wet oxidation. Sauvé et al. (1998a) concluded that these equations were relatively successful in predicting the chemical speciation of contaminated soil solutions, and provide a preliminary approach for the derivation of soil quality criteria based on estimated solution-free metal activity. These equations will be used to help predict free metal ion activity in Chino soil and sediment, in addition to actual laboratory-measured free metal ion activities, as outlined in the following section.

3.0 DETERMINATION OF BIOAVAILABLE METAL FRACTION IN CHINO SOILS

As outlined in Section 4 of TM-1, soil and sediment samples will be analyzed for parameters that influence plant growth, COC bioavailability and toxicity, and COC fate and transport. These analyses include total soil metals, water-soluble metals, free metal ions, water-soluble anions, nutrients, total soil organic carbon (TOC), dissolved organic carbon (DOC), texture, grain size, and pH. The analytical approach proposed here is based on procedures outlined by Sauvé et al. (1998a) and McBride et al. (1997) with modifications, and standard EPA analytical methods for soil and water (Figure C-6).

Prior to analysis, all soil and sediment samples will be air dried, sieved to less than 2 mm, homogenized and weighed. Each sample will be split into 5 sub-samples, of which 4 will be subjected to laboratory analysis and the fifth sample archived for future use if necessary. Sample splits will be used to determine soil and sediment properties (TOC, nutrients, pH, texture and grain size), water-soluble metal content, total soil metal content, and water-soluble anions. These data will be used as input to equations 1 through 5 to calculate the free metal ion fraction in Chino soil and sediment based on Sauvé et al. (1998a) and McBride et al. (1997). The calculated free metal contents will then be compared to measured and model-derived free metal ion content (discussed below) and the results of laboratory phytotoxicity tests to derive site-specific empirical equations describing the relationship between toxicity and metal content at Chino.

3.1 Total Metal Content

The analyses used for total soil metals include EPA Methods 6010 (inductively coupled plasma (ICP) spectrometry), Method 6020 (ICP-mass spectrometry), or Method 7000 series (graphite furnace atomic absorption-GFAA). All of these methods utilize an aggressive digestion using hot nitric acid that dissolves all but the most insoluble soil/sediment components. The total metal content of the digestate is determined using ICP, ICP-MS, or GFAA depending on required DQOs (Table C-1). The concentration in the sample is quantified by the instrument response relative to standards of known metal concentration. It is important to note that these methods vastly overestimate the bioavailable metal fraction in the sample because sample digestion releases the entire metal pool to solution regardless of solubility or form. All soil and sediment samples will be digested prior to analysis using either EPA Method 3050 (nitric acid hot plate) or EPA Method 3051 (nitric acid microwave). The total metal content will be used in equations 1 through 5 in conjuction with the sample pH and total organic carbon content to derive the calculated free metal ion content.

3.2 Water-soluble Metal Content

Soil and sediment sample splits will be used to determine the water-soluble fraction of metals at the Chino site. Dilute aqueous soil and sediment solutions will be prepared using 1:2 soil:water extracts of 0.01 *M* $CaCl_2$ after the method of Sauvé et al. (1995). In this method, 5 g of soil will be mixed with 10 mL of 0.01 *M* $CaCl_2$ prepared using deionized water. The soil extract will be filtered using a 0.45 μm nitrocellulose filter to remove suspended particulates prior to analysis. Sample pH and conductivity will be measured following filtering. Total dissolved metals in the filtrate will then be analyzed using either EPA Methods 6010 (ICP), 6020 (ICP-MS), or 7000 series (GFAA).

3.3 Free Metal Ion Analysis

The free metal content of the soil/sediment samples will be determined using ion-selective electrodes (ISEs) or EPA-approved wet chemistry methods on sample splits of the filtered water-soluble samples (Section 3.2). Analysis of soil/sediment solutions using ISEs yields the free ion activities of target metals in solutions over a range of ionic strengths with few interferences. Potentially interfering elements such as Fe^{3+} and Al^{3+} occur mainly in acidic soil solutions, and therefore should not pose a problem for Chino soil/sediment, where pHs generally range from 4 to 8. However, results of the total metal content obtained from analysis of the water-soluble metal content (Section 3.2) will be used to reveal which elements (if any) occur in high enough concentrations to pose potential interferences. The exact methods that will be used to determine free metal ion content will depend on those results. In some cases, wet chemical methods such as the porphyrin method or EPA's bicinchoninate method for free copper may be more accurate if potentially interfering elements are present. Subtracting the free metal ion concentration from the total metal concentration will provide the concentration of complexed metals in the soil/sediment solutions.

3.4 Water-Soluble Anions

Soil and sediment samples will also be analyzed for water-soluble anions and dissolved organic carbon to characterize the presence and amount of inorganic and organic ligands available for metal complexing. Analysis for anions will include sulfate (SO_4^{2-}), chloride (Cl^-), orthophosphate (as P), fluoride (F^-), nitrate and nitrite (as N), and alkalinity (bicarbonate + carbonate). Exact methodologies used will depend on required DQO's for this site. Sulfate will be determined using either EPA Method M375.3 (Gravimetric), or method M300 (Ion Chromatography). Chloride will be determined using EPA Method M325.2 (Automated Colorimetric) or M300 (Ion Chromatography). Orthophosphate will be determined using EPA Method M365.1 (Automated Colorometric); fluoride by M340.2 (Ion Specific Electrode) or M300 (Ion Chromatography); nitrate and nitrite by M353.2 (Automated Colorometric); and alkalinity by M310.1 (Titration). Water-soluble dissolved organic carbon (DOC) will be analyzed using EPA Method 415.1 Combustion/IR Detection.

4.0 GEOCHEMICAL MODELING

As an additional characterization and quality-control step, geochemical modeling using the EPA-code MINTEQA2 (Allison et al. 1991) will be used to corroborate the results obtained from laboratory analysis of total dissolved and free metal ion content, and the calculated free metal ion concentration obtained using the empirical equations of Sauvé et al. (1998a). The model will use the analytical results obtained from the water-soluble analyses as input. These will consist of total cations and anions, total dissolved organic carbon, and solution pH. Geochemical models are used to speciate aqueous solutions by using the total measured metal and ligand (anion) concentration (e.g., by ICP, GFAA or ISE) in conjunction with the system pH and Eh (oxidation-reduction potential), if appropriate. The model first determines the ionic strength of the solution, calculates activity coefficients and ion activities, and then speciates the bulk solution according to established equilibrium constants for free ion and metal complexes. The model can also used to simulate changes in soil/sediment solution chemistry caused by *in situ* mass transfer processes, such as dissolution/precipitation, ingassing/outgassing, and ion exchange/adsorption reactions and thus provide insight into fate and transport processes that may occur at the site (Langmuir 1997).

Regression equations will be developed that correlate the results of the water soluble and free metal ion concentrations determined by laboratory analyses (Sections 3.0 through 3.3) to the theoretical equilibrium speciation predicted by MINTEQA2 and to the calculated free metal ion concentrations predicted using the equations of Sauvé et al. 1998 (discussed in Section 4 of TM-1). If correlation between the model, predictive equations, and laboratory results is satisfactory, then the model and predictive equations may be used as a proxy additional field sampling and laboratory tests thereby reducing sampling and analytical costs.

5.0 REFERENCES

Allen, H.E. and D.J. Hansen. 1996. The importance of trace metal speciation to water quality criteria. Water Environ. Res. 68:42-54.

Allison, J.D., D.S. Brown, and K.H. Novo-Gradac. 1993. MINTEQA2: A Geochemical Assessment Model for Environmental Systems. International Groundwater Modeling Center, Golden, CO.

Campbell, P.G.C. 1995. Interactions between trace metals and aquatic organisms: A critique of the free-ion activity model. In: Tessier, A., D.R. Turner, Eds. Metal Speciation and Bioavailability in Aquatic Systems. John Wiley & Sons, New York. pp. 45-102.

Drever, J.I. 1988. The Geochemistry of Natural Waters. 2nd Ed. Prentice-Hall, Englewood Cliffs, NJ. 437 pp.

Dzombak, D. and F.M.M. Morel. 1990. Surface Complexation modeling. Hydrous Ferric Oxide. Wiley-Interscience, New York.

Garrels, R.M. and C.L. Christ. 1965. Solutions, Minerals, and Equilibria. Harper & Row Publishing Co., New York.

Hem, J.D. 1985. Study and Interpretation of the Chemical Characteristics of Natural Waters. USGS Water-Supply Paper 2254. 3rd Ed.

Langmuir, D. 1997. Aqueous Environmental Geochemistry. Prentice Hall, New Jersey.

McBride, M.B., S. Sauvé, and W.H. Hendershot. 1997. Solubility control of Cu, Zn, Cd, and Pb in contaminated soils. Eur. J. Soil Sci. 48:337-346.

McGrath, S.P. 1994. Effects of heavy metals from sewage sludge on soil microbes in agricultural systems. In: Ross, S.M., ed. Toxic Metals in Soil-plant Systems. John Wiley & Sons, New York. pp. 247-274.

Stumm, W. and J.J. Morgan. 1981. Aquatic Chemistry, 2nd Ed. John Wiley & Sons, New York.

Sauvé, S., M.B. McBride, and W.H. Hendershot. 1995. Ion-selective electrode measurements of copper(II) activity in contaminated soils. Arch. Environ. Contam. Toxicol. 29:373-379.

Sauvé, S., M.B. McBride, and W.H. Hendershot. 1997. Speciation of lead in contaminated soils. Environ. Pollut. 98:149-155.

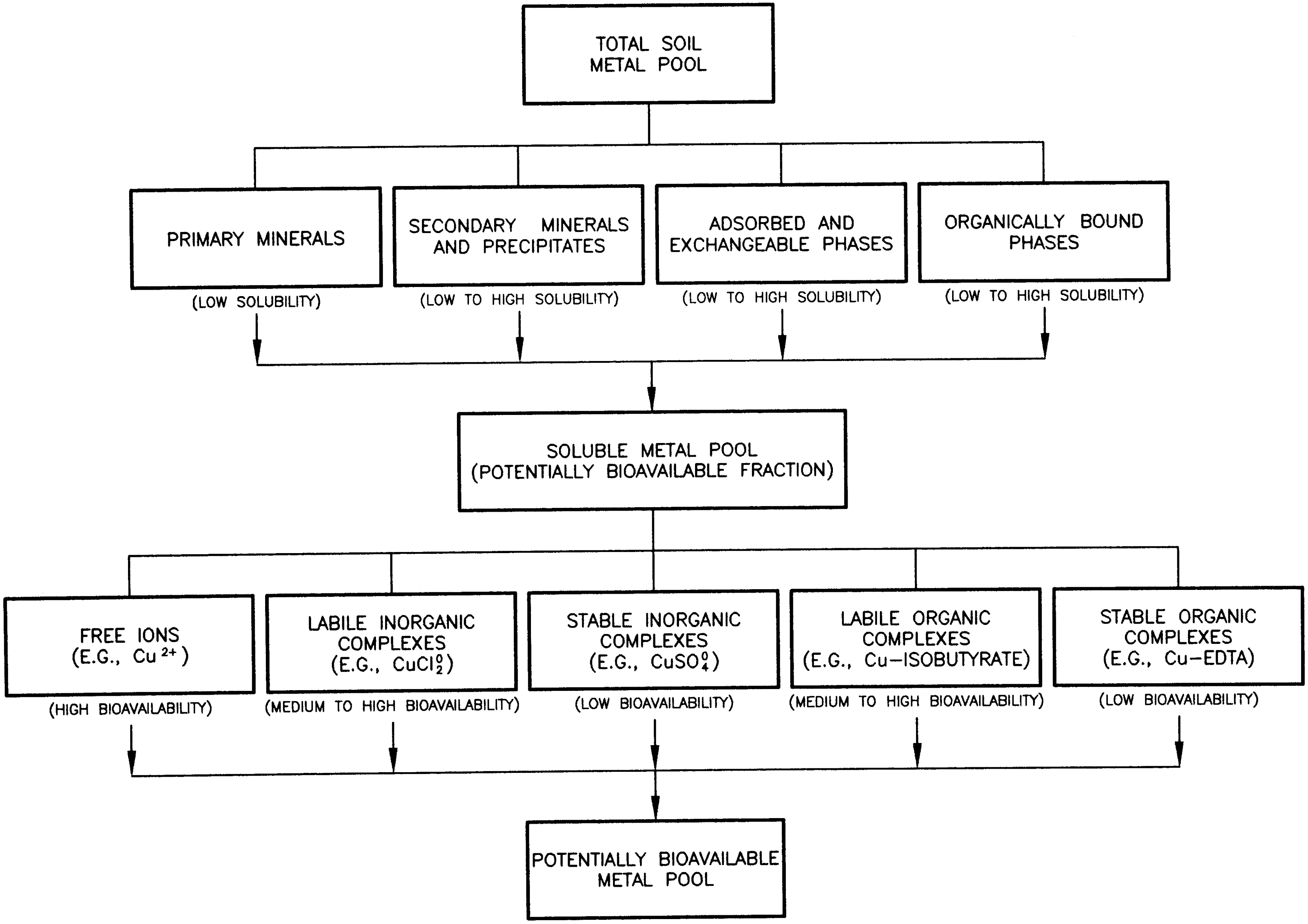
Sauvé S., A. Dumestre, M. McBride, and W. Hendershot. 1998a. Derivation of soil quality criteria using predicted chemical speciation of Pb^{2+} and Cu^{2+}. Environ. Tox. Chem. 19:481-1489.

Sauvé, S., M. McBride, and W. Hendershot. 1998b. Lead phosphate solubility in water and soil suspensions. Environ. Sci. Technol. 32:388-393.

Tessier, A., P.G.C. Campbell, and M. Bisson. 1979. Sequential extraction procedure for the speciation of particulate trace metals. Anal. Chem. 51:844-851.

Thurman, E.M. 1985. Organic Geochemistry of natural Waters. Martinus Nijhoff/Dr. W. Junk, Dordrecht (Netherlands). 497 pp.

Figures and Tables



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Designed: M.L.
Checked: M.L.
Date: 9-29-98
Dwg. No. 270-1

Figure A-1

ISSUED FOR CLIENT REVIEW.

Revision: REV 1

PRINCIPAL FORMS OF METALS IN SOIL SOLID PHASES AND RELATIVE CONTRIBUTION TO BIOAVAILABLE FRACTION

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Schafer

Figure A-2
Distribution of Dissolved Lead Species as a Function of pH

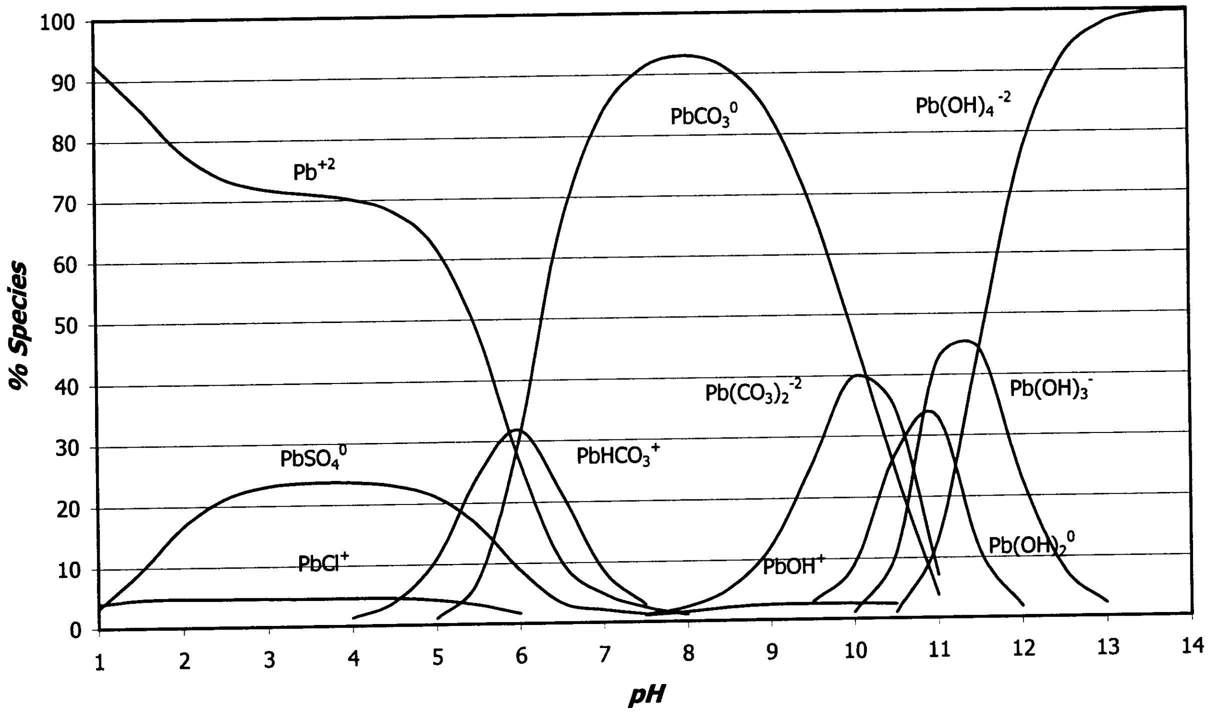


Figure A-3
Distribution of Dissolved Copper Species as a Function of pH

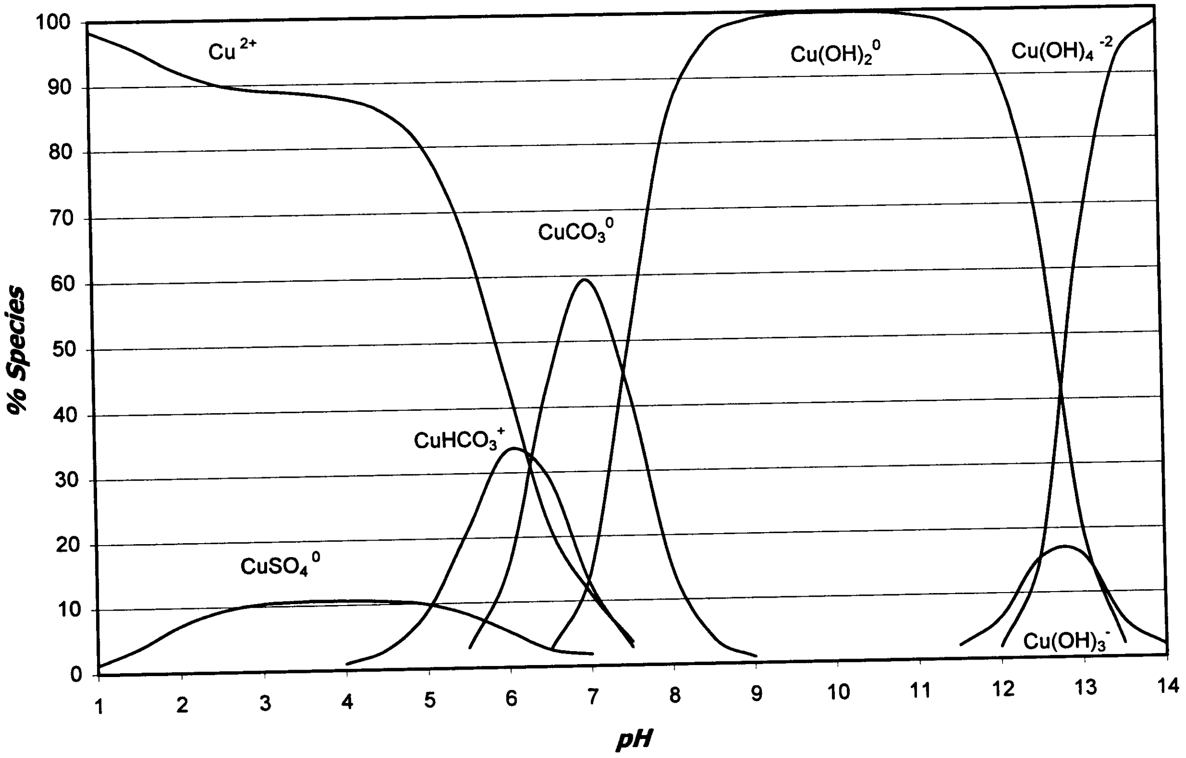


Figure A-4
Distribution of Dissolved Cadmium Species as a Function of pH

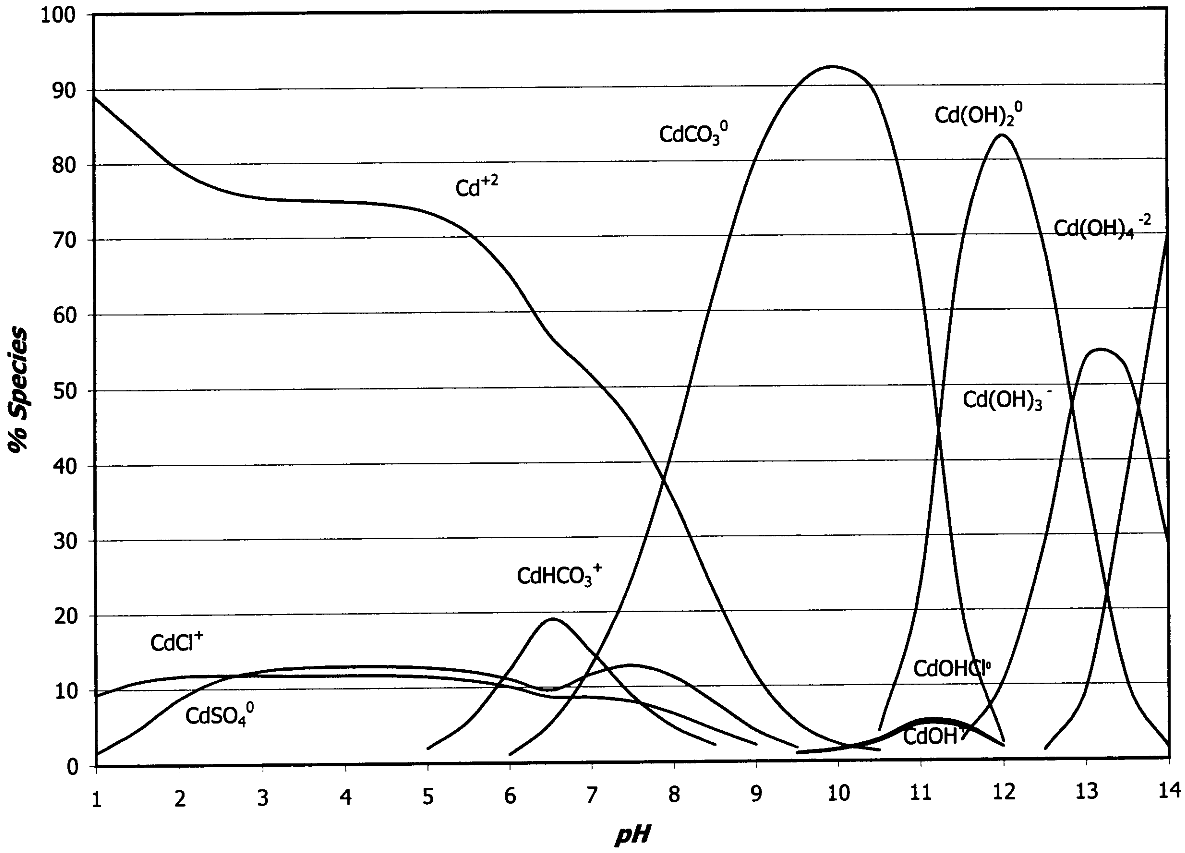
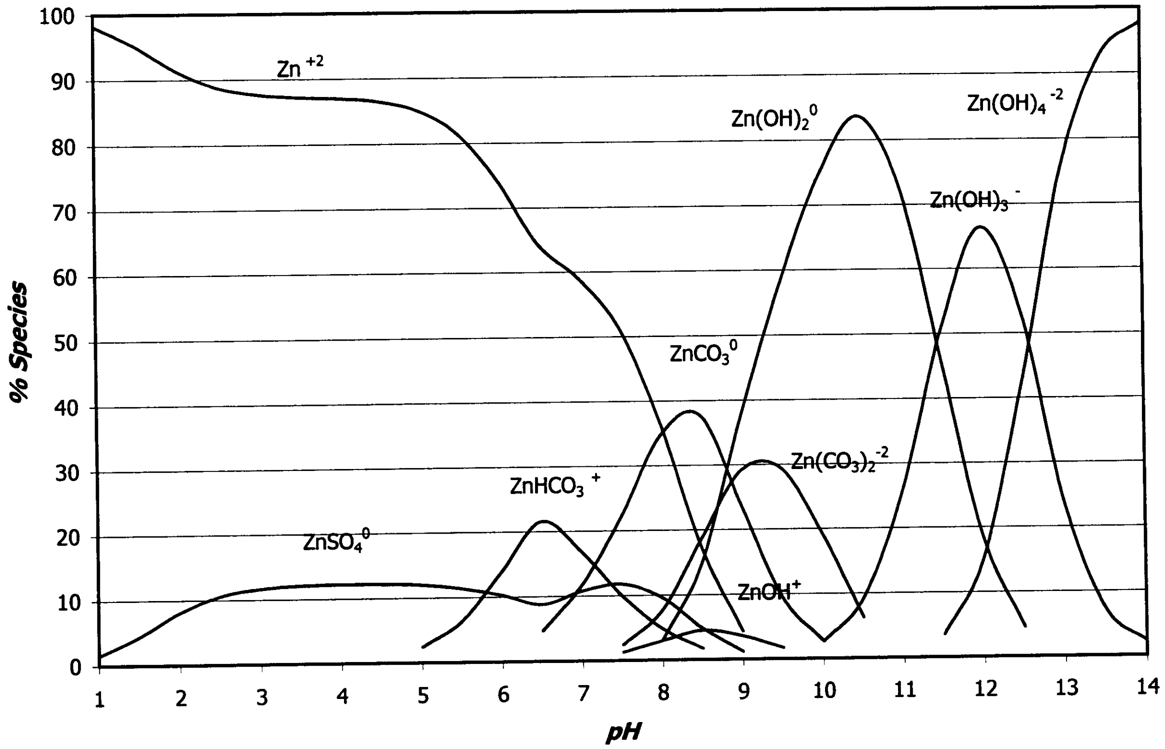


Figure A-5
Distribution of Dissolved Zinc Species as a Function of pH



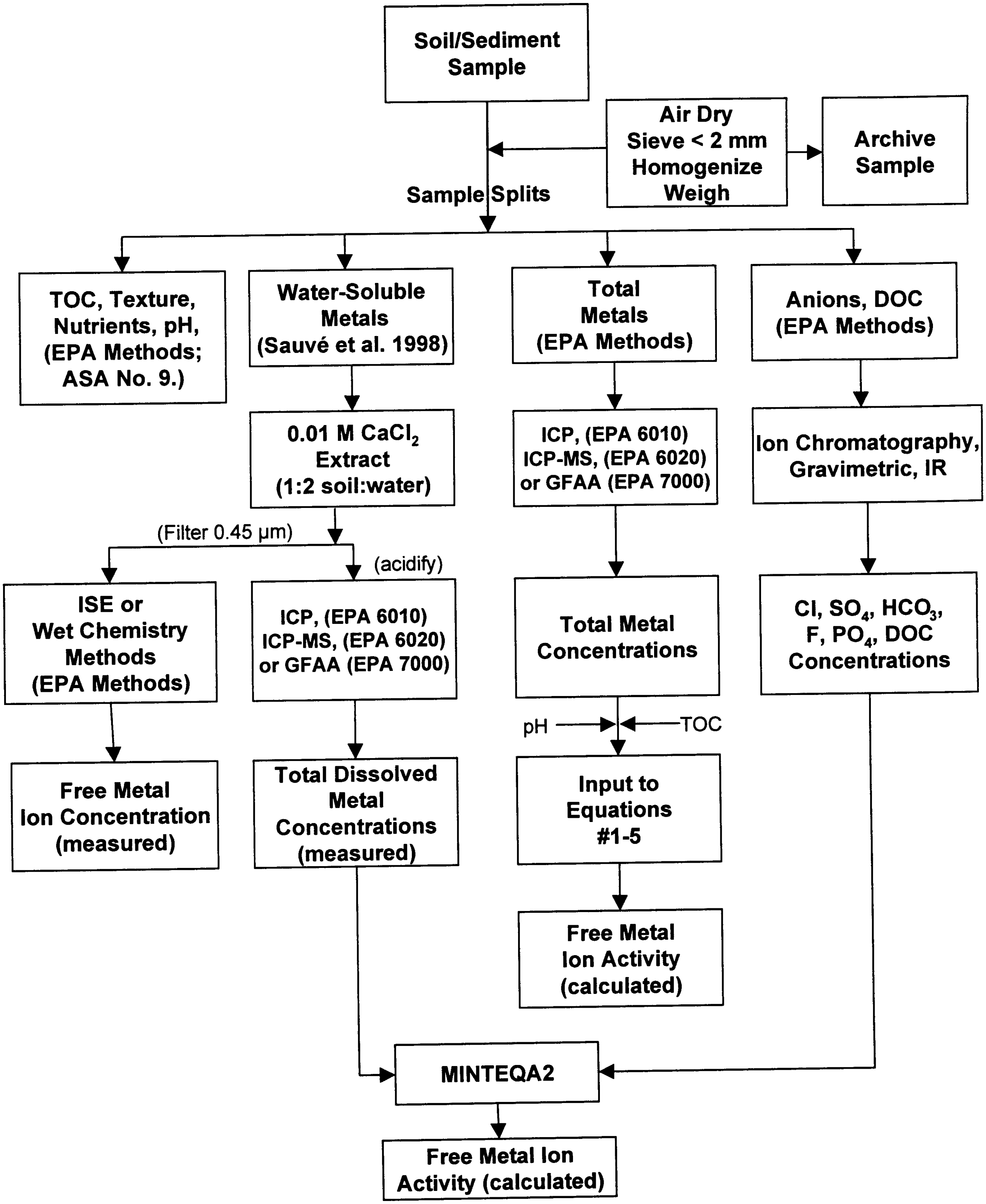


Figure A-6
Proposed Analytical Routine for Chino Mine Site Soil and Sediment Bioavailability Analysis

Table A-1
EPA Methods and Analyte Detection Limits

METHOD DETECTION LIMIT (mg/kg)*			
Instrument	**ICP**	**GFAA**	**ICP/MS**
EPA Method	**6010**	**7000 Series**	**6020**
Aluminum (Al)	3	-	-
Antimony (Sb)	-	0.2	0.02
Arsenic (As)	-	0.1	0.1
Barium (Ba)	0.3	-	0.02
Beryllium (Be)	0.2	0.05	0.01
Bismuth (Bi)	10	-	-
Boron (B)	1	-	-
Cadmium (Cd)	0.3	0.05	0.02
Calcium (Ca)	20	-	-
Chromium (Cr)	1	0.1	0.02
Cobalt (Co)	1	0.1	0.01
Copper (Cu)	1	0.1	0.1
Gallium (Ga)	10	-	-
Gold (Au)	-	0.1	-
Iron (Fe)	1	0.1	-
Lanthanum (La)	10	-	-
Lead (Pb)	2	0.1	0.01
Lithium (Li)	2	-	-
Magnesium (Mg)	20	-	-
Manganese (Mn)	0.5	0.1	0.04
Mercury (Hg)	-	-	-
Molybdenum (Mo)	0.5	0.1	0.01
Nickel (Ni)	1	0.1	0.06
Potassium (K)	30	-	-
Selenium (Se)	4	0.2	0.3
Silica (Si)	20	-	-
Silver (Ag)	0.5	0.05	0.01
Sodium (Na)	30	-	-
Strontium (Sr)	1	-	0.01
Thallium (Tl)	20	0.2	0.01
Tin (Sn)	10	0.1	-
Titanium (Ti)	0.5	-	0.1
Tungsten (W)	10	-	-
Uranium (U)	-	-	0.01
Vanadium (V)	0.5	-	0.03
Zinc (Zn)	1	0.1	0.2

* MDL based on 3050 or 3051 digestion of a dry sample that is relatively free of matrix interferences.

APPENDIX B

EXPOSURE PARAMETERS

Exposure Parameters for the Omnivorous Rodent
Representative species : Deer Mouse *(Peromyscus maniculatus)*

Parameter	**Value and Comments**	**Reference**
Habitat	Ubiquitous in central and western US	Armstrong (1972), Fitzgerald et al. (1995)
Body Weight	18.7 g	EG&G data from RFETS
Diet Composition	seeds 43% forbs 5.4% grasses and sedges 3.6% shrubs 2.1% beetles 13% leafhoppers 4.9% lepidopterans 9.4% spiders 2.0%	Flake (1973)*
Total Dietary Ingestion Rate	0.21 g/g body weight/day	Cronin and Bradley (1988)*
Water Ingestion Rate	0.19 g/g body weight/day	Ross (1930), Dice (1922)*
Fraction Soil in Diet	2 % of food ingestion rate (from white footed mouse)	EPA (1993)

*cited in USEPA (1993)

Exposure Parameters for Large Ground-feeding Bird
Representative Species: Gambel's quail *(Callipepla gambelii)*

Parameter	**Value and Comments**	**Reference**
Habitat	Desert scrublands and thickets of southwestern US and northern Mexico	Bent (1963), National Geographic Society (1987)
Body Weight	Approximately 300 g; slightly larger than northern bobwhite quail	EPA (1993)
Diet Composition	Invertebrates 0.5% seeds, grains 36% leaves, shoots 63.5%	Bent (1963)
Total Dietary Ingestion Rate	0.09 g/g body weight/day (for northern bobwhite)	Koerth & Guthery (1990)*
Water Ingestion Rate	0.13 g/g body weight/day (for northern bobwhite)	Calder and Braun (1983)*
Fraction Soil in Diet	10 % of food ingestion rate (from American woodcock data)	EPA (1993)*

*cited in USEPA (1993)

Exposure Parameters for Small Ground-feeding Bird
Representative species: Dark-eyed Junco *(Junco hyemalis)*

Parameter	**Value and Comments**	**Reference**
Habitat	Mixed woodlands	National Geographic Society (1987)
Body Weight	12 grams	EPA (1993)
Diet Composition	Primarily seeds	National Geographic Society (1987)
Total Dietary Ingestion Rate	0.9 g/g body weight/day (for the similar-sized marsh wren)	EPA (1993)*
Water Ingestion Rate	0.28 g/g body weight/day (for the similar-sized marsh wren)	Calder and Braun (1983)*
Fraction Soil in Diet	10 % of food ingestion rate (from American woodcock data)	EPA (1993)

*cited in USEPA (1993)

Exposure Parameters for Mammalian Predator
Representative species: Red Fox *(Vulpes vulpes)*

Parameter	**Value and Comments**	**Reference**
Habitat	Upland woodland, agricultural areas, shrublands while hunting	Ables (1974), Samuel and Nelson (1982)*
Body Weight	4,530g	Storm et al. (1976)
Diet Composition	rabbits 44% small mammals 30% birds 19%	Seargeant (1978)
Total Dietary Ingestion Rate	0.11 g/g body weight/day	Seargeant (1978)
Water Ingestion Rate	0.085 g/g body weight/day (estimated)	Calder and Braun (1983)
Fraction Soil in Diet	2.8 % of food ingestion rate	EPA (1993)

*cited in USEPA (1993)

Exposure Parameters for Ruminant
Representative species: Mule deer *(Odocoileus hemionus)*

Parameter	Value and Comments	Reference
Habitat	All major habitat types except deserts and tundra	Anderson and Wallmo (1984)
Body Weight	70 kg for adults	Anderson et al. (1974)
Diet Composition	shrubs 58% forbs 29% grass 6% other 7%	Carpenter et al. (1979), Kufeld et al. (1973)
Total Dietary Ingestion Rate	0.022 kg air dry forage/kg body weight/day	Alldredge et al. (1974)
Water Ingestion Rate	44 ml/kg body mass/day	Bissell et al. (1955)
Fraction Soil in Diet	5 % of food ingestion rate	EPA (1993)

Exposure Parameters for the Avian Predator
Representative species: Red-tailed hawk *(Buteo jamicensis)*

Parameter	Value and Comments	Reference
Habitat	Woodlands, wetlands, prairie	EPA (1993)
Body Weight	1 kg for adults	EPA (1993)
Diet Composition	Small mammals 78% Birds 8.5%	EPA (1993)
Total Dietary Ingestion Rate	0.3 kg /kg body weight/day (conservative value used from kestrel)	EPA (1993)
Water Ingestion Rate	0.05 kg/kg body mass/day	EPA (1993)
Fraction Soil in Diet	5 % of food ingestion rate	EPA (1993)

APPENDIX C

TOXICITY PROFILES FOR POTENTIAL CONSTITUENTS OF CONCERN

ALUMINUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Aluminum is a Group III metal found only in the trivalent state; it is the most abundant metal in the earth's crust at 8.1% (Bodek et al. 1988). The free metal is not found in the ambient environment, however, due to its reactivity. In weathered soils, aluminum precipitates as hydrated aluminum oxides. In surface water with a neutral pH and the presence of minimal complexing agents, naturally occurring dissolved aluminum concentrations are low. Flow rate also influences aluminum concentrations in surface water, and under turbulent conditions of high flow, aluminum can dissolve from suspended minerals. In the presence of silica, aluminum precipitates from water as poorly crystallized clay mineral species (Bodek et al. 1988). Acidity increases the mobility of aluminum, and tends to produce elevated concentrations.

TOXICITY TO TERRESTRIAL ORGANISMS

Aluminum toxicity data were available for ruminants (cattle and sheep) and rodents (mice). The studies cited in Table A-A provided a No Observed Adverse Effect Level (NOAEL) for cattle (Valdivia et al. 1978), a Lowest Observed Adverse Effect Level for sheep (LOAEL) (Valdivia et al. 1978), and a No Effect Level (NEL) based on the impairment of growth in mice (Ondreicka et al. 1966). These studies were used to represent toxicity to the mammalian receptors selected for the CMC risk assessment. A long-term study that provided a NEL based on reproductive and growth endpoints was available for the ring dove (Carriere et al. 1986). Aluminum at 1000 ppm in diet had no effect on reproduction or growth over 4 month period when Ca and P levels in diet were normal.

Although some sources recommend a criterion of 50 mg/kg in soil as protective of plants (Will and Suter 1995), this value may be below CMC background conditions. Therefore, a value of 730 mg/kg in soils based on a decrease of root length in wheat grown in acidic soils (Wright et al. 1989) is a more appropriate value for western states; under the alkaline soil conditions that predominate within the CMC boundaries, toxicity to plants at this level is not expected. This soil concentration would also be protective of soil fauna, which appear to exhibit effects at higher concentrations of aluminum. Data indicate that woodlice have high survival rates (50-75%) over a 6-12 week period when exposed to 2800 mg/kg soil (ICF 1989).

TOXICITY TO AQUATIC ORGANISMS

The LC50 for *Bufo americanus* was 0.627 to > 1.762 mg/L for a 96 h exposure (Freda et al. 1990). The LC50 for *Rana pipiens* was similar at 0.403 to > 1 mg/L for a 96 h exposure (Freda et al. 1990; Freda and McDonald 1990). Toxicity decreased with increasing amount of natural as compared to artificial water, indicating that the higher end of the concentration range would be appropriate for protecting wild amphibian populations.

An Ambient Water Quality Criterion (AWQC) was unavailable for aluminum, as were sediment quality criteria (EPA 1996a; Persaud et al. 1992; Batts and Cubbage 1995).

ANTIMONY

BIOLOGICAL AND ENVIRONMENTAL FATE

Antimony occurs in the ambient environment primarily as sulfides or oxides (Bodek et al. 1988). The chemical properties are similar to arsenic in that it exhibits numerous oxidation states. In surface water, antimony exists as hydroxo-complexed species under typical environmental conditions, such as antimonous acid ($Sb(OH)_3$)and antimonic acid ($Sb(OH)_5$). Antimony oxides are highly soluble; alkylated forms are found in the environment, and biomethylation has been suggested due to antimony's similarity to other elements that have been shown to biomethylate, such as arsenic, selenium, and lead (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

Data for the toxicity of antimony to avian species were unavailable in the literature reviewed. Data were available for the toxicity to mammals (Friberg et al. 1979). The study documented a 15% reduction in the life-span of rats exposed to antimony in drinking water. This study was applied to represent the mammalian receptors at the CMC site.

A recommended benchmark value for the protection of plants was 5 mg/kg soil (Will and Suter 1995). Data were unavailable regarding the toxicity of antimony to terrestrial invertebrates in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

Data regarding the toxicity of antimony to amphibians were unavailable in the literature reviewed.

An Ambient Water Quality Criterion (AWQC) was unavailable for antimony, however, an unpublished EPA document reported proposed acute and chronic criteria of 0.088 and 0.030 mg/l, respectively (EPA 1992a).

Sediment quality criteria were unavailable from EPA (1996a); Persaud et al. (1992); and Batts and Cubbage (1995). Long and Morgan (1991) present an Effects Range Low (ERL) of 2 mg/kg, and an Effects Range Median (ERM) of 25 mg/kg, where the concentrations are expressed on a dry weight basis. The ERL is the lower 10th percentile of the distribution of biological effects data, whereas the ERM is the median value.

ARSENIC

BIOLOGICAL AND ENVIRONMENTAL FATE

The environmental chemistry of arsenic is very complex; arsenic can assume numerous oxidation states (Bodek et al. 1988). Arsenic is usually associated with sulfide ores, although over 100 minerals and ores contain arsenic. Arsenic is sorbed to soils and sediments by adsorption onto iron and aluminum hydrous oxides, clays, and carbonates, co-precipitation with iron oxides, and isomorphic substitution with phosphorus in minerals. These sorption mechanisms are controlled by redox conditions, pH, the presence of competing anions and complexing ions, salinity, hydrous oxide content, and clay content. In aerobic waters, the chemistry of arsenic is strongly influenced by pH; because it can form anions in solution, arsenic does not complex with simple anions like chloride, but instead behaves like a ligand and complexes with organic sulfur, nitrogen, and carbon. The highly toxic form arsine is only produced in extremely anoxic environments (Bodek et al. 1988), which are not expected to occur within the CMC boundaries.

TOXICITY TO TERRESTRIAL ORGANISMS

A limited study with sheep (James et al. 1966) was used to represent the toxicity to wild ruminants; this study is limited due to the small sample size (n=3) and the fact that the dose vehicle was by capsule instead of diet (Table A-A). This study was used only for the mule deer due to the relatively close phylogenetic relationship between the sheep and mule deer. A multi-generation study with mice was used to represent the toxicity to other mammals (Pershagen and Vahter 1979). A chronic study with mallard ducks (*Anas platyrhynchos*) exposed to arsenic in diet (Stanley et al. 1994) was used to predict toxicity to the avian receptors.

One recommended benchmark value for the protection of plants was 10 mg/kg (Will and Suter 1995; CEPA 1993). Pentavalent arsenic at this concentration may suppress growth in some species, however. Trivalent arsenic suppresses growth at 25 mg/kg (Will and Suter 1995; CEPA 1993). Another recommended criterion is <25 mg total arsenic/kg soil (Eisler 1988). The maximum soil concentration from unimpacted field collected soils used in a laboratory phytotoxicity bioassay was 224 mg/kg (Kapustka et al. 1995). The recommended benchmark for arsenic for protection of invertebrates is 60 mg/kg (Will and Suter 1995).

TOXICITY TO AQUATIC ORGANISMS

The 5 h LC50 for exposure of *Gastrophryne carolinensis* from fertilization to 4 days post-hatch was 0.04 mg/l (Birge et al. 1979).

The chronic AWQC for trivalent arsenic is 0.190 mg/l (EPA 1996a). The acute freshwater criterion for trivalent arsenic is 0.360 mg/l (EPA 1992a).

Sediment quality benchmarks from EPA and Canada range from a low of 6 mg/kg for the Lower Effect Level (LEL) to a high of 33 mg/kg for the Severe Effect Level (SEL) (EPA 1996a; Persaud et al. 1992). The LEL is indicative of relatively clean or marginally polluted sediments, with a potential to affect some sensitive water uses. The SEL is indicative of grossly polluted sediments, and is a concentration that will significantly affect the use of sediment by benthic organisms (Persaud et al. 1992). The NOAA ERL for arsenic in sediment is 33 and the ERM is 85 mg/kg dry weight (Long and Morgan 1991).

BARIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Barium occurs in the ambient environment primarily as the sulfate (barite) and carbonate (witherite) forms (Bodek et al. 1988); only the +2 oxidation state is of environmental importance. Certain barium species are soluble in water; these include the chloride, hydroxide and nitrate forms. Other barium salts have low solubility.

TOXICITY TO TERRESTRIAL ORGANISMS

The barium study used to obtain a NOAEL for mammalian receptors was based on an endpoint of hypertension in rats; this value was also a NEL for overt toxicity and survival. (Perry et al. 1989. This daily intake was estimated from exposure to 100 mg/l in drinking water and a daily ingestion rate of 15 ml water per 45 g body weight. A clear link between hypertension and impaired population success has not been made, but the study is appropriate since overt toxicity and survival are clearly linked with population success. This study exposed animals by dosing their drinking water, which may alter bioavailability as compared to dietary exposure. The LOAEL for mammals was based on a study with rats (Borzelleca et al. 1988), although the exposure route was oral gavage (Table A-A). A study based on a endpoint of growth depression in chickens exposed to 1000 mg/kg barium in diet (Johnson et al. 1960) was used to represent toxicity to avian receptors (Table A-A). This dietary exposure resulted in a slight growth depression.

The recommended benchmark for barium protective of plants is 500 mg/kg (Will and Suter 1995). Data regarding barium toxicity to soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

Toxicity data were unavailable for amphibians in the literature reviewed.

An acute or chronic AWQC is not available for barium. However, a Tier II value of 0.0039 mg/l was proposed by EPA (1996a).

Sediment quality criteria or benchmarks were unavailable in the literature reviewed (Long and Morgan 1991; EPA 1996a; Persaud et al. 1992). EPA has reported a guideline of <20 mg/kg as representative of nonpolluted sediments for the classification of harbor sediments (Batts and Cubbage 1995).

BERYLLIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Beryllium is typically found in conjunction with silicates in the environment (Bodek et al. 1988). There is relatively little information on the sorption of beryllium by soils or sediments. Certain clay minerals (montmorillonite and illite-like minerals) can strongly sorb beryllium; however, kaolinite does not sorb beryllium. In water, beryllium exhibits only the +2 oxidation state. Beryllium hydroxo-complexes are relatively insoluble, although they dominate as the dissolved species under certain conditions. The sulfates and chlorides are more soluble. The pH of the ambient water affects the relative solubility of the various beryllium compounds.

TOXICITY TO TERRESTRIAL ORGANISMS

A chronic lifetime study with rats that resulted in only mild weight loss was used to represent toxicity to mammalian receptors (WHO 1990). This study dosed animals by dietary exposure (Table A-A). Beryllium in the diet of poultry at a concentration of 0.5% resulted in ricketts (Friberg et al. 1979). This study was used to represent toxicity to avian receptors (Table A-A).

The recommended benchmark protective of plant populations is 10 mg/kg (Will and Suter 1995). Data regarding the toxicity of beryllium to soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

The LC50 for mortality for 24, 48, and 96 h exposures at a water hardness of 400 mg/L ranged from 18.2 to 31.5 mg/L (Slonim and Ray 1975). The species tested were *Ambystoma maculatum* and *A. opacum*.

An AWQC is not currently available for beryllium (EPA 1996a). The acute and chronic lowest observed adverse effect levels (LOAEC) were 0.130 and 0.0053 mg/l, respectively (EPA 1992a). A Tier II value of 0.0051 mg/l was proposed by EPA (1996a).

Sediment quality criteria or benchmarks were unavailable in the literature reviewed (EPA 1996a; Persaud et al. 1992; Batts and Cubbage 1995; Long and Morgan 1991).

BORON

BIOLOGICAL AND ENVIRONMENTAL FATE

Data regarding the biological or environmental fate of boron were unavailable in the literature reviewed.

TOXICITY TO TERRESTRIAL ORGANISMS

A dietary concentration of 1000 mg/kg caused cellular and enzyme changes in male rats (Lee et al. 1978), whereas a lower test concentration of 500 mg/kg diet had no effect (Table A-A). The study duration was not specified as the primary source for this information was unavailable. The dietary test concentrations were converted to a daily intake or dose with a dietary ingestion rate of 0.085 kg/kg-d (Groton et al. 1991).

Mallard ducklings were exposed to boron in diet (Whitworth et al. 1993). The LOAEL for endpoints that included mortality, growth rate, behavior, and blood chemistry changes in ducklings was 1600 mg/kg diet (Table A-A). A dietary concentration of 400 mg/kg was the NOAEL, although some behavior alterations may have occurred at this dose. The duration of the study was 9 weeks. The dietary concentrations were converted to an intake or dose normalized to duckling body weight by using a conversion factor for dietary ingestion rate of 0.14 kg diet/kg-d (Camardese et al. 1990).

Data regarding the toxicity of boron to plants or soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

Data regarding the toxicity of boron to amphibians were unavailable in the literature reviewed.

An AWQC is not available for boron. No other potential criteria or guidelines were proposed in the literature reviewed (EPA 1996a; EPA 1992a).

Sediment quality criteria or benchmarks were unavailable in the literature reviewed (EPA 1996a; Persaud et al. 1992; Batts and Cubbage 1995; Long and Morgan 1991).

CADMIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Cadmium, a Group II metal, is commonly associated with zinc in sulfide and carbonate ores, and can occur as the byproduct of the refining of other metals (Bodek et al. 1988). The cation exchange capacity (CEC) of soil influences adsorption. Cadmium sorbs to iron and aluminum oxides, clay minerals, carbonate minerals, and to some extent organic matter in soils. Water hardness is a driving factor in determining the speciation and toxicity of cadmium in ambient waters, although pH can influence the concentration of dissolved cadmium (Bodek et al. 1988). Below a water pH of 6.0, cadmium tends to remain in solution. Although speciation of cadmium compounds is not affected by the concentration of organic ligands typically found in ambient waters, high levels of organic pollution can influence speciation; this situation is not expected to occur within the CMC boundaries. The concentration of dissolved cadmium decreases with increasing water hardness (Bodek et al. 1988), as reflected in the equation for the Ambient Water Quality Criterion (AWQC).

Some of the mechanisms for cadmium toxicity include erythrocyte, renal, and pulmonary damage (Bodek et al. 1988). Cadmium is readily bioavailable, and its toxicity results from a strong affinity for sulfhydryl groups, causing it to compete with zinc in binding with enzymes *in vivo*. It is not readily excreted, and is therefore bioaccumulated. Bioaccumulation factors for plants ranged from 0.04 to 2.24 on a wet weight basis (Carlson and Rolfe 1979; Miller et al. 1977), and those for soil invertebrates ranged from 1.25 to 10.57 (Diercxsens et al. 1985; Morgan and Morgan 1988; McKenna et al. 1992). Bioconcentration factors for various aquatic species ranged from a low of 8 to a high of 910 on a wet-weight basis (Jorgensen 1991).

TOXICITY TO TERRESTRIAL ORGANISMS

A study with sheep (Mills and Dalgamo 1972) was used to represent toxicity to mule deer (Table A-A), while a chronic study with rats exposed through diet was used to represent the toxicity to other mammalian receptors (Wilson et al. 1941). A 100 day study with mallard ducks was used to obtain a LOAEL and an NOAEL for avian receptors (Wilson et al. 1941)

Cadmium concentrations of 100 mg/kg soil decreased yield by 76% relative to controls in soybeans (Wallace 1989a). The maximum soil concentration from unimpacted field collected soils used in a laboratory phytotoxicity bioassay was 8.6 mg/kg (Kapustka et al. 1995). Recommended benchmark concentrations are as low as 3 mg/kg (Will and Suter 1995). The NEL for adverse effects on populations of isopods was 9 mg/kg soil, although individual effects of fewer gravid females and decreased size were observed (Donker et al. 1993).

TOXICITY TO AQUATIC ORGANISMS

The 48 h LC50 for *X. laevis* was 32 mg/L CdNO3, while 23 mg/L was the NOEC for mortality (Slooff and Baerselman 1980; Slooff et al. 1983). The 100 d LC50 for *Xenopus laevis* was 1.5 mg/L CdCl2, while the NOEC for mortality, body weight changes, somatic index was 0.03 mg/L (Canton and Slooff 1982). The EC50 for inhibition of larval development was 0.65 mg/L

(Canton and Slooff 1982). The 48 h LC50 for *Ambystoma mexicanum* was 1.3 mg/L CdNO3, while the NOEC for mortality was 1.10 mg/L (Slooff et al. 1983).

The acute AWQC for cadmium is hardness dependent, and the equation used to calculate the acute AWQC cadmium concentration for total metal in surface water is as follows (EPA 1990):

$$Chronic\ AWQC = e^{(1.128 * \ln hardness - 3.828)}$$

The chronic AWQC for cadmium is hardness dependent, and the equation used to calculate the chronic AWQC cadmium concentration for dissolved metal in surface water is as follows (EPA 1996a):

$$Chronic\ AWQC = e^{(0.7852 * \ln hardness - 3.490)} * 0.909$$

Sediment quality benchmarks range from a low of 0.6 mg/kg for the LEL (Persaud et al. 1992) to 10 mg/kg for the SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) reported an ERL of 5 and an ERM of 9 mg/kg on a dry weight basis.

CHROMIUM(III)

BIOLOGICAL AND ENVIRONMENTAL FATE

Chromium is widely found in the earth's crust, and adsorption to soil particles is influence by redox conditions, organic matter, and the presence of oxides. Adsorption of trivalent chromium increases with increasing pH. The speciation of chromium is highly dependent on pH of the ambient water.

Trivalent chromium is a necessary trace element for mammals. Hexavalent chromium is not only more toxic, but carcinogenic; however, hexavalent chromium is not expected to predominate under the environmental conditions occurring within the CMC boundaries. For example, in municipal waste waters, 97-99% of the dissolved chromium was in the trivalent form (Bodek et al. 1988). Chromium is bioaccumulated by aquatic organisms, and bioaccumulation is influenced by water hardness, temperature, and pH.

TOXICITY TO TERRESTRIAL ORGANISMS

A toxicity study with cats that resulted in a NEL (NAS 1974) was used to represent toxicity to all mammalian receptors except rodents (Table A-A). Toxicity to deer mice was represented by a study with rats that provided a NOAEL (NAS 1974). A field study with wild tern populations was used to represent toxicity to avian receptors (CEPA 1994a). The chromium concentration in major dietary items was measured as 7.6 mg/kg, and compared to population metrics of reproduction and population success. The data were used to estimate a daily dose of 1.28 mg/kg-d (CEPA 1994a).

Under certain soil conditions, 150 mg/kg chromium may inhibit growth in sensitive plants (CEPA 1994a). A chromium concentration of 200 mg/kg inhibited growth by 23-36% in grasses, lettuce, and radish (CEPA 1994a). A benchmark value of 1 mg/kg has been suggested as a threshold concentration protective of plant species (Will and Suter 1995). A benchmark value of 0.4 mg/kg has been suggested as protective of soil fauna (Will and Suter 1995)

TOXICITY TO AQUATIC ORGANISMS

The 5 h LC50 for *G. carolinensis* was 0.03 mg/l (Birge et al. 1979). The 96 h LC50 for *Bufo melanostictus* was 49.29 mg/l (Khangarot and Ray 1987).

The acute AWQC is 1.7 mg/l at a hardness of 100 mg/l CaCO3 (EPA 1992a). The AWQC for chromium is hardness dependent. The equation used to calculate the acute AWQC total chromium concentration is as follows (EPA 1990):

$$\mathit{Chronic\ AWQC} = e^{(0.8190 * \ln \mathit{hardness} + 3.688)}$$

Toxicity Summary

The equation used to calculate the chronic AWQC dissolved chromium concentration is as follows (EPA 1996a):

$$Chronic\ AWQC = e^{(0.8190 * \ln hardness + 1.561)} * 0.860$$

Sediment quality benchmarks for chromium range from a low of 26 mg/kg for the LEL to 110 mg/kg for the SEL (Persaud et al. 1992). Long and Morgan (1991) reported an ERL of 80 and an ERM of 145 mg/kg on a dry weight basis.

COBALT

BIOLOGICAL AND ENVIRONMENTAL FATE

Data regarding the biological or environmental fate of cobalt were unavailable in the literature reviewed.

TOXICITY TO TERRESTRIAL ORGANISMS

A short-term study with rats exposed to cobalt in diet resulted in a LOAEL for survival of offspring (Domingo 1994); this value was used to represent toxicity to mammalian receptors (Table A-A). This study provided a NOAEL for maternal toxicity that was based on endpoints of weight gain and food consumption; however, adverse effects on pup survival and development were observed when cobalt was given on days 14-21 gestation. Pup survival was affected at a maternal dose of 12 mg/kg-d. (Domingo 1994).

A study with ducklings exposed to cobalt at a concentration of 0.2% of the diet (Friberg et al. 1979) was used to represent toxicity to avian receptors. The endpoint evaluated in this study was a lowest effect level (LEL) for growth (Table A-A). The LEL for adverse effects on growth was 0.2% of diet. The dietary concentration was converted with an ingestion rate of 1.26 kg/kg-d estimated from an allometric equation (EPA 1993) and assumed body weight of 100 g as follows:

$$Ingestion\ Rate\,(kg\,/\,kg-d) = \frac{0.495 * BW^{(0704)}}{BW}$$

A soil concentration of 75 mg/kg decreased yield by 50% and produced iron deficiency in corn (Wallace 1989b). Data regarding the toxicity of cobalt to soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

The 5 h LC50 for *G. carolinensis* exposed from fertilization to 4 days post-hatching was 0.05 mg/l (Birge et al. 1979).

An acute or chronic AWQC is unavailable for cobalt (EPA 1992a). However, a Tier II value of 0.003 mg/l has been suggested (EPA 1996a).

Sediment quality benchmarks were unavailable in the literature reviewed (EPA 1996a; Persaud et al. 1992; Long and Morgan 1991). Batts and Cubbage (1995) report a value of 50 mg/kg on a dry weight basis which is equivalent to a LEL.

COPPER

BIOLOGICAL AND ENVIRONMENTAL FATE

Copper occurs most often as the sulfide or oxide form in the ambient environment, and is strongly sorbed by hydrous iron and manganese oxides, clay minerals, and organic matter in soils. Copper complexes with inorganic and organic ligands in aqueous environments, where the +2 valence is most common at a pH less than 7 and also under aerobic conditions (Bodek et al. 1988). Copper sulfates, oxysulfates, and phosphates are very soluble, whereas the oxides and sulfides are not. Water hardness is a driving factor influencing toxicity and availability of copper, as indicated the hardness dependent AWQC.

Copper is an essential nutrient for aquatic and terrestrial plants and animals. Because it is an essential element, it is readily absorbed by aquatic and terrestrial biota. Plants and animals have regulatory mechanisms that regulate internal concentrations according to metabolic needs. However, copper can be toxic at concentrations that are not much greater than those required for nutrition.

The bioavailability of copper depends greatly on the chemical species present. For example, the cupric ion is considered more bioavailable and, therefore, potentially more toxic than other copper species. For many metals, free metal activity in soil solution is thought to be responsible for most toxic effects in terrestrial environments (Sauvé et al. 1998). The free metal ion activity is typically a small fraction of the total metal content of soils. Free Cu^{+2} activity depends on the adsorption-desorption from surfaces of soil solids, exchange reactions with solid organic matter, and complexation with dissolved organic and inorganic ligands (Sauvé et al. 1998). These processes are, in turn, highly dependent upon pH.

The bioavailability of copper from plants to herbivores is affected by the fiber and mineral content (other than copper) of the ingested material (Suttle 1986). Copper availability is lowest in high content forage materials such as grasses, and higher in lower fiber foods such as cereals. Approximately 10 percent of copper content is absorbed by sheep from cereals, whereas 2.5 percent is available from grasses (Suttle 1974 as cited in Suttle 1986). Molybdenum, sulfur, and iron inhibit absorption and metabolism of copper.

Copper has limited potential for bioaccumulation in terrestrial species. Bioaccumulation factors of 0.01 to 2.32 on a wet weight basis were reported for various species of soil invertebrates (Morgan and Morgan 1988; Diercxsens et al. 1985; Jorgensen 1991; Donker et al. 1993).

The chemical speciation of copper in the aquatic environment is important because the chemical species may have differing toxicity to aquatic organisms. The divalent cupric ion is often considered the most toxic form to aquatic animals (Florence 1982; Chakoumakos et al. 1979; Howarth and Sprague 1978; Lauren and McDonald 1986; Pagenkopf et al. 1974; Stiff 1971a, b; Sylva 1976. Copper-hydroxy complexes are also considered to be toxic (Florence 1982). Bioconcentration factors expressed on a wet weight basis for aquatic plants ranged from 100 to 4830, and for aquatic invertebrates the bioconcentration factors ranged from 167 to 3800 (Jorgensen 1991).

TOXICITY TO TERRESTRIAL ORGANISMS

Exposure to copper has been found to cause injuries to liver and kidney tissue in birds and mammals (Walsh et al. 1977, Forman et al. 1980). Copper accumulations in liver and kidney tissues is associated with histopathological and functional effects (Haywood et al. 1985, Sawaki et al. 1994). Developmental effects of copper are indicated by reduced motility of human spermatozoa after *in vitro* incubation with copper (Friberg et al. 1986), toxicity of copper to the blastocyst stage of mouse embryos (Shepard 1986).

A study using sheep was used to represent the toxicity of copper to ruminants (James et al. 1966)(Table A-A). This chronic (46-145 days) study evaluated reproductive effects due to 10 mg/kg bw copper sulfate exposure by daily ingestion of copper sulfate in gelatin capsules. One of four ewes aborted at this dosage. The 10 mg/kg bw dosage is identified as the NOAEL for ruminants.

A study exposing mice by dosing by copper sulfate in food and water (*ad libitum*) was used to represent toxicity to deer mice (Hebert et al. 1993).

A chronic study with mink was used to represent toxicity to mammalian carnivorous receptors (Aulerich et al. 1982). A dietary exposure of 25-50 mg copper/kg diet is a NEL, and may even be beneficial. However, 100 mg/kg in diet resulted in decreased weight gain, and may have increased kit mortality due to effects on lactation (Aulerich et al. 1982). Control animals tested in this study were exposed to 60.5 mg/kg in diet for a study duration of 357 days. The dietary concentrations were converted with a dietary ingestion rate of 0.13 kg/kg-d (EPA 1993).

Sublethal copper phytotoxicity is manifested as decreased yields in crop plants, reduced root and shoot growth, chlorotic foliage, reduced generation of root laterals, and early leaf fall (Påhlsson 1989). Seed germination is relatively insensitive to copper, but early growth of root and shoot from newly germinated seeds is sensitive to copper concentrations. Adverse physiological responses to elevated copper include reduced rates of photosynthesis and respiration and increased permeability of root cells and subsequent loss of potassium (Påhlsson 1989).

Concentrations of 100 mg/kg have been reported to be phytotoxic in soils (ICF 1989). Other studies indicate a recommended benchmark value of 100 mg/kg (Will and Suter 1995). Higher concentrations may be tolerated, however, as indicated by data reported in Kapustka et al. (1995). In this study, soils reported as being unimpacted and nonphytotoxic contained copper concentrations as high as 1062 mg/kg. Local populations of some plant species are known to develop tolerance to copper concentrations areas around copper mines and on copper mine spoils (Kruckeberg and Wu 1992, Lin and Wu 1994)

Copper concentrations of 84 mg/kg resulted in a NEL for adverse effects on populations of isopods, although individual effects such as fewer gravid females and decreased size were observed (Donker et al. 1993). Soil copper concentrations of 100 mg/kg altered the structure of the soil microarthropod community; 400 mg/kg would decrease function as total numbers decreased (Parmelee et al. 1993).

TOXICITY TO AQUATIC ORGANISMS

Copper exposures have been found to cause liver cell necrosis, including fragmentation of the endoplasmic reticulum, damage to nuclear components, and rupture of plasma and nuclear membranes in fish (Leland 1983). Skeletal calcification of brown trout fry is impaired at copper concentrations of 5μg/L (Reader et al. 1989). The exposure of trout to copper can also result in lethality (Reader et al. 1989, McKim and Benoit 1971), behavioral avoidance (Giattina et al. 1982), and reduced growth (Finlayson and Verrue 1980, Dixon and Spraygue 1981).

The 48 h LC50 for *X. laevis* was 1.7 mg/L CuSO4 (de Zwart and Slooff 1987). Data for other amphibian species were unavailable in the literature reviewed. The acute and chronic AWQC for copper are hardness dependent. The equation used to calculate the acute AWQC total copper concentration is as follows (EPA 1990):

$$Chronic\ AWQC = e^{(0.9422*\ln hardness-1.464)}$$

The equation used to calculate the chronic AWQC dissolved copper concentration is as follows (EPA 1996a):

$$Chronic\ AWQC = e^{(0.8545*\ln hardness-1.465)} * 0.960$$

Sediment quality benchmarks range from a low of 16 for the LEL (Persaud et al. 1992) to 110 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) present an ERL of 70 mg/kg and an ERM of 390 mg/kg expressed on a dry weight basis.

LEAD

BIOLOGICAL AND ENVIRONMENTAL FATE

Lead occurs in many forms in the environment, but the predominant one is galena, a fairly insoluble ore (Bodek et al. 1988). Lead sorbs to soils and sediments by ion exchange, incorporation into cationic lattices, and other mechanisms. In soils, lead sorbs to organic matter, clay minerals, and oxides of manganese and iron. Sorption in soils tends to increase with increasing pH (Bodek et al. 1988). In surface water, the dominant species under acidic conditions is Pb^{+2}, whereas $PbOH^{+}$ occurs at typical pH ranges. Complexation with organic and inorganic ligands in aquatic systems occurs (Bodek et al. 1988). Water hardness is a driving factor influencing toxicity and availability of lead, as indicated the hardness dependent AWQC.

Lead is not a necessary nutrient, and lead bioaccumulates to some extent in terrestrial species. Bioaccumulation factors of 0.02 to 2.1 on a wet weight basis were reported for various plant species (Bodek et al. 1988; Carlson and Rolfe 1979; Miller et al. 1977; Motto et al. 1970). Bioaccumulation factors ranged from 0.002 to 36.5 for various species of soil invertebrates (Morgan and Morgan 1988; Morgan and Morgan 1991; Diercxsens et al. 1985; Donker et al. 1993; Beyer and Cromartie 1987). Birds accumulate relatively low concentrations of lead into soft tissues, as evidenced by bioaccumulation factors of 0.0001 to 0.0005 for kestrel liver (Pattee 1984). Mammals also accumulate relatively low amounts of lead into soft tissue; bioaccumulation factors ranged from 0.000001 to 0.19 for mouse and vole livers (Kisssberth et al. 1984; Shore 1995; Stansley and Roscoe 1996).

There is little evidence that lead methylates under ambient environmental conditions; however, organic lead compounds have been detected in fish. Bioconcentration factors expressed on a wet weight basis for aquatic plants ranged from 41000 to 70000 (Jorgensen 1991), and for aquatic invertebrates the bioconcentration factors ranged from 75 to 197000 (Jorgensen 1991). Bioconcentration factors for fish ranged from 65 to 75 (Jorgensen 1991). Any concentration data on a dry weight basis were converted to fresh weight by dividing invertebrate bioconcentration factors by a factor of 6, and plant concentration factors by a conversion factor of 6.7.

TOXICITY TO TERRESTRIAL ORGANISMS

A subacute study with calves was used to represent the toxicity of copper to mule deer (Table A-A). This seven day study provided a NOAEL due to exposure to lead in diet (Zmudzki et al 1984). A multigenerational study exposing rats by dosing with lead in diet was used to represent toxicity to deer mice (Stowe and Goyer 1971); a LOAEL but no NOAEL was reported for this study. A chronic study with dogs exposed to lead in the diet was used to represent toxicity to mammalian carnivorous receptors (Horwitt and Cowgill 1939). This study was limited by small sample sizes (n=14), and because animals were fed lead acetate. However, sensitive endpoints including clinical chemistry, hematology, and growth were tracked from about 30 to 200 days of age. A 10 week study exposing chickens to lead acetate in diet was used to represent the toxicity of lead to birds (Edens and Garlich 1983). This study examined the reproductive effects of lead exposure, and provided a NOAEL. However, the study is limited since lead acetate has a higher bioavailability than inorganic lead, and test concentrations were nominal.

Phytotoxicity of soils is probable above 494 mg/kg, which is a median from studies reported by EPA (1992b). Lower concentrations may be more appropriate as a conservative screening benchmark, as indicated by data reported in Kapustka et al.(1995). In this study, soils reported as being unimpacted and nonphytotoxic contained lead concentrations only as high as 179 mg/kg. Lead concentrations of 606 mg/kg resulted in a NEL for adverse effects on populations of isopods, although individual effects such as fewer gravid females and decreased size were observed (Donker et al. 1993). Soil lead concentrations of 95 mg/kg were reported as a no observed effect concentration (NOEC) for unspecified effects in soil invertebrates (ICF 1989).

TOXICITY TO AQUATIC ORGANISMS

The 48 h LC50 for the toad *Bufo arenarum* ranged from 0.47 to 0.9 mg/L Pb as PbNO3. There were no survivors at concentrations of 8 mg/L and higher (Perez-Coll et al. 1988). Data for other amphibian species were unavailable in the literature reviewed.

The acute and chronic AWQC for lead are hardness dependent. The equation used to calculate the acute AWQC total lead concentration is as follows (EPA 1990):

$$ChronicAWQC = e^{(1.273*\ln hardness-1.460)}$$

The equation used to calculate the chronic AWQC dissolved lead concentration is as follows (EPA 1996a):

$$ChronicAWQC = e^{(1.273*\ln hardness-4.705)} * 0.791$$

Sediment quality benchmarks range from a low of 31 for the LEL (Persaud et al. 1992) to 250 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) present an ERL of 35 mg/kg and an ERM of 110 mg/kg expressed on a dry weight basis.

MANGANESE

BIOLOGICAL AND ENVIRONMENTAL FATE

Manganese is a necessary nutrient found in all plants and animals (Bodek et al. 1988). Manganese occurs in multiple valence states, but Mn^{+2} is the most stable aqueous form. Redox conditions and pH affect the speciation in surface water.

Manganese is easily translocated to the vegetative portion of plants (Bodek et al. 1988). The predicted bioaccumulation factor for the vegetative portion in food crops and forage is 0.25 (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

The toxicity benchmarks for manganese were derived from a study with mice (NTP 1993). Mice were fed manganese (II) sulfate monohydrate in diet for up to 2 years. There were 70 male and 70 female mice fed either a 0, 1500, 5000, or 15000 ppm diet. This was equivalent to 0, 160-200, 540-700, and 1800-2250 mg/kg bw/d. Survival, body weight, food consumption, and clinical chemistry were among the endpoints that were reported. Body weight was lower in the 1500, 5000, and 15000 ppm groups by 6, 9, and 13%, respectively, relative to the controls. The study provided both NOAELs and LOAELs, and was used to represent all mammalian receptors (Table A-A). A chronic study with Japanese quail that provided a NOAEL was used to represent toxicity to avian species (Laskey and Edens 1985).

A recommended benchmark for plants is 500 mg/kg (Will and Suter 1995).

TOXICITY TO AQUATIC ORGANISMS

A NOAEL and LOAEL for slight effects on morphogenesis were obtained for *Xenopus laevis* (Miller and Landesman 1978). Elevated magnesium concentrations reduced manganese toxicity. At a Mg level of only 20 mg/L, there were no effects on percent survival and a NOAEL of 5 mg/L was obtained. The corresponding LOAEL for manganese at a Mg concentration of 20 mg/L was 10 mg/L. However, at higher levels of magnesium (i.e., 200 mg/L Mg), higher concentrations of manganese were tolerated, and 10 mg/L Mn was the NOAEL.

A Tier II value of 80 µg/L was available as the only water quality criterion (EPA 1996a).

Sediment quality benchmarks range from a low of 460 for the LEL (Persaud et al. 1992) to 1100 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). A sediment effect concentration (SEC) of 726 mg/kg on a dry weight basis was also available (EPA 1996b). This value was an ERL obtained from a 14 and 28 day test with *Hyalella azteca*.

MERCURY

BIOLOGICAL AND ENVIRONMENTAL FATE

Mercury tends to occur in conjunction with sulfides (Bodek et al. 1988). The environmental chemistry of mercury is complex due to the various oxidation states, the fact that under certain conditions methylation occurs, and because some forms are also volatile. Mercury sorbs to soils and sediments onto iron and manganese oxides, organics, and clays (Bodek et al. 1988). Methylation occurs under both anaerobic and aerobic conditions, although methylation rates are lower aerobically than anaerobically. The pH, temperature, redox potential, and the microbial population all influence methylation rates (Bodek et al. 1988).

Bioaccumulation factors of 0.005 to 0.009 on a wet weight basis were reported for plants, and bioaccumulation factors ranging from 0.88 to 2.35 were reported for various species of soil invertebrates (John 1972; Talmage and Walton 1993). The relationship of plant mercury content to the concentration in soil is complex; twinberry (*Lonicera involucrata*), *Spiraea douglasii*, and squaw current (*Ribes cereum*) showed relatively high (>55%) correlations of root and/or leaf mercury content with the surrounding soils (Ellis and Eslick 1997). Orchard grass (*Dactyliis glomerata*) roots and stems exhibited a high correlation to soil mercury, whereas only the root concentrations were highly correlated to soil concentrations for redtop bentgrass (*Agrostis alba*) and wheatgrass (*Agropyron spicatum*). Other species including Kentucky bluegrass (*Poa pratensis*) and willow (*Salix sp.*) did not show strong correlations with mercury in soil (Ellis and Eslick 1997). This study suggests that soil concentrations exceeding 0.99 mg/kg could influence species composition. This ultimately could lead to changes in ecological structure or function, depending on the plant species affected.

Uptake data for avian species were available for both duck and hawk liver and muscle. Bioaccumulation factors for bird tissues ranged from 0.103 to 7.69 (Finley and Stendall 1978; Lindsay and Dimmick 1983; Fimreite and Karstad 1971). The avian liver accumulated more mercury than muscle by factors ranging from approximately 2 to 5. Reported values for sheep, mouse, or shrew liver or kidney ranged from 0.57 to 4.4 (Eisler 1987; Talmage and Walton 1993).

Bioconcentration factors expressed on a wet weight basis for aquatic plants ranged from 104 to 2435, and for aquatic invertebrates the bioconcentration factors ranged from 20 to 579 (Jorgensen 1991). Fish concentrate mercury as well; reported bioconcentration factors ranged from 4 to 3400 on a wet weight basis (Jorgensen 1991). Any concentration data on a dry weight basis were converted to fresh weight by dividing invertebrate bioconcentration factors by a factor of 6, and plant concentration factors by a conversion factor of 6.7.

TOXICITY TO TERRESTRIAL ORGANISMS

Revis et al. 1989 reported a NOAEL for a 20 month exposure of mice to mercuric sulfide (Table A-A). This NOAEL was used to represent toxicity to both mule deer and deer mouse. A NOAEL for mink for mercuric chloride based on various reproductive endpoints for a six month exposure was reported by Aulerich et al.(1974). This study is limited since only one dietary concentration of inorganic mercury and one of organic mercury were tested. Organic mercury is more toxic than inorganic mercury. The 5 ppm organic mercury in diet was toxic to mink from the 25th day of exposure, and caused clinical symptoms and death. The 10 ppm inorganic mercury produced no effects throughout the duration of the study. There were 15 animals tested at each dose and the control. The authors estimated the daily dose of organic mercury at 18 mg, and inorganic mercury at 233 mg, for female mink that weighed approximately 1 kg. This study was used as the basis of the toxicity benchmark for mammalian carnivores. A 1 year reproductive study with Japanese quail provided both a NOAEL and a LOAEL for exposure to mercuric sulfide (Hill and Schaffner 1976). This study was used to represent toxicity to avian receptors.

A concentration of mercury in sludge identified as potentially toxic to plants and animals was 10 mg/kg (ICF 1989). Ellis and Eslick (1997) suggest that soil concentrations exceeding 1 mg/kg on a dry weight basis could influence plant species composition or abundance, as supported by data for relative frequency per square meter along selected transects. Abundance decreased by as much as 80% for *Spirea douglasii* and *Poa pratensis* above 1 mg/kg mercury in soil (Ellis and Eslick 1997). The no effect level (NEL) for normal regeneration in earthworms was 1 mg/kg, and the lower effect level (LEL) was 5 mg/kg (Eisler 1987).

TOXICITY TO AQUATIC ORGANISMS

The 96 h LC50 for the toad *Bufo punctatus* was 0.0368 mg/L; a 96 h LC50 for *B. fowleri* of 0.0659 mg/L was also reported (Birge et al. 1983). Slooff and Baerselman (1980) provided a 48 h LC50 of 0.4 mg/L for exposure to mercuric chloride for the salamander *Ambystoma mexicanum*. Data for other amphibian species were unavailable in the literature reviewed.

The final chronic value (FCV) for inorganic mercury is 1.3 μg/L (EPA 1996a). Organic mercury is more toxic as indicated by a Tier II value of only 0.003 μg/L (EPA 1996a).

Sediment quality benchmarks range from a low of 0.2 for the LEL (Persaud et al. 1992) to 2 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) present an ERL of 0.15 mg/kg and an ERM of 1.3 mg/kg expressed on a dry weight basis.

MOLYBDENUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Molybdenum occurs in soluble forms, and also adsorbed to iron in soils (Bodek et al. 1988). Molybdenum solubility decreases in acid soils.

Molybdenum is a necessary nutrient that is found in most tissues (Bodek et al. 1988) Concentration factors for food crops and forage plants are expected to be 0.25 (Bodek et al. 1988). Uptake by vegetation is enhanced by high pH and saturated soils, and is decreased when the pH is <5.5 or in the presences of iron oxides (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

A multi-generational study with mice was used to represent the toxicity of molybdenum to mammals (Schroeder and Mitchner 1971) (Table A-A). This study provided a LOAEL with reproductive effects as the endpoint. This study is limited since the route of administration was by drinking water, which could alter bioavailability in comparison to exposure via the diet. The authors reported that molybdenum at 10 ppm in drinking water was only slightly toxic as evidenced by significantly more mortality in the F1 and F3 generations, and more runts in the F3 generation, relative to controls. A 21 day study with chickens was used to represent toxicity to avian receptors (Lepore and Miller 1965). This study provided a LOAEL with reproductive effects as the endpoint (Table A-A).

Plant deficiency occurs at plant tissue concentrations of <0.1 to 0.3 mg/kg on a dry weight basis, whereas toxic levels are in the range of 10 to 100 mg/kg (Bodek et al. 1988). Other data on the toxicity of molybdenum to plants or soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

Data on the toxicity of molybdenum to amphibians were unavailable in the literature reviewed. A Tier II value for the protection of freshwater aquatic life of 240 µg/L was reported (EPA 1996a). Sediment quality benchmarks were unavailable in the literature reviewed.

NICKEL

BIOLOGICAL AND ENVIRONMENTAL FATE

In the ambient environment, nickel ores occur in the form of oxides, silicates, and sulfides (Bodek et al. 1988). Nickel is a very mobile heavy metal, although it will sorb to soils, iron and manganese oxides, and clay minerals. Humic acids alter the chemical speciation of nickel in surface water, and can enhance the amount in solution. Toxicity is influenced by water hardness, as indicated by the hardness dependent AWQC.

Nickel may be essential at very low levels (Bodek et al. 1988). Bioaccumulation factors for plants are predicted to be 0.06 for food crops and forage plants (Bodek et al. 1988). The soluble forms of nickel are readily taken up by roots, and nickel can accumulate in seeds and leaves of plants. A pH below 6.0 promotes uptake by plants, whereas lime, iron, and organic matter appear to decrease uptake (Bodek et al. 1988). Bioconcentration factors in freshwater invertebrates and fish are expected to be about 0.1 to 78, where uptake is higher for whole body measurements than for muscle (Bodek et al. 1988). Bioconcentration factors of about 10 have been reported for freshwater algae (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

A multi-generation study with rats provided a NOAEL and a LOAEL (Ambrose et al. 1976) and was used to represent the toxicity of nickel to all mammalian receptors (Table A-A). The endpoint measured in this study was the body weight of offspring. A study by Cain and Pafford (1981) with mallard ducklings provided a NOAEL and LOAEL for behavior and growth in a 90 day exposure (Table A-A). This study was used to represent all avian receptors.

Nickel is toxic to lettuce and wheat at levels of 250 to 270 mg/kg in soil, where 25% decreased growth was observed in calcareous soils. In acid soil, the same effect was observed at only 75 to 110 mg/kg (CEPA 1994b). Soil concentrations of 50 mg/kg decreased yield in oats by 16 to 31% at pH 5.7 (CEPA 1994b). Reduced growth and reproduction was observed in earthworms at soil concentrations of 500 mg/kg (ICF 1989).

TOXICITY TO AQUATIC ORGANISMS

Data regarding the toxic effects of nickel to amphibians were unavailable in the literature reviewed. The acute and chronic AWQC for nickel are hardness dependent. The equation used to calculate the acute AWQC total nickel concentration is as follows (EPA 1990):

$$Chronic\ AWQC = e^{(0.8460*\ln \mathit{hardness}+3.3612)}$$

The equation used to calculate the chronic AWQC dissolved nickel concentration is as follows (EPA 1996a):

$$Chronic\ AWQC = e^{(0.8460*\ln \mathit{hardness}+1.1645)} * 0.997$$

Toxicity Summary

Sediment quality benchmarks range from a low of 16 for the LEL (Persaud et al. 1992) to 75 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) present an ERL of 30 mg/kg and an ERM of 50 mg/kg expressed on a dry weight basis.

SELENIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Naturally occurring sulfides or sulfur deposits often contain selenium, and sulfides of iron, mercury, copper, silver, lead and zinc can contain up to 20% selenium (Bodek et al. 1988). Selenium in the environment occurs in four oxidation states, and exhibits anionic behavior in aqueous solutions. Elemental selenium is the most stable form over a wide range of pH and redox conditions, and is relatively insoluble in water. Low pH and reducing conditions enhance the presence of elemental selenium.

Selenium is a micronutrient in animals; its role in plant nutrition is uncertain (Bodek et al. 1988). Water soluble forms of selenium in soils are bioavailable since they do not tend to sorb to iron oxides (Bodek et al. 1988). Plant roots take up water soluble species, and thereby concentrate selenium, which is then distributed to shoots and seeds (Bodek et al. 1988). Uptake is enhanced by alkaline, well oxidized soil conditions, and decreased under acidic or neutral conditions where selenites or selenides fail to form (Bodek et al. 1988). While some plants are hyperaccumulators, typical bioaccumulation factors are expected to be about 0.025 (Bodek et al. 1988). Selenium is concentrated at relatively low rates by aquatic animals; bioconcentration factors are expected to be 0.167 (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

The TRV for mule deer was derived from the concentration tolerated in the diet of grazing animals, which is reported as 2 mg/kg dry weight (Bodek et al. 1988)(Table A-A). This value was converted to a daily intake of 0.08 mg/kg bw/d by applying an ingestion rate of 0.04 kg/kg bw/d (Sax 1984). A study providing a dietary LOAEL based on reproductive effects was used to represent toxicity to the other mammalian receptors (Rosenfeld and Beath 1954). Heinz et al.(1989) reported a NOAEL and a LOAEL for effects on mallard ducks and their offspring. Adult ducks fed 16 mg/kg selenium in diet exhibited 100% decreased hatching success in a 100 d study, 8 mg/kg increased the number of malformed embryos. A NOAEL of 4 mg/kg and a LOAEL of 8 mg/kg selenium in diet was obtained. Dietary concentrations were converted to daily intakes by multiplying by a daily food ingestion rate of 0.063 kg/kg bw/d from EPA (1993). This study was used as the TRV for all avian receptors.

The concentration of selenium in plant tissues that is considered toxic to the plant ranges from 5 to 100 mg/kg dry weight (Bodek et al. 1988). Soil concentrations as low as 1 mg/kg have been observed to reduce growth rate in wheat and buckwheat (ICF 1989). A recommended soil threshold for the protection of soil fauna is 70 mg/kg (Will and Suter 1995).

TOXICITY TO AQUATIC ORGANISMS

The 48 h LC50 for *Gastrophryne carolinensis* embryo-larval stage was 0.09 mg/L; test included exposure from fertilization to 4 days post-hatch (Birge et al. 1979). Browne and Dumont (1979) reported a 48 hr LC50 for *Xenopus laevis* tadpoles of 8.04 mg/L. For longer exposures, median survival times were 4.7, 4.0, and 2.5 d in 2, 5, and 10 mg/L, respectively.

Toxicity Summary

The acute AWQC is 20 µg/L, and the chronic AWQC is 5 µg/L (EPA 1990; EPA 1996a). Sediment quality criteria were unavailable in the literature reviewed (EPA 1996a; Long and Morgan 1991; Persaud et al. 1992).

SILVER

BIOLOGICAL AND ENVIRONMENTAL FATE

In the environment, silver is typically associated with sulfides, such as those of copper, lead, and arsenic (Bodek et al. 1988). Most silver compounds, with the exception of the silver nitrate compounds, are relatively insoluble. Bioaccumulation factors for plants are expected to be about 0.4 (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

A NOAEL for behavior and drinking water ingestion was obtained from a study with rats that were exposed to silver in their drinking water (Walker 1971) and from a study with pigs where overt toxicity and growth depression were observed (Van Vleet 1976) (Table AA). The NOAEL was 6 mM AgNO3, whereas 24 mM was the LOAEL for appearance, behavior, fluid consumption, and mortality for the 12 week study. For longer exposure durations, 12 mM AgNO3 was toxic after 76 weeks, with rats exhibiting poor clinical condition (Walker 1971). The pigs were fed a diet containing 2% silver acetate, which was converted to a daily intake with an ingestion rate of 0.034 kg/kg bw/d (Wiseman 1987). The toxicity to avian species was represented by data for turkeys which provided a LEL for cardiac effects and mortality in an 18-week exposure (Friberg et al. 1979).

A recommended benchmark for the protection of terrestrial plants is 2 mg/kg (Will and Suter 1995). Data were unavailable in the literature reviewed for soil fauna.

TOXICITY TO AQUATIC ORGANISMS

The 96 h LC50 for the toad *Bufo melanostictus* is 0.0041 mg/L (Khangarot and Ray 1987). Data for other amphibian species were unavailable in the literature reviewed.

The chronic AWQC (EPA 1980) was 0.12 μg/L for silver; this value was withdrawn by the EPA (EPA 1997). For sediment criteria, Long and Morgan (1991) suggest an ERL of 1 mg/kg and an ERM of 2.2 mg/kg on a dry weight basis.

THALLIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Thallium is found as an impurity to other metal ores (Bodek et al. 1988). It occurs in two oxidation states, and strongly sorbs to clay minerals in soils. Thallium was banned as a rodenticide in 1972 (Bodek et al. 1988). Thallium compounds can be water soluble and adsorption and biotic uptake are primary removal mechanisms of thallium from solution.

Thallium acts by replacing potassium in enzymes (Bodek et al. 1988). The expected bioaccumulation factor for terrestrial plants is 5.5×10^{-3} (Bodek et al. 1988). Thallium is concentrated by aquatic life; algae *Ulothrix sp* concentrated thallium by factors of 127 to 220 (Bodek et al. 1988). Thallium appeared to be irreversibly bound to the algal tissues, indicating a potential for herbivores to be exposed by dietary ingestion.

TOXICITY TO TERRESTRIAL ORGANISMS

The toxicity of thallium to mammalian receptors was represented by a study with rats that provided a LOAEL (Roll and Matthiaschk 1981) (Table A-A). Rats that were gavaged on gestational days 6-15 had a slight increase in postimplantation fetal loss. Hudson et al. (1984) provided an LD50 value for ring-necked pheasant that was used to represent toxicity to all avian receptors.

A recommended benchmark for the protection of terrestrial plants is 1 mg/kg (Will and Suter 1995). Data were unavailable in the literature reviewed for soil fauna.

TOXICITY TO AQUATIC ORGANISMS

The 48 hr LC50 for *Gastrophryne carolinensis* was 0.11 mg/L; exposure was from fertilization to 4 days post-hatch (Birge et al. 1979). Frogs died upon emergence following exposure to thallium (Dilling and Healey 1926). Data for other amphibian species were unavailable in the literature reviewed.

AWQC for thallium are unavailable (EPA 1996a). Sediment quality criteria were unavailable in the literature reviewed (EPA 1996a; Long and Morgan 1991; Persaud et al. 1992).

URANIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

Uranium occurs in several valence states, although in nature it exists primarily in the +4 and +6 states (Bodek et al. 1988). There are two naturally occurring isotopes of uranium of primordial origin, ^{235}U and ^{238}U (Bodek et al. 1988). There is also a daughter of ^{238}U (^{234}U), and a decay product (^{235}U). ^{238}U decays by alpha emission. The expected bioaccumulation factor for terrestrial plants is 8.5×10^{-3} (Bodek et al. 1988) as indicated by modeled results for forage and food crops.

TOXICITY TO TERRESTRIAL ORGANISMS

A study with mice that exposed the animals prior to and during gestation, through delivery and lactation (Paternain et al. 1989), was used to represent the toxicity to mammalian receptors. The route of administration was by oral gavage, which may have enhanced bioavailability and thus toxicity relative to that expected due to dietary exposure. All of the available studies for birds exposed the test subjects by injection (Kupsh et al. 1991). This route of administration was considered inappropriate as the basis of a TRV since bioavailability is 100% by injection. Therefore, a TRV for avian receptors is unavailable.

Data regarding the toxicity of uranium to plants or soil fauna were unavailable in the literature reviewed.

TOXICITY TO AQUATIC ORGANISMS

Uranyl ion had no effect on the electrical properties of amphibian cells *in vitro* (Kanno et al. 1978). Other toxicity data for amphibians were unavailable in the literature reviewed.

AWQC for uranium are unavailable (EPA 1996a).). Sediment quality criteria were unavailable in the literature reviewed (EPA 1996a; Long and Morgan 1991; Persaud et al. 1992).

VANADIUM

BIOLOGICAL AND ENVIRONMENTAL FATE

The expected bioaccumulation factor for terrestrial plants is 5.5×10^{-3} (Bodek et al. 1988) as indicated by modeled results for forage and food crops. Bioconcentration factors for aquatic invertebrates are expected to be about 3, whereas those for freshwater fish are expected to be only 0.01 (Bodek et al. 1988).

TOXICITY TO TERRESTRIAL ORGANISMS

A chronic study with rats cited in Friberg et al. (1979) was used to represent toxicity to mammalian receptors (Table A-A). This study reported a LEL for effects on reflexes; rats were exposed by ingestion of drinking water. A study with chickens exposed to vanadium in diet was used to represent toxicity to birds (Kubena et al.,1986). This study was 4 weeks long, and reported that 25 mg/kg vanadium in diet had no effect on growth rate.

A recommended benchmark for the protection of terrestrial plants is 2 mg/kg (Will and Suter 1995). The suggested threshold for the protection of soil fauna is 20 mg/kg (Will and Suter 1995).

TOXICITY TO AQUATIC ORGANISMS

Toxicity data for amphibians were unavailable in the literature reviewed. AWQC for vanadium are unavailable; however, a Tier II value of 19 μg/L has been proposed (EPA 1996a). Sediment quality criteria were unavailable in the literature reviewed (EPA 1996a; Long and Morgan 1991; Persaud et al. 1992).

ZINC

BIOLOGICAL AND ENVIRONMENTAL FATE

Zinc occurs as various species, but predominant forms are carbonates, oxides, silicates, and sulfides (Bodek et al. 1988). Zinc sorbs to soils, but is readily bioavailable to plants. The aquatic toxicity of zinc is hardness dependent.

Zinc is a necessary nutrient. Zinc deficient animals absorbed a higher proportion of dietary zinc than normal animals, and retention of zinc in tissues was higher and excretion lower than in normal animals (Miller et al. 1968). Zinc deficiency in cattle occurred at dietary concentrations of 2 mg/kg, a typical diet contained 18.7 mg ZnO/kg diet, whereas a zinc enriched diet contained 36 mg/kg zinc (Miller et al. 1968).

Uptake by plants is enhanced by low soil pH, and reduced by high clay content, high cation exchange capacity, high pH, and high phosphate levels in soil (Bodek et al. 1988). The expected bioaccumulation factor for terrestrial plants is 15 as indicated by modeled results for forage and food crops (Bodek et al. 1988); data indicate a range of bioaccumulation factors in plants from 0.13 to 2.24 on a wet weight basis (Bodek et al. 1988). Bioaccumulation factors for soil fauna including earthworms and isopods ranged from 0.12 to 1.97 on a wet weight basis (Beyer et al. 1987; Donker et al. 1993; Diercxsens et al. 1985; Morgan and Morgan 1988). Mallard ducks accumulated relatively low amounts (bioaccumulation factors of 0.0001 to 0.011) of zinc from diet into soft tissues (Gasaway and Buss 1972). Bioaccumulation factors for mink liver ranged from 0.038 to 0.85 (Aulerich et al. 1991).

TOXICITY TO TERRESTRIAL ORGANISMS

Toxicity to mule deer was predicted with data from sheep that were exposed by dietary ingestion for 10 weeks (Ott et al. 1966) (Table A-A). Toxicity to deer mice was predicted with data from a chronic study with rats (Schlicker and Cox 1968) that were fed high levels of zinc in diet. Zinc was incorporated at 0.2 and 0.4% of the diet; reproductive effects occurred in the offspring of the 0.4% treatment. Aulerich et al. (1991) reported a dietary NOAEL for mink that was used to represent toxicity to mammalian carnivores. Mink were exposed to dietary concentrations of 0, 500, 1000, and 1500 mg/kg $ZnSO_4$ for 144 days. Endpoints evaluated in this study were body weight, food consumption, hematology, fur quality, and survival. The authors concluded that mink tolerate at least 1500 mg/kg in diet.

Gasaway and Buss (1972) reported both a dietary NOAEL and a LOAEL for mortality in mallard ducks. This study was used to represent toxicity to avian receptors (Table A-A). Ducks were fed 0, 3000, 6000, 9000, and 12000 added ZnCO3 in diet. High mortality was observed in the zinc treated diets, food consumption was decreased, and the gonads were so reduced in size that cessation of function was considered likely. Zinc carbonate is more toxic than oxide or sulfate forms.

Chrysanthemums exposed to 93 mg/kg in soil had decreased growth (ICF 1989). The maximum concentration from a plant bioassay indicating unimpacted or nonphytotoxic soils was 379 mg/kg (Kapustka et al. 1995). The NEL for adverse effects on isopod populations was 864 mg/kg, although individual effects (i.e., fewer gravid females, decreased size) were observed at this concentration (Donker et al. 1993). The NOEC for woodlice was 100 mg/kg, and reported LC50 values for earthworms were 662 mg/kg (ICF 1989).

TOXICITY TO AQUATIC ORGANISMS

A study with *Xenopus laevis* indicated that 34.5 mg/L Zn as ZnSO4 was the 96 h LC50 at a water hardness of 100 mg/L (Dawson et al. 1988). In the same study, the 96 h EC50 for malformations was 3.6 mg/L, also at a water hardness of 100 mg/L. The 7 d LC50 for *Gastrophryne carolinensis* embryo-larval stage was 0.01 mg/L; test included exposure from fertilization to 4 days post-hatch at a water hardness of 195 mg/L as $CaCO_3$ (Birge et al. 1979). Tadpoles of the toad *Bufo melanostictus* were exposed at a water hardness ranging from 120 to 160 mg/L; the 7 d LC50 ranged from 19.86 to 47.26 mg/L (Khangarot and Ray 1987).

The acute and chronic AWQC for zinc are hardness dependent. The equation used to calculate the acute AWQC total zinc concentration is as follows (EPA 1990):

$$Chronic\ AWQC = e^{(0.8473*\ln hardness+0.8604)}$$

The equation used to calculate the chronic AWQC dissolved zinc concentration is as follows (EPA 1996a); the chronic AWQC for total zinc in water is the same (EPA 1990) except there is no correction factor of 0.986 following the natural log:

$$Chronic\ AWQC = e^{(0.8473*\ln hardness+0.7614)} * 0.986$$

Sediment quality benchmarks range from a low of 120 mg/kg for the LEL (Persaud et al. 1992) to 820 mg/kg for the Ontario SEL (Persaud et al. 1992; EPA 1996a). Long and Morgan (1991) present an ERL of 120 mg/kg and an ERM of 270 mg/kg expressed on a dry weight basis.

REFERENCES

Ambrose, A.M.; Larson, P.S.; Borzelleca, J.F.; Hennigar, G.R. Jr. 1976. Long-term toxicologic assessment of nickel in rats and dogs. *J. Food Sci. Tech.* 13:181-187.

Aulerich, R.J.; R.K. Ringer, M.R. Bleavins, and A. Napolitano. 1982. Effects of supplemental dietary copper on growth, reproductive performance, and kit survival of standard dark mink and the acute toxicity of copper to mink. *J. Animal Sci.* 55,(2):337-343.

Aulerich, R. J., S.J. Bursian, R.H. Poppenga, W.E. Braselton, and T.P. Mullaney. 1991. Toleration of high concentrations of dietary zinc by mink. *J. Vet Diagn. Invest.* 3:232-237.

Aulerich, R.J., R.K. Ringer, and S. Iwamoto. 1974. Effects of dietary mercury on mink. *Arch. Environ. Contam. Toxicol.* 2:43-51.

Batts, D. and J. Cubbage. 1995. Summary of Guidelines for Contaminated Freshwater Sediments. Environmental Investigations and Laboratory Services Program. Washington State Department of Ecology. Olympia, Washington. February. Publication No. 95-308.

Beyer, W.N. and E. Cromartie. 1987. Environmental Monitoring and Assessment 8:27-36. Provided to Carolyn Fordham, Terra Technologies, by Ned Black, USEPA Region 9 Nov. 6 1996.

Birge, W.J., J.A. Black, A.G. Westerman, and B.A. Ramey. 1983. *Fund. Appl. Toxicol.* 3:237-242. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Birge, W.J., J.A. Black, and A.G. Westerman. 1979. Evaluation of Aquatic Pollutants Using Fish and Amphibian Eggs as Bioassay Organisms. In: Neilsen, S.W., G. Migaki, and D.G. Scarpelli (eds). *Animals as Monitors of Environmental Pollutants*, National Academy of Science, Washington, D.C. pp. 108-118. In: Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Environment Canada, Canadian Wildlife Service. 2nd printing.

Bodek, I., W.J. Lyman, W.F. Reehl, and D.H. Rosenblatt (eds.). 1988. Environmental Inorganic Chemistry: Properties, Processes, and Estimation Methods. A Special Publication of SETAC. Pergamon Press New York, NY.

Borzelleca, J.F., L.W. Condie, Jr., and J.L. Egle, Jr. 1988. Short-term toxicity (one- and ten-day gavage) of barium chloride in male and female rats. *J. Amer. College Toxicol.* 7:675-685.

Browne, C., and J.N. Dumont. 1979. Toxicity of selenium to developing *Xenopus* tadpoles (*Xenopus laevis*). *Arch. of Environ. Contam. Toxicol.* 9:181-191. In: Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Environment Canada, Canadian Wildlife Service. 2nd printing.

Cain, B.W. and E.A. Pafford. 1981. Effects of dietary nickel on survival and growth of mallard ducklings. *Arch. Environ. Contam. Toxicol.* 10: 737-745.

Camardese, M.; Hoffman, D.; LeCaptain, L.; Pendleton, G. 1990. Effects of arsenate on growth and physiology in mallard ducklings. *Environ. Toxicol. Chem.* 9:785-795.

Canada Environmental Protection Agency (CEPA). 1993. Arsenic. Priority Substances List Assessment Report, Health Canada, Canada Communications Group Publishing, Ottawa, Canada.

Canada Environmental Protection Agency (CEPA). 1994a. Chromium and its Compounds. Priority Substances List Assessment Report, Health Canada, Canada Communications Group Publishing, Ottawa., Canada.

Canada Environmental Protection Agency (CEPA). 1994b. Nickel. Priority Substances List Assessment Report, Health Canada, Canada Communications Group Publishing, Ottawa., Canada.

Canton, J.H. and W. Slooff. 1982. *Ecotoxicol. Environ. Safety* 6:113-128. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Carlson, R.W., and G.L. Rolfe. 1979. Growth of rye grass and fescue as affected by lead-cadmium-fertilizer interaction. *J. Environ. Qual.* 8:348-352.

Carriere, D.; K. Fischer; D. Peakall; and P. Angehrn. 1986. Effects of dietary aluminum in combination with reduced calcium and phosphorus on the ring dove (*Streptopellia risoria*). *Water, Air, and Soil Poll.* 30:757-764.

Chakoumakos, C., R.C. Russo, and R.V. Thurston. 1979. Toxicity of copper to cutthroat trout (*Salmo Clarki*) under different conditions of alkalinity, pH, and hardness. *Environ. Sci. Technol.* 13(2):213-219.

Dawson, D.A., E.F. Stebler, S.L. Burks, and J.A. Bantle. 1988. *Environ. Toxicol. Chem.* 7:27-34. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

de Zwart, D. and W. Slooff. 1987. *Bull. Environ. Contam. Toxicol.* 38:345-351. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Diercxsens, P., D. de Weck, N. Borsinger, B. Rosset, and J. Tarradellas. 1985. Earthworm contamination by PCBs and heavy metals. *Chemosphere* 14:511-522

Dilling, W.J. and C.W. Healey. 1926. Influence of lead and the metallic ions of copper, zinc, thorium, beryllium and thallium on the germination of frogs' spawn and on the growth of tadpoles. *Annal. of Appl. Biol.* 13:177-188. In: Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Environment Canada, Canadian Wildlife Service. 2nd printing.

Dixon, D.G. and J.B. Sprague. 1981. Acclimation to copper by rainbow trout (*Salmo gairdneri*) – A modifying factor in toxicity. *Can. J. Fish. Aquat. Sci.* 38:880-888.

Domingo, J.L. 1994. Metal-induced developmental toxicity in mammals: A Review. *J. Toxicol. Environ. Health.* 43: 123-141.

Donker, M.H., H.E. van Capelleveen, and N.M. van Straalen. 1993. Metal Contamination Affects Size-Structure and Life-History Dynamics in Isopod Field Populations. Chapter 19. Ecotoxicology of Metals in Invertebrates. Lewis Publishers. 384-399.

Edens, F.W.; Garlich, J.D. 1983. Lead-induced egg production decrease in leghorn and Japanese quail hens. *Poult. Sci.* 62:1757-1763.

Eisler, R. 1987. Mercury Hazards to Fish, Wildlife, and invertebrates: A Synoptic Review. Fish and Wildlife Service, U.S. Dept. of the Int. Biological Report No. 85(1.10).

Eisler, R. 1988. Arsenic Hazards to Fish, Wildlife and Invertebrates: A Synoptic Review. Fish and Wildlife Service, U.S. Dept. of the int. Biological Report No. 85(1.12).

Ellis, R.W. and L. Eslick. 1997. Variation and range of mercury uptake into plants at a mercury-contaminated abandoned mine site. *Bull. Environ. Contam. Toxicol.* 59:763-769.

Fimreite, N. and L. Karstad. 1971. Effects of dietary methyl mercury on red-tailed hawks. *J. Wildl. Mgt.* 35:293-300.

Finlayson, B.J. and K.M. Verrue. 1980. Estimated safe zinc and copper levels for chinook salmon (*Oncorhynchus tshawytscha*) in the Upper Sacramento River, California. *Calif. Fish & Game* 66(2):68-82.

Finley, M. and R.C. Stendall. 1978. Survival and reproductive success of black ducks fed methyl mercury. *Environ. Poll.* 16:51-63.

Forman, S.J. and K.S. Jumar. 1980. Hemolytic anemia in Wilson's disease: Clinical findings and biochemical mechanisms. *Amer. J. Hematol.* 9:269-275.

Freda, J. , V. Cavdek, and D.G. McDonald. 1990. *Can. J. Fish. Aquat. Sci.* 47:217-224. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Freda, J. and D.G. McDonald. 1990. *Can. J. Fish. Aquat. Sci.* 47:210-216. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Friberg, L., *et al.* 1979. Handbook on the Toxicology of Metal. Elsevier/North Holland Biomedical Press.

Friberg, L., F.G. Nordberg, and V.B. Vouk (eds). 1986. Handbook on the Toxicology of Metals, 2nd Ed. Elsevier/North Holland Biomedical Press.

Gasaway, W.C. and I.O. Buss. 1972. Zinc toxicity in the mallard. *J. Wildl. Mgt.* 36: 1107-1117.

Giattina, J.D., R.R. Garton, and D.G. Stevens. 1982. Avoidance of copper and nickel by rainbow trout as monitored by a computer-based data acquisition system. *Trans. Amer. Fish. Soc.* 111:491-504.

Groton, J.P., E.J. Sinkeldam, J.B. Luten, and P.J. van Bladeren. 1991. Composition of the toxicity of inorganic and liver-incorporated cadmium: a 4-week feeding study in rats. *Food Chem. Toxicol.* 28:435-442.

Haywood, S. and M. Loughran. 1985. Copper toxicosis and tolerance in the rat. II. Tolerance – a liver protective adaptation. *Liver* 5(5):267-275.

Hebert, C.D., M.R. Elwell, G.S. Travlos, C.J. Fitz, and J.R. Bucher. 1993. Subchronic toxicity of cupric sulfate administered in drinking water and feed to rats and mice. *Fund. Appl. Technol.* 21: 461-475.

Heinz, G.H.; D.J. Hoffman; and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. *J. Wildl. Mgt.* 53: 418-428.

Hill, E.F. and C.S. Schaffner. 1976. Sexual maturation and productivity of Japanese quail fed graded concentrations of mercuric chloride. *Poult. Sci.* 55: 1449-1459.

Horwitt, M.K. and G.R. Cowgill. 1939. The effects of ingested lead on the organism: II. Studies on the dog. *J. Pharmacol. Exper. Therapy* 66: 289-301.

Howarth, R.S. and J.B. Sprague. 1978. Copper lethality to rainbow trout in waters of various hardness and pH. *Water Res.* 12:455-462.

Hudson, R.H.; R.K. Tucker; and M.A. Haegele. 1984. Handbook of Toxicity of Pesticides to Wildlife. U.S. Fish and Wildl. Serv. Resource. Publ. 153. 90 pp.

ICF. 1989. Scoping Study of the Effects of Soil Contamination on Terrestrial Biota. Volume I. Project Summary. Prepared for Office of Toxic Substances. USEPA. Washington, D.C.

James, L.F.; V.A. Lazar, V.A.; W. Binns. 1966. Effects of sublethal doses of certain minerals on pregnant ewes and fetal development. *Am. J. Vet. Res.* V. 17 No. 11. 132-135.

John, M.K. 1972. Mercury uptake from soil by various plant species. *Bull. Environ. Contam. Toxicol.* 8:77-80.

Johnson, D., Jr., A.L. Mehring, Jr., and H.W. Titus. 1960. Tolerance of chickens for barium. *Proc. Soc. Exp. Biol. Med.* 104: 436-438.

Jorgensen, S.E., S.N. Nielson, and L.A. Jorgensen 1991. Handbook of Ecological Parameters and Ecotoxicology. Elsevier Press. 1262 pp.

Kanno, Y., T. Yamami, Y. Muneoka, and Y. Shiba. 1978. Effects of heavy metal ions on the electrical properties of mucous epithelial cells in the newt stomach. *J. Toxicol. Sci.* 3:39-50. In: Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Environment Canada, Canadian Wildlife Service. 2nd printing.

Kapustka, L.A., J. Lipton, H. Galbraith, D. Cacela, and K. LeJeune. 1995. Metal and arsenic impacts to soils, vegetation communities and wildlife habitat in southwest Montana uplands contaminated by smelter emissions: II. Laboratory Phytotoxicity Studies. *Environ. Toxicol. Chem.* 14:1905-1912.

Khangarot, B.S. and P.K. Ray. 1987. Sensitivity of toad tadpoles, *Bufo melanostictus* (Schneider), to heavy metals. *Bull. Environ. Contam. Toxicol.* 38:523-527. In: Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Environment Canada, Canadian Wildlife Service. 2nd printing.

Kisssberth, W.C., J.P. Sundberg, R.W. Nyboer, J.D. Reynolds, S.C. Kasten, V.R. Beasley. 1984. Industrial lead contamination of an Illinois wildlife refuge and indigenous small mammals. *J. Amer. Vet. Med. Assoc.* 11:1309-1313.

Kruckeberg, A.L. and L Wu. 1992. Copper tolerance and copper accumulation of herbaceous planta colonizing inactive California copper mines. *Ecotoxicol. Env. Saf.* 23: 307-319.

Kubena, I.F.; R.B. Harvey; T.D. Phillips; and O.J. Fletcher. 1986. Influence of ochratoxin A and vanadium on various parameters in growing chicks. *Poult. Sci.* 65:1671-1678.

Kupsh, C.C., R.J. Julian, V.E.O. Valli, and G.A. Robinson. 1991. Renal damage induced by uranyl nitrate and oestradiol-17 in Japanese quail and Wistar rats. *Avian Pathol.* 20:25-34.

Laskey, J.W. and F.W. Edens. 1985. Effects of chronic high-level manganese exposure on male behavior in the Japanese quail (*Coturnix coturnix japonica*). *Poult. Sci.* 64:579-584.

Lauren, D.J. and D.G. McDonald. 1986. Influence of water hardness, pH and alkalinity on the mechanisms of copper toxicity in juvenile rainbow trout, *Salmo gairdneri*. *Can. J. Fish. Aquat. Sci.* 43:1488-1496.

Lee, I.P., R.T. Sherins, and R.L. Dixon. 1978. Evidence for introduction of germinal aplasia in male rats by environmental exposure to boron. *Toxicol. Appl. Pharmacol.* 45:577-590. In: Whitworth, M.R., G.W. Pendleton, D.J. Hoffman, and M.B. Camardese. 1991. Effects of dietary boron and arsenic on the behavior of mallard ducklings. *Environ. Toxicol. Chem.* 10:911-916.

Leland, H.V. 1983. Ultrastructural changes in the hepatocytes of juvenile rainbow trout and mature brown trout exposed to copper or zinc. *Environ. Toxicol. Chem.* 2:353-368.

Lepore, P.D. and R.F. Miller. 1965. Embryonic viability as influenced by excess molybdenum in chicken breeder diets. *Proc. Soc. Exp. Biol. Med.* 118: 155-157.

Lin, S and L. Wu. 1994. Effect of copper concentration on mineral nutrient uptake and copper accumulation in protein of copper–tolerant and nontolerant *Lotus purshianus* (L). *Ecotoxicol. Env. Saf.* 29: 214-218.

Lindsay, R.C. and R.W. Dimmick. 1983. Mercury residues in wood ducks and wood duck foods in eastern Tennessee. *J. Wildl. Diseases* 19:114-117.

Long, E.R. and L.G. Morgan. 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. 2nd Printing. August 1991. NOAA, Department of Commerce.

McKenna, I.M., R.L. Chaney, S.H. Tao, R.M. Leach, Jr., and F.M. Williams. 1992. Interactions of pant zinc and plant species on the bioavailability of plant cadmium to Japanese quail fed lettuce and spinach. *Environ. Res.* 57:73-87.

McKim, J.M. and D.A. Benoit. 1971. Effects of long-term exposures to copper on survival, growth, and reproduction of brook trout (*Salvelinus fontinalis*). *J. Fish. Res. Board Can.* 28(5):655-622.

Miller, J.C. and R. Landesman. 1978. Reduction of heavy metal toxicity to *Xenopus* embryos by magnesium ions. *Bull. of Environ. Contam. Toxicol.* 20:93-95.

Miller, J.E., J.J. Hassett, and D.E. Koeppe. 1977. Interactions of lead and cadmium on metal uptake and growth of corn plants. *J. Environ. Qual.* 6:18-20.

Miller, W.J., Y.G. Martin, R.P. Gentry and D.M. Blackmon. 1968. Zn and stable zinc absorption, excretion and tissue concentrations as affected by type of diet and level of zinc in normal calves. *J. Nutr.* 94:391-401.

Mills, C.F. and A.C. Dalgarno. 1972. Copper and zinc status of ewes and lamb receiving increased dietary concentrations of cadmium. *Nature.* 239:171-173.

Morgan, J.E., and A.J. Morgan. 1988. Earthworms as biological monitors of cadmium, copper, lead and zinc in metalliferous soils. *Environ. Poll.* 54:123-138.

Morgan, J.E., and A.J. Morgan. 1991. *Bull. Environ. Contam. Toxicol.* 47:296-301. Provided to Carolyn Fordham, Terra Technologies, by Ned Black, USEPA Region 9, Nov. 6 1996.

Motto, H.L., R.H. Daines, D.M. Chilko, and C.K. Motto. 1970. Lead in soils and plants: Its Relationship to traffic volume and proximity to highways. *Environ. Sci. Technol.* 4:231-237.

National Academy of Science (NAS). 1974. Medical and biologic effects of environmental pollutants. Chromium. National Academy of Sciences, Division of Medical Science, National Resources Council. Washington, D.C.

National Toxicology Program (NTP) 1993. The Toxicology and Carcinogenesis Studies of Manganese (II) Sulfate Monohydrate (CAS No. 10034-96-5) in F344/N Rats and B6C3F$_1$ Mice (Feed Studies). National Toxicology Program, Research Triangle Park, NC. December 1993. NTP TR 428. NIH Publication No. 94-3159. US. Department of Health and Human Services. Public Health Service. National Institutes of Health.

Ondreicka, R.; E. Ginter; and J. Kortus. 1966. Chronic toxicity of aluminum in rats and mice and its effects on phosphorus metabolism. *Brit. J. Indust. Med.* 23: 305-313.

Ott, E.A.; W.H. Smith, R.B. Harrington, W.M. Beeson. 1966. Zinc toxicity in ruminants. II. Effects of high levels of dietary zinc on gains, feed consumption and feed efficiency of beef cattle. *J. Animal Sci.* 25:419-423.

Pagenkopf, G.K., R.C. Russo, and R.V. Thurston. 1974. Effect of complexation on toxicity of copper to fishes. *J. Fish. Res. Board Can.* 31(4):462-465.

Påhlsson, A.B. Toxicity of heavy metals (Zn, Cu, Cd, Pb) to vascular plants. *Water, Air, Soil Poll.* 47:287-319.

Parmelee, R.W., R.S. Wentsel, C.T. Phillips, M. Simini, and R.T. Checkai. 1993. Soil microcosm for testing the effects of chemical pollutants on soil fauna communities and trophic structure. *Environ. Toxicol. Chem.* 1477-1486.

Paternian, J.L.; J.L. Domingo; A. Ortega; and J.M. Llobet. 1989. The effects of uranium on reproduction, gestation, and postnatal survival in mice. *Ecotoxicol. Environ. Saf.* 17: 291-296.

Pattee, O.H. 1984. Eggshell thickness and reproduction in american kestrels exposed to chronic dietary lead. *Arch. Environ. Contam. Toxicol.* 13:29-34.

Pérez-Coll, C.S., J. Herkovits, and A. Salibian. 1988. *Bull. Environ. Contam. Toxicol.* 41:247-252. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Persaud, D., R. Jaagumagi, and A. Hayton. 1992. Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario. March 1993 Revision. Water Resources Branch. Ontario Ministry of the Environment.

Pershagen, G. and M. Vahter. 1979. Arsenic—a toxicological and epidemiological appraisal. Naturvardsverket Rapp. SNV PM 1128, Liber Tryck, Stockholm. 265 pp.

Perry, H.M., E.F. Perry, M.M. Erlanger, and S.J. Kopp. 1983. Cardiovascular effects of chronic barium ingestion. In: Proc 17th Annual Conference Trace Substances in Environ. Health, Vol. 17. Univ. of Missouri Press. Columbia, MO.

Reader, J.P., N.C. Everall, M.D.J. Sayer, and R. Morris. 1989. The effects of eight trace metals in acid soft water on survival, mineral uptake and skeletal calcium deposition in the yolk-sac fry of brown trout, *Salmo trutta* L. *J. Fish. Biol.* 35:187-198.

Revis, N.; G. Holdsworth; G. Bingham; A. King; and J. Elmore. 1989. An Assessment of Health Risk Associated with Mercury in Soil and Sediment from East Fork Poplar Creek, Oak Ridge, Tennessee. Oak Ridge Research Institute, Final Report, 58 pp.

Roll, R. and G. Matthiaschk. 1981 (as cited in Borges & Daugherty 1996). Investigations on embryotoxic effects of thallium chloride and thallium acetate in mice and rat. *Teratol.* 24:46A-47A.

Rosenfeld, I. and O.A. Beath. 1954. Effect of selenium on reproduction in rats. *Proc. Soc. Exp. Biol. Med.* 87: 295-297.

Sauvé, S., M. McBride, and W. Hendershot. 1998. Lead phosphate solubility in water and soil suspensions. *Environ. Sci. Technol.* 32:388-393.

Sax. I. 1984. Sax's Dangerous Properties of Industrial Materials, 4th Edition; Lewis, R.J., ed., Van Nostrand Reinhold, New York, NY.

Schlicker, S. and D. Cox. 1968. Maternal dietary zinc, and development and zinc, iron, and copper content of the rat fetus. *J. Nutr.* 95:287-294.

Schroeder, H.A. and M. Mitchener. 1971. Toxic effects of trace elements on the reproduction of mice and rats. *Arch. Environ. Health* 23:102-106.

Shepard, T.H. 1986. Catalog of Teratogenic Agents, 5th Edition, John Hopkins University Press, Baltimore, MD.

Shore, R.F. 1995. Predicting cadmium, Lead, and fluoride levels in small mammals from soil residues and by species-species extrapolation. *Environ. Poll.* 88:333-340.

Slonim, A.R. and E.E. Ray. 1975. *Bull. Environ. Contam. Toxicol.* 13:307-312. . In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Slooff, W. and R. Baerselman. 1980. *Bull. Environ. Contam. Toxicol.* 24:439-443. In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Slooff, W., J.H. Canton, and J.L.M. Hermens. 1983. *Aquat. Toxicol.* 4:113-128. . In: Devillers, J. and J.M Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers. 351 pp.

Stanley, T.R. Jr.; J.W. Spann; G.J. Smith; and R. Rosscoe. 1994. Main and interactive effects of arsenic and selenium on mallard reproduction and duckling growth and survival. *Arch. Environ. Contam. Toxicol.* 26: 444-451.

Stansley, W., and D.E. Roscoe. 1996. The uptake and effects of lead in small mammals and frogs at a trap and skeet range. *Arch. of Environ. Contam. Toxicol.* 30:220-226.

Stiff, M.J. 1971a. The chemical states of copper in polluted fresh water and a scheme of analysis to differentiate them. *Water Res.* 5:585-599.

Stiff, M.J. 1971b. Copper/bicarbonate equilibria in solutions of bicarbonate ion at concentrations similar to those found in natural water. *Water Res.* 5:171-176.

Sylva, R.N. 1976. The environmental chemistry of copper (II) in aquatic systems. *Water Res.* 10:789-792.

Stowe, H.D. and R.A. Goyer. 1971. The reproductive ability and progeny of F1 lead-toxic rats. *Fertil. Steril.* 22: 110-115.

Suttle, N.F. 1974. Title Unavailable. *Brit. J. Nutr.*

Suttle, N.F. 1986. Copper deficiency in ruminants; recent developments. *The Vet. Rec.*, November 22.

Talmage, S.S., and B.T. Walton. 1993. Food chain transfer and potential renal toxicity of mercury to small mammals at a contaminated terrestrial field site. *Ecotoxicol.* 2:243-256.

U.S. Environmental Protection Agency (EPA). 1980. Ambient Water Quality Criteria for Silver.

U.S. Environmental Protection Agency (EPA). 1990. Region 8 List of Priority Pollutants Indicating EPA 304(a) Published Criteria, Modified Criteria, Recommended Criteria for 303 (c)(2)(B), and Drinking Water Maximum Contaminant Levels (MCLS). Water Management Division. August 1990.

U.S. Environmental Protection Agency (EPA). 1992a. Water Quality Criteria Summary. Office of Science and Technology. Health and Ecological Criteria Division. Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1992b. Ecological Effects of Soil Lead Contamination. Toxics Integration Branch, Office of Emergency and Remedial Response. U.S. EPA September 9 1992.

U.S. Environmental Protection Agency (EPA). 1993. Handbook of Wildlife Exposure Factors.

U.S. Environmental Protection Agency (EPA). 1996a. ECO Update: Ecotox Thresholds. Office of Solid Waste and Emergency Response. EPA 540/F-95/038. January 1996.

U.S. Environmental Protection Agency (EPA). 1996b. Assessment and Remediation of Contaminated Sediments (ARCS) Program. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod *Hyalella Azteca* and the Midge *Chironomus Riparius*. EPA 905-R96-008. September 1996. Great Lakes National Program Office. Chicago Ill.

U.S. Environmental Protection Agency (EPA). 1997. Personal Communication between Dr. Jon Rauscher, USEPA Region VI, and Carolyn Fordham, Terra Technologies, regarding the withdrawal of the silver AWQC. November 24.

Valdivia, R.; C.B. Ammerman; C.J. Wilcox; and P.R. Henry. 1978. Effect of dietary aluminum on animal performance and tissue mineral levels in growing Steers. *J. Animal Sci.* 47:1351.

Van Vleet, J.F. 1976. Induction of lesions of selenium-vitamin E deficiency in pigs fed silver. *Amer. J. Vet. Res.* 37: 1415-1420.

Walker, F. 1971. Experimental argyria: A model for basement membrane studies. *Brit. J. Exper. Pathol.* 52: 589-593.

Wallace, A. 1989a. Effect of liming on trace-element interactions in plants. *Soil Sci.* 47:416-421.

Wallace, A. 1989b. Phytotoxicity of cobalt when uniformly mixed in soil versus localized spot placement in soil. *Soil Sci.* 147:449-450.

Walsh, F.M. F.J. Crossen, J. Bayley, et al. 1977. Acute copper intoxication. *Amer. J. Diseases in Children.* 131:149-151.

Whitworth, M.R., G.W. Pendleton, D.J. Hoffman, and M.B. Camardese. 1991. Effects of dietary boron and arsenic on the behavior of mallard ducklings. *Environ. Toxicol Chem.* 10:911-916.

WHO. 1990. Beryllium. Environmental Health Criteria 106, Geneva, Switzerland.

Will, M.E. and G.W. Suter II. 1995. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants: 1995 Revision. Environmental Restoration Risk Assessment Program U.S. Dept of Energy. ES/ER/TM-85/R2.

Wilson, R.H.; DeEds, F.; Cox, A.J. Jr. 1941. Effects of continued cadmium feeding. Sandford Univ. School of Med.; Food Research Unit. 222-235.

Wiseman, J. 1987. Feeding of Nonruminant Livestock; Butterworths, Boston MA. 41-201.

Wright, R.J., V.C. Baligar, and J.L. Ahlrichs. 1989. The influence of extractable and soil solution aluminum on root growth of wheat seedlings. *Soil Sci.* 148:293-302.

Zmudzki, J.; G.R. Bratton; C. Womac; and L.D. Rowe. 1984. The influence of milk diet, grain diet and method of dosing on lead toxicity in young calves. *Toxicol. Appl. Pharmacol.* 76:490-497.

APPENDIX D

SOIL SCREENING CRITERIA CALCULATIONS

EXPLANATION OF SOIL SCREENING CRITERIA CALCULATION

Soil Screening Criteria (SSCs) were calculated for each receptor type based on the concentration of a given chemical in soil that would result in a daily intake rate equal to a ecotoxicological benchmark called the Toxicity Reference Value (TRV). The procedure was based on the following general equation for intake of chemicals from food (forage or prey) and incidental ingestion of soil (EPA 1989, 1997):

[Equation 1]

$$DI = [((FIR*C_{food}*B) + (SIR*C_{soil}*B)]*AUF$$

where:

DI = Daily Intake of chemical (mg of chemical/kg body weight/day)

FIR = daily food ingestion rate (kg/kg body weight/day)

SIR = daily rate for incidental ingestion of soil or sediment (kg/kg body weight/day)

C_{food} *= concentration of COC in food (forage or prey)*

C_{soil} *= concentration of COC in soil and/or sediment (mg/kg)*

B = bioavailability of metal from ingested material (food or soil)

AUF = area use factor; the proportion of the daily intake obtained from the study area (unitless)

Soil ingestion rates (SIR) were obtained using estimates provided in the *Wildlife Exposure Factors Handbook* (EPA 1993) which are expressed as a fraction of total dietary ingestion rates (DIR) (shown in Appendix A). The DIR include all ingested material including soils and food. Thus, SIR was estimated from:

[Equation 2]

$$SIR = DIR\ (kg/kg\ bw/d) * (soil\ fraction\ in\ diet\ [unitless])$$

The FIR was then calculated as:

[Equation 3]

$$FIR = DIR - SIR$$

Data were not available on concentrations of chemicals in biota at the Chino Mine site. Therefore, C_{food} was estimated using bioaccumulation factors (BAFs) and the C_{soil}:

[Equation 4]

$$C_{food} = BAF * C_{soil}$$

The BAFs used in the screening level calculation approximate the maximum BAFs observed at other mining and industrial sites (Table 3).

The AUF and B were conservatively assumed to be 1 (100 %) for all receptors in the screening-level assessment.

To obtain the SSC, Equation 1 was solved for the C_{soil} that would result in a total daily intake equal to the TRV for a given receptor type. In other words, the C_{soil} that results in a hazard quotient (HQ) = 1 was designated as the SSC.

The following tables show details of the data and parameters used in calculating the SSC for each receptor/chemical combination. The column labeled "Soil Screening Criterion" in each table shows the value for C_{soil} that results in an HQ = 1 and, therefore, is the designated SSC.

Table 3-8
Summary of Bioaccumulation Factors Used in Calculating Soil Screening Criteria

Chemical	Soil-Vegetation		Soil-Terrestrial Invertebrates		Soil-Small Mammals	
Aluminum	0.02	c	na		na	
Antimony	na		na		na	
Arsenic	0.06	a	0.4	a	0.003	d
Barium	na		na		0.06	d
Beryllium	na		na		na	
Boron	na		na		na	
Cadmium	0.14	a	0.6	a	0.3	d
Chromium	0.03	c	na		0.08	d
Cobalt	0.03	c	na		0.02	d
Copper	0.2	a	0.3	a	0.2	d
Lead	0.03	a	0.04	a	0.05	a
Manganese	0.1	c	na		na	
Mercury	na		na		0.05	d
Molybdenum	6	b	1.8	b	0.2	b
Nickel	0.1	c	na		na	
Selenium	0.8	a	0.93	a	0.2	d
Silver	na		na		na	
Thallium	na		na		0.1	d
Vanadium	0.01	c	na		na	
Zinc	0.1	b	1.6	a	0.8	d

na = not available

Sources:
- a. Ecological Risk Assessment, North Oquirrh Mountains
- b. Lincoln Park Ecological Risk Assessment
- c. Toele Army Depot Ecological Risk Assessment
- d. Sample et. al. 1998

Table 3-9
Literature-based Soil Screening Criteria (SSCs) for Chino Mines Investigation Area (units = mg/kg)

***Based on Site Soil Data and Bioaccumulation Factors*[1]**

	Reference Area Concentrations		Soil Screening Criterion						
Analyte	Mean	UCL_{95}	Lg. Ground-feeding Bird	Sm. Ground-feeding Bird	Predatory Bird	Omnivorous Rodent	Mammalian Predator	Ruminant	Vegetation
Aluminum	21,500	24,186	90,678	9,068	3,210	164	309	21,573	730
Antimony	3	4	na	na	na	0.5	2	10	5
Arsenic	2.3	3	1,263	126	2,089	7	176	215	16
Barium	118	141	2,156	216	8,303	8	537	255	500
Beryllium	0.7	24	544	54	163	12	39	215	10
Boron	6.6	8	na	na	na			na1	--
Cadmium	0.6	1	84	8	18	16	71	105	3
Chromium	32	40	114	11	42	9	1,719	11,338	150
Cobalt	12.0	14	219	22	187	6	230	680	20
Copper	136	179	1,111	111	424	1,585	736	202	46
Lead	22	25	350	35	181	3,683	154	2,835	250
Manganese	537	596	57,135	5,713	3,257	894	1,818	64,935	500
Mercury	nd (0.1)	--	6	1	23	33	119	650	0.3
Molybdenum	nd (0.6)	--	7	1	53	0.2	12	3	2200
Nickel	16	18	4,503	450	257	100	364	12,987	250
Selenium	1.1	1.3	4	0.4	5	0.1	1	6	4
Silver	nd (0.2)	--	100	10	30	162	591	3,250	10
Thallium	7	8	22	2.2	54	0.5	22	15	1
Vanadium	101	121	245	24	8	0.05	0.1	7	2
Zinc	69	75	573	57	41	352	3,543	9,741	380

[1] Calculated using bioaccumulation factors from other sites.

Shaded cells indicate literature-based soil screening-criterion is less than UCL_{95} of mean for site-specific reference concentration for Hurley Soils IU.

Calculation of Preliminary Remediation Goals For COCs in Soils

Ground-feeding Bird--Quail

	Vegetation						
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	1813.6	0.02	0.09	0.08	1	1	146.9
Antimony	--	1	0.09	0.08	1	1	--
Arsenic	75.8	0.06	0.09	0.08	1	1	6.1
Barium	2155.6	1	0.09	0.08	1	1	174.6
Beryllium	544.4	1	0.09	0.08	1	1	44.1
Boron	--	1	0.09	0.08	1	1	--
Cadmium	11.7	0.14	0.09	0.08	1	1	0.9
Chromium	3.4	0.03	0.09	0.08	1	1	0.3
Cobalt	6.6	0.03	0.09	0.08	1	1	0.5
Copper	222.2	0.2	0.09	0.08	1	1	18.0
Lead	10.5	0.03	0.09	0.08	1	1	0.9
Manganese	5713.5	0.1	0.09	0.08	1	1	462.8
Mercury	5.6	1	0.09	0.08	1	1	0.5
Molybdenum	42.4	6	0.09	0.08	1	1	3.4
Nickel	450.3	0.1	0.09	0.08	1	1	36.5
Selenium	3.3	0.8	0.09	0.08	1	1	0.3
Silver	100.0	1	0.09	0.08	1	1	8.1
Thallium	22.2	1	0.09	0.08	1	1	1.8
Vanadium	2.4	0.01	0.09	0.08	1	1	0.2
Zinc	57.3	0.1	0.09	0.08	1	1	4.6

	Soil							
Chemical	Soil Ingestion Rate (mg/kg bw/day)[2]	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV$_{NOEL}$ (mg/kg bw/day)	HQ[4]	Soil Screening Criterion (mg/kg)[3]
Aluminum	0.009	1	1	816.1	9.6E+02	963	1.0	90,678
Antiomony	0.009	1	1	--	--	na	na	na[1]
Arsenic	0.009	1	1	11.4	17.5	17.5	1.0	1,263
Barium	0.009	1	1	19.4	194.0	194	1.0	2,156
Beryllium	0.009	1	1	4.9	49.0	49	1.0	544
Boron	0.009	1	1	--	--	na	na	na[1]
Cadmium	0.009	1	1	0.8	1.7	1.7	1.0	84
Chromium	0.009	1	1	1.0	1.3	1.3	1.0	114
Cobalt	0.009	1	1	2.0	2.5	2.5	1.0	219
Copper	0.009	1	1	10.0	28.0	28	1.0	1,111
Lead	0.009	1	1	3.2	4.0	4	1.0	350
Manganese	0.009	1	1	514.2	977.0	977	1.0	57,135
Mercury	0.009	1	1	0.1	0.5	0.5	1.0	6
Molybdenum	0.009	1	1	0.1	3.5	3.5	1.0	7
Nickel	0.009	1	1	40.5	77.0	77	1.0	4,503
Selenium	0.009	1	1	0.0	0.3	0.3	1.0	4
Silver	0.009	1	1	0.9	9.0	9	1.0	100
Thallium	0.009	1	1	0.2	2.0	2	1.0	22
Vanadium	0.009	1	1	2.2	2.4	2.4	1.0	245
Zinc	0.009	1	1	5.2	9.8	9.8	1.0	573

Calculation of Preliminary Remediation Goals For COCs in Soils

Small Ground-feeding Bird--Junco

	Vegetation						
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	181.4	0.02	0.9	0.8	1	1	146.9
Antimony	--	1	0.9	0.8	1	1	--
Arsenic	7.6	0.06	0.9	0.8	1	1	6.1
Barium	215.6	1	0.9	0.8	1	1	174.6
Beryllium	54.4	1	0.9	0.8	1	1	44.1
Boron	--	1	0.9	0.8	1	1	--
Cadmium	1.2	0.14	0.9	0.8	1	1	0.9
Chromium	0.3	0.03	0.9	0.8	1	1	0.3
Cobalt	0.7	0.03	0.9	0.8	1	1	0.5
Copper	22.2	0.2	0.9	0.8	1	1	18.0
Lead	1.1	0.03	0.9	0.8	1	1	0.9
Manganese	571.3	0.1	0.9	0.8	1	1	462.8
Mercury	0.6	1	0.9	0.8	1	1	0.5
Molybdenum	4.2	6	0.9	0.8	1	1	3.4
Nickel	45.0	0.1	0.9	0.8	1	1	36.5
Selenium	0.3	0.8	0.9	0.8	1	1	0.3
Silver	10.0	1	0.9	0.8	1	1	8.1
Thallium	2.2	1	0.9	0.8	1	1	1.8
Vanadium	0.2	0.01	0.9	0.8	1	1	0.2
Zinc	5.7	0.1	0.9	0.8	1	1	4.6

	Soil							
Chemical	Soil Ingestion Rate (mg/kg bw/day)	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV$_{NOEL}$ (mg/kg bw/day)	HQ	Soil Screening Criterion (mg/kg)
Aluminum	0.09	1	1	816.1	9.6E+02	963	1.0	9068
Antiomony	0.09	1	1	--	--	na	na	na[1]
Arsenic	0.09	1	1	11.4	17.5	17.5	1.0	126
Barium	0.09	1	1	19.4	194.0	194	1.0	216
Beryllium	0.09	1	1	4.9	49.0	49	1.0	54
Boron	0.09	1	1	--	--	na	na	na[1]
Cadmium	0.09	1	1	0.8	1.7	1.7	1.0	8
Chromium	0.09	1	1	1.0	1.3	1.3	1.0	11
Cobalt	0.09	1	1	2.0	2.5	2.5	1.0	22
Copper	0.09	1	1	10.0	28.0	28	1.0	111
Lead	0.09	1	1	3.2	4.0	4	1.0	35
Manganese	0.09	1	1	514.2	977.0	977	1.0	5713
Mercury	0.09	1	1	0.1	0.5	0.5	1.0	1
Molybdenum	0.09	1	1	0.1	3.5	3.5	1.0	1
Nickel	0.09	1	1	40.5	77.0	77	1.0	450
Selenium	0.09	1	1	0.0	0.3	0.3	1.0	0.4
Silver	0.09	1	1	0.9	9.0	9	1.0	10
Thallium	0.09	1	1	0.2	2.0	2	1.0	2
Vanadium	0.09	1	1	2.2	2.4	2.4	1.0	24
Zinc	0.09	1	1	5.2	9.8	9.8	1.0	57

[1] SSC not calculated because TRV not available.

Calculation of Preliminary Remediation Goals For COCs in Soils

Avian Predator -- Harrier

	Small Mammals						
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	3210.0	1	0.3	0.29	1	1	938.9
Antimony	--	1	0.3	0.29	1	1	--
Arsenic	6.3	0.003	0.3	0.29	1	1	1.8
Barium	498.2	0.06	0.3	0.29	1	1	145.7
Beryllium	163.3	1	0.3	0.29	1	1	47.8
Boron	--	1	0.3	0.29	1	1	--
Cadmium	5.4	0.3	0.3	0.29	1	1	1.6
Chromium	3.4	0.08	0.3	0.29	1	1	1.0
Cobalt	3.7	0.02	0.3	0.29	1	1	1.1
Copper	84.8	0.2	0.3	0.29	1	1	24.8
Lead	9.0	0.05	0.3	0.29	1	1	2.6
Manganese	3256.7	1	0.3	0.29	1	1	952.6
Mercury	1.1	0.05	0.3	0.29	1	1	0.3
Molybdenum	10.6	0.2	0.3	0.29	1	1	3.1
Nickel	256.7	1	0.3	0.29	1	1	75.1
Selenium	0.9	0.2	0.3	0.29	1	1	0.3
Silver	30.0	1	0.3	0.29	1	1	8.8
Thallium	5.4	0.1	0.3	0.29	1	1	1.6
Vanadium	8.0	1	0.3	0.29	1	1	2.3
Zinc	33.1	0.8	0.3	0.29	1	1	9.7

	Soil							
Chemical	Soil Ingestion Rate (mg/kg bw/day)	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV$_{NOEL}$ (mg/kg bw/day)	HQ	Soil Screening Criterion (mg/kg)
Aluminum	0.008	1	1	24.08	9.6E+02	963	1.0	3210
Antiomony	0.008	1	1	--	--	na	na	na[1]
Arsenic	0.008	1	1	15.67	17.5	17.5	1.0	2089
Barium	0.008	1	1	62.28	208.0	208	1.0	8303
Beryllium	0.008	1	1	1.22	49.0	49	1.0	163
Boron	0.008	1	1	--	--	na	na	na[1]
Cadmium	0.008	1	1	0.13	1.7	1.7	1.0	18
Chromium	0.008	1	1	0.32	1.3	1.3	1.0	42
Cobalt	0.008	1	1	1.40	2.5	2.5	1.0	187
Copper	0.008	1	1	3.18	28.0	28	1.0	424
Lead	0.008	1	1	1.36	4.0	4	1.0	181
Manganese	0.008	1	1	24.42	977.0	977	1.0	3257
Mercury	0.008	1	1	0.17	0.5	0.5	1.0	23
Molybdenum	0.008	1	1	0.40	3.5	3.5	1.0	53.0
Nickel	0.008	1	1	1.92	77.0	77	1.0	257
Selenium	0.008	1	1	0.03	0.3	0.3	1.0	5
Silver	0.008	1	1	0.23	9.0	9	1.0	30
Thallium	0.008	1	1	0.41	2.0	2	1.0	54
Vanadium	0.008	1	1	0.06	2.4	2.4	1.0	8
Zinc	0.008	1	1	0.31	10.0	10	1.0	41

[1] SSC not calculated because TRV not available.

Calculation of Preliminary Remediation Goals For COCs in Soils

Omnivorous Rodent -- Deer Mouse

	Vegetation								Invertebrates							
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Diet	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)	Fraction Diet	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	3.3	0.02	0.4	0.39	0.5	1	1	0.6	163.5	1	0.4	0.39	0.5	1	1	32.1
Antiomony	0.5	1	0.4	0.39	0.5	1	1	0.1	0.5	1	0.4	0.39	0.5	1.0	1.0	0.1
Arsenic	0.4	0.06	0.4	0.39	0.5	1	1	0.1	2.8	0.4	0.4	0.39	0.5	1.0	1.0	0.5
Barium	7.5	1	0.4	0.39	0.5	1	1	1.5	7.5	1	0.4	0.39	0.5	1.0	1.0	1.5
Beryllium	12.5	1	0.4	0.39	0.5	1	1	2.4	12.5	1	0.4	0.39	0.5	1.0	1.0	2.4
Boron	--	1	0.4	0.39	0.5	1	1	--	--	1	0.4	0.39	0.5	1.0	1.0	--
Cadmium	2.3	0.14	0.4	0.39	0.5	1	1	0.4	9.8	0.6	0.4	0.39	0.5	1.0	1.0	1.9
Chromium	0.3	0.03	0.4	0.39	0.5	1	1	0.1	8.6	1	0.4	0.39	0.5	1.0	1.0	1.7
Cobalt	0.2	0.03	0.4	0.39	0.5	1	1	0.0	5.7	1	0.4	0.39	0.5	1.0	1.0	1.1
Copper	317.0	0.2	0.4	0.39	0.5	1	1	62.1	475.5	0.3	0.4	0.39	0.5	1.0	1.0	93.2
Lead	110.5	0.03	0.4	0.39	0.5	1	1	21.7	147.3	0.04	0.4	0.39	0.5	1.0	1.0	28.9
Manganese	89.4	0.1	0.4	0.39	0.5	1	1	17.5	894.5	1	0.4	0.39	0.5	1.0	1.0	175.3
Mercury	32.5	1	0.4	0.39	0.5	1	1	6.4	32.5	1	0.4	0.39	0.5	1.0	1.0	6.4
Molybdenum	1.2	6	0.4	0.39	0.5	1	1	0.2	0.4	1.8	0.4	0.39	0.5	1.0	1.0	0.1
Nickel	10.0	0.1	0.4	0.39	0.5	1	1	2.0	100.0	1	0.4	0.39	0.5	1.0	1.0	19.6
Selenium	0.1	0.8	0.4	0.39	0.5	1	1	0.0	0.1	0.93	0.4	0.39	0.5	1.0	1.0	0.0
Silver	162.5	1	0.4	0.39	0.5	1	1	31.9	162.5	1	0.4	0.39	0.5	1.0	1.0	31.9
Thallium	0.5	1	0.4	0.39	0.5	1	1	0.1	0.5	1	0.4	0.39	0.5	1.0	1.0	0.1
Vanadium	0.0	0.01	0.4	0.39	0.5	1	1	0.0	0.1	1	0.4	0.39	0.5	1.0	1.0	0.0
Zinc	35.2	0.1	0.4	0.39	0.5	1	1	6.9	562.7	1.6	0.4	0.39	0.5	1.0	1.0	110.3

	Soil							
Chemical	Soil Ingestion Rate (mg/kg bw/day)	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV_{NOEL} (mg/kg bw/day)	HQ	Soil Screening Criterion (mg/kg)
Aluminum	0.008	1	1	1.3	3.4E+01	34	1.0	164
Antiomony	0.008	1	1	0.0	2.0E-01	0.2	1.0	1
Arsenic	0.008	1	1	0.1	0.7	0.6	1	7
Barium	0.008	1	1	0.1	3.0	5.1	1	8
Beryllium	0.008	1	1	0.1	5.0	4.3	1	12
Boron	0.008	1	1	--	--	na	na	na[1]
Cadmium	0.008	1	1	0.1	2.5	2.5	1	16
Chromium	0.008	1	1	0.1	1.8	1.8	1	9
Cobalt	0.008	1	1	0.0	1.2	1.2	1	6
Copper	0.008	1	1	12.7	168.0	168	1	1585
Lead	0.008	1	1	29.5	80.0	80	1.00	3683
Manganese	0.008	1	1	7.2	200.0	200	1	894
Mercury	0.008	1	1	0.3	13.0	13	1	33
Molybdenum	0.008	1	1	0.0	0.3	0.3	1	0.2
Nickel	0.008	1	1	0.8	22.4	40	1	100
Selenium	0.008	1	1	0.0	0.0	0.03	1	0.1
Silver	0.008	1	1	1.3	65.0	65	1	162
Thallium	0.008	1	1	0.0	0.2	0.3	1	0.5
Vanadium	0.008	1	1	0.0	0.0	0.01	1	0.1
Zinc	0.008	1	1	2.8	120.0	120	1	352

[1] SSC not calculated because TRV not available.

Calculation of Preliminary Remediation Goals For COCs in Soils

Mammalian Predator -- Coyote

	Small Mammals						
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	309.1	1	0.11	0.107	1	1	33.0
Antimony	1.8	1	0.11	0.107	1	1	0.2
Arsenic	0.5	0.003	0.11	0.107	1	1	0.1
Barium	32.2	0.06	0.11	0.107	1	1	3.4
Beryllium	39.1	1	0.11	0.107	1	1	4.2
Boron	--	1	0.11	0.107	1	1	#VALUE!
Cadmium	21.3	0.3	0.11	0.107	1	1	2.3
Chromium	137.5	0.08	0.11	0.107	1	1	14.7
Cobalt	4.6	0.02	0.11	0.107	1	1	0.5
Copper	147.2	0.2	0.11	0.107	1	1	15.7
Lead	7.7	0.05	0.11	0.107	1	1	0.8
Manganese	1818.2	1	0.11	0.107	1	1	194.4
Mercury	5.9	0.05	0.11	0.107	1	1	0.6
Molybdenum	2.5	0.2	0.11	0.107	1	1	0.3
Nickel	363.6	1	0.11	0.107	1	1	38.9
Selenium	0.2	0.2	0.11	0.107	1	1	0.0
Silver	590.9	1	0.11	0.107	1	1	63.2
Thallium	2.2	0.1	0.11	0.107	1	1	0.2
Vanadium	0.1	1	0.11	0.107	1	1	0.0
Zinc	2834.7	0.8	0.11	0.107	1	1	303.1

	Soil							
Chemical	Soil Ingestion Rate (mg/kg bw/day)	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV$_{NOEL}$ (mg/kg bw/day)	HQ	Soil Screening Criterion (mg/kg)
Aluminum	0.003	1	1	1.0	3.4E+01	34	1	309
Antiomony	0.003	1	1	0.0	2.0E-01	0.2	1	2
Arsenic	0.003	1	1	0.5	0.6	0.6	1	176
Barium	0.003	1	1	1.7	5.1	5.1	1	537
Beryllium	0.003	1	1	0.1	4.3	4.3	1	39
Boron	0.003	1	1	--	--	na	na	na[1]
Cadmium	0.003	1	1	0.2	2.5	2.5	1	71
Chromium	0.003	1	1	5.3	20.0	20	1	1719
Cobalt	0.003	1	1	0.7	1.2	1.2	1	230
Copper	0.003	1	1	2.3	18.0	18	1	736
Lead	0.003	1	1	0.5	1.3	1.3	1	154
Manganese	0.003	1	1	5.6	200.0	200	1	1818
Mercury	0.003	1	1	0.4	1.0	1	1	119
Molybdenum	0.003	1	1	0.0	0.3	0.3	1	12.3
Nickel	0.003	1	1	1.1	40.0	40	1	364
Selenium	0.003	1	1	0.0	0.03	0.03	1	1
Silver	0.003	1	1	1.8	65.0	65	1	591
Thallium	0.003	1	1	0.1	0.3	0.3	1	22
Vanadium	0.003	1	1	0.0	0.0	0.01	1	0.10
Zinc	0.003	1	1	10.9	314.0	314	1	3543

[1] SSC not calculated because TRV not available.

Calculation of Preliminary Remediation Goals For COCs in Soils

Ruminant -- Mule Deer

	Vegetation						
Chemical	Conc. In Food (mg/kg)[1]	Soil-food Uptake Factor	Dietary Ingestion Rate (kg/kg bw/d)	Food Ingestion Rate (kg/kg bw/day)[2]	Fraction Food Ingested onsite	Bioavail-ability	Intake from Food (mg/kg bw/day)
Aluminum	431.5	0.02	0.02	0.02	1	1	8.1
Antimony	10.0	1	0.02	0.02	1	1	0.2
Arsenic	12.9	0.06	0.02	0.02	1	1	0.2
Barium	255.0	1	0.02	0.02	1	1	4.8
Beryllium	215.0	1	0.02	0.02	1	1	4.0
Boron	--	1	0.02	0.02	1	1	--
Cadmium	14.6	0.14	0.02	0.02	1	1	0.3
Chromium	340.1	0.03	0.02	0.02	1	1	6.4
Cobalt	20.4	0.03	0.02	0.02	1	1	0.4
Copper	40.3	0.2	0.02	0.02	1	1	0.8
Lead	85.1	0.03	0.02	0.02	1	1	1.6
Manganese	6493.5	0.1	0.02	0.02	1	1	122.1
Mercury	650.1	1	0.02	0.02	1	1	12.2
Molybdenum	15.8	6	0.02	0.02	1	1	0.3
Nickel	1298.7	0.1	0.02	0.02	1	1	24.4
Selenium	5.0	0.8	0.02	0.02	1	1	0.1
Silver	3250.0	1	0.02	0.02	1	1	61.1
Thallium	15.0	1	0.02	0.02	1	1	0.3
Vanadium	0.1	0.01	0.02	0.02	1	1	0.0
Zinc	974.1	0.1	0.02	0.02	1	1	18.3

	Soil					Risk Estimate		
Chemical	Soil Ingestion Rate (mg/kg bw/day)	Fraction Soil Ingested Onsite	Bioavail-ability	Intake from Soil (mg/kg bw/day)	Total Intake from Site (mg/kg bw/day)	TRV_{NOAEL} (mg/kg bw/day)	HQ	Soil Screening Criterion (mg/kg)
Aluminum	0.001	1	1	25.9	3.4E+01	34	1.0	21573
Antiomony	0.001	1	1	0.0	2.0E-01	0.2	1.0	10
Arsenic	0.001	1	1	0.3	5.0E-01	0.5	1.0	215
Barium	0.001	1	1	0.3	5.1E+00	5.1	1.0	255
Beryllium	0.001	1	1	0.3	4.3E+00	4.3	1.0	215
Boron	0.001	1	1	--	--	na	--	na[1]
Cadmium	0.001	1	1	0.1	4.0E-01	0.4	1.0	105
Chromium	0.001	1	1	13.6	2.0E+01	20	1.00	11338
Cobalt	0.001	1	1	0.8	1.2E+00	1.2	1.0	680
Copper	0.001	1	1	0.2	1.0E+00	1	1.0	202
Lead	0.001	1	1	3.4	5.0E+00	5	1.0	2835
Manganese	0.001	1	1	77.9	2.0E+02	200	1.0	64935
Mercury	0.001	1	1	0.8	1.3E+01	13	1.00	650
Molybdenum	0.001	1	1	0.0	3.0E-01	0.3	1.0	3
Nickel	0.001	1	1	15.6	4.0E+01	40	1.0	12987
Selenium	0.001	1	1	0.0	1.0E-01	0.1	1.00	6
Silver	0.001	1	1	3.9	6.5E+01	65	1.0	3250
Thallium	0.001	1	1	0.0	3.0E-01	0.3	1.0	15
Vanadium	0.001	1	1	0.0	1.0E-02	0.01	1.0	7
Zinc	0.001	1	1	11.7	3.0E+01	30	1.0	9741

[1] SSC not calculated because TRV not available.

APPENDIX E

Equations for Interval and Equivalence Tests

Calculation of the *F*-statistic associated with the interval and equivalence tests is (Kilgour et al. 1998):

$$F = (MEAN_{ref} - Value_{site})^2 \div (\sigma_{ref} * \sqrt{(1/n_{ref})})^2$$

Where:

$Mean_{ref}$ = mean of reference population

$Value_{site}$ = value for individual site

σ_{ref} = standard deviation for reference population

n_{ref} = number of reference samples

Calculations for the interval and equivalence tests are identical. However, different null hypotheses require different critical values for the non-central *F*-distribution and results of the comparisons are complimentary (Kilgour et al. 1998, McBride et al. 1993).

Tests will be first conducted using the interval test based on the hypothesis:

$H_{\varnothing 1}$: $|MEAN_{ref} - Value_{site}| \leq [psd]$ *vs* H_{A1}: $|MEAN_{ref} - Value_{site}| > [psd]$

The power of the interval test is low unless the sample size for the reference population is large (i.e., >50). Therefore, if results indicate rejection of $H_{\varnothing 1}$, the site is assumed to be outside the natural range of the reference value. If $H_{\varnothing 1}$ is not rejected, an equivalence test will be conducted:

$H_{\varnothing 2}$: $|MEAN_{ref} - Value_{site}| \geq [psd]$ *vs* H_{A2}: $|MEAN_{ref} - Value_{site}| < [psd]$

If $H_{\varnothing 2}$ is rejected, the site is assumed to be equivalent to reference. If $H_{\varnothing 2}$ is accepted, the site falls in a "gray" area, and the status of the site will be reviewed by risk assessors and managers.

This approach is similar to comparison of individual sites to descriptive parameters such as tolerance intervals. However, the interval and equivalence tests allow evaluation of a practical significant difference and errors associated with the comparison.