Cal. Gulch 318850
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ADMINISTRATIVE RECORD
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FILE PLAN
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DRAFT FINAL BASELINE AQUATIC ECOLOGICAL RISK ASSESSMENT

CALIFORNIA GULCH, LEADVILLE, COLORADO

OCTOBER 1994

Prepared for:

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Region VIII
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EPA Work Assignment No. 31-8L29
Document Control No. 4800-31-0078

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ACRONYMS

AG	Airport Gulch
ANOVA	Analysis of Variance
AR	Arkansas River
ARARs	Applicable or Relevant and Appropriate Requirements
AWQC	Ambient Water Quality Criteria
BCF	bioconcentration factor
BERA	Baseline Aquatic Ecological Risk Assessment
CCC	Criterion Continuous Concentration
CDH	Colorado Department of Health
CDL	Colorado Department of Law
CDOW	Colorado Division of Wildlife
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cfs	cubic feet per second
CG	California Gulch
cm	centimeter
CMC	Criterion Maximum Concentration
COCs	contaminants of concern
CRL	Contract Reporting Limit
EF	East Fork (of the Arkansas River)
Eh	reduction-oxidation potential
EG	Evans Gulch
EM	Empire Gulch
EPA	U.S. Environmental Protection Agency
ERL	Effects Range Low
ERM	Effects Range Medium
GG	Georgia Gulch
HC	Halfmoon Creek
HI	hazard index
Hg	mercury
HQ	hazard quotient
IDL	instrument detection limit
kg	kilogram
kg/ha	kilograms (of fish) per hectare
km	kilometer
LC_{50}	median lethal concentration
LF	Lake Fork
LMDT	Leadville Mine Drainage Tunnel
LOAEL	Lowest Observed Adverse Effect Level
LSD	Least Significant Differences
MG	Malta Gulch

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LIST OF ACRONYMS (Continued)

mg/kg	milligrams per kilogram
mg/ℓ	milligram per liter
mph	miles per hour
N/ha	number of individuals per hectare
NELs	No Effect Levels
NOAEL	No Observed Adverse Effect Level
OG	Oregon Gulch
PG	Pawnee Gulch
PRG	preliminary remedial goals
RBP	Rapid Bioassessment Protocol
RI	Remedial Investigation
RME	Reasonable Maximum Exposure
ROD	Record of Decision
SD	Starr Ditch
SG	Stray Horse Gulch
TC	Tennessee Creek
UCL95	upper 95th percent confidence interval for the mean
USGS	U.S. Geological Survey
USFWS	U.S. Fish and Wildlife Service
WESTON	Roy F. Weston, Inc.
μg/ℓ	micrograms per liter
μg/g	micrograms per gram

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EXECUTIVE SUMMARY

Roy F. Weston, Inc. (WESTON®) and its subcontractor, Terra Technologies, conducted a Baseline Aquatic Ecological Risk Assessment (BERA) for the California Gulch Superfund site in Lake County, Colorado. Using guidance established by the U.S. Environmental Protection Agency (EPA), the BERA identifies the impact of mine waste contamination on the plants and animals found at the Site. This information will be of assistance in determining the type and extent of remediation that should take place in accordance with Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) mandates. The BERA consists of a site characterization, an exposure assessment, a stress response assessment, a toxicity assessment, and a risk characterization.

The site characterization details the historical information regarding the site, specifically, the mining activities and waste materials from mining processes in the Leadville area that have impacted the California Gulch aquatic ecosystem. Waste materials considered important in this assessment include waste rock and tailing piles, and smelter wastes in the forms of slag, flue dust, and stack emissions. These source materials have caused increased metal loading to surface water and sediments in the California Gulch area and the Arkansas River.

The exposure assessment identifies the primary exposure pathways for the aquatic ecological receptors. A complete exposure pathway includes a receptor (animal) that is exposed, a route or means by which exposure occurs, and a contaminant in the media to which the receptor is exposed. While there are many potential exposure pathways, only some of them can actually be quantified. The potential exposure pathways for the aquatic receptors are ingestion of surface water, sediments, and dietary items, and direct contact with surface water and sediments. The amount of sediment ingested by fish and aquatic invertebrates is unknown, as well as the amount of contamination received in the diet compared to the amount received directly from water. For these reasons, only the direct contact pathways were evaluated quantitatively (i.e., potential adverse effects were determined due to exposure to surface water and sediment). The abiotic data available for this report include surface water (dissolved inorganics) and sediment data. These media provide the most direct and significant means of evaluating exposure for aquatic

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ecological receptors. The ecological receptors include aquatic plants, benthic macroinvertebrates, and fish.

The stress response assessment describes the biological data available for the site in relation to location. Sample locations included stations on California Gulch, the Arkansas River, and along several tributaries of the Arkansas River. Site-related effects on populations and ecosystems were identified. The results of the stress response assessment indicated that California Gulch is totally devoid of aquatic life forms that are typically found in a mountain stream of its size. The only invertebrates observed in California Gulch were *Daphnia*, a lentic (still-water) genus that originated from the sewage treatment plant. The fauna of the Arkansas River are impacted downstream from California Gulch, as documented by reduced populations of benthic macroinvertebrates and fish.

The toxicity assessments summarize the results of a literature review of the adverse effects of metals and inorganics on aquatic species. Water quality criteria are reported in this section, as well as bioconcentration factors. The Ambient Water Quality Criteria (AWQC) are values, which provide a concentration of a given metal or inorganic parameter (i.e., hardness) in water, above which adverse effects on aquatic life become likely. Concentrations below the AWQC are unlikely to affect populations of fish or macroinvertebrates.

The risk characterization utilizes toxicity values from the toxicity assessment and exposure intakes from the exposure assessment to determine risk to ecological receptors. The hazard quotient approach was used, where the exposure intakes were divided by the appropriate toxicity value. A hazard quotient exceeding one indicates ecological risk. Hazard quotients were higher at the on-site locations (California Gulch and its tributaries, and Evans Gulch) than at off-site locations (East Fork, Lake Fork, and other tributaries of the Arkansas River outside the influences of the source areas on-site). The hazard quotients for on-site locations were often two or three orders of magnitude higher than the hazard quotients for the off-site locations.

California Gulch, Malta Gulch, Stray Horse Gulch, and Oregon Gulch are all streams that originate on-site. Hazard quotients for cadmium, copper, lead, and zinc were higher at the

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Malta Gulch sampling location compared to most stations on California Gulch. Concentrations of cadmium, copper, and lead were higher in Malta Gulch samples than in samples from Oregon Gulch. Hazard quotients based on average cadmium and lead concentrations were highest at Malta Gulch, copper hazard quotients were highest at Stray Horse Gulch, and zinc hazard quotients were highest at Oregon Gulch. Airport Gulch, another stream originating on-site, also had high hazard quotients for cadmium, copper, lead, and zinc. In contrast, hazard quotients for streams originating off-site, such as Iowa or Empire Gulch, were typically less than one. Hazard quotients for streams originating off-site were generally at least two orders of magnitude lower than streams originating on-site.

The results of the risk characterization, combined with the results of the stress response assessment, provide strong evidence that California Gulch is adversely impacted by on-site mining activities. California Gulch and its tributaries and the Arkansas River downstream of California Gulch all have hazard quotients for exposure of aquatic biota to surface water and sediment higher than those at off-site reference locations. In addition, California Gulch lacks a benthic macroinvertebrate community. This is indicative of the extreme stress being placed on this ecosystem due to high metal concentrations. The Arkansas River macroinvertebrate community is adversely affected at locations downstream of California Gulch. Populations of trout are severely reduced in the Arkansas River from California Gulch downstream to the sampling stations identified as AR4 and AR5.

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SECTION 1
INTRODUCTION

The preliminary Baseline Aquatic Ecological Risk Assessment (BERA) was conducted for the California Gulch Superfund site in Lake County, Colorado. Using guidance established by the U.S. Environmental Protection Agency (EPA), the BERA identifies the impact of California Gulch site contamination on the plants and animals found at the Site; this information will be of assistance in determining the type and extent of remediation that should take place in accordance with Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) guidelines. The BERA includes a site characterization, an exposure assessment, a stress response assessment, a toxicity assessment, and a risk characterization section. The BERA summarizes and interprets data collected for the completion of current remedial investigation (RI) activities and data from other investigations pertinent to the assessment of aquatic ecological risk.

1.1 AQUATIC RISK ASSESSMENT SCOPE AND ORGANIZATION

Many areas within the California Gulch RI study area have been identified as potentially impacted by mining activity. These areas include tailings and waste rock piles and the surrounding vicinity. In addition, surface water and sediments in streams originating on the Site carry high concentrations of metals, particularly during periods of high water flow, such as storm events and spring runoff. The Arkansas River receives the flow from California Gulch, and higher concentrations of metals occur downstream of this confluence than above. The BERA presents the ecological risks (i.e., the chance of adverse effects on the environment or on plants and animals living in the environment) associated with these areas under baseline conditions in the absence of any additional remedial action (the no-action alternative). Data collected and available prior to February 1, 1994, have been incorporated into this report. This includes 1991 data (previous to Yak Tunnel remediation) and July 1992 to August 1993 data (subsequent to Yak Tunnel remediation). The data collected from March to June 1992, the period during which the Yak Tunnel treatment facility began operation, were not used in the risk assessment due to the dynamic (i.e., fluctuating) conditions in California Gulch at that time.

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Data collected prior to the operation of the Yak Tunnel are evaluated separately from data collected after water treatment at the Yak Tunnel facility began to compare the water quality in California Gulch before and after the Yak Tunnel water treatment plant went on-line.

Data for this risk assessment include EPA Level III and IV data from the current RI activities. These data have passed the rigorous quality assurance/quality control requirements of the EPA Contract Laboratory Program. Only analytical data that were collected in 1991 and later were used in the quantitative exposure assessment. A complete list of the analytical data utilized in the aquatic risk assessment, definitions of terms relating to data quality, the raw data, and the data qualifiers are provided in Appendix B of this report.

Using the results of the BERA, EPA will identify the areas within the Site that will be targeted for potential remediation and/or further analysis. If the results of the baseline BERA indicate that remediating certain areas will improve surface water quality and the health of the aquatic ecosystem, then the risk assessment will be used to rank these areas in order of priority. Preliminary remedial goals (PRGs) will also be used to set priorities for cleanup. The PRGs are levels of metals in the surface water that are protective of the aquatic community and environment.

The steps used to perform the BERA are listed below:

- Evaluate analytical data obtained during the current RI activities.
- Identify site-related contaminants of concern (COCs).
- Identify potential exposure pathways for ecological receptors for the COCs.
- Evaluate actual or potential adverse impacts associated with the exposure of these receptors to the average and reasonable maximum concentration of site-related contaminants.
- Evaluate biological data (i.e., periphyton, fish tissue) collected from the site.

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Guidance available in the documents listed in Table 1-1 provided the methodology used to assess ecological risk. In addition, the following federal and Colorado state agencies assisted in performing the BERA for the California Gulch Site:

- U.S. Environmental Protection Agency.
- U.S. Geological Survey (USGS).
- U.S. Fish and Wildlife Service (USFWS).
- Colorado Division of Wildlife (CDOW).
- Colorado Department of Health (CDH).

The following introductory sections include historical background, physical conditions, and habitat characteristics of the California Gulch area, as well as a summary of significant findings from previous investigations at the Site.

1.2 SITE CHARACTERIZATION

Mining activities in the Leadville area have visibly impacted the California Gulch site. Waste materials from the mining processes that occurred include waste rock, tailing piles, and smelter wastes in the forms of slag, flue dust, and stack emissions. The following sections describe how these contaminant sources were produced and the overall characterization of the study Site.

1.2.1 Study Area Description and Site Background

The California Gulch site is located in Lake County, Colorado, approximately 100 miles southwest of Denver in the Colorado mineral belt of the Rocky Mountains (Figure 1-1). The Site boundary includes the watershed that drains California Gulch and flows west to the Arkansas River (Figure 1-1). The Site ranges in elevation from 9,570 to 12,250 feet above mean sea level and incorporates the city of Leadville, Colorado (Res-ASARCO, 1991a).

Mining, milling, and smelting operations recovering gold, silver, lead, and zinc have been active for more than 130 years. The Leadville Mining District has produced an estimated 26 million

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Color Map(s)

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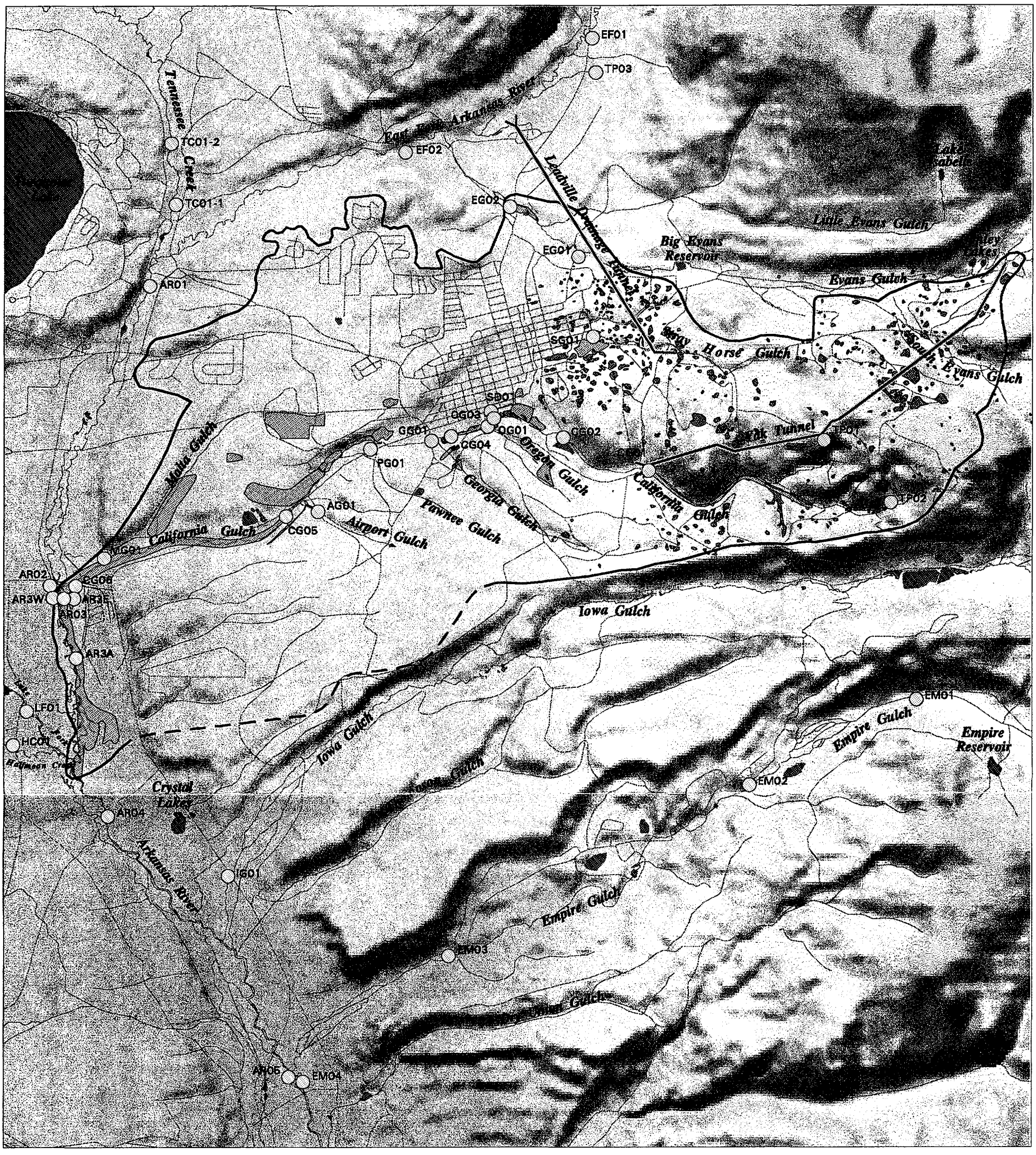


Legend

- Transportation
- Hydrography
- RI/FS Study Area Boundary
- Boundary Pending RI Completion
- Tailing Impoundments
- Waste Rock
- Slag Piles
- Areas Containing Fluvial Tailing
- Sampling Stations

SCALE 1:50000

1 0 1 2

MILES

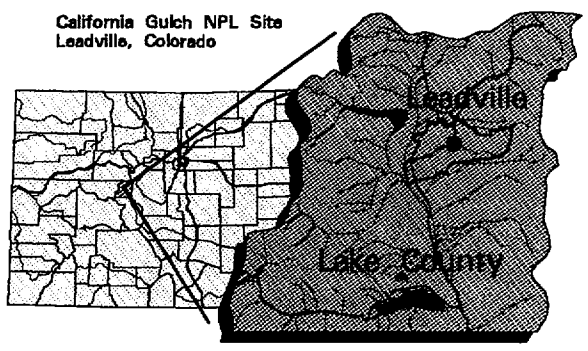
Elevation Key - feet

9200 9600 10000 10400 10800 11200 11600 12000 12400 12800 13200 13600

N

S

California Gulch RI/FS Study Area Surface Water/Sediment Sample Stations and Arkansas River Drainages



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Hydrography from IntraSearch, Inc., 1992 and Roy F. Weston, Inc. Transportation From USGS 1:100,000 DLG data. Topography from USGS 1:24,000 DEM data. Study Area boundary defined by EPA. Tailing Impoundments, Waste Rock Piles, and Slag Piles compiled by Roy F. Weston, Inc., from Steffan,Robertson and Kirsten (SRK), 1989; and Walsh and Associates 1993. Fluvial areas by Woodward – Clyde Consultants, 1993. Surface Water Sampling and Hazard Index data compiled by Roy F. Weston, Inc. and Terra Technologies, Inc. This map may be revised following the resolution of outstanding data issues and completion of remedial investigations.

Figure 1-1

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DCN: 4800-31-M001

Revision Date: 27-OCT-1994

Table 1-1

Guidance Documents Applicable to the Assessment of Ecological Risk

Risk Assessment Guidance for Superfund: Human Health Evaluation Manual (EPA, 1989a)
Risk Assessment Guidance for Superfund: Environmental Evaluation Manual (EPA, 1989b)
Superfund Exposure Assessment Manual (EPA, 1988a)
Ecological Risk Assessment (EPA, 1986)
Ecological Assessment of Hazardous Waste Sites — A Field and Laboratory Reference (EPA, 1989c)
Guidance for Data Useability in Risk Assessment (EPA, 1990a)
Review of Ecological Risk Assessment Methods (EPA, 1988b)
Framework for Ecological Risk Assessment (EPA, 1992a)
A Review of Ecological Assessment Case Studies from a Risk Assessment Perspective (EPA, 1993)

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tons of ore between 1859 and 1986 (ASARCO, 1987). Native gold was discovered in 1859 and was initially mined by placer methods; later, ores of gold and other metals were mined by underground methods. W.H. Stevens and A.B. Wood began mining carbonate ore in 1875, which they discovered to be rich in silver and lead. When the surface veins of ore diminished, miners tunneled deeper into the mountains. Groundwater flooding in the tunnels became a costly problem, and mining activity decreased. In 1889, the Yak Tunnel was constructed to drain the mines under Iron Hill. The Yak Tunnel currently extends a total distance of 3.5 miles.

As additional underground mines were created, more waste rock was excavated with the ore. This waste rock was placed near mine entrances, while ore was transported to the mills. At the mills, ore was crushed and separated into metallic concentrates and waste products by physical processes. The metallic concentrates were further processed at a smelter in the area or shipped elsewhere. The waste products or mill tailings were generally placed near the mill in a tailing pond. At the smelter, high-grade ores were refined and concentrated into higher-grade products. Waste products from the smelters included slag, dust, and off-gases (JEG, 1991).

Recent activity in the Leadville area is restricted to a few moderate-size mining and reprocessing operations (EPA, 1987). The Black Cloud Mine and Mill, located in Iowa Gulch, has been operating continuously since 1971 (ASARCO, 1987).

The Yak Tunnel and Leadville drainage tunnels were developed to drain existing mines and explore unmined areas. The Yak Tunnel is approximately 3.5 miles long and discharges mine water into California Gulch at a point 4.5 miles upstream of the confluence of California Gulch with the Arkansas River. A surge pond and diversion ditches have recently been constructed below the Yak Tunnel portal to contain any surges emanating from the Yak Tunnel and to divert rainfall around the surge pond (Res-ASARCO, 1991b).

Yak Tunnel flow was diverted to the surge pond beginning on November 8, 1989, and water water was discharged from the filter unit to California Gulch beginning December 5, 1989. Water from the Yak Tunnel and hillside springs flows into the surge pond through ditches, channels, and pipelines. The water collects in the surge pond and sediments settle out. Vertical

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pumps, located on a floating barge, pumped the water to the filter unit, and the effluent from the filter unit was discharged into South Ditch. A water treatment plant was designed in accordance with the Record of Decision (ROD) for the Yak Tunnel Operable Unit and Administrative Order (AO 89-20) to treat the Yak Tunnel discharge (Res-Asarco, 1991a). This became operable in February 28, 1992. The filter unit was shut down permanently concurrent with the start-up of the Yak Tunnel water treatment plant.

1.2.2 Physical Characteristics

The climate at the California Gulch Superfund Site is typical of central Colorado mountainous areas, with normal temperature extremes from -30°F to 86°F (Topielec et al., 1977). The wind is predominantly from the northwest, ranging from calm to 30 miles per hour (mph) (Res-Asarco, 1991a), and the average annual precipitation is 18 inches (SCS, 1965).

The geological setting and soils of the California Gulch site are summarized in the Human Health Preliminary Baseline Risk Assessment (EPA, 1991a) and the Soils Inventory and Map Report (Walsh, 1992). The terrain in the study area is highly variable, encompassing sloping to steep fans in California Gulch, to level or gently sloping terraces, swales, and flood plains along the Arkansas River. Soils consist of gravelly and sandy loams and clay.

Over the last 100 years, mining practices have disrupted the naturally occurring soils through physical disturbance (i.e., movement of rock and ore from underlying geological strata to the surface), as well as by inducing chemical changes in soil metals concentrations due to waste piles and smelter emissions. Mining activities also mixed the alluvial materials in stream beds and flood plains with mine waste products. In disturbed areas, the major surface materials consist of mine dumps, mine workings, slag piles resulting from smelter operations, and mill tailings. Over 2,000 waste dumps occur in the study area (EPA, 1989d) and comprise approximately 5% of the study area (EPA, 1987).

Surface water drainages analyzed in this report (Figure 1-1) include California Gulch, Stray Horse Gulch, Evans Gulch, Tennessee Gulch, and other tributaries of California Gulch and the

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Arkansas River. The Arkansas River is the major surface water body in the study area. Wetlands occur along the East Fork of the Arkansas River and Tennessee Creek, and along the Arkansas River just above the confluence with California Gulch. Other wetland areas occur below the confluence of the Arkansas River and Lake Creek, along Evans Gulch, and below the confluence with Empire Gulch.

1.2.3 Summary of Historical Data for California Gulch and Vicinity

Many of the previous investigations conducted for ecological parameters contain useful information for the identification of ecological characteristics of the site and exposure routes, as well as for the identification of potential receptors and determination of bioaccumulation in on-site food chains. Several studies including EPA (1987), Levy (1990), and the Colorado Department of Law (CDL) (1986) describe the flora and fauna common to the area. The investigation conducted by EPA (1987) describes threatened and sensitive plant communities, while a study conducted by ERT (1987) provides data on brown trout habitat suitability of the East Fork of the Arkansas River (East Fork) and upper Arkansas River. The studies described in this section were conducted by different private organizations and public agencies, and there are different levels of data quality and quantity. In this report, no attempt has been made to critique these investigations since the data and results are reported only to provide background information pertinent to the site.

LaBounty et al. (1975) investigated water quality, accumulation of metals in river sediments, species diversity indices, fish populations, and concentrations of five metals (copper, iron, lead, zinc, and manganese) in macroinvertebrates from an 18-mile stretch of the upper Arkansas River between East Fork and Lake Creek. Metal concentrations in macroinvertebrates ranged from 20 to 8,870 milligrams per kilogram body weight (mg/kg body weight) on a dry weight basis for copper; 820 to 47,700 mg/kg for iron; 20 to 152,000 mg/kg for lead; 630 to 9,150 mg/kg for zinc; and 40 to 9,100 mg/kg for manganese. These results indicated that the three main sources of heavy-metal input were the Leadville Drainage Tunnel and the sewage outflow from a trailer park, the California Gulch outflow, and the diffuse sources between the confluence of the Arkansas River with Halfmoon Creek and Lake Creek.

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Roline and Boehmke (1981) investigated the effects of metal pollution on the distribution of aquatic macrofauna in a 10-kilometer (km) section of the upper Arkansas River between East Fork and the town of Granite. Water quality parameters were also collected in conjunction with the collection of macroinvertebrates and fish. Macroinvertebrates and fish tissues were analyzed for metal content. Conclusions indicate that metal pollution in the upper Arkansas River near Leadville is severe, and that water quality as well as fish and macroinvertebrate populations have been impacted.

CDOW examined brown trout populations in the upper Arkansas River between East Fork and Lake Creek and its tributaries in 1982 (CDOW, 1982). Results of the investigations demonstrated a reduction in the abundance of trout downstream from the Leadville Drainage Tunnel. The authors observed that the metal pollution from California Gulch correlated with the fish population decrease observed immediately downstream from the discharge point.

EPA (1983) summarized existing literature documenting potential effects of metals on aquatic life. The metals investigated include zinc, manganese, cadmium, iron, and copper. The authors reported harmful concentrations of metals to aquatic biota in surface water within the California Gulch site.

In 1986, CDOW evaluated chronic lead, zinc, copper, and cadmium contamination effects in brown trout in the Arkansas River from above Leadville to Coaldale (CDOW, 1986). Results indicated that lead bioaccumulation in liver and kidney tissues was not significant. Zinc occurred at levels that fell within an acute toxicity range of 0.3 to 1.0 milligram per liter (mg/ℓ) for four species of trout (rainbow, brown, cutthroat, and brook). Results also indicated bioaccumulation of zinc in liver and kidney tissues. Copper was observed to bioaccumulate in the liver with both increasing exposure concentrations and exposure time. Copper bioaccumulation appeared to be occurring through food chain uptake rather than osmotically across the gills or other absorptive tissues. Cadmium appeared to pose the most significant risk since it bioaccumulates in both liver and kidney tissues with increasing exposure concentration and exposure time.

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ERT (1987) conducted habitat suitability studies in the upper Arkansas River to better understand the ecology of brown trout. Results indicate that habitat suitability was good near the confluence with Tennessee Creek and 0.5 miles downstream from the confluence with Lake Fork. Habitat suitability was moderate approximately 1,500 feet downstream from the confluence with California Gulch. The habitat suitability was poor on the East Fork of the Arkansas River near State Highway 91 and on the Arkansas River approximately 3.5 miles downstream of the confluence with Lake Fork. The predominant spawning areas for brown trout appeared to be in the lower Lake Fork and in the middle reaches of the East Fork of the Arkansas River (ERT, 1987).

ENSR (Drotter, 1989) conducted laboratory bioassays with Arkansas River water to determine acute toxicity to fathead minnows (*Pimephales promelas*) and daphnia (*Cerodaphnia dubia*). Water used for the bioassays was collected from the East Fork of the Arkansas River, Tennessee Creek, the Arkansas River above and below California Gulch, Lake Fork, Chalk Creek, and the Arkansas River at Malta. The criterion for effect was death. Results indicate that water collected from California Gulch was the most acutely toxic to both test species. These data are discussed in greater detail in Section 3.

Roline (EPA, 1990b) investigated the effects of the water diversion from the Frying Pan River to the Turquoise Reservoir on the macroinvertebrate populations in the upper Arkansas River in 1983. Results demonstrated that macroinvertebrate density as well as diversity have increased since the diversion was installed, except in the areas of the river that are most affected by the Leadville Drainage Tunnel and California Gulch.

CDOW (1990) reported that the water quality of the Arkansas River was adversely impacted by California Gulch. The report concludes that fish and macroinvertebrates were negatively impacted downstream of California Gulch.

The USFWS (1993) assessed trout populations in the Upper Arkansas River Basin. This report states that fewer fingerling brown trout survive during years of high spring runoff. The report also documents the reduced alkalinity in California Gulch, and the metal concentrations in the

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Arkansas River downgradient of California Gulch. Above the confluence of the Arkansas River and California Gulch, most metals occurred at concentrations below acute and chronic standards for trout, with the exception of zinc, which exceeded chronic criteria. However, downstream of California Gulch, cadmium, copper, manganese, and zinc were elevated by over two orders of magnitude. The number of brown trout collected upstream of the confluence of Lake Creek and the Arkansas River was much lower than downstream. In addition, 3- to 4-year old brown trout were virtually absent downstream of California Gulch.

Levy (1990) investigated metals contamination in soils and plant species and the subsequent adverse impacts to livestock health and plant growth. The distribution of lead, copper, zinc, and cadmium were characterized and forage metal concentrations were determined for individual plant species consumed by livestock in the meadows that were studied. Concentrations ranged from 3.4 to 21 mg/kg for copper, 0.11 to 21 mg/kg for cadmium, 0.25 to 0.52 mg/kg for lead, and 26 to 630 mg/kg for zinc. Results indicated that forage metal concentrations commonly exceeded the maximum levels chronically tolerated by cattle according to the National Research Council, resulting in detrimental impacts to livestock health. Iron deficiencies in several plant species were found to be related to excessive zinc.

1.2.4 Conceptual Site Exposure Model

A conceptual site exposure model consists of a series of working hypotheses that reflect how stressors may affect the ecological receptors in the environment. The model includes descriptions of the ecosystem that are potentially at risk as well as the relationship between assessment and measurement endpoints selected for the assessment (EPA, 1992a). The stressors at California Gulch are the inorganics released to the environment as a result of mining-related activities. EPA (1990b; 1991b) summarized the available historical analytical data from previous studies in a series of documents entitled Site Management Plan, Volumes I, II, and III. These data were considered during development of the site conceptual model.

The conceptual model was developed in meetings that included EPA, CDOW, CDH, Resurrection Mining Company, and ASARCO and their contractors. During the model

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development, discussion involved a preliminary analysis of the ecosystem, the toxicological characteristics of inorganics present at the site, and ecological effects that could result due to exposure to the inorganics. Potential exposure scenarios were developed that described how the ecological receptors were likely to interact with the stressors (i.e., inorganics). The exposure scenarios include a source, environmental transport mechanism, and potential routes of exposure. Although many scenarios were considered, only the most likely to contribute to ecological risk were analyzed further in the risk assessment.

The conceptual site exposure model was developed in order to focus the risk assessment on complete exposure pathways. A complete exposure pathway includes a contamination source, release and transport processes, a receptor for contaminant exposure, and an exposure route whereby contamination enters the receptor. The conceptual model (Figure 1-2) assists in focusing RI efforts to ensure that data pertinent to the ecological risk assessment are collected and/or evaluated. The model outlines potential exposure pathways for general groupings of ecological receptors. In this manner, media that are likely to present a source of exposure to site-generated contaminants are identified.

The abiotic media of concern for the BERA include surface water and sediments. These media provide the most direct and significant means of exposure to ecological receptors. Waste rock and tailings surface soils are source materials that provide the contaminant load or input of inorganics to the aquatic ecosystem. Ecological effects in the aquatic ecosystem are addressed by consideration of measured or modeled water and sediment concentrations. Migration of contaminants to shallow groundwater, surface water, or sediments is possible by several means, including surface runoff and percolation. Storm events and spring runoff are events causing an increased volume of surface runoff that are likely to wash large amounts of source material into the streams.

Groundwater can interact with the environment via plant uptake from shallow aquifers or by seeps. Oral ingestion of vegetation by herbivores, or consumption of herbivores by carnivores, can then result in dietary intake of inorganics due to food web contamination. Benthic invertebrates may contact contaminated vegetation decaying on the stream bottom; invertebrates

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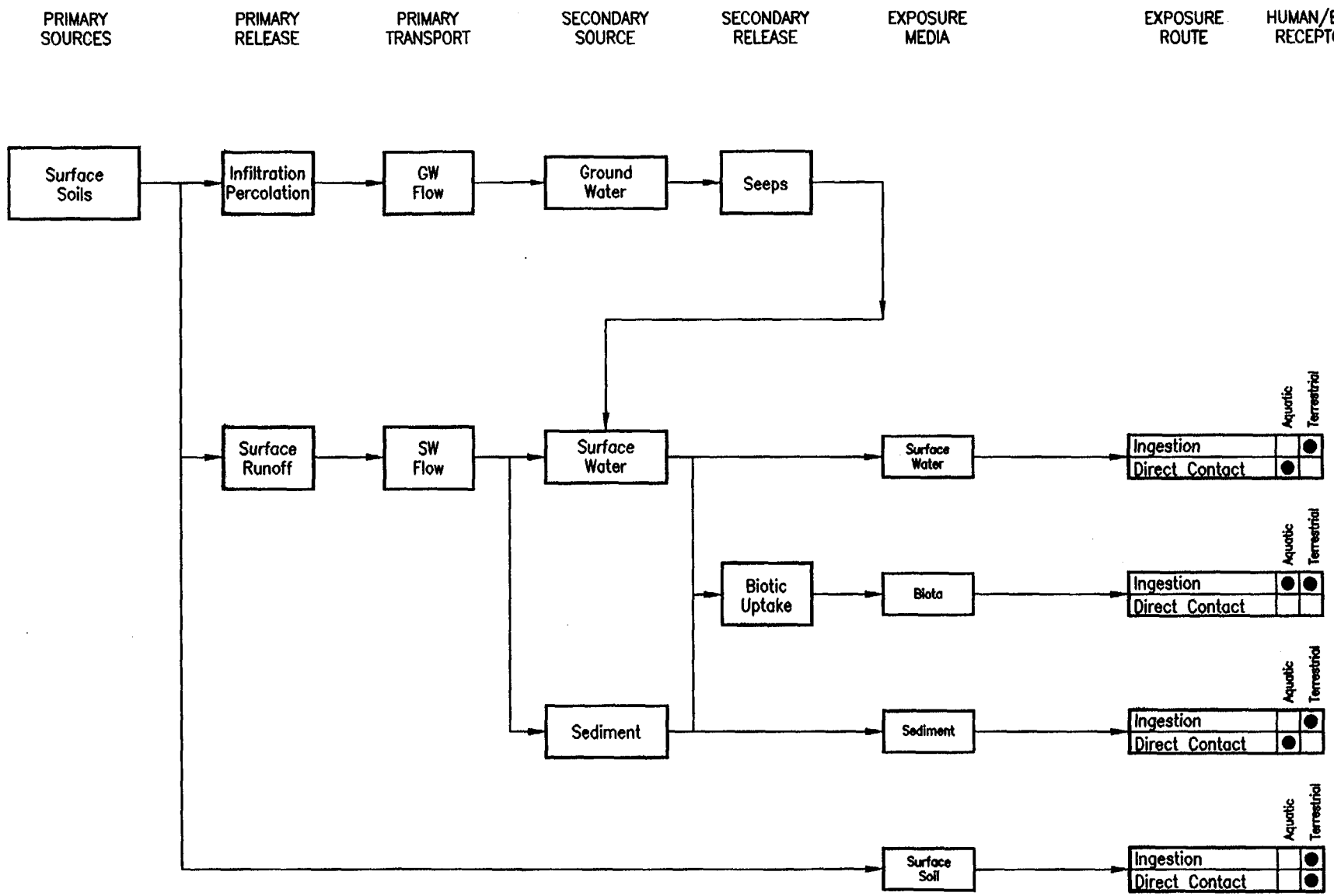

FIGURE 1-2

CONCEPTUAL MODEL FOR CALIFORNIA GULCH ECOLOGICAL ASSESSMENT
(10/31/93)

are then consumed by fish. Direct or dermal contact with surface water, or oral ingestion of surface water contaminated by groundwater seeps, is also possible. Risk due to contact with groundwater was analyzed indirectly by using measured inorganic concentrations in surface water as it is uncertain to what extent aquatic life contact groundwater discharging to surface water.

Surface water and sediments may be contaminated by runoff and percolation from soil, waste rock, or tailing piles. Aquatic life may then be exposed directly to inorganics in surface water or sediments; terrestrial life may be exposed by ingestion of surface water containing dissolved inorganics or suspended sediments as a drinking water source. Some of the inorganics have bioaccumulation tendencies; these inorganics may be taken up by aquatic plants, invertebrates, and fish and enter the aquatic food web as described above for groundwater. Terrestrial species that feed upon aquatic life may also be exposed by food web contamination; however, risk to terrestrial species will be addressed in the terrestrial risk assessment.

1.2.5 Applicable or Relevant and Appropriate Requirements

The Applicable or Relevant and Appropriate Requirements (ARARs) identified at this time for ecological risk assessment are the Federal Ambient Water Quality Criteria for the Protection of Freshwater Aquatic Life and Their Uses (AWQC) and the State Water Quality Standards. The criteria for inorganics are summarized in the Toxicity Assessment section (Table 4-3 through 4-5). The most current AWQC values currently available were used (EPA, 1992b).

1.2.6 Ecological Assessment Endpoints

Assessment endpoints are defined as those ecological values or parameters for which protection is considered critical. Assessment endpoints are defined for different ecological receptors in Table 1-2. An ideal assessment endpoint has social or biological relevance, has an unambiguous definition, is measurable or predictable, is susceptible to the hazard, and assists the risk manager in making the decisions regarding ecological risk (EPA, 1989c).

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Table 1-2

Summary of Endpoints for the BERA

Assessment Endpoint	Measurement Endpoint
Productive capability of California Gulch	Invertebrate biomass, invertebrate density, invertebrate diversity, periphyton diversity, species occurrence, abiotic data
Productive capability of the Arkansas River	Fish biomass, invertebrate biomass, invertebrate density, invertebrate diversity, periphyton diversity, species occurrence
Extinction of trout species in the Arkansas River	Species occurrence, prey data (invertebrate diversity/density), fish and invertebrate bioassay data, inorganic concentrations in water and sediment
Abundance of fish in the Arkansas River and its tributaries	Abundance, age/size class data

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Measurement endpoints are the specific variables that are measured in order to predict affects on the assessment endpoints. The measurement endpoints must correspond to or predict an assessment endpoint, either directly or via a mathematical or statistical model (EPA, 1989c). If neither of the above are possible, then the measurement endpoint should be so protective or sensitive to the expected adverse effects relative to the site that if the measurement endpoints show no effect, it is likely that nothing will be affected (EPA, 1989c). An ideal measurement endpoint corresponds to an assessment endpoint, is easily measured, is appropriate to the site scale and exposure pathways, has appropriate temporal dynamics, is diagnostic and has low innate variability, is broadly applicable and standardized, and is represented by existing site or literature data. Measurement endpoints considered for the BERA for the California Gulch site are listed in Table 1-2.

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SECTION 2
EXPOSURE ASSESSMENT

For this Exposure Assessment, contaminant concentrations in water and sediment were evaluated, from data collected both prior to completion of the Yak Tunnel and following remedial actions, to determine exposure by the ecological receptors. For the aquatic baseline risk assessment, ecological receptors are the aquatic species likely to be exposed to metals. Exposure point concentrations were calculated at each sampling station where surface water or sediment data were collected. Both the arithmetic mean and the upper 95th percent confidence interval for the mean (UCL95) are used as exposure point concentrations. The data summarized in the Exposure Assessment are compared to the EPA Ambient Water Quality Criteria for the Protection of Freshwater Aquatic Life (Section 4) in order to evaluate risk to aquatic life (Section 5).

Data in the quantitative exposure assessment are referred to by station; stations are the specific sites that were sampled, such as AR01. In addition to stations on stream systems, data for springs and other surface water features sampled at the site are evaluated. A description of station locations is provided in Table 2-1.

The sample locations within the site boundaries or receiving flow from streams within the site boundaries are located in the lower Arkansas River (AR03, AR04, AR05, AR06), California Gulch (CG), and the tributaries of California Gulch (Figure 1-1). The tributaries include Starr Ditch (SD), Airport Gulch (AG), Stray Horse Gulch (SG), Georgia Gulch (GG), Oregon Gulch (OG), Malta Gulch (MG), and Pawnee Gulch (PG). Evans Gulch (EG) also receives input from source areas on-site.

Data collected from January 1991 through January 1992 as part of the RI activities preceded the Yak Tunnel remediation. Data collected during the period that water treatment was initiated (March 1992) were not used because of the potentially dynamic nature of the water quality at this time. Data collected from July 1992 through October 1993, following initiation of water treatment, are discussed as well. The data collected after the water treatment plant became

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Table 2-1

Description of Sampling Station Locations

Sampling Station		Location
New Sampling Station ID (Summer/Fall 1991 and 1992)	Old Sampling Section ID (Ice-Off/Spring 1991)	
AR05	AR01	Arkansas R. above confluence with Empire Gulch
AR04	AR02	Arkansas R. below confluence with Halfmoon Creek - Lake Fork
AR3A	NEW	Arkansas R. below confluence with Cal Gulch
AR3E	AR03	East 1/3 Arkansas R. in mixing zone with Cal Gulch water
AR3W	AR03	West 2/3 Arkansas R. in mixing zone with Cal Gulch water
AR02	AR04	Arkansas R. above confluence with Cal Gulch
AR01	AR05	Arkansas R. below confluence with Arkansas R.
TC01	TC01	Tennessee C. above confluence with Arkansas R.
EF02	EF01	East Fork below confluence with Evans Gulch
EF01	EF02	East Fork above confluence with Evans Gulch
EM04	EM01	Empire Gulch above confluence with Arkansas R.
EM03	EM02	Empire Gulch above 1-3/4 mi. above EM04
EM02	EM03	Empire Gulch above Beaver Lakes subdivision
EM01	EM04	Empire Gulch at old boiler
IG01	IG01	Iowa Gulch above confluence with Arkansas R.
HC01	HC01	Halfmoon Creek above confluence with Arkansas R.
LF01	LF01	Lake Fork above confluence with Arkansas R.
CG06	CG01	Cal Gulch above confluence with Arkansas R.
CG05	CG02	Cal Gulch below confluence with Airport Gulch
CG04	CG03	Cal Gulch below confluence with Oregon Gulch
CG03	CG04	Cal Gulch above confluence with Starr Ditch
CG02	CG05	Cal Gulch below Resurrection Tailings
CG01	CG06	Cal Gulch above confluence with Yak Tunnel
MG01	MG01	Malta Gulch above confluence with Cal Gulch
AG01	AG01	Airport Gulch above confluence with Cal Gulch
PG01	PG01	Pawnee Gulch above confluence with Cal Gulch
CG01	CG01	Georgia Gulch above confluence with Cal Gulch
OG01	OG01	Oregon Gulch above confluence with Cal Gulch
SD01	SD01	Starr Ditch above confluence with Cal Gulch
SG01	SG01	Stray Horse Gulch above culvert inlet at 5th St.
EG02	EG01	Evans Gulch below confluence with L. Evans Gulch
EG01	EG02	Evans Gulch above confluence with L. Evans Gulch
LE01	LE01	Little Evans Gulch above confluence with Evans Gulch

Note: Some of the sampling locations have changed over time. The drainage areas estimated to each sampling location are approximate and are used to estimate areal loading rates only. Source: Water, Waste and Land, Inc.

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operative are the most pertinent for the risk assessment as these data are more likely to reflect current and future conditions at the site. The 1991 data are evaluated for comparative purposes.

The data utilized in the Exposure Assessment are classified as EPA Data Quality Level III or IV. This classification takes into consideration the level of Quality Control/Quality Assurance associated with the data. The data used in the Exposure Assessment are more consistent in collection methods and sample locations than the analytical data collected prior to 1991 for this site. These data are thus more likely to provide comparable and representative values for risk assessment purposes.

2.1 ECOLOGICAL RECEPTORS

Ecological receptors are considered to be exposed to contaminants via exposure pathways. A complete exposure pathway consists of a contaminant source, a receptor, a transport mechanism, and an exposure route. The exposure pathways were presented in the site conceptual model (Figure 1-2). Direct contact with surface water or sediments, or ingestion of contaminated prey or food items, are considered to be the primary exposure pathways for aquatic life.

The receptors in the aquatic ecosystem include algae and plankton, macrophytes, macroinvertebrates, amphibians, and fish, primarily trout species. The benthic macroinvertebrates are important because they serve as prey for trout species. Many other species depend on the aquatic ecosystem for their survival, although they are not strictly aquatic. These include waterfowl and other avian species that inhabit riparian or wetland areas. Many mammals also occur in and depend on wetland areas as habitat. Risk to ecological receptors that utilize the aquatic ecosystem other than strictly aquatic life is addressed in the terrestrial risk assessment.

2.2 STRESSORS

The stressors considered critical at the California Gulch site are the contaminants of concern (COCs). The COCs were selected by evaluating current data for surface water and sediments.

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An inorganic analyte was considered a potential COC in the aquatic ecosystem if the analyte was a potential threat to environmental quality. Analytes that are considered nontoxic but for which there were analytical data (i.e., calcium and magnesium) were screened out in this manner.

The COCs for this evaluation are: aluminum, arsenic, barium, cadmium, copper, iron, manganese, nickel, lead, antimony, selenium, and zinc.

Other analytes were rejected from further evaluation in the quantitative exposure assessment due to the high number of nondetects. It should be noted that the fact that an analyte is not detected does not necessarily indicate that it is not a potential threat in the environment; for example, detection limits for mercury exceed the chronic AWQC. Technical limitations may result in detection limits that exceed the levels of concern. The presence of matrix interferences or chemical interferences can elevate the detection limits substantially, producing high numbers of nondetects. Dissolved mercury, molybdenum, silver, and chromium were not evaluated in surface water in the quantitative exposure assessment due to the high proportion of nondetects.

2.3 EXPOSURE PATHWAYS

A complete exposure pathway consists of the following elements:

- Contaminant source and release mechanism.
- Transport medium.
- Exposure point for the ecological receptors.
- Route of exposure at the point of contact.

Aquatic biota are continually exposed to contaminants in surface water and sediments. For fish and aquatic invertebrates, the primary source of exposure for inorganic analytes in water likely occurs via intake across the gill membranes. Analytes adsorbed to suspended sediments also contribute to exposure, because particulates are ingested during feeding. Exposure can also occur as a result of ingestion of contaminated prey, surface water, and sediments. Exposure due to transfer across dermal membranes may also be possible, as some inorganics may concentrate

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in either the fish mucus layer or the chitin layer of invertebrates. The relative importance of the different routes of exposure for fish and aquatic invertebrates is difficult to assess individually; therefore, the primary exposure pathways considered for fish and aquatic invertebrates are:

- Direct contact with dissolved contaminants in surface water.
- Direct contact with dissolved contaminants in sediment pore water (the water filling the interstitial spaces between sediment solids).

Aquatic macroinvertebrates and fish are assumed to be exposed directly to the average and the Reasonable Maximum Exposure (RME) (lower of the observed maximum or UCL95 concentrations) point concentrations. Although aquatic life are exposed to the complete range of metals in the aquatic ecosystem, it is typical for risk assessment to address the average as a typical exposure and the RME as a usual worst case. Actual concentrations in sediment were converted to estimated pore water concentrations to estimate exposure point concentrations because contaminants in pore water are likely to be more bioavailable for uptake. Surface water dissolved metals were evaluated as exposure point concentrations to address adsorption and absorption by invertebrates and fish in the water column.

Aquatic plants are also exposed to contaminants in water and sediment. Uptake from water may be possible across root membranes or other portions of the plant. The dissolved fraction in sediment pore water is likely to be more bioavailable than sediment solids to plants. The primary exposure pathways for aquatic plants are:

- Direct contact with dissolved contaminants in surface water.
- Direct contact with dissolved contaminants in sediment pore water.

Aquatic plants are assumed to be exposed directly to the average and RME exposure point concentrations. Actual concentrations in sediment were converted to pore water concentrations to estimate exposure point concentrations because contaminants in pore water are likely to be

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more bioavailable for root uptake. Surface water dissolved metals were evaluated as exposure point concentrations to address adsorption and absorption to submerged vegetative surfaces.

Exposure intakes (milligram contaminant/kilogram body weight/day) were not estimated for the truly aquatic species such as fish and benthic invertebrates. Species-specific data for water or dietary ingestion rates were therefore not required. The water and sediment pore water exposure point concentrations used as in the exposure assessment are compared to federal criteria (i.e., EPA AWQC), which are designed to represent all aquatic species.

2.4 EXPOSURE POINT CONCENTRATIONS

The average concentration at each station (exposure point) represents a minimal average exposure expected to occur during the period data were collected, and the UCL95 is utilized as the RME. This is consistent with current guidance from EPA regarding risk assessment (EPA, 1989). It is important to consider the UCL95 as an RME for the California Gulch Ecological Risk Assessment because the data indicate both seasonal highs and variability with sampling events, and the average concentration does not reflect the exposure during seasonal highs or periods of high runoff. The UCL95 is applicable unless it exceeds the maximum observed value for the station and analyte, in which case, the maximum value is used to represent the RME.

Uncertainty is associated with estimating a true average concentration in abiotic media. Therefore, the UCL95 is used to represent the true average because there is reasonable confidence that underestimation of the true average will not occur (EPA, 1992). Use of the UCL95 as the RME accounts for uncertainty and variability in the contaminant concentration which may not be fully assessed due to limited sampling data.

The UCL95 was predicted from the Least Significant Differences (LSD) method using a pooled standard error. Pooled standard error was used because the internal standard error for each station was often zero as a result of limited sample sizes and many nondetect values. However, the confidence interval based on the LSD method is narrower than that based on the *H* statistic recommended by Gilbert (1987) and EPA (1992) if the data are lognormal. When applied to

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subsets of the surface water data, the *H* statistic produced such high UCL95 values that the maximum would have been used in every case to represent the upper bounds on exposure. Therefore, calculation of the *H* statistic did not seem useful in approximating a conservative estimate of the average. As can be determined from the tables in this section, the maximum observed value is used more often than the UCL95 to represent exposure even when the H statistic is not applied.

The purpose in using the RME is to estimate the high end of exposure. As can be determined later in this section, seasonal high concentrations exceed the RME (CDOW, 1994). Therefore, risk to aquatic life may be underestimated periodically.

2.4.1 Data Analysis

Prior to evaluating the analytical data, all of the values that received a "U" laboratory data qualifier (i.e., below detection) were divided by 2 to obtain one-half of the detection limit for that sample. The "B" qualified data were not divided by 2; the "B" qualifier refers to the data reported between the Contract Reporting Limit (CRL) and the instrument detection limit (IDL). Data between the CRL and IDL are valid. Data that were rejected by the validation process (R qualified) were not used in the quantitative exposure assessment.

In addition, each field duplicate sample was averaged with its corresponding field sample to obtain an arithmetic mean. This was determined to be the least biased way of utilizing the data for several reasons:

- One value could be reported below detection and the other above detection. To always select the value above detection presents a risk of biasing the data towards the higher concentrations.
- To discard all the field duplicates regardless of concentration may also bias the data because the field duplicate samples represent measurement error; there is no reason to expect that the field duplicates are more in error than the field samples.

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Using the arithmetic mean of the duplicates accounts for the presence of error in either sample results, providing a better estimate of actual concentration at the site. The complete data set used in the quantitative exposure assessment, including data qualifiers and field duplicates, is presented in Appendix B.

2.4.2 Data Collected Prior to Initiation of Yak Tunnel Water Treatment

2.4.2.1 Dissolved Inorganic Analytes in Surface Water

Current water quality criteria for metals utilize dissolved inorganic analytes in water as the basis for determining exposure and ecological risk. Total metals in surface water include metals bound to particulates, which often are not biologically available. Use of total metals data may therefore be too conservative (i.e., the water concentrations and associated predicted ecological risk may be too high). The dissolved fraction can correlate more closely with observed toxicity in many aquatic species, and is recommended as the basis for current water quality criteria (EPA, 1993).

Appendix B-1 provides the raw data for dissolved inorganic analytes in surface water. Cyanide, molybdenum, mercury, chromium, and silver were predominantly below detection in surface water. These elements were not analyzed further in the quantitative exposure assessment.

Table 2-2 presents arithmetic average concentrations of analytes in surface water for samples collected prior to March 1992. These values were obtained by averaging data for all sampling events at each station. For the 1991 to 1992 data, this included:

- Ice-off (April to May, 1991).
- Summer (June to August, 1991).
- Fall (September to October, 1991).

The average concentrations represent the water concentration to which aquatic life are usually exposed. However, storm events can produce significantly higher concentrations in stream systems that could reasonably last for several days to several weeks. These periods of elevated

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Table 2-2

Average Concentrations of Dissolved Inorganics in Surface Water - 1991 Data

ug/l

Station	Al	As	Ba	Be	Cd	Cu	Fe	Mn	Ni	Pb	Sb	Se	Tl	Zn
AG01	93.80	8.40	5.00	NA	5.20	38.40	121.00	113.00	10.00	55.80	20.00	0.50	NA	171.00
AR01	25.00	0.50	38.06	NA	0.29	2.95	146.90	55.53	10.00	0.50	20.00	0.50	NA	21.70
AR02	25.00	0.68	48.50	NA	1.23	4.00	146.20	52.53	10.00	0.63	15.78	0.65	NA	58.90
AR03	25.00	0.50	41.85	NA	1.40	5.30	199.60	196.00	10.00	0.50	20.00	0.50	NA	258.00
AR03A	25.00	0.50	56.00	NA	1.00	1.45	43.50	138.50	10.00	0.50	14.90	0.50	NA	66.30
AR03E	25.00	0.80	53.25	NA	0.85	3.30	61.40	259.50	10.00	0.50	13.60	0.50	NA	90.95
AR03W	25.00	0.50	53.45	NA	0.23	3.15	68.00	68.15	10.00	0.50	12.70	0.50	NA	8.10
AR04	25.00	0.50	30.15	NA	1.38	6.95	144.40	219.50	10.00	0.68	17.45	0.50	NA	86.03
AR05	25.00	8.00	31.93	NA	2.25	14.70	116.20	100.00	10.00	9.70	20.00	6.05	NA	97.00
CG01	11600.50	0.50	22.35	NA	173.50	372.05	1270.00	4350.00	19.75	842.00	20.00	5.25	NA	21450.00
CG02	4001.00	0.50	22.95	NA	210.00	182.80	4950.00	10825.00	18.40	263.55	20.00	2.75	NA	26000.00
CG03	93.75	0.68	37.70	NA	122.84	220.55	695.50	9191.25	18.87	13.10	17.02	1.00	NA	33251.00
CG04	3135.25	0.83	42.63	NA	155.10	460.88	28191.30	40350.00	54.98	652.63	18.20	2.88	NA	44780.00
CG05	1620.75	0.50	38.43	NA	119.48	233.98	6962.50	23475.00	32.50	181.20	16.25	1.50	NA	32192.50
CG06	25.00	0.50	26.78	NA	49.20	11.75	12.50	9696.67	14.65	1.13	16.25	0.68	NA	11337.25
EF01	25.00	0.50	71.03	NA	0.13	1.67	32.70	11.43	10.00	0.50	14.40	0.50	NA	5.00
EF02	25.00	0.50	53.18	NA	0.46	2.27	20.00	179.87	10.00	0.50	14.63	0.50	NA	9.97
EG01	25.00	0.50	59.45	NA	0.39	2.05	10.00	5.00	10.00	0.50	12.05	0.50	NA	5.00
EG02	25.00	0.50	50.80	NA	0.54	4.00	10.00	5.00	10.00	0.50	20.00	0.50	NA	5.00
EM01	25.00	0.50	71.63	NA	0.05	1.70	37.30	5.00	10.00	0.50	17.17	0.50	NA	5.00
EM02	42.65	0.50	86.23	NA	0.07	1.33	45.50	16.23	10.00	0.93	16.80	0.50	NA	5.00
EM03	25.00	0.68	90.23	NA	0.07	1.65	82.00	18.30	10.00	0.50	16.53	0.50	NA	5.00
EM04	25.00	0.73	99.38	NA	0.43	7.13	50.80	15.55	10.00	0.50	17.35	0.38	NA	5.00
GG01	85.80	11.20	5.00	NA	1.90	9.70	61.00	5.00	10.00	15.50	20.00	0.50	NA	34.00
HC01	25.00	0.50	16.43	NA	0.07	7.45	28.90	6.33	10.00	0.50	16.50	0.50	NA	5.00
IG01	25.00	0.50	45.18	NA	0.15	2.93	23.30	14.08	10.00	1.25	16.30	0.50	NA	4.35
LF01	25.00	0.50	31.58	NA	0.80	2.60	248.00	49.75	12.50	0.50	20.00	0.50	NA	6.68
MG01	77.30	1.20	143.00	NA	63.20	43.40	48.80	448.00	10.00	226.00	20.00	0.50	NA	3280.00
OG01	143516.67	141.90	140.27	NA	443.00	3861.00	2392666.70	1025670.00	536.00	23.90	506.66	6.67	NA	379000.00
PG01	88.10	3.80	35.40	NA	2.90	13.20	105.00	57.90	10.00	12.90	20.00	0.50	NA	177.00
SD01	3816.67	0.80	42.60	NA	183.57	309.87	3602.50	5962.67	13.47	302.70	20.00	0.50	NA	19572.33
SG01	21508.33	370.33	42.43	NA	497.33	2906.63	365883.20	22986.70	45.70	647.00	80.00	1.17	NA	53600.00

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Table 2-2 (continued)

Average Concentrations of Dissolved Inorganics in Surface Water - 1991 Data

ug/l

Station	Al	As	Ba	Be	Cd	Cu	Fe	Mn	Ni	Pb	Sb	Se	Tl	Zn
SPR01	100.00	NA	400.00	5.00	27.00	50.00	17300.00	23400.00	80.00	66.90	120.00	5.00	29.20	47300.00
SPR02	100.00	10.00	400.00	5.00	78.85	50.00	118.00	9035.00	80.00	26.35	60.00	10.00	40.80	41000.00
SPR03	2480.00	25.00	1000.00	12.50	76.90	170.00	28200.00	95000.00	200.00	1010.00	150.00	12.50	68.20	77200.00
SPR04	100.00	10.00	400.00	5.00	50.30	25.00	50.00	35.30	80.00	6.00	120.00	5.00	10.00	2610.00
SPR05	100.00	10.00	400.00	5.00	52.70	25.00	50.00	30.00	80.00	6.00	120.00	10.00	33.60	17700.00
SPR07	250.00	25.00	1000.00	12.50	25.00	62.50	326000.00	152000.00	130.50	25.70	150.00	12.50	116.00	100000.00
SPR08	3330.00	20.00	800.00	10.00	50.50	101.00	168000.00	139000.00	222.00	860.00	120.00	20.00	94.60	88100.00
SPR09	100.00	10.00	400.00	5.00	149.00	50.00	5440.00	13100.00	80.00	99.50	120.00	10.00	36.60	55000.00
SPR10	100.00	10.00	400.00	5.00	10.00	25.00	57000.00	3450.00	80.00	10.20	120.00	5.00	10.00	431.00
SPR11	100.00	10.00	400.00	5.00	5.00	25.00	100.00	95.30	40.00	3.00	120.00	5.00	10.00	NA
SPR12	100.00	10.00	400.00	5.00	5.00	37.50	100.00	328.50	40.00	3.00	120.00	5.00	5.00	40.00
SPR13	100.00	10.00	400.00	5.00	10.00	25.00	100.00	30.00	80.00	3.00	120.00	5.00	14.80	70.70
SPR14	100.00	10.00	449.00	5.00	5.00	50.00	100.00	538.00	40.00	3.00	120.00	5.00	5.00	20.00
SPR15	21500.00	10.00	400.00	10.00	28.00	369.00	874.00	2100.00	80.00	50.60	120.00	5.00	10.00	6230.00
SPR16	27500.00	10.00	400.00	5.00	29.10	1860.00	17100.00	1270.00	80.00	3.00	120.00	5.00	10.60	1350.00
SPR17	18900.00	10.00	400.00	10.00	35.70	162.00	1310.00	2520.00	80.00	24.10	120.00	5.00	10.00	7980.00
TC01-1	45.60	0.50	8.95	NA	0.49	7.35	419.50	42.70	10.00	0.50	20.00	0.50	NA	58.45
TC01-2	25.00	0.50	11.50	NA	0.05	7.70	169.00	20.60	10.00	1.60	3.60	NA	NA	10.60
TP01	9660.00	10.00	400.00	5.00	10.00	337.00	7240.00	1870.00	80.00	3.00	120.00	5.00	10.00	505.00
TP02	334.00	10.00	400.00	5.00	5.80	50.00	100.00	1670.00	80.00	8.80	120.00	5.00	5.00	1390.00
TP03	100.00	10.00	400.00	5.00	5.00	25.00	50.00	30.00	40.00	1.50	120.00	5.00	5.00	40.00

Note: 1991 data only

concentrations may adversely impact the aquatic community when no risk is apparent from average water concentrations. Therefore, the RME was used to represent maximum exposure concentrations (Table 2-3). As stated previously, the RME is the lower of the UCL95 or the maximum observed concentration.

The RME is intended to represent storm events or other periods where water concentrations are elevated. The UCL95 is typically higher than the average concentration. The UCL95 reflects variability within the samples, and the greater the variability, the higher the UCL95. For this reason, the UCL95 may exceed maximum measured concentrations at any given station. Table 2-3 presents the maximum concentration at each station as well as the UCL95. The maximum is compared to the UCL95, and the lower of the two values is used to represent the exposure point concentration (i.e., when maximum value at a station exceeds the UCL95, the UCL95 is used; when the maximum is less than the UCL95, the maximum is used). The logic behind this selection of maximum versus UCL95 as the RME is that the UCL95 may be inappropriately conservative when it exceeds maximum observed concentrations, and measured concentrations may never be as high as the UCL95. Validation of this approach was performed by comparing the RME concentrations to data collected by CDOW (Woodling, 1994); the validation is described in Section 2.4.4.

The UCL95 was calculated using the pooled standard error for each analyte as opposed to the internal standard error. This provided a wider confidence interval than would be obtained by consideration of each station individually. This was necessary because many stations had no internal standard error due to missing data, sample quantities of one, or because all of the data were below detection for a given station. When the error term is zero, the UCL95 calculation reduces to the average, and an RME can not be obtained.

Most concentrations of inorganics in California Gulch were elevated relative to other tributaries of the Arkansas River (Table 2-2 and 2-3). For example, average zinc concentrations were over three orders of magnitude higher in samples from California Gulch compared to samples from East Fork (EF01, EF02), the Arkansas River upgradient of California Gulch (AR01, AR02), Lake Fork (LF01), or Iowa Gulch (IG01) (Table 2-2). Other streams originating on-site also

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Table 2-3

RME Concentrations of Dissolved Inorganic COCs in Surface Water - 1991 Data

Station	UCL 95 Al	Max Al	Exp.Pt Al	UCL95 As	Max As	Exp.Pt. As	UCL95 Ba	Max Ba	Exp.Pt. Ba	UCL95 Cd	Max Cd	Exp.Pt. Cd
AG01	51080.62	93.80	93.80	177.00	8.40	8.40	40.77	5.00	5.00	168.13	5.20	5.20
AR01	22008.08	25.00	25.00	84.80	0.50	0.50	55.95	63.40	55.95	81.76	0.38	0.38
AR02	22008.08	25.00	25.00	85.00	1.20	1.20	66.39	65.80	65.80	82.69	3.90	3.90
AR03	31113.77	25.00	25.00	84.80	0.50	0.50	67.14	60.10	60.10	116.61	1.60	1.60
AR03A	31113.77	25.00	25.00	119.80	0.50	0.50	81.29	59.80	59.80	163.93	1.00	1.00
AR03E	31113.77	25.00	25.00	120.00	1.10	1.10	78.54	60.10	60.10	116.06	1.20	1.20
AR03W	31113.77	25.00	25.00	85.00	0.50	0.50	78.74	63.00	63.00	115.44	0.32	0.32
AR04	22008.00	25.00	25.00	84.80	0.50	0.50	48.04	36.10	36.10	82.84	1.50	1.50
AR05	25408.87	25.00	25.00	105.40	23.00	23.00	52.58	43.10	43.10	117.46	2.70	2.70
CG01	42689.27	22500.00	22500.00	119.80	0.50	0.50	47.64	31.00	31.00	288.71	225.00	225.00
CG02	35089.77	7800.00	7800.00	119.80	0.50	0.50	48.24	28.20	28.20	325.21	257.00	257.00
CG03	22076.83	300.00	300.00	85.00	1.20	1.20	55.58	44.20	44.20	204.30	327.00	204.30
CG04	25118.33	8920.00	8920.00	85.20	1.30	1.30	60.51	65.70	60.51	236.57	313.00	236.57
CG05	23603.83	6110.00	6110.00	84.80	0.50	0.50	56.31	61.10	56.31	200.94	254.00	200.94
CG06	22008.00	25.00	25.00	84.80	0.50	0.50	44.66	37.10	37.10	130.67	137.00	130.67
EF01	25408.87	25.00	25.00	97.90	0.50	0.50	91.68	104.00	91.68	94.20	0.30	0.30
EF02	25408.87	25.00	25.00	97.90	0.50	0.50	73.83	77.00	73.83	94.53	0.88	0.88
EG01	31113.77	25.00	25.00	119.80	0.50	0.50	84.74	69.30	69.30	115.60	0.46	0.46
EG02	43991.16	25.00	25.00	169.20	0.50	0.50	86.57	50.80	50.80	163.47	0.54	0.54
EM01	25408.87	25.00	25.00	97.90	0.50	0.50	92.29	83.00	83.00	115.26	0.05	0.05
EM02	22025.73	95.60	95.60	84.80	0.50	0.50	104.11	96.00	96.00	94.14	0.12	0.12
EM03	22008.08	25.00	25.00	85.00	1.20	1.20	108.11	111.00	108.11	94.14	0.12	0.12
EM04	22008.08	25.00	25.00	85.10	1.40	1.40	117.26	108.00	108.00	81.90	1.50	1.50
GG01	44051.96	85.80	85.80	179.87	11.20	11.20	40.77	5.00	5.00	164.83	1.90	1.90
HC01	22008.08	25.00	25.00	84.83	0.50	0.50	34.31	21.70	21.70	94.14	0.10	0.10
IG01	22008.08	25.00	25.00	84.83	0.50	0.50	63.06	59.30	59.30	94.22	0.19	0.19
LF01	22008.08	25.00	25.00	84.83	0.50	0.50	49.46	100.00	49.46	82.26	1.50	1.50
MG01	44043.46	77.30	77.30	169.87	1.20	1.20	178.77	143.00	143.00	226.13	63.20	63.20
OG01	168900.54	331000.00	168900.54	261.16	268.00	261.16	160.92	250.00	160.92	537.07	664.00	537.07
PG01	44054.26	88.10	88.10	172.47	3.80	3.80	71.17	35.40	35.40	165.83	2.90	2.90
SD01	29200.54	11400.00	11400.00	98.18	1.40	1.40	63.25	48.30	48.30	277.64	410.00	277.64
SG01	46892.20	52200.00	46892.20	467.71	1110.00	467.71	63.08	50.00	50.00	591.40	934.00	591.40
SPR01	44066.16	100.00	100.00	NA	NA	NA	435.77	400.00	400.00	189.93	27.00	27.00
SPR02	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	241.78	78.85	78.85
SPR03	46446.16	2480.00	2480.00	193.67	25.00	25.00	1035.77	1000.00	1000.00	239.83	76.90	76.90
SPR04	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	213.23	50.30	50.30
SPR05	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	215.63	52.70	52.70
SPR07	44216.16	250.00	250.00	193.67	25.00	25.00	1035.77	1000.00	1000.00	187.93	25.00	25.00
SPR08	47296.16	3330.00	3330.00	188.67	20.00	20.00	835.77	800.00	800.00	213.43	50.50	50.50
SPR09	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	311.93	149.00	149.00
SPR10	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	172.93	10.00	10.00
SPR11	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	167.93	5.00	5.00
SPR12	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	167.93	5.00	5.00
SPR13	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	172.93	10.00	10.00
SPR14	44066.16	100.00	100.00	178.67	10.00	10.00	484.77	449.00	449.00	167.93	5.00	5.00
SPR15	65466.16	21500.00	21500.00	178.67	10.00	10.00	435.77	400.00	400.00	190.93	28.00	28.00
SPR16	71466.16	27500.00	27500.00	178.67	10.00	10.00	435.77	400.00	400.00	192.03	29.10	29.10
SPR17	62866.16	18900.00	18900.00	178.67	10.00	10.00	435.77	400.00	400.00	198.63	35.70	35.70
TC01-1	31134.37	66.20	66.20	119.76	0.50	0.50	34.24	12.90	12.90	115.70	0.56	0.56
TC01-2	43991.16	25.00	25.00	169.17	0.50	0.50	47.27	11.50	11.50	162.98	0.05	0.05
TP01	53626.16	9660.00	9660.00	178.67	10.00	10.00	435.77	400.00	400.00	172.93	10.00	10.00
TP02	44300.16	334.00	334.00	178.67	10.00	10.00	435.77	400.00	400.00	168.73	5.80	5.80
TP03	44066.16	100.00	100.00	178.67	10.00	10.00	435.77	400.00	400.00	167.93	5.00	5.00

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Table 2-3 (continued)

RME Concentrations of Dissolved Inorganic COCs in Surface Water - 1991 Data

Station	UCL95 Cu	Max Cu	Exp.Pt. Cu	UCL95 Fe	Max Fe	Exp.Pt. Fe	UCL95 Mn	Max Mn	Exp.Pt. Mn	UCL95 Ni	Max Ni	Exp.Pt. Ni
AG01	1573.19	38.40	38.40	420151.70	121.00	121.00	176682.40	113.00	113.00	105.45	10.00	10.00
AR01	770.35	4.50	4.50	148447.48	381.00	381.00	88340.20	93.90	93.90	57.72	10.00	10.00
AR02	771.40	8.20	8.20	148447.48	340.00	340.00	88337.20	92.60	92.60	57.72	10.00	10.00
AR03	1090.56	5.50	5.50	297206.10	310.00	310.00	125049.40	202.00	202.00	77.49	10.00	10.00
AR03A	1086.71	2.40	2.40	297050.00	45.10	45.10	124991.90	165.00	165.00	77.49	10.00	10.00
AR03E	1088.56	5.20	5.20	297068.00	78.70	78.70	125112.90	324.00	324.00	77.49	10.00	10.00
AR03W	1088.41	3.80	3.80	297074.50	72.90	72.90	124921.60	87.60	87.60	77.49	10.00	10.00
AR04	774.35	13.10	13.10	210159.70	254.00	254.00	88504.20	493.00	493.00	57.72	10.00	10.00
AR05	900.81	33.40	33.40	242621.00	202.00	202.00	102042.40	142.00	142.00	65.11	10.00	10.00
CG01	1457.31	713.00	713.00	298276.60	2340.00	2340.00	129203.40	6570.00	6570.00	87.24	29.50	29.50
CG02	1268.06	355.00	355.00	301956.60	8700.00	8700.00	135678.40	13800.00	13800.00	85.89	26.80	26.80
CG03	987.95	874.00	874.00	210710.00	2320.00	2320.00	97476.00	17300.00	17300.00	73.97	36.60	36.60
CG04	1228.27	888.00	888.00	238206.00	90600.00	90600.00	128634.70	59800.00	59800.00	102.70	96.10	96.10
CG05	1001.37	580.00	580.00	216977.90	22900.00	22900.00	111759.70	36400.00	36400.00	80.22	43.00	43.00
CG06	779.15	20.70	20.70	210027.90	20.00	20.00	111639.10	15100.00	15100.00	62.37	28.60	28.60
EF01	887.78	2.60	2.60	242537.50	44.90	44.90	101953.80	18.70	18.70	65.11	10.00	10.00
EF02	888.38	3.80	3.80	242524.90	40.10	40.10	102122.30	425.00	425.00	65.11	10.00	10.00
EG01	1087.31	3.60	3.60	297016.60	10.00	10.00	124858.40	5.00	5.00	77.49	10.00	10.00
EG02	1538.79	4.00	4.00	420040.70	10.00	10.00	176574.40	5.00	5.00	105.45	10.00	10.00
EM01	887.81	3.20	3.20	242542.20	45.00	45.00	101947.40	5.00	5.00	65.11	10.00	10.00
EM02	768.72	2.90	2.90	242550.40	98.10	98.10	101958.60	33.50	33.50	57.73	10.00	10.00
EM03	769.04	3.00	3.00	210097.00	129.00	129.00	88303.00	29.30	29.30	57.73	10.00	10.00
EM04	774.52	25.00	25.00	210066.20	75.60	75.60	88300.30	30.40	30.40	57.73	10.00	10.00
GG01	1544.49	9.70	9.70	420091.70	61.00	61.00	176574.40	5.00	5.00	105.45	10.00	10.00
HC01	774.84	23.80	23.80	210044.30	63.20	63.20	88291.00	10.30	10.30	57.73	10.00	10.00
IG01	770.32	7.50	7.50	210038.70	63.20	63.20	88298.80	28.60	28.60	57.73	10.00	10.00
LF01	770.00	4.50	4.50	210263.40	292.00	292.00	88334.50	70.00	70.00	60.23	20.00	20.00
MG01	1578.19	43.40	43.40	420079.50	48.80	48.80	177017.40	448.00	448.00	105.45	10.00	10.00
OG01	4747.11	8420.00	4747.11	2635171.50	3650000.00	2635171.50	1127609.10	1530000.00	1127609.10	591.11	916.00	591.11
PG01	1547.99	13.20	13.20	420135.70	105.00	105.00	176627.30	57.90	57.90	105.45	10.00	10.00
SD01	1195.98	910.00	910.00	246107.40	10500.00	10500.00	107905.10	15600.00	15600.00	68.57	20.40	20.40
SG01	3792.75	7880.00	3792.75	608388.10	1080000.00	608388.10	124929.10	50600.00	50600.00	100.81	100.00	100.00
SPR01	1584.79	50.00	50.00	437330.70	17300.00	17300.00	199969.40	23400.00	23400.00	175.45	80.00	80.00
SPR02	1584.79	50.00	50.00	420148.70	118.00	118.00	185604.40	9035.00	9035.00	175.45	80.00	80.00
SPR03	1704.79	170.00	170.00	448230.70	28200.00	28200.00	271569.40	95000.00	95000.00	295.45	200.00	200.00
SPR04	1559.79	25.00	25.00	420080.70	50.00	50.00	176604.70	35.30	35.30	175.45	80.00	80.00
SPR05	1559.79	25.00	25.00	420080.70	50.00	50.00	176599.40	30.00	30.00	175.45	80.00	80.00
SPR07	1597.29	62.50	62.50	746030.70	326000.00	326000.00	328569.40	152000.00	152000.00	225.95	130.50	130.50
SPR08	1635.79	101.00	101.00	588030.70	168000.00	168000.00	315569.40	139000.00	139000.00	317.45	222.00	222.00
SPR09	1584.79	50.00	50.00	425470.70	5440.00	5440.00	189669.40	13100.00	13100.00	175.45	80.00	80.00
SPR10	1559.79	25.00	25.00	477030.70	57000.00	57000.00	180019.40	3450.00	3450.00	175.45	80.00	80.00
SPR11	1559.79	25.00	25.00	420130.70	100.00	100.00	176664.70	95.30	95.30	135.45	40.00	40.00
SPR12	1572.29	37.50	37.50	420130.70	100.00	100.00	176897.90	328.50	328.50	135.45	40.00	40.00
SPR13	1559.79	25.00	25.00	420130.70	100.00	100.00	176599.40	30.00	30.00	175.45	80.00	80.00
SPR14	1584.79	50.00	50.00	420130.70	100.00	100.00	177107.40	538.00	538.00	135.45	40.00	40.00
SPR15	1903.79	369.00	369.00	420904.70	874.00	874.00	178669.40	2100.00	2100.00	175.45	80.00	80.00
SPR16	3394.79	1860.00	1860.00	437130.70	17100.00	17100.00	177839.40	1270.00	1270.00	175.45	80.00	80.00
SPR17	1696.79	162.00	162.00	421340.70	1310.00	1310.00	179089.40	2520.00	2520.00	175.45	80.00	80.00
TC01-1	1092.61	8.60	8.60	297426.10	720.00	720.00	124896.10	52.10	52.10	77.49	10.00	10.00
TC01-2	1542.49	7.70	7.70	420199.70	169.00	169.00	176590.00	20.60	20.60	105.45	10.00	10.00
TP01	1871.79	337.00	337.00	427270.70	7240.00	7240.00	178439.40	1870.00	1870.00	175.45	80.00	80.00
TP02	1584.79	50.00	50.00	420130.70	100.00	100.00	178239.40	1670.00	1670.00	175.45	80.00	80.00
TP03	1559.79	25.00	25.00	420080.70	50.00	50.00	176599.40	30.00	30.00	135.45	40.00	40.00

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Table 2-3 (continued)

RME Concentrations of Dissolved Inorganic COCs in Surface Water - 1991 Data

Station	UCL95 Pb	Max Pb	Exp.Pt. Pb	UCL95 Sb	Max Sb	Exp.Pt. Sb	UCL95 Se	Max Se	Exp.Pt. Se	UCL95 Zn	Max Zn	Exp.Pt. Zn
AG01	482.24	55.80	55.80	150.80	20.00	20.00	3.17	0.50	0.50	72765.56	171.00	171.00
AR01	213.89	0.50	0.50	95.52	20.00	20.00	2.04	0.50	0.50	36318.98	45.80	45.80
AR02	213.89	1.00	1.00	81.18	20.00	20.00	1.98	1.10	1.10	36356.21	157.00	157.00
AR03	302.04	0.50	0.50	112.49	20.00	20.00	2.39	0.50	0.50	51590.11	376.00	376.00
AR03A	302.04	0.50	0.50	107.39	20.00	20.00	3.17	0.50	0.50	51398.41	122.00	122.00
AR03E	426.94	0.50	0.50	106.09	20.00	20.00	2.39	0.50	0.50	51423.06	167.00	167.00
AR03W	426.94	0.50	0.50	105.19	20.00	20.00	2.39	0.50	0.50	51340.21	11.20	11.20
AR04	213.89	1.20	1.20	82.85	20.00	20.00	2.04	0.50	0.50	41998.52	202.00	202.00
AR05	255.90	18.60	18.60	95.52	20.00	20.00	7.94	11.60	7.94	51429.11	189.00	189.00
CG01	1143.54	997.00	997.00	112.49	20.00	20.00	7.14	10.00	7.14	72782.11	30500.00	30500.00
CG02	565.09	492.00	492.00	112.49	20.00	20.00	4.64	5.00	4.64	77332.11	34100.00	34100.00
CG03	259.30	37.80	37.80	82.43	20.00	20.00	2.33	2.50	2.33	69548.53	71400.00	69548.53
CG04	865.84	2240.00	865.84	83.60	20.00	20.00	4.21	7.00	4.21	81077.28	71900.00	71900.00
CG05	427.40	433.00	427.40	81.65	20.00	20.00	2.83	2.50	2.50	68489.78	59400.00	59400.00
CG06	214.34	2.50	2.50	82.55	20.00	20.00	2.01	1.20	1.20	47634.53	33400.00	33400.00
EF01	246.70	0.50	0.50	89.92	20.00	20.00	2.39	0.50	0.50	41917.49	5.00	5.00
EF02	246.70	0.50	0.50	90.15	20.00	20.00	2.39	0.50	0.50	41922.46	19.90	19.90
EG01	302.04	0.50	0.50	104.54	20.00	20.00	3.17	0.50	0.50	51337.11	5.00	5.00
EG02	426.94	0.50	0.50	150.80	20.00	20.00	3.17	0.50	0.50	72599.56	5.00	5.00
EM01	246.70	0.50	0.50	92.69	20.00	20.00	2.39	0.50	0.50	41917.49	5.00	5.00
EM02	214.47	2.20	2.20	82.20	20.00	20.00	2.04	0.50	0.50	36302.28	5.00	5.00
EM03	213.72	0.50	0.50	81.93	20.00	20.00	2.04	0.50	0.50	36302.28	5.00	5.00
EM04	213.72	0.50	0.50	82.75	20.00	20.00	1.71	0.50	0.50	36302.28	5.00	5.00
GG01	441.94	15.50	15.50	150.80	20.00	20.00	3.17	0.50	0.50	72628.56	34.00	34.00
HC01	213.72	0.50	0.50	81.90	20.00	20.00	2.04	0.50	0.50	36302.28	5.00	5.00
IG01	214.47	3.50	3.50	81.70	20.00	20.00	2.04	0.50	0.50	36301.63	5.00	5.00
LF01	213.72	0.50	0.50	95.52	20.00	20.00	2.04	0.50	0.50	36303.96	10.00	10.00
MG01	652.44	226.00	226.00	150.80	20.00	20.00	3.17	0.50	0.50	75874.56	3280.00	3280.00
OG01	325.44	47.30	47.30	582.19	1000.00	582.19	8.21	10.00	8.21	430332.11	644000.00	430332.11
PG01	439.34	12.90	12.90	150.80	20.00	20.00	3.17	0.50	0.50	72771.56	177.00	177.00
SD01	548.90	899.00	548.90	95.52	20.00	20.00	2.04	0.50	0.50	61484.82	52400.00	52400.00
SG01	893.20	1420.00	893.20	155.52	20.00	20.00	2.71	2.50	2.50	95512.49	94800.00	94800.00
SPR01	493.34	66.90	66.90	250.80	120.00	120.00	7.67	5.00	5.00	119894.56	47300.00	47300.00
SPR02	452.79	26.35	26.35	190.80	60.00	60.00	12.67	10.00	10.00	113594.56	41000.00	41000.00
SPR03	1436.44	1010.00	1010.00	280.80	150.00	150.00	15.17	12.50	12.50	149794.56	77200.00	77200.00
SPR04	432.44	6.00	6.00	250.80	120.00	120.00	7.67	5.00	5.00	75204.56	2610.00	2610.00
SPR05	432.44	6.00	6.00	250.80	120.00	120.00	12.67	10.00	10.00	90294.56	17700.00	17700.00
SPR07	452.14	25.70	25.70	280.80	150.00	150.00	15.17	12.50	12.50	172594.56	100000.00	100000.00
SPR08	1286.44	860.00	860.00	250.80	120.00	120.00	22.67	20.00	20.00	160694.56	88100.00	88100.00
SPR09	525.94	99.50	99.50	250.80	120.00	120.00	12.67	10.00	10.00	127594.56	55000.00	55000.00
SPR10	436.64	10.20	10.20	250.80	120.00	120.00	7.67	5.00	5.00	73025.56	431.00	431.00
SPR11	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	NA	NA	NA
SPR12	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	72634.56	40.00	40.00
SPR13	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	72665.26	70.70	70.70
SPR14	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	72614.56	20.00	20.00
SPR15	477.04	50.60	50.60	250.80	120.00	120.00	7.67	5.00	5.00	78824.56	6230.00	6230.00
SPR16	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	73944.56	1350.00	1350.00
SPR17	450.54	24.10	24.10	250.80	120.00	120.00	7.67	5.00	5.00	80574.56	7980.00	7980.00
TC01-1	302.04	0.50	0.50	92.89	20.00	20.00	2.39	0.50	0.50	51390.56	64.90	64.90
TC01-2	428.04	1.60	1.60	134.40	3.60	3.60	NA	5.00	NA	72605.16	10.60	10.60
TP01	429.44	3.00	3.00	250.80	120.00	120.00	7.67	5.00	5.00	73099.56	505.00	505.00
TP02	435.24	8.80	8.80	250.80	120.00	120.00	7.67	5.00	5.00	73984.56	1390.00	1390.00
TP03	427.94	1.50	1.50	250.80	120.00	120.00	7.67	5.00	5.00	72634.56	40.00	40.00

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had high metal concentrations (i.e., Oregon Gulch [OG01] and Airport Gulch [AG01]) relative to stream systems originating off-site.

2.4.2.2 Inorganics in Sediment

Aquatic life are exposed to sediments in the environment as well as to surface water. Because health effects criteria are unavailable for sediments, the concentrations of inorganics in sediment pore water are used as the exposure point concentrations. The concentrations in sediment pore water are compared to the EPA AWQC for the protection of aquatic life.

Sediment pore water concentrations at the California Gulch Superfund site were not measured during the 1991 sampling events. Therefore, sediment pore water concentrations must be estimated for this time period. In order to predict toxicity to benthic macroinvertebrates or other aquatic life, sediment pore water concentrations were estimated from the sediment data by using the MINTEQA2 model (Medine, 1994). Pore water concentrations were estimated for several locations on the Arkansas River and for several locations on California Gulch. Pore water concentrations were estimated for different dates, as pore water metals concentrations are expected to vary seasonally (Medine, 1994).

Copper and lead are predicted to be more highly adsorbed to sediment solids than cadmium and zinc. Therefore, cadmium and zinc are expected to occur in the dissolved phase.

The estimated pore water concentrations based on measured concentrations in sediment solids are presented in Table 2-4 and 2-5, respectively. Table 2-4 presents average pore water concentrations by date and station. Table 2-5 presents the estimated minimum and maximum pore water concentrations by date and station.

The estimated pore water concentration values are not comparable to the actual pore water concentration measurements collected in May 1994 by Chadwick & Associates for Resurrection, because sediment metal concentrations could have altered over time.

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Table 2-4

Estimated Average Concentration of Metals in Sediment Pore Water (Dissolved Metals in mg/ℓ)

Station	Date	Cadmium (Average)	Copper (Average)	Lead (Average)	Zinc (Average)
ARKANSAS RIVER					
AR03	04/30/91	0.080	0.035	0.050	2.6
AR3A	09/16/91	0.043	0.002	0.010	2.400
AR3A	07/24/91	0.105	0.005	0.010	2.130
AR04	09/16/91	0.236	0.004	0.122	17.200
AR04	04/29/91	0.772	0.003	0.023	18.600
AR05	09/16/91	0.105	0.001	0.007	3.700
AR05	04/29/91	0.181	0.002	0.008	4.380
CALIFORNIA GULCH					
CG03	03/24/92	0.020	0.009	0.008	0.685
CG03	09/17/91	0.298	0.020	0.042	22.600
CG03	07/24/91	0.412	0.013	0.020	11.850
CG04	03/24/92	0.207	0.004	0.030	9.581
CG04	07/24/91	0.379	0.015	46.100	1845.000
CG05	03/24/92	0.012	0.014	0.029	6.090
CG05	09/17/91	0.496	0.022	0.131	27.700
CG05	07/24/91	0.993	22.440	0.088	71.200
CG06	03/24/92	0.133	0.059	0.115	4.910
CG06	09/17/91	0.048	0.031	0.029	2.900
CG06	07/24/91	0.493	0.390	0.117	32.100

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Table 2-5

Estimated RME Concentration of Metals in Sediment Pore Water (Dissolved Metals in mg/ℓ)

Station	Date	Cadmium		Copper		Lead		Zinc	
		Min	Max	Min	Max	Min	Max	Min	Max
Arkansas River									
AR03	04/30/91	0.013	0.326	0.006	0.147	0.007	0.22	0.442	11.00
AR3A	09/16/91	0.017	0.079	0.001	0.003	0.003	0.020	0.965	4.430
AR3A	07/24/91	0.035	0.282	0.002	0.013	0.002	0.026	0.643	5.26
AR04	09/16/91	0.092	0.468	0.002	0.009	0.059	0.265	6.430	35.800
AR04	04/29/91	0.285	1.366	0.001	0.006	0.007	0.048	7.260	35.100
AR05	09/16/91	0.084	0.175	0.001	0.001	0.006	0.012	3.050	6.150
AR05	04/29/91	0.064	0.368	0.001	0.003	0.002	0.018	1.580	8.910
California Gulch									
CG03	03/24/92	0.007	0.043	0.003	0.018	0.002	0.019	0.228	1.400
CG03	09/17/91	0.103	0.458	0.007	0.039	0.010	0.093	5.800	31.800
CG03	07/24/91	0.171	0.741	0.006	0.023	0.006	0.041	4.880	20.460
CG04	03/24/92	0.070	0.435	0.002	0.009	0.008	0.069	3.289	20.240
CG04	07/24/91	0.162	0.755	0.006	0.028	11.200	0.074	6.600	30.800
CG05	03/24/92	0.033	0.273	0.002	0.017	0.011	0.075	1.790	14.400
CG05	09/17/91	0.219	0.919	0.006	0.024	0.006	0.249	11.600	48.700
CG05	07/24/91	0.327	1.898	0.010	0.036	0.042	0.167	31.600	48.700
CG06	03/24/92	0.016	0.747	0.007	0.349	0.012	0.68	0.620	28.500
CG06	09/17/91	0.009	0.253	0.010	0.283	0.005	0.135	0.485	13.800
CG06	07/24/91	0.075	2.005	0.005	0.130	0.014	0.491	4.660	136.000

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2.4.3 Data Collected After Initiation of Yak Tunnel Water Treatment

2.4.3.1 Dissolved Inorganics in Surface Water

Different stations were sampled during the 1992 to 1993 sampling events as compared to the 1991 sampling events. Sampling during 1992 and 1993 focused on California Gulch, the Arkansas River, and the various springs (Appendix B-1). Data for tributaries to the Arkansas River that originate off-site are lacking. Concentrations of inorganics in California Gulch and the Arkansas River below the confluence with California Gulch can be compared to data for AR02, the station upgradient of the confluence. The sampling events in 1992 to 1993 that were included in this risk assessment were:

- Summer (July to August, 1992).
- Fall (September to October, 1992).
- Fall (October, 1993).
- Winter (November to December, 1992 and January, 1993).

The period of high flow during the spring ice-off period was not included for the 1992 data because the Yak Tunnel Treatment Facility had just become operative and the aquatic ecosystem was expected to be in a state of flux. The ice-off period is generally the period when metal concentrations are highest; this skews this data set and makes the average and RME values artificially low compared to the concentrations observed for the 1991 data set.

Data for the summer and fall sampling events for the pre- and post-treatment periods were compared for several of the inorganics that are toxic to aquatic life and are present in the aquatic ecosystem at California Gulch. These inorganics were arsenic, cadmium, copper, lead, and zinc. The results of this comparison are presented in Figures 2-1 to 2-5. Concentrations of inorganics in surface water have been greatly reduced by the Yak Treatment Plant, but as indicated below, inorganics in the aquatic ecosystem are still elevated with respect to upgradient areas.

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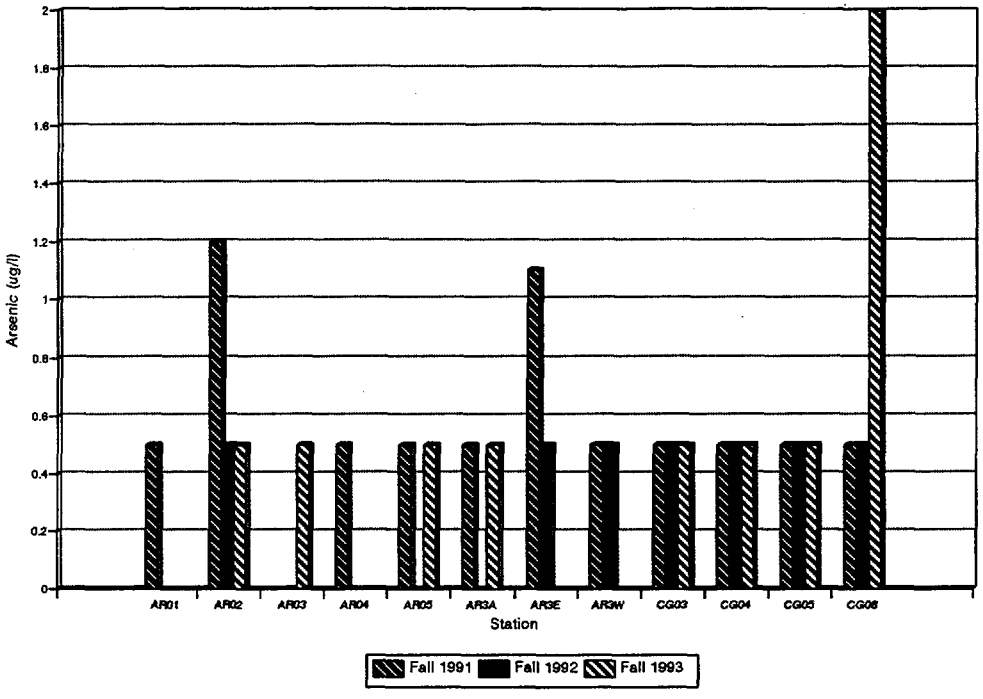
Figure 2-1

Arsenic Concentration by Year



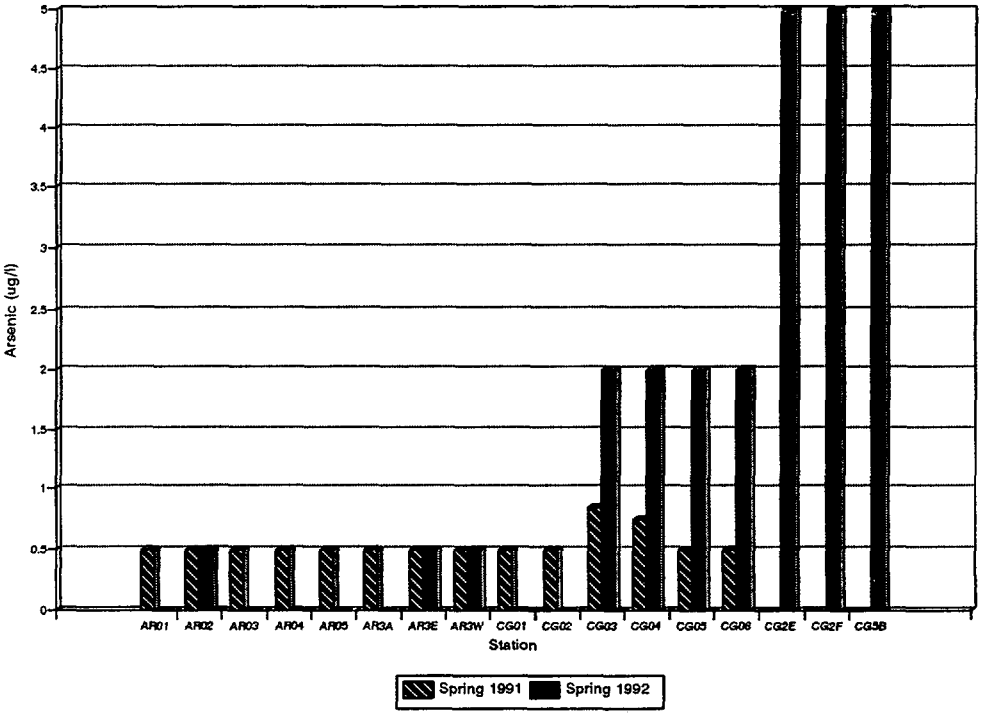


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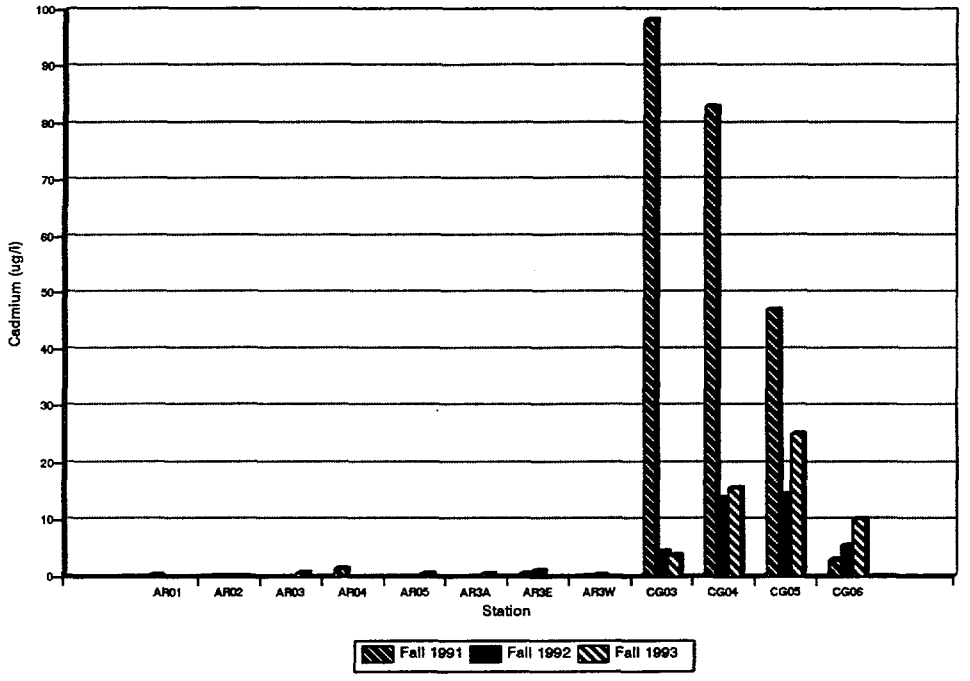
Figure 2-2

Cadmium Concentration by Year



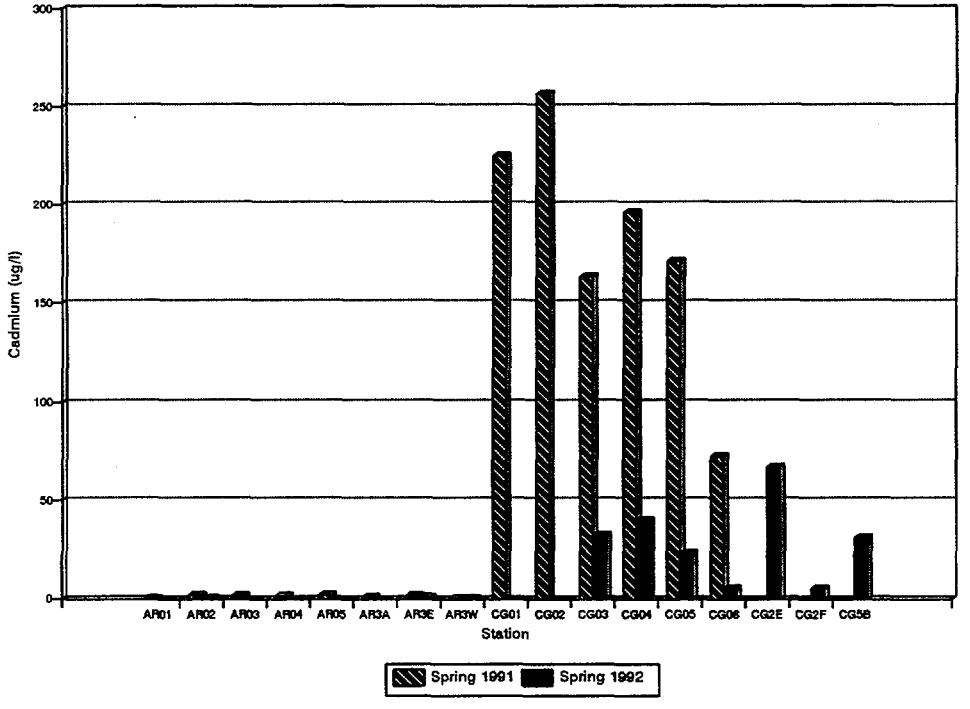


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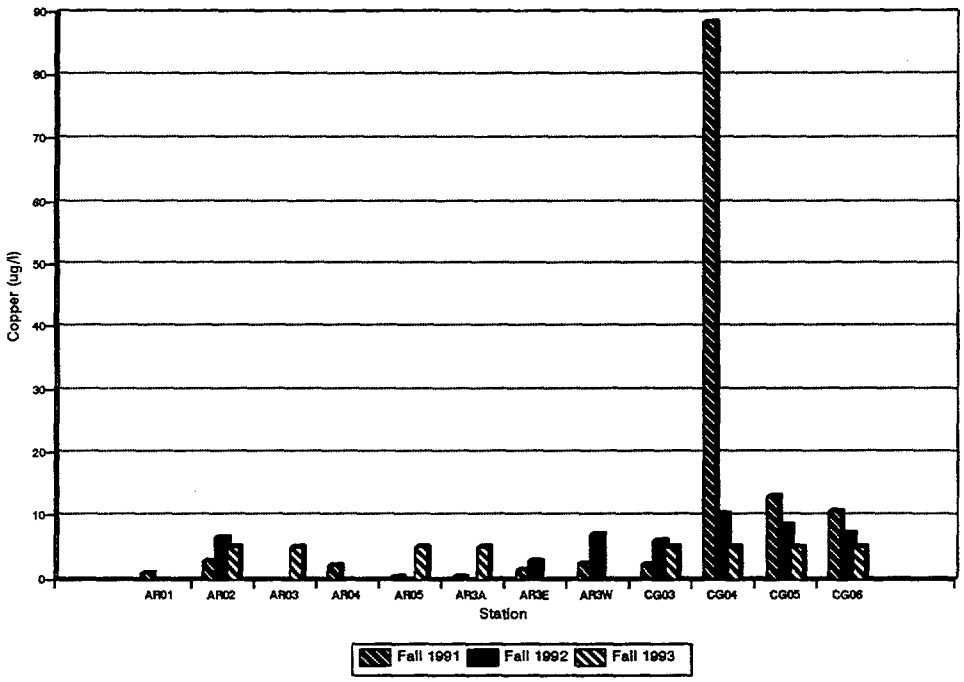
Figure 2-3

Copper Concentration by Year



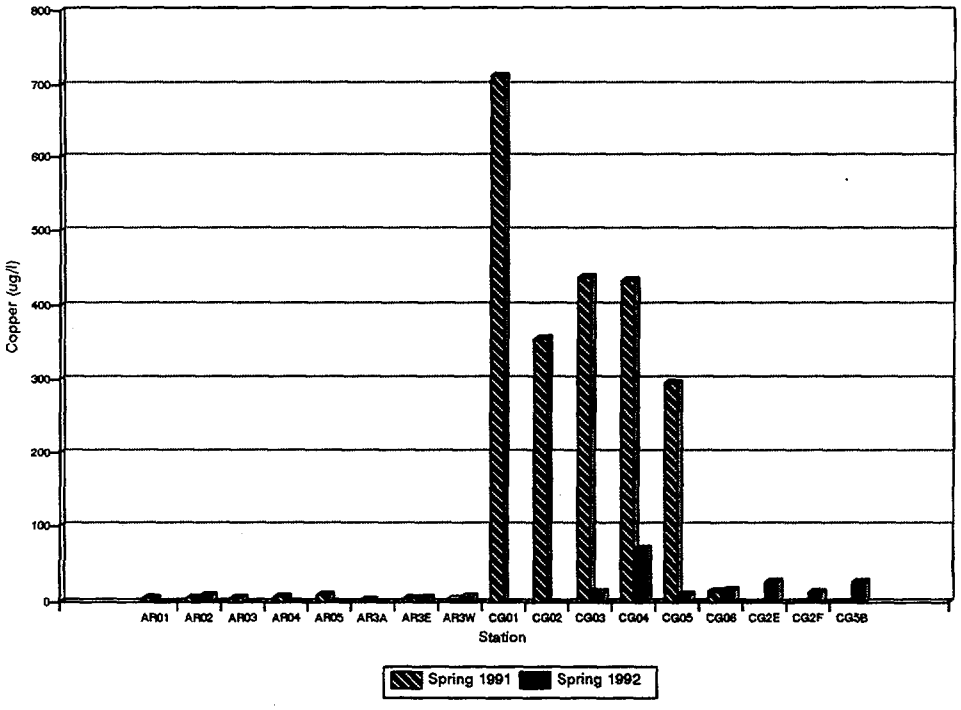


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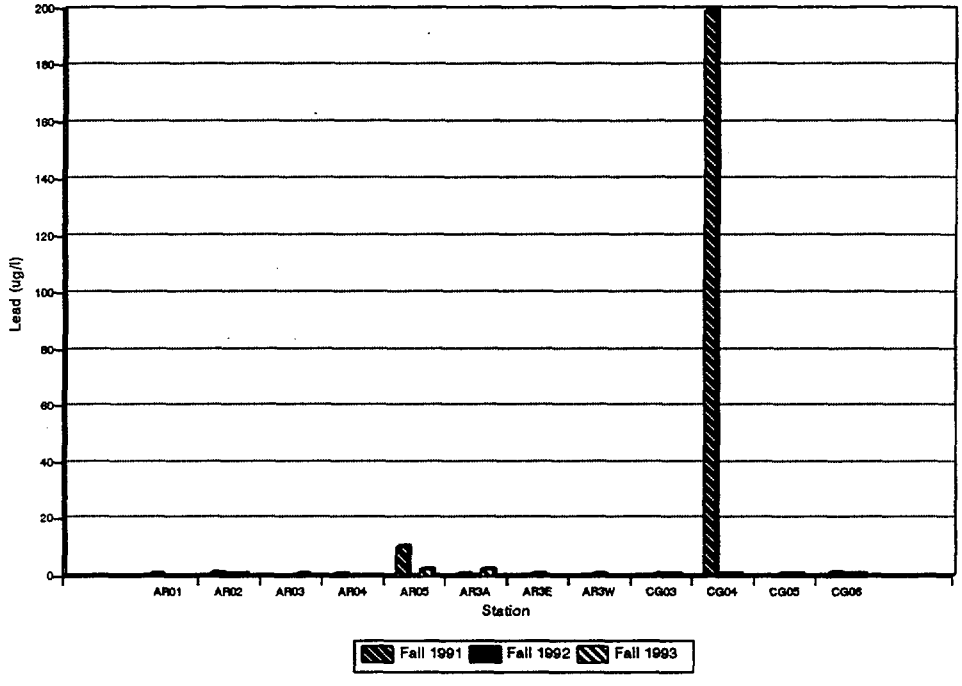
Figure 2-4

Lead Concentration by Year



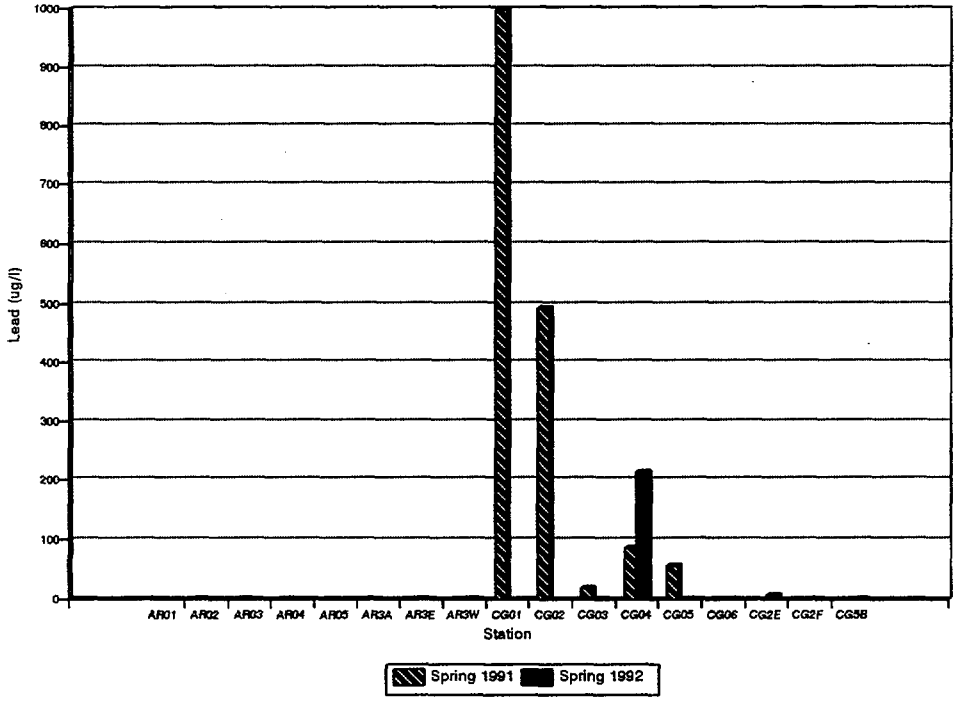


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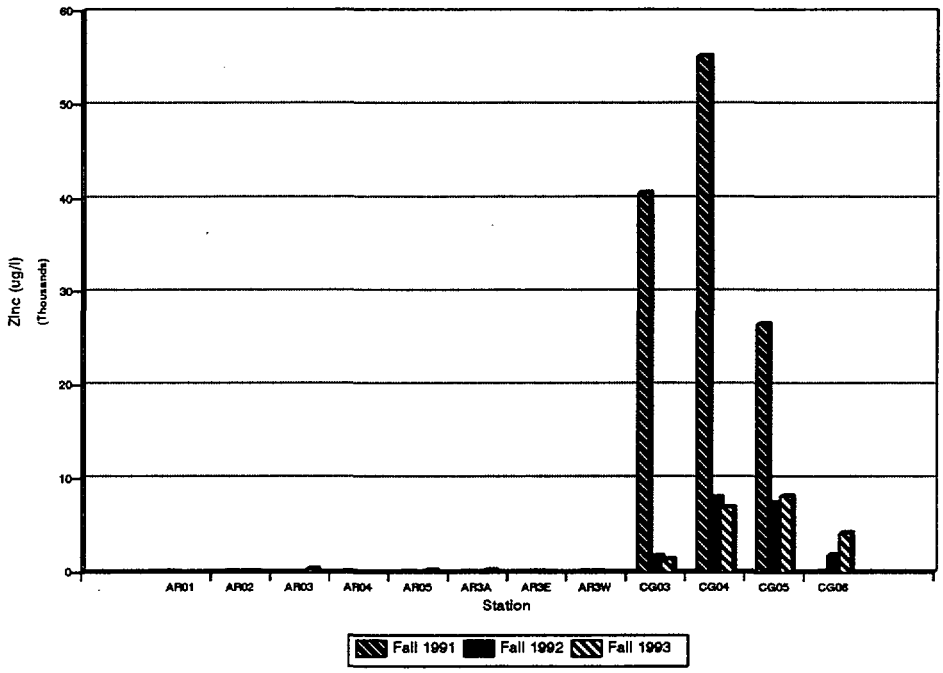
Figure 2-5

Zinc Concentration by Year



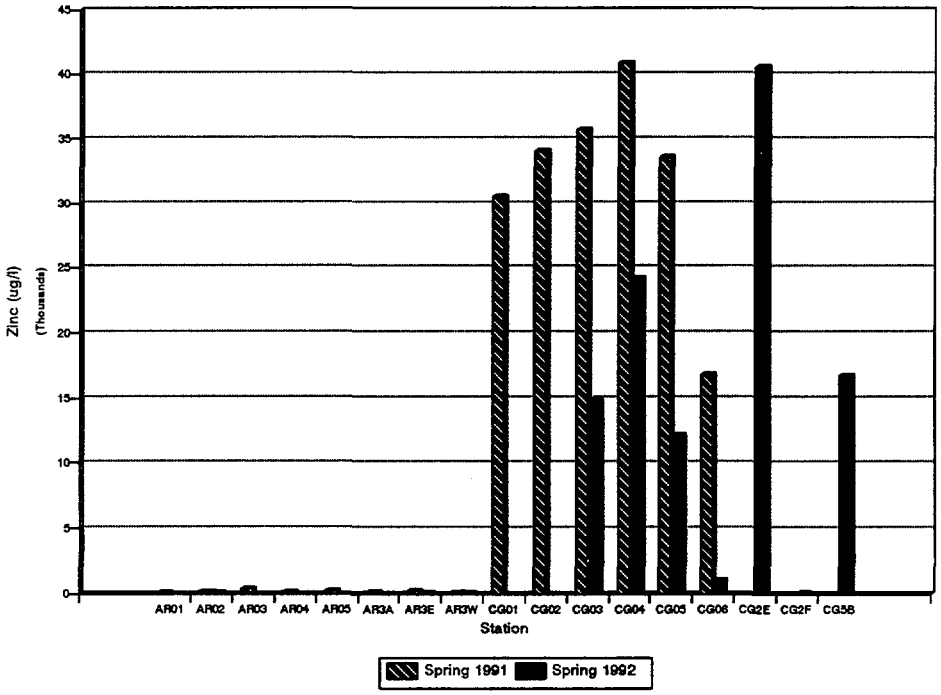


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Table 2-6 presents the average concentration of dissolved inorganics in surface water. The RME values are presented in Table 2-7. As with data prior to initiation of water treatment, the average concentration represents typical conditions at each station, whereas the RME represents periods of high input to the stream system. Average concentrations of aluminum and barium were similar in the Arkansas River and California Gulch (Table 2-7). Average concentrations of cadmium and nickel are higher at all California Gulch stations and at stations downgradient of the confluence of California Gulch with the Arkansas River. Copper and antimony concentrations are higher in California Gulch than in the Arkansas River, although concentrations in the Arkansas River downgradient of California Gulch do not appear elevated relative to AR02. Lead concentrations were high at CG04 and AR03A (the station where California Gulch empties into the Arkansas River), but at other stations lead concentrations were similar to AR02. Selenium concentrations were twice as high in California Gulch as in the Arkansas River. Zinc concentrations were higher in California Gulch and at AR03, but by AR05 the average concentrations of zinc were similar to AR02. The springs had high concentrations of metals relative to AR02.

The RME concentrations are provided in Table 2-7, which reports the UCL95, maximum, and exposure point concentrations at each station. To evaluate whether or not the RME was sufficiently protective of aquatic life, the RME values were compared to data collected by the CDOW (Section 2.4.3).

2.4.3.2 Inorganics in Sediment

Hyporheic sampling was conducted by Resurrection (1994) in order to obtain measured concentrations of metals in sediment pore water. Table 2-8 presents the measured concentrations of dissolved metals in sediment pore water and calculated water hardness. One sample was collected at each location using a shallow groundwater pump, which extracted interstitial pore water from the *in situ* sediments. The August samples were collected at only two locations and were intended to provide insight as to within station variability. Cadmium, lead, and zinc concentrations in sediment pore water were higher in California Gulch and in the Arkansas River downgradient of California Gulch than at reference locations.

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Table 2-6
Average Concentrations of Dissolved Inorganics in Surface Water
July 1992 through August 1993 Data

Station	Antimony	Barium	Cadmium	Copper	Iron	Lead	Manganese	Nickel	Silver	Zinc
AR02	10.13	35.20	0.12	4.60	44.60	0.50	24.15	0.83	0.05	9.08
AR03E	13.30	53.90	0.92	4.35	22.95	0.50	353.25	1.65	0.06	12.88
AR03W	10.20	48.70	0.20	7.18	33.28	0.50	30.78	0.98	0.06	5.00
AR3A	8.00	28.44	1.16	7.07	90.61	1.00	521.86	1.14	0.06	363.67
CG03	13.30	41.80	5.16	6.84	31.94	0.50	444.40	5.90	0.06	1664.60
CG04	10.78	37.72	12.64	12.20	1371.20	0.50	8748.00	15.76	0.06	8180.00
CG05	8.96	34.74	17.32	9.60	697.50	0.50	9056.00	14.16	0.06	9098.00
CG06	8.34	18.08	21.13	61.31	2684.11	32.89	7182.50	9.62	0.06	6490.58

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Table 2-7

RME Concentrations of Dissolved Inorganics in Surface Water

July 1992 through August 1993 Data

Station	Aluminum		Antimony		Arsenic		Barium		Cadmium		Copper		Iron	
	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max
AR02	430.51	62.40	12.19	13.65	0.55	0.50	39.84	67.20	7.47	0.24	31.29	7.60	1897.07	132.00
AR03E	691.83	25.00	16.71	23.40	0.58	0.50	61.60	65.30	13.11	1.50	48.62	6.40	3094.91	61.80
AR03W	691.83	25.00	13.61	13.20	0.58	0.50	56.40	53.90	12.39	0.35	51.45	12.80	3105.24	93.10
AR3A	451.01	58.80	10.15	12.45	0.55	0.50	33.31	54.00	8.87	5.10	35.07	15.35	2033.49	630.00
CG03	621.43	25.00	16.35	21.00	0.58	0.50	48.69	45.70	16.06	9.00	46.44	12.60	2779.59	77.20
CG04	621.43	25.00	13.83	13.00	0.58	0.50	44.61	41.00	23.54	20.40	51.80	14.00	4118.85	2820.00
CG05	621.43	25.00	12.01	16.40	0.58	0.50	41.63	43.10	28.22	25.20	49.20	23.40	3445.15	1710.00
CG06	1086.66	6800.00	10.30	15.00	0.68	1.20	22.52	30.10	28.17	109.00	86.87	416.00	4457.71	31200.00

Station	Lead		Manganese		Mercury		Nickel		Selenium		Silver		Zinc	
	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max	UCL95	Max
AR02	17.50	0.50	2049.83	49.40	0.12	0.20	3.11	1.95	1.00	1.00	0.06	0.10	2161.09	40.30
AR03E	28.69	0.50	3712.46	483.00	0.11	0.10	5.44	2.00	0.72	0.50	0.08	0.10	3581.57	24.70
AR03W	28.69	0.50	3389.99	43.50	0.11	0.10	4.76	1.70	0.72	0.50	0.08	0.10	3573.70	5.00
AR3A	18.83	4.10	2646.41	1780.00	0.11	0.10	3.53	1.80	1.14	1.00	0.07	0.12	2620.71	1430.00
CG03	25.71	0.50	3448.97	633.00	0.13	0.20	9.29	12.60	0.80	1.00	0.07	0.10	4856.54	2850.00
CG04	25.71	0.50	11752.57	12400.00	0.11	0.10	19.15	21.30	0.92	1.10	0.07	0.10	11371.94	11400.00
CG05	25.71	0.50	12060.57	14600.00	0.11	0.10	17.55	18.70	1.32	2.60	0.07	0.10	12289.94	14300.00
CG06	49.16	275.00	9121.94	36300.00	0.11	0.10	11.81	40.90	1.00	1.00	0.07	0.10	8550.97	35900.00

Note: All metal concentrations in ug/l

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Table 2-8

Porewater Data from Ressurection

Station	Date	Dissolved Metal Concentrations in Porewater (ug/l)								Hardness
		Al	As	Cd	Cu	Fe	Mn	Pb	Zn	
AR01	05/25/94	NA	NA	0.30	5.00	NA	NA	1.00	70.00	64.68
AR02	05/25/94	NA	NA	0.40	5.00	NA	NA	0.50	70.00	173.06
AR03	05/26/94	NA	NA	6.60	5.00	NA	NA	2.00	1,930.00	239.93
AR04	05/25/94	NA	NA	1.60	5.00	NA	NA	0.50	330.00	165.02
AR05	05/26/94	NA	NA	1.20	5.00	NA	NA	15.00	310.00	67.82
CG04	05/25/94	60.00	NA	31.00	NA	140.00	7,310.00	10.00	11,100.00	NA
CG05	Data Missing	NA	NA	NA	NA	NA	NA	NA	NA	NA
CG06	05/25/94	25.00	0.50	34.50	5.00	270.00	7,810.00	NA	12,100.00	NA
EF02	05/26/94	NA	NA	0.60	5.00	NA	NA	1.00	100.00	173.06
LF01	05/26/94	NA	NA	0.10	5.00	NA	NA	0.50	70.00	163.70

Note: all "<" values divided by 2 to obtain 1/2 the detection limit
Total Ca and Mg used to obtain hardness; 2.495[Ca] + 4.114[Mg] = hardness

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Sediment pore water concentrations predicted by the model (Medine, 1994) are not comparable to those measured by Resurrection (1994) because the model is based on 1991 data, which predate the Yak Tunnel Treatment Plant. The measured data represent current, or post-treatment, conditions.

2.4.4 CDOW Data Used to Validate the Risk Assessment Approach

2.4.4.1 Validation of the RME and Episodic Sampling Events

CDOW (Woodling, 1994) has collected extensive surface water data from the Arkansas River both up and downgradient of California Gulch. These data are summarized in Table 2-9. The data are separated into 1991 (pre Yak Tunnel treatment of surface water in California Gulch) and 1992 and later data sets (post Yak Tunnel treatment of surface water in California Gulch). These data were collected on a more frequent basis (i.e., often daily) than the monthly data collected under the current RI activities by ASARCO.

The CDOW data clearly demonstrate that while water quality in upper California Gulch has improved as a result of water treatment by the Yak Tunnel treatment facility, certain metals remain elevated in surface water in the Arkansas River downgradient of California Gulch. These elevated metals are cadmium, copper, iron, manganese, lead, and zinc (Table 2-9).

Figures 2-6 and 2-7 present the dissolved cadmium and zinc data collected in 1991 over time. The gaps in the lines represent data gaps. Cadmium and zinc were much higher below the confluence of California Gulch with the Arkansas. Figures 2-8 and 2-9 present the dissolved cadmium and zinc data collected from April 1992 to September 1993. Cadmium and zinc concentrations decrease in May as a result of the water treatment by the Yak Tunnel treatment facility. However, from April through June of 1993 excessive metal concentrations were observed downgradient of the confluence of California Gulch with the Arkansas River. This may be the result of high runoff during this period. In addition, cadmium and zinc concentrations in the Arkansas remain elevated downstream from the confluence relative to upgradient locations.

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Table 2-9

Summary of CDOW Surface Water Data

River	Location	Date Range		pH	Temp C	Total Alkalinity	Total Hardness	Dissolved Oxygen mg/l
Arkansas	Above Cal Gulch	05/06/91	Max	8.16	18.00	78.00	100.00	8.00
		10/15/91	Min	7.46	4.00	30.00	40.00	8.00
			Mean	7.87	10.75	44.85	54.46	8.00
			Count	25	26	26	26	1
	Below Cal Gulch	05/06/91	Max	7.97	18.00	73.00	150.00	8.00
		10/15/91	Min	7.03	4.00	30.00	46.00	8.00
			Mean	7.54	11.45	46.65	75.70	8.00
			Count	24	26	26	27	1
	Above Cal Gulch	06/01/92	Max	8.20	18.50	74.00	162.00	9.00
		11/23/93	Min	7.31	3.00	17.00	38.00	9.00
			Mean	7.73	10.61	43.81	67.58	9.00
			Count	78	70	78	78	1
	Below Cal Gulch	06/01/92	Max	8.27	18.50	78.00	414.00	11.00
		12/03/93	Min	6.65	0.50	8.00	18.00	7.00
			Mean	7.60	9.60	46.71	109.58	9.00
			Count	93	87	93	95	2

River	Location	Date Range		Cadmium Dissolved	Cadmium Total	Copper Dissolved	Copper Total	Iron Dissolved	Iron Total	Manganese Dissolved	Manganese Total	Lead Dissolved	Lead Total	Zinc Dissolved	Zinc Total
Arkansas	Above Cal Gulch	05/06/91	Max	1.31	2.11	4.90	6.10	283.00	877.00	88.00	146.00	27.70	2.70	187.00	209.00
		10/15/91	Min	0.10	0.11	1.20	2.40	111.00	132.00	20.00	37.00	27.70	2.70	50.00	62.00
			Mean	0.48	0.61	2.74	4.28	199.25	410.00	47.36	70.36	27.70	2.70	100.27	124.00
			Count	24	27	9	4	8	10	11	11	1	1	11	11
	Below Cal Gulch	05/06/91	Max	4.83	9.99	8.10	34.70	329.00	869.00	871.00	892.00	0.00	29.30	1441.00	1612.00
		10/15/91	Min	0.84	1.29	1.90	1.80	100.00	320.00	182.00	251.00	0.00	0.60	270.00	448.00
			Mean	2.48	3.96	3.93	12.96	180.67	593.40	482.30	506.40	0.00	13.85	705.10	911.60
			Count	23	27	10	10	6	10	10	10	0	6	10	10
	Above Cal Gulch	06/01/92	Max	1.54	3.17	16.70	19.40	640.00	4342.00	373.00	588.00	0.00	34.70	558.00	624.00
		11/23/93	Min	0.10	0.10	1.00	1.00	101.00	107.00	10.00	10.00	0.00	25.70	16.00	18.00
			Mean	0.31	0.50	2.51	3.03	200.22	381.45	26.85	57.61	0.00	30.20	55.11	78.85
			Count	53	60	59	64	32	73	66	77	0	2	75	78
	Below Cal Gulch	06/01/92	Max	30.00	36.00	68.00	156.00	7240.00	14110.00	7770.00	7220.00	594.00	347.00	7800.00	7450.00
		12/03/93	Min	0.14	0.21	1.00	1.10	100.00	127.00	13.00	28.00	12.60	10.30	47.00	55.00
			Mean	2.37	3.92	6.47	16.41	647.62	1217.46	508.16	623.37	85.03	74.12	642.70	885.37
			Count	89	93	78	87	47	91	89	93	10	33	89	93

17 Table 2-9 Summary of CDOW Surface Water Data (Woodling, 1994)

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FIGURE 2-6

Cadmium Concentrations in the Arkansas River

Above and Below California Gulch - 1991 CDOW Data

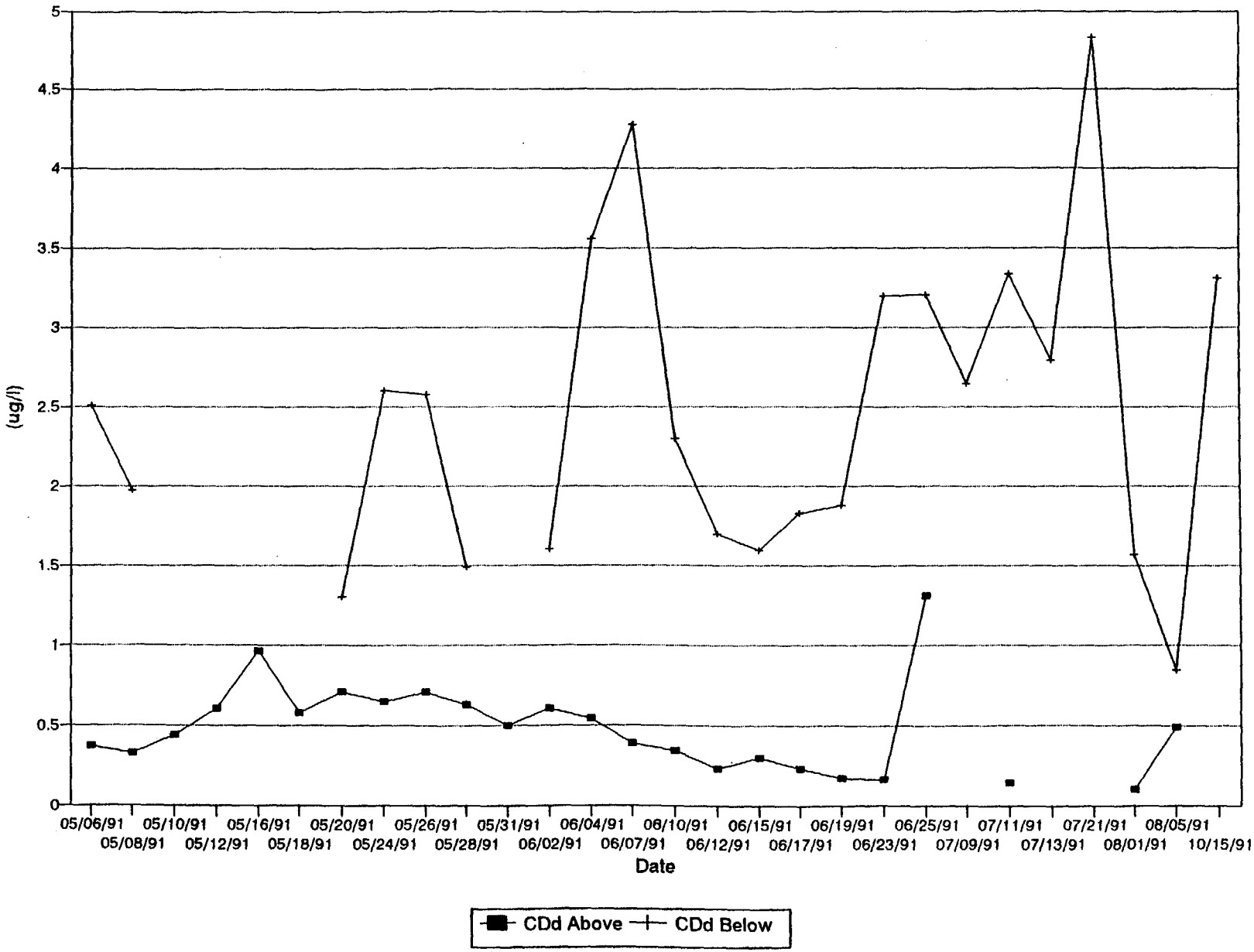

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FIGURE 2-7

Zinc Concentrations in the Arkansas River

Above and Below California Gulch - 1991 CDOW Data

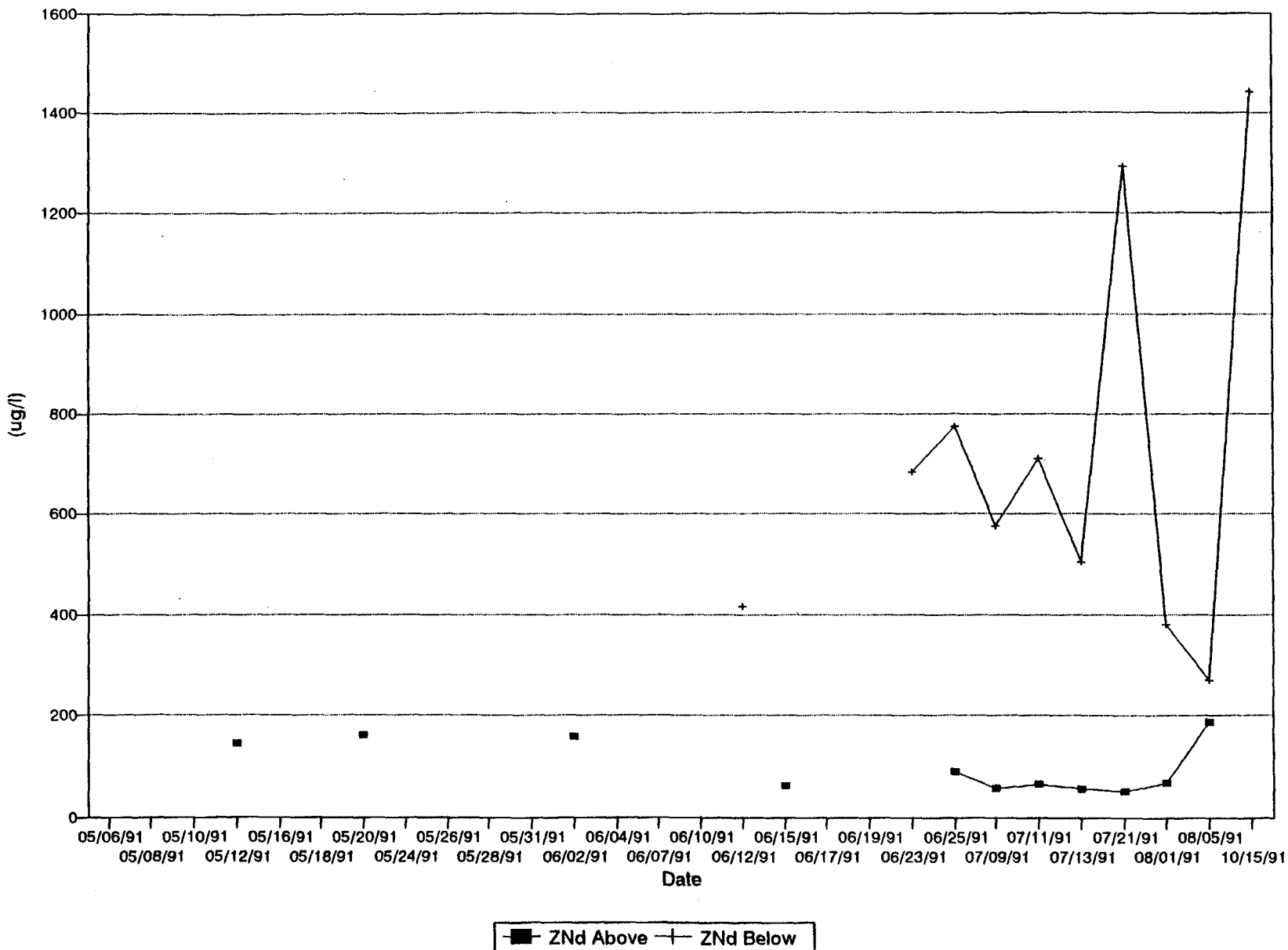

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FIGURE 2-8

Cadmium Concentrations in the Arkansas River

Above and Below California Gulch - 1992 - 1993 CDOW Data

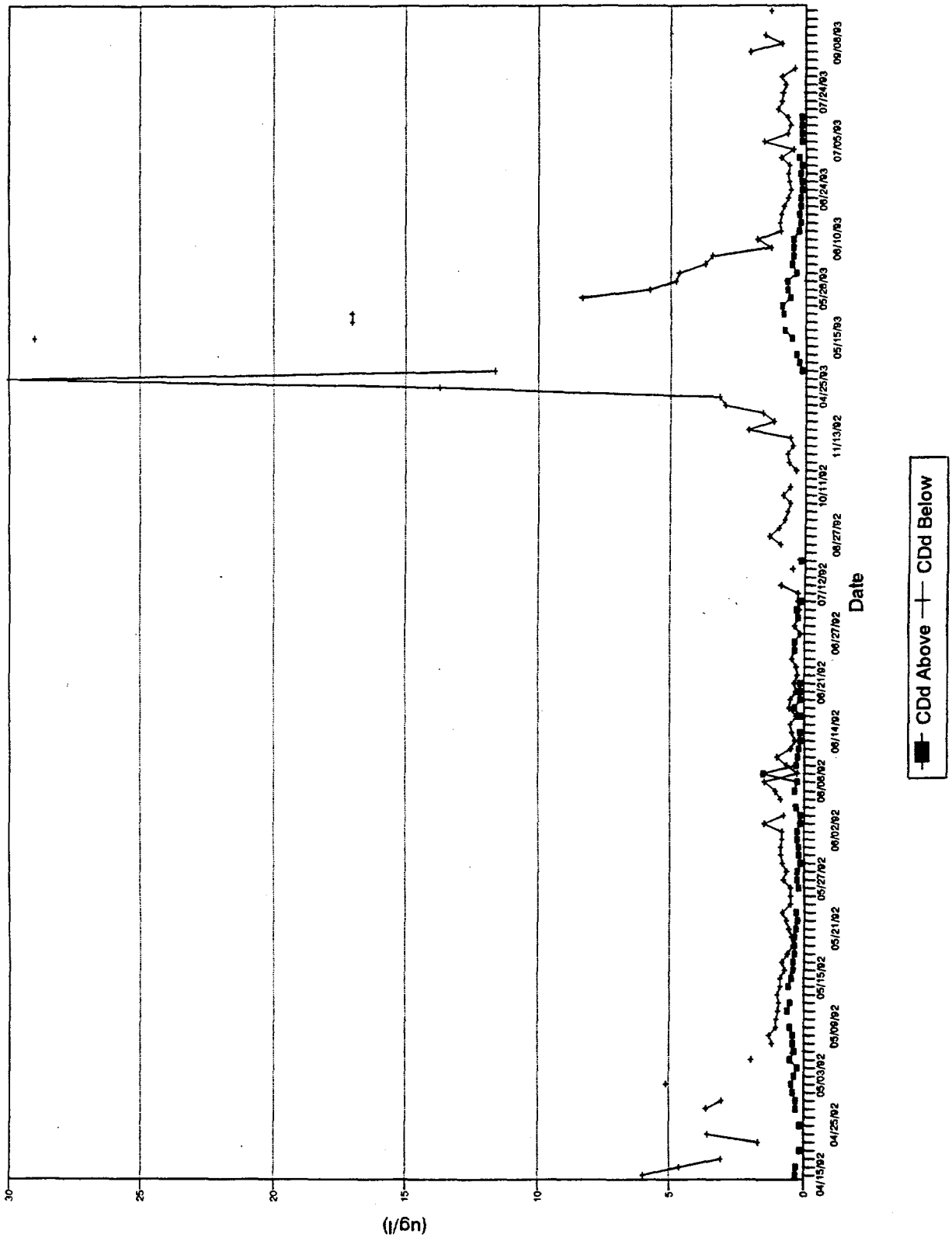

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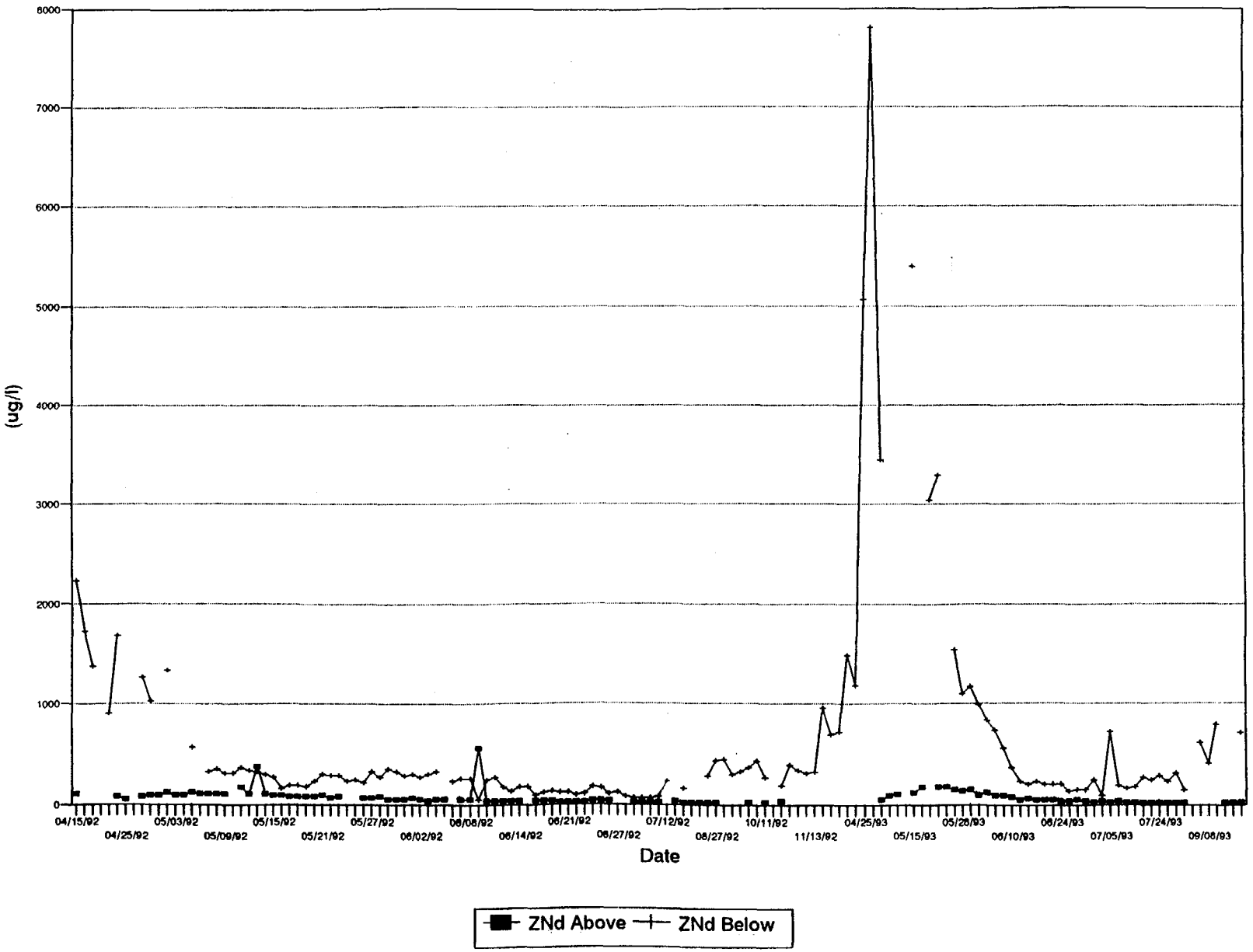
FIGURE 2-9

Zinc Concentrations in the Arkansas River

Above and Below California Gulch - 1992 - 1993 CDOW Data



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These data indicate that excess metals are continuing to enter the Arkansas River from California Gulch. The data were also used to verify whether or not the RME concentrations based on ASARCO's monthly surface water data are sufficiently conservative to protect most aquatic life. The RME was compared to the mean and maximum surface water dissolved metal concentrations observed by CDOW for data collected in 1991, and in 1992 through 1993 (Table 2-10).

The results indicate that the RME for AR03 for 1991 was lower than average values for cadmium, manganese, and zinc from the CDOW data, and lower than maximum values for cadmium, copper, iron, manganese, and zinc. This indicates that the RME fails to conservatively represent concentrations of metals at the station below the confluence of California Gulch with the Arkansas River during the 1991 sampling period. Table 2-10 presents RME concentrations for all of the stations directly below the confluence, for comparative purposes.

The RME for 1992 was lower than observed average concentrations based on CDOW data for cadmium, copper, iron, lead, and zinc (Table 2-10). The RME was lower than maximum concentrations for all metals. This demonstrates that:

- Monthly sampling failed to adequately monitor changes in water quality that could adversely affect the aquatic ecosystem.
- Episodic storm events or periods of high runoff may greatly elevate the metal concentrations in the Arkansas River.
- The risk assessment may underestimate the negative effects that California Gulch is having on the aquatic ecosystem (i.e., is not adequately conservative).
- Metal concentrations entering the Arkansas River are higher than expected, despite the Yak Tunnel treatment facility.

It is acceptable for the RME to be below maximum values because the RME is intended to represent high exposures that could occur on a regular basis. However, aquatic communities can be damaged by high concentrations entering surface water for relatively short durations.

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Table 2-10
Comparison of RME Values Based on ASARCO Data from the Arkansas River Downgradient of California Gulch with Average and Maximum Values from CDOW Data (Woodling, 1994)

1991 Data

Analyte	AR03	AR03E	AR03A	CDOW Mean	CDOW Max
Cadmium	1.6	1.2	1.0	2.5	4.8
Copper	5.5	5.2	2.4	3.9	8.1
Iron	310.0	78.7	45.1	180.7	329.0
Manganese	202.0	324.0	165.0	482.3	871.0
Lead	0.5	0.5	0.5	--	--
Zinc	376.0	167.0	122.0	705.1	1441.0

1992 - 1993 Data

Analyte	AR03	AR03E	AR03A	CDOW Mean	CDOW Max
Cadmium	0.7	1.5	1.3	2.4	30.0
Copper	5.0	6.4	11.8	6.5	68.0
Iron	40.0	61.8	68.6	647.6	7240.0
Manganese	600.0	483.0	340.0	508.2	7770.0
Lead	0.5	0.5	2.0	85.0	594.0
Zinc	230.0	24.7	315.0	642.7	7800.0

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The acute EPA AWQC are based on exposures of only 1 hour, whereas the chronic criteria are based on exposure durations of only 4 days. The CDOW data (Woodling, 1994) indicate that metal concentrations are likely to remain elevated for longer than 4 days (Figures 2-8 and 2-9).

2.4.5 General Water Quality Parameters

Data for general water quality were collected in addition to the chemical data collected. The calcium, magnesium, and pH data are reported in Appendix B-1. Hardness was calculated from the calcium and magnesium data according to the following equation:

$$[\,(\textit{Calcium}\ (mg/\ell)\ x\ 2.497) + (\textit{Magnesium}\ (mg/\ell)\ x\ 4.116)\,]$$

The water quality data are summarized in Table 2-8. The pH concentrations range from a low of 2.7 from Oregon Gulch in 1991, to a high of 8.0 in several of the springs that were sampled. In 1992, the pH ranged from 3.0 at a tailings pile seep to 8.0 at AR05. In the stream systems, lowest pH values occurred in water samples from California Gulch, while the Arkansas River had higher pH values.

Water hardness was also compared by station for pre- and post-water treatment data (Table 2-11). Water from California Gulch had higher hardness values than water from the Arkansas River. Samples collected from springs varied widely in hardness.

2.4.6 Concentration of Metals in Biota

Not all the inorganic COCs are predicted to bioaccumulate or biomagnify. Inorganic analytes that are also nutrients can be metabolized and regulated, as long as ambient concentrations do not exceed an animal's ability to regulate intake. Cadmium, copper, mercury, lead, and zinc may accumulate in tissues as a result of excessive concentrations in water, or because they have bioaccumulative properties. Cadmium and mercury have long depuration times; therefore, even low water concentrations may result in high tissue levels.

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Table 2-11

Average Water Quality Parameters

	1991 Data					1992 - 1993 Data				
Station	pH	Hardness	+/- Std. Dev	Calcium	Magnesium	pH	Hardness	+/- Std. dev	Calcium	Magnesium
AG01	6.00	9.11	0.00	2.00	1.00	NA	NA	NA	NA	NA
AR01	7.53	76.04	31.45	18.50	7.25	NA	NA	NA	NA	NA
AR02	7.70	81.62	33.84	19.50	8.00	7.63	92.93	25.31	21.83	9.33
AR03	7.10	82.06	53.92	20.50	7.50	7.90	111.08	0.00	28.00	10.00
AR03A	7.54	92.42	2.91	23.00	8.50	7.55	103.59	0.00	27.75	11.50
AR03E	7.54	103.21	0.52	26.50	9.00	7.70	103.38	25.01	25.13	9.88
AR03W	7.55	87.42	19.33	21.00	8.50	7.75	81.00	24.56	19.25	8.00
AR04	7.19	149.33	169.14	49.50	6.25	NA	NA	NA	NA	NA
AR05	7.20	81.43	19.12	20.25	7.50	8.00	90.36	0.00	23.00	8.00
CG01	4.05	156.32	55.49	42.00	12.50	NA	NA	NA	NA	NA
CG02	4.80	686.64	314.27	183.50	55.50	NA	NA	NA	NA	NA
CG2A	NA	NA	NA	NA	NA	6.70	659.59	0.00	218.00	28.00
CG03	7.58	608.38	243.41	174.00	42.25	7.05	445.22	155.80	127.75	30.67
CG04	4.75	738.97	151.07	163.25	80.50	6.64	531.08	108.87	145.35	40.85
CG05	5.68	586.30	33.27	134.25	61.00	6.77	510.10	120.31	131.29	44.29
CG06	6.79	351.70	137.58	89.75	31.00	7.43	321.68	102.29	84.55	26.86
EF01	7.60	81.50	31.18	20.00	7.67	NA	NA	NA	NA	NA
EF02	7.70	130.38	79.41	31.33	12.67	NA	NA	NA	NA	NA
EG01	7.65	91.10	25.77	20.00	10.00	NA	NA	NA	NA	NA
EG02	7.50	75.38	0.00	17.00	8.00	NA	NA	NA	NA	NA
EM01	7.27	66.02	7.15	16.00	6.33	NA	NA	NA	NA	NA
EM02	7.78	94.66	16.78	22.25	9.50	NA	NA	NA	NA	NA
EM03	7.93	103.80	14.36	25.50	9.75	NA	NA	NA	NA	NA
EM04	7.68	114.38	11.00	28.50	10.50	NA	NA	NA	NA	NA
GG01	6.90	14.54	14.54	5.00	0.50	NA	NA	NA	NA	NA
HC01	7.13	34.60	8.72	8.50	3.25	NA	NA	NA	NA	NA
IG01	7.85	316.87	81.05	87.75	23.75	NA	NA	NA	NA	NA
LF01	7.13	31.51	5.45	8.50	2.50	NA	NA	NA	NA	NA
MG01	4.90	20.28	20.28	4.00	2.50	NA	NA	NA	NA	NA
OG01	2.70	3885.94	3758.11	196.33	825.00	NA	NA	NA	NA	NA
PG01	6.70	33.20	33.20	10.00	2.00	NA	NA	NA	NA	NA
SD01	5.57	218.44	107.29	50.67	22.33	NA	NA	NA	NA	NA
SG01	3.40	487.94	491.60	122.33	44.33	NA	NA	NA	NA	NA
SPR01	6.40	889.41	889.41	193.00	99.00	5.90	847.37	0.00	191.00	90.00
SPR02	6.60	1210.05	14.65	283.50	122.00	4.60	1199.56	0.00	276.00	124.00
SPR03	3.30	1288.96	1288.96	264.00	153.00	3.60	1188.75	0.00	242.00	142.00
SPR04	8.00	241.99	241.99	59.00	23.00	6.90	247.86	0.00	63.00	22.00
SPR05	6.80	876.81	876.81	244.00	65.00	6.50	711.95	0.00	206.00	48.00
SPR07	4.50	1669.33	1669.33	296.00	226.00	5.00	1923.39	0.00	345.00	258.00
SPR08	3.50	1445.36	1445.36	264.00	191.00	3.50	1725.41	5.82	330.00	219.00
SPR09	6.70	740.27	740.27	158.00	84.00	5.40	758.63	0.00	167.00	83.00
SPR10	6.70	174.85	174.85	42.00	17.00	5.70	108.58	0.00	27.00	10.00
SPR11	8.20	236.86	236.86	52.00	26.00	7.60	187.19	0.00	42.00	20.00
SPR12	7.40	137.39	137.39	27.00	17.00	7.30	37.76	0.00	11.00	2.50
SPR13	7.90	416.56	416.56	91.00	46.00	7.20	458.00	0.00	101.00	50.00
SPR14	8.00	121.06	121.06	32.00	10.00	7.30	102.84	0.00	28.00	8.00
SPR15	3.60	87.86	87.86	22.00	8.00	3.60	111.08	0.00	28.00	10.00
SPR16	3.00	66.27	66.27	15.00	7.00	3.60	68.76	0.00	16.00	7.00
SPR17	3.70	125.18	125.18	32.00	11.00	3.80	176.60	0.00	46.00	15.00
TC01-1	7.05	27.99	22.55	7.50	2.25	NA	NA	NA	NA	NA
TC01-2	NA	32.32	32.32	8.00	3.00	NA	NA	NA	NA	NA
TP01	7.90	64.65	64.65	16.00	6.00	3.00	89.48	0.00	21.00	9.00
TP02	3.20	78.01	78.01	23.00	5.00	4.70	75.21	0.00	26.00	2.50
TP03	7.90	98.59	0.00	23.00	10.00	7.60	98.59	0.00	23.00	10.00

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Data for *aufwuchs* (the abiotic and biotic material, including the periphyton or algae community, that accumulates on submerged rocks and other surfaces) and benthic invertebrates in the Arkansas River were available (Kiffney and Clements, 1993). The metals analyzed in this study were cadmium, copper, and zinc. Kiffney and Clements (1993) observed that whole body metal concentrations for invertebrates were higher at AR03 and AR05 (stations downgradient of the Arkansas River confluence with California Gulch) compared to EF01, AR01, and AR02 (stations upgradient of California Gulch confluence with the Arkansas River). Concentrations in *aufwuchs* were also elevated.

Metal concentrations were highest in grazers and collector–filter feeding groups (mayfly *[Baetis spp.]*, stonefly *[Pteronarcella badia]*, and caddisfly *[Arctopsyche grandis])*, and lowest in predatory invertebrates (*Skwala americana* and *Rhyacophila spp.*) (Kiffney and Clements, 1993). Metal concentrations varied seasonally, by station, and among taxa. Cadmium concentrations ranged from approximately 5 micrograms per gram (μg/g) (dry weight) to nearly 30 μg/g in benthic invertebrates from AR03 and AR05 during October 1990 to August 1991. Cadmium concentrations at EF01 ranged from approximately 2.5 to 5 μg/g for October 1990 to August 1991. Cadmium concentrations in invertebrates from AR01 and AR02 ranged from approximately 2.5 to 10 μg/g during the same time period. *Aufwuchs* collected from AR03 and AR05 had cadmium concentrations of 50 to over 100 μg/g, whereas *aufwuchs* collected from EF01, AR01, and AR02 contained less than 50 μg/g (dry weight).

Copper concentrations ranged from approximately 20 μg/g (dry weight) to nearly 150 μg/g in benthic invertebrates from AR03 and AR05 during October 1990 to August 1991. Copper concentrations at EF01 ranged from approximately 15 to 50 μg/g for October 1990 to August 1991. Copper concentrations in invertebrates from AR01 and AR02 ranged from approximately 10 to 50 μg/g during the same time period. For any given sampling period, copper concentrations were higher in invertebrates collected from stations below the confluence with California Gulch compared to those collected from stations upgradient of California Gulch, with only one exception. *Rhyacophila* collected from AR05 during October of 1990 had similar copper concentrations as those collected from EF01, AR01, and AR02 during the same time period.

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Aufwuchs collected from AR03 and AR05 had copper concentrations of approximately 180 to over 500 μg/g; *aufwuchs* collected from EF01, AR01, and AR02 contained copper at less than 100 μg/g to nearly 400 μg/g (dry weight) during October 1990 to August 1991. During the May 1991 sampling period, *aufwuchs* from EF01 had higher copper concentrations than those collected from AR03, AR05, or AR08, although this difference was not statistically significant.

Zinc concentrations ranged from approximately 600 μg/g (dry weight) to 7,500 μg/g in benthic invertebrates from AR03 and AR05 during October 1990 to August 1991. Zinc concentrations at EF01 ranged from approximately 200 to 1,000 μg/g for October 1990 to August 1991. Concentrations in invertebrates from AR01 and AR02 ranged from approximately 1,500 to 7,000 μg/g during the same time period. For any given sampling period, the highest zinc concentrations were observed in invertebrates collected at stations below California Gulch (AR03, AR05, or AR08) relative to the upgradient sampling stations. However, these differences were not always statistically significant, and there were instances where zinc concentrations in invertebrates from one of the upgradient stations would exceed one or more of the downgradient sampling locations.

Aufwuchs collected from AR03 and AR05 had zinc concentrations of 2,500 to nearly 15,000 μg/g, whereas *aufwuchs* collected from EF01, AR01, and AR02 contained approximately 2,000 μg/g (dry weight) to 6,000 μg/g. There was a general trend for zinc concentrations to be elevated in samples from stations downgradient of California Gulch compared to stations upgradient of California Gulch. In October 1990, samples from AR03 and AR05 were significantly elevated with respect to EF01, and samples from AR03 were significantly elevated with respect to AR01. During the May 1991 sampling period, samples from AR08 were significantly elevated with respect to EF01 and AR02, but not AR01.

Fish residue data were collected to provide data for the Human Health Risk Assessment as part of the on-going Remedial Investigation activities (Table 2-12). Residues in fish from the Arkansas River were low, and it does not appear as if excessive biomagnification (the transfer of contaminants between trophic levels) is occurring. However, only the fillets (muscle) of the fish, which are consumed by humans, were analyzed. Muscle is rarely a site of action for

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Table 2-12

Fish Tissue Residue Data

Fish Tissue Sample ID	Date Collected	Metal Concentration (mg/kg)						
		Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Zinc
A-LF01F-01-910813	08/13/91	0.04	ND	0.4	0.4	0.2	0.04	5.3
A-IG01F-01-910813	08/13/91	ND	ND	0.3	0.4	0.2	ND	5.1
A-EM02F-01-910814	08/14/91	0.02	ND	0.3	0.4	0.2	0.04	5.5
A-EM03F-01-910814	08/14/91	0.02	ND	0.3	0.3	0.2	0.03	4.5
A-EM04F-01-910814	08/14/91	ND	ND	0.2	0.4	0.2	0.10	3.9
A-TC01F-01-910814	08/14/91	ND	ND	ND	0.3	0.2	0.11	5.1
A-EF01F-01-910812	08/12/91	0.02	ND	0.2	0.3	0.2	0.04	4.7
A-EF02F-01-910815	08/12/91	0.02	ND	0.2	0.4	0.2	0.05	4.8
A-EF02F-01-910815	08/15/91	0.06	ND	0.3	0.4	0.2	0.08	9.3
A-HC01F-01-910813	08/13/91	0.02	ND	0.4	0.4	0.2	0.07	4.0
A-AR01F-01-910813	08/13/91	0.10	ND	0.4	0.6	0.2	0.09	5.9
A-AR02F-01-910813	08/13/91	0.02	ND	0.3	0.4	0.2	0.12	8.0
A-AR03F-01-910813	08/13/91	ND	ND	0.3	0.5	0.2	ND	8.9
A-AR04F-01-910813	08/14/91	ND	ND	0.4	0.2	0.2	0.20	3.5
A-AR05F-01-910814	08/14/91	ND	ND	0.2	0.3	0.4	ND	5.8

ND = Not detected above the reporting limit.

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toxicity or a target organ for accumulation, especially for metals. Liver, kidney, and other internal organs are typically the tissues likely to accumulate metals as well as be adversely affected by metal toxicity. Therefore, use of fillet data may underestimate the potential for bioaccumulation by trout species. Predators of fish (i.e., larger fish) consume smaller fish whole, as opposed to eating only the fillets; therefore, risk to predators of fish may be underestimated due to the absence of metal concentration data on a whole-body basis.

Concentrations of metals in fish collected from areas downgradient of California Gulch were similar to those in fish collected from upgradient areas. Fish were collected during the summer season, and data were collected only during one sampling season; therefore, any variability due to year or season is unknown. The Aquatic Ecosystem Data Collection Workplan (ASARCO, 1991) called for fish in the 2- to 3-year age class, although it allows for use of smaller fish if 2- to 3-year fish were unavailable. The raw data does not specify what age class the fish that actually made up each sample fall into. Age is a potential confounding variable when considering tissue concentrations, because young fish grow more rapidly than older fish, and total accumulation of certain metals is proportional to exposure time. If the age class differed between one location and another, the data for each location are not comparable.

The tissue of interest was fillet (ASARCO, 1991). However, the work plan states small fish would be frozen whole; thus, the work plan may have allowed for the use of whole fish if only small fish were collected. This is another source of variability that makes comparison of the location-specific data unreliable, as metals data for whole fish are likely to vary from fillet. The single sample collected at each location is potentially not comparable to a sample from another location.

Fish are mobile and are more likely to integrate their exposure over a wider area and range of concentrations than are macroinvertebrates and plants. Because fish may move in their environment, thus altering their exposure, it is difficult with the small sample sizes available to determine if there are any trends in the data between locations upgradient of the confluence of the Arkansas River with California Gulch and downgradient locations. It appears that bioaccumulation of metals is not occurring in fish fillets; there may be sufficient variability in

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the data that there really is no significant difference in tissue concentrations between the different locations. However, if the sample size was larger, or consisted of whole fish only, it is possible that the data could indicate different conclusions.

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SECTION 3
STRESS-RESPONSE ASSESSMENT

As part of the ecological risk assessment, a stress-response assessment was conducted to further characterize potential ecological effects due to site-related contamination. This section describes biological data such as population density, species occurrence, and biomass. Most of the biological data were collected prior to initiation of water treatment at the Yak Tunnel, which commenced in February and March of 1992, and therefore do not reflect current conditions at California Gulch.

3.1 DESCRIPTION OF AQUATIC HABITATS AND AQUATIC LIFE

The major surface water resource in the study area is the Arkansas River and its tributaries. The headwaters of the East Fork of the Arkansas River are located approximately 9 miles north of Leadville at an elevation of 12,540 feet. The confluence of the East Fork and Tennessee Creek forms the beginning of the Arkansas River about 2 miles northwest of Leadville. The northern Arkansas River drains the Sawatch Range to the west and the Mosquito Range to the east. The Arkansas River flows generally south through a valley that varies in width from 1 to 4 miles (EPA, 1990).

The California Gulch site is drained by several gulches that form tributaries to the Arkansas River, including Oregon and California gulches (Figure 1-1). These gulches flow mostly in response to precipitation events and snow melt. Drainage tunnels constructed to drain underground mines discharge metal-contaminated groundwater to the gulches. These tunnels include the Canterbury, Leadville, and Yak tunnels. The Leadville Tunnel discharges into the East Fork of the Arkansas River upstream of sampling station EF2. The Yak Tunnel is the most extensive of these tunnels, and collects groundwater from the Iron Hill, Ibex, and Breece Hill Mine complexes and discharges it into California Gulch.

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3.1.1 California Gulch

In general, upper California Gulch does not provide adequate habitat to sustain fish populations as locations above the Yak Tunnel become dry during the fall season (CDOW, 1990). Locations downstream of the Yak Tunnel have permanently flowing water provided by the tunnel itself (Davies, 1991). Flow in lower California Gulch is dependent upon the Yak Tunnel, surfacing groundwater, and discharge from the Leadville Waste Water Treatment Facility into California Gulch (Figure 1-1) (CDOW, 1990). Due to the potential for surface runoff and groundwater discharge into California Gulch and the other drainages to California Gulch, aquatic habitats and aquatic life are likely to be adversely impacted by source-related contaminants in the vicinity of the site.

The physical habitat of California Gulch has been described as homogenous. Width of the stream ranges between 2.8 and 4.4 feet with a depth from 0.1 to 0.5 feet. The stream is comprised of a series of runs and riffles with few meandering sections and, therefore, few pools are able to form. The pools that do exist are very small and generally less than 6 inches deep. Stream bottom substrate ranges from small gravel to 8-inch cobbles. In some areas, ferric hydroxide deposits have cemented the substrate materials together.

Riparian areas are comprised of sparse mixed grasses and sedges. Slag deposits, waste rock, tailings, parking areas, and buildings are located next to California Gulch. Orange silt (ferric hydroxide deposits) lines some of the bank areas (CDOW, 1990). California Gulch flows around four tailings ponds located within the drainage (EPA, 1990).

Aquatic habitat in the lower reaches of California Gulch has slightly different characteristics than the upper reaches. In the lower reaches the stream is narrower and deeper, approaching 1 foot, and flows over fluvial tailings and waste rock. There are some small pools, but California Gulch in the lower reaches is comprised primarily of runs and riffles. The bottom substrate is a larger cobble, and ferric hydroxides and silt cover the entire substrate. Riparian vegetation primarily consists of dense grasses and sedges, with shrubs present along the bank at the lowest areas. Stock grazing is a land use in this lower area (CDOW, 1990).

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There are minimal aquatic populations residing in the California Gulch tributary (CDOW, 1990; Davies, 1991). Multiple sampling efforts and stream characterizations performed by several investigators revealed the presence of virtually no invertebrate taxa in California Gulch. During a spring sampling investigation, CDOW (1990) observed only two invertebrate taxa. These two taxa were *Limnophora* (Muscidae) and *Diplocladius cultriger* (Chironomidae). CDOW (1990) observed only three invertebrate taxa during a fall sampling investigation. The 3 species included 1 adult Hemipteran, 1 adult beetle *(Coleoptera)*, and 91 *Daphnia*. Results of the investigation indicated that these organisms are typical of small ponds and probably originated in the discharge from the sewage treatment plant adjacent to the site. During the fall and spring investigations, CDOW (1990) characterized the habitat in California Gulch and sampled for any existing fish populations. Results of the field efforts yielded no occurrence of any fish species.

Decay of organic material in the aquatic environment, possibly enriched by the disposal of sewage effluent, together with detritus formed by natural weathering processes, provides a rich source of nutrients in both the bottom sediments and the overlying water body. Therefore, there should be an adequate food supply for aquatic plants, algae, microbes, and invertebrates in California Gulch, and the lack of aquatic invertebrates in California Gulch is atypical.

3.1.2 Arkansas River

The Arkansas River flow is characterized as high by late spring and early summer. Many of the tributaries to the Arkansas River dry up during the summer and fall (LaBounty et al., 1975). Monthly average flows in the Arkansas River at Leadville Junction (above California Gulch) ranged from 13.6 to 350 cubic feet per second (cfs) for water years 1968 through 1983 (USGS, 1985). The Arkansas River has been characterized as typical of a soft-water mountain stream in the Rocky Mountain Region with respect to water quality parameters of hardness (44 to 98 mg/ℓ), pH (7.6 to 8.0), and alkalinity (34 to 72 mg/ℓ) (Clements and Kiffney, 1994).

The resident fish community of the Arkansas River in the immediate vicinity of California Gulch is dominated by brown trout *(Salmo trutta)* and brook trout *(Salvelinus fontinalis)*. On occasion, rainbow trout *(Onchorhynchus mykiss)* and cutthroat trout *(Salmo clarki)* have also been caught

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in the area (Anderson, 1991). The rainbow and cutthroat trout are believed to have originated from neighboring CDOW hatcheries from the Lake Fork Creek (Anderson, 1991). The fish community appears to have been severely affected by inflows from California Gulch, and possibly other sources, in that a 70 to 90% decrease in trout populations was observed between the confluence of California Gulch and the Arkansas River downstream towards Salida (CDOW, 1990). Brown trout populations exhibit reduced growth, density, and survival beyond 3 to 4 years of age and at lengths greater than 35 centimeters (cm) (14 inches) (USFWS, 1993). The specific reasons for these differences (i.e., direct or indirect effects of metal contamination in water, or other reasons) have not been determined. Populations of aquatic organisms in temperate streams are adapted for living in a dynamic environment that fluctuates on a daily as well as seasonal basis. Therefore, most stream fishes can maintain populations under a variety of physical and chemical conditions. Stream fish populations are often capable of interacting with a diversity of other species, and are quick to colonize new areas of suitable habitat (Moyle and Cech, 1982). Therefore, the results of the CDOW and USFWS studies are indicative of atypical stressors in the Arkansas River near California Gulch.

3.2 CURRENT BIOLOGICAL DATA

This section of the report describes data obtained from the following sources:

- Aquatics Associates (1991) for ASARCO, as cited from ASARCO (1993) and Aquatics Associates (1991).
- USFWS (1993).
- CDOW (1990).
- ENSR (1991).
- EPA (Willingham, 1993).
- Clements (in press), Clements and Kiffney (1994), and Kiffney and Clements (1993).

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Figure 1-1 indicates the sampling locations on the Arkansas River and its tributaries where the data were collected. Most of the studies discussed consider AR1 and AR2 to be reference or uncontaminated areas for comparison to AR3 and other stations downstream of California Gulch. In addition, EF1 is considered a reference area for the East Fork, whereas EF2 is downstream of the drainage from the Leadville Mine Drainage Tunnel (LMDT). However, because the East Fork feeds the mainstem of the Arkansas River, it is possible that AR1 and AR2 may be affected by upstream impacts to the water quality. There are also other source areas outside of the California Gulch Superfund Site. It is the purpose of this risk assessment to determine adverse environmental effects due to source areas within the site boundaries; therefore, AR1 and AR2 will be considered reference areas for comparison to the downstream locations.

3.2.1 Biological Data Collected by EPA as Part of RI Activities

Sublette (1991) observed the presence of considerable amounts of algal and plant fibrous material in samples collected from below the confluence of California Gulch in the Arkansas River. Analysis of invertebrate taxa collected from sample locations that spanned the width of the river indicated a significant gradient of species diversity and numbers across the width of the stream. This suggests effects from heavy metal loading from California Gulch to the east bank of the Arkansas River. According to Sublette (1991), of the macrobenthic species that occurred with the greatest frequency, the following species appear to be most tolerant of heavy metal adverse effects:

- *Baetis* sp.
- *Brachycentrus americanus*
- *Epeorus* sp.
- *Podmosta* sp.
- *Heterlimnius* sp.

Sublette reported that of those invertebrate species that occur with greatest frequency, the least tolerant to metal effects include:

- *Rhithrogena* sp.

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- *Cinugmula* sp.
- *Diamese leona*
- *Eukiefferiella* sp.
- *Orthocladius frigidus*

Sublette noted that at EF2 there was no effect of stream position on benthic invertebrate data; however, at AR3 there was strong variation with position. Sublette (1991) noted changes in the dipteran *(Chironomidae)* diversity and numbers that could be related to heavy metal loading. The highest numbers of Plecoptera and Chironomidae were on the west side (left bank). These taxa were lower in the middle, and were fewest on the east (right bank) where California Gulch loading to the Arkansas occurs. *Trichoptera* and *Ephemeroptera* differed by stream position as well. Sublette stated that *Ephemeroptera* were the most sensitive to heavy metals and that AR3 had decreased species diversity that only partially recovered by AR4.

Sublette did not analyze the data extensively due to some sampling problems that may have biased the data (Sublette, 1991). These problems included lack of replicates and large net mesh size.

EPA (Willingham, 1993) collected water samples and conducted acute toxicity tests with the invertebrate *Ceriodaphnia sp.* and fathead minnows *(Pimephales promelas)*. These toxicity tests evaluate the toxicity of water from the site relative to the toxicity of water from reference areas, or reconstituted water used as a control. Percent mortality and the percent of site water on a volume to volume basis causing death in 50% of the exposed population (LC_{50}) for invertebrates are presented in Figures 3-1 and 3-2. Mortality in invertebrates was highest for invertebrates exposed to water collected from EF2, AR4, and AR5. Water from CG6 was the most toxic to invertebrates as indicated by the LC_{50} value of less than 10% site water required to produce the LC_{50}. Mortality in fish (Figure 3-3) was highest for fish exposed to water collected from AR3. The LC_{50} with lower and upper confidence limits (95%) for fish exposed to water from CG6 was 54.8% (44.7 to 67.2%) site water in September 1991, and was 33.6% (18.0 to 63.8%) site water in March 1992. LC_{50} values were not calculated for the other sites. The Yak Tunnel Water Treatment facility became operational in February to March of 1992, and it is likely that the stream water quality was in a state of flux; therefore, it is not surprising that acute toxicity

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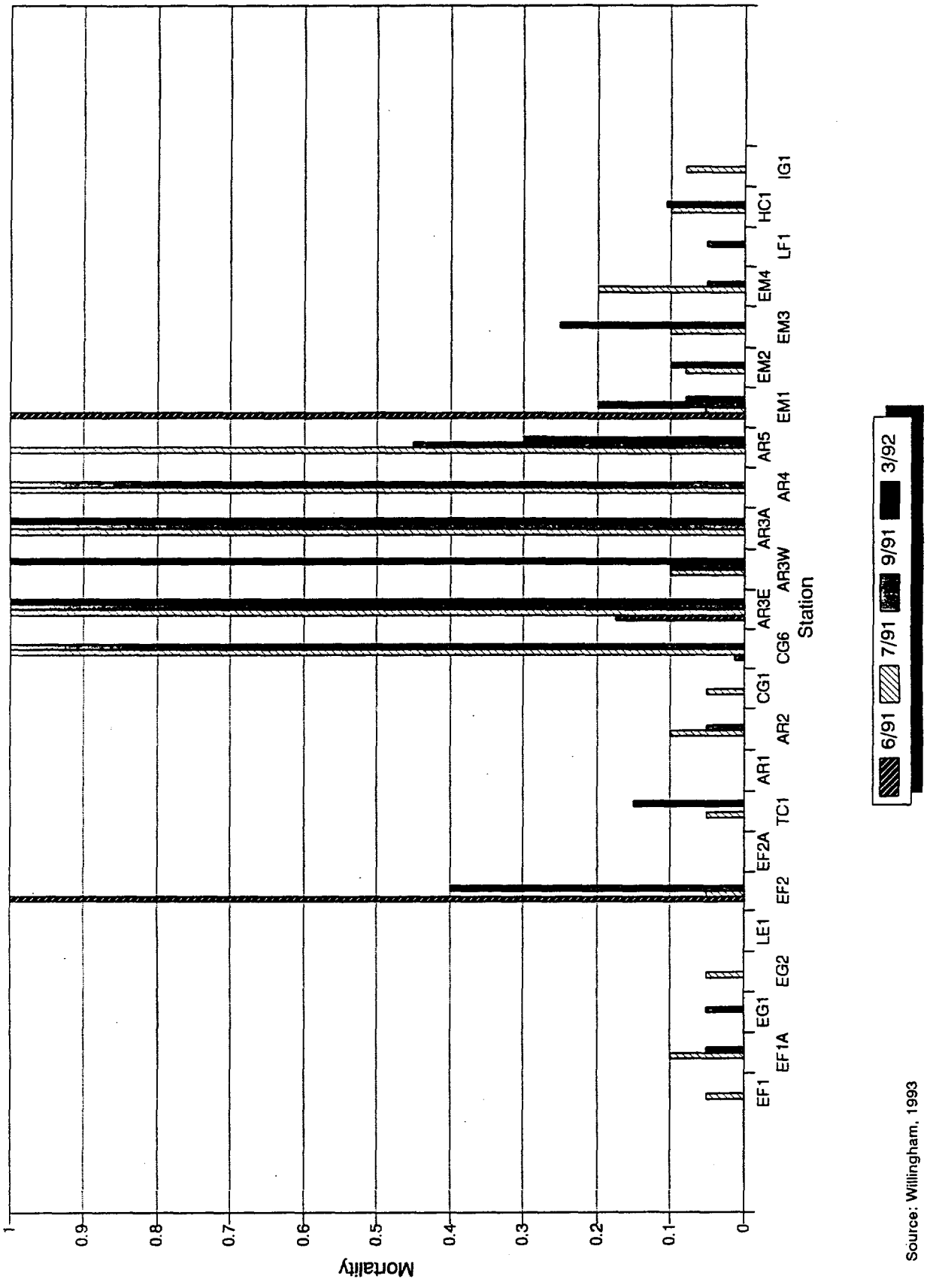
Figure 3-1

Invertebrate Mortality in 100% Site Water



Source: Willingham, 1993

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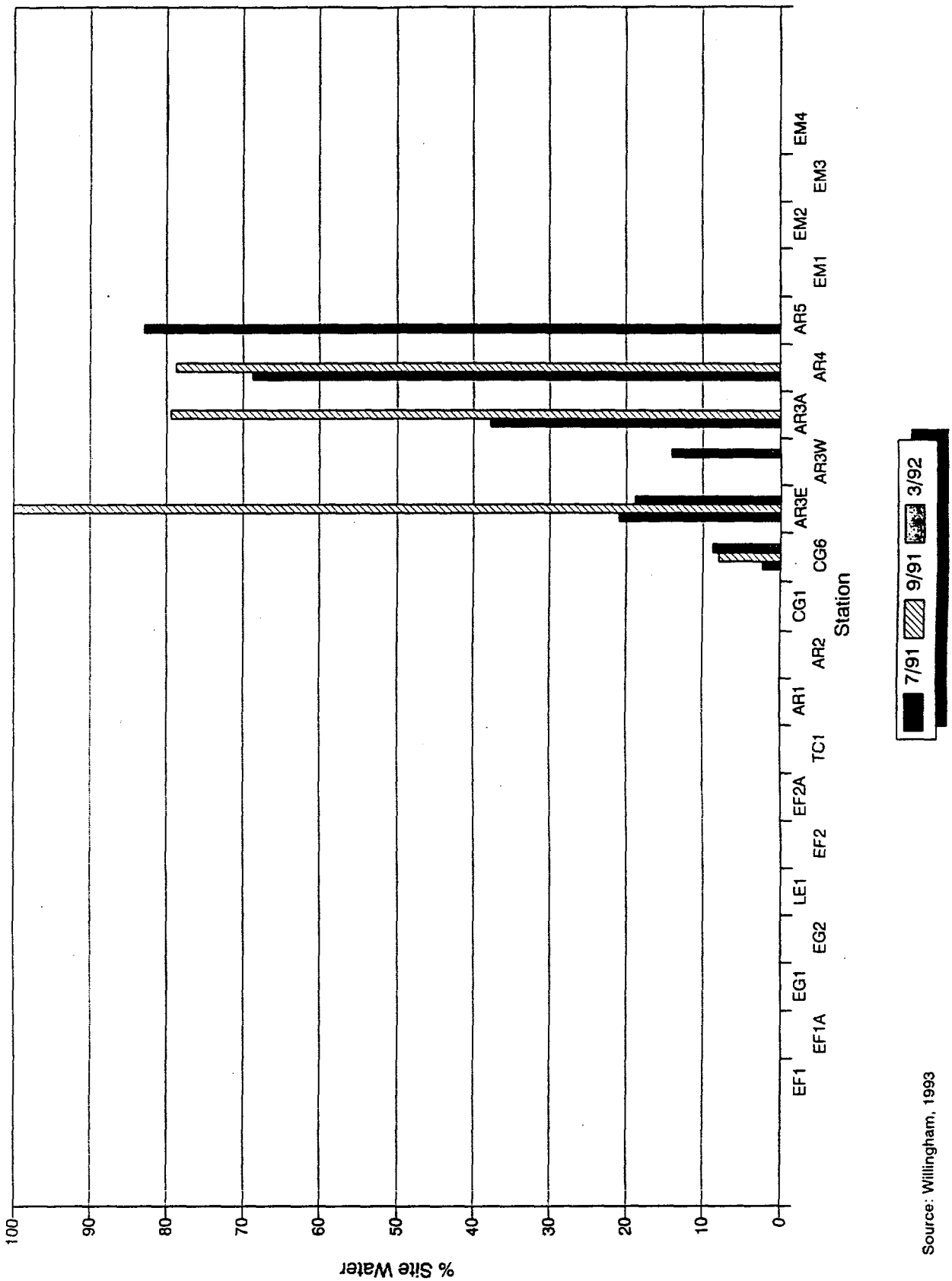
Figure 3-2

Percent Site Water resulting in a 48-hour LC50 for Invertebrates



Source: Willingham, 1993

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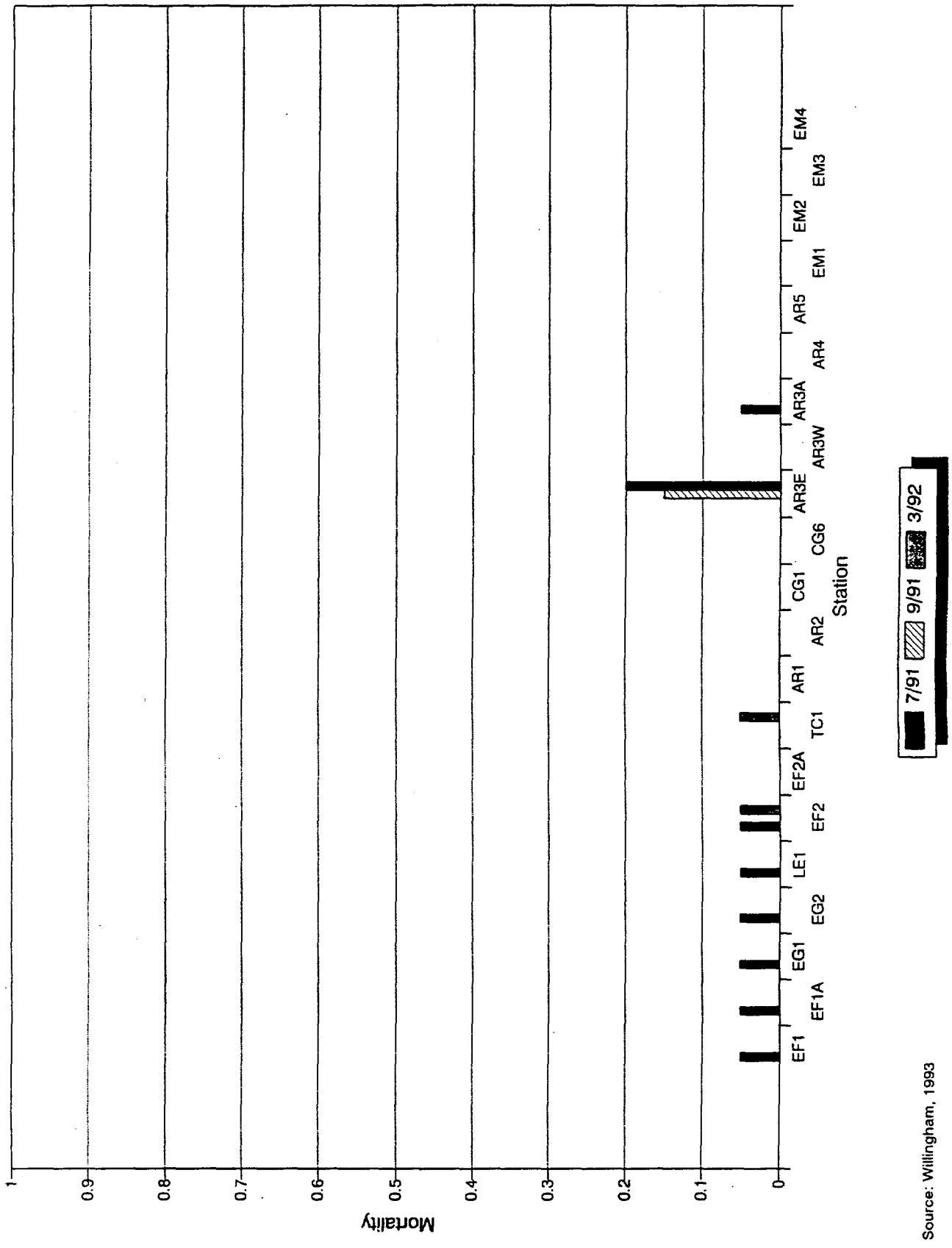
Figure 3-3

Fish Mortality in 100% Site Water



Source: Willingham, 1993

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is similar for the Pre-Yak (i.e., sampling events prior to March 1992) and Post-Yak period. For the acute toxicity test data, mortality exceeding 20% is considered a significant adverse effect (Willingham, 1993). Tests where more than 10% of the control animals die are rejected.

Sediment toxicity tests were conducted with samples from the California Gulch Site (Willingham, 1993). Stations where mortality was significantly higher than controls were AR03A, AR03, and C606. Samples from East Fork, Empire Gulch, Hamms Mill, AR06, and Tennessee Creek did not differ significantly from controls. Dunnetts ANOVA, with an alpha level of 0.05, was the statistical test used. The only station sampled on California Gulch was CG06.

3.2.2 ENSR Toxicity Test Results

The results of ENSR's laboratory bioassay (ENSR, 1991) with fathead minnows and invertebrates *(Ceriodaphnia dubia)* indicated that water from AR3 and CG1 was highly toxic to invertebrates and fish. LC_{50}s ranged from 1 to 12.5% site water. Water from AR4 was moderately toxic to invertebrates; the LC_{50} was 37% site water. Some fish died (10 to 35% mortality) when exposed to water from AR4 at 50% and higher ratios of site to dilution water. By contrast, no control fish in these studies died. Water from EM1 was nontoxic as no mortality was observed for either fish or invertebrates. Water from EF2 produced minimal mortality in invertebrates (10%) and no mortality in fish. Figures 3-4 and 3-5 present the ENSR toxicity data for invertebrates and fish. Table 3-1 presents the LC_{50} values for invertebrates and fish.

3.2.3 ASARCO 1991 Biological Data

As part of the RI, data were collected from May 7 through September 18, 1991, for attached algae (periphyton), benthic macroinvertebrates, and fish populations (ASARCO, 1993). Data are lacking for other components of the aquatic ecosystem (e.g., emergent plants, waterfowl, amphibians). The locations that were sampled were from the Arkansas River (AR1, AR2, AR3, AR4, AR5), the East Fork of the Arkansas River (EF1, EF2), Tennessee Creek (TC1), Lake Fork (LF1), Halfmoon Creek (HC1), California Gulch (CG6), Iowa Gulch (IG1), and Empire

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Figure 3-4

Invertebrate Cumulative Mortality at 48-hour

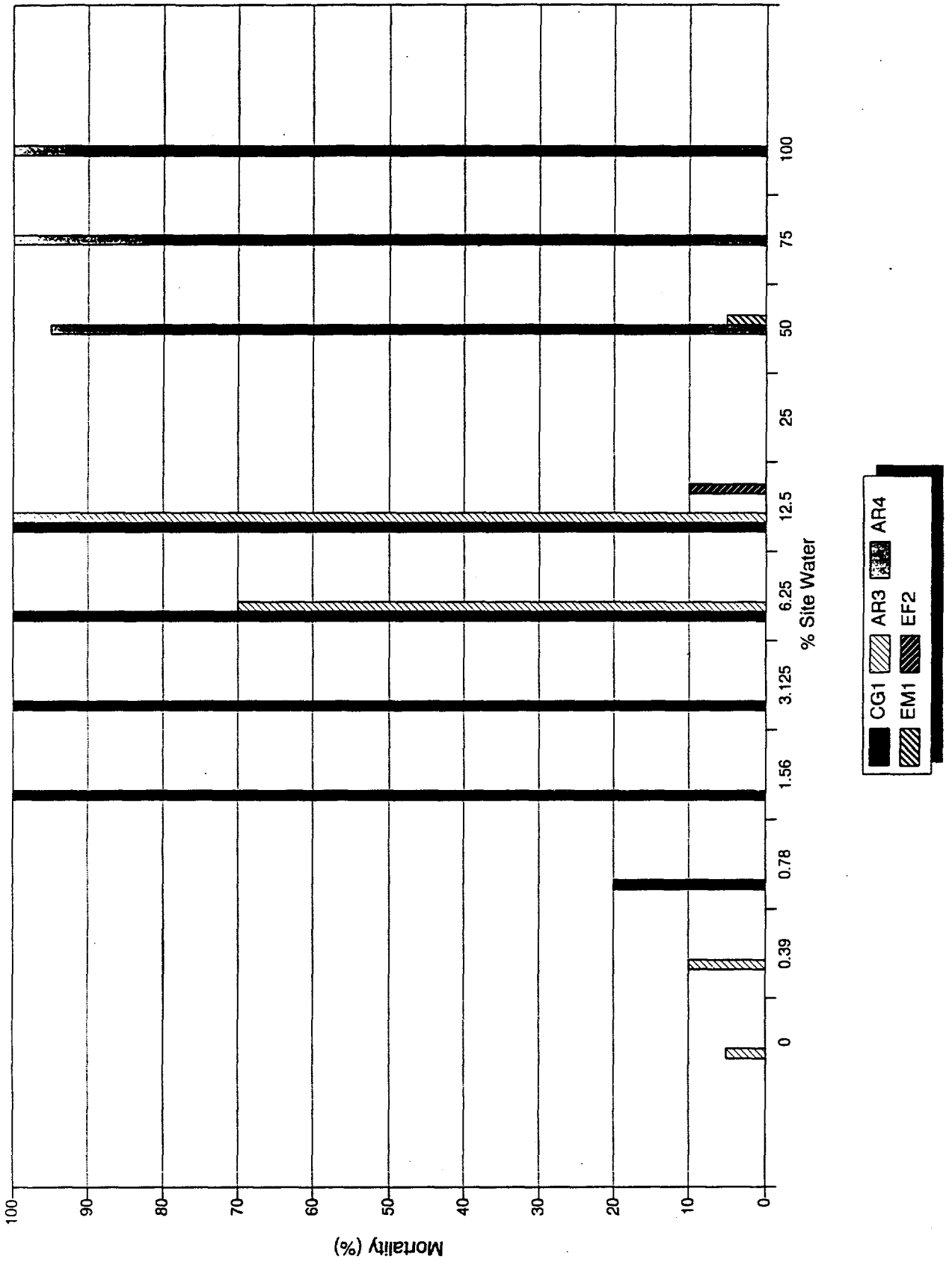

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Figure 3-5

Fish Cumulative Mortality at 96-hour

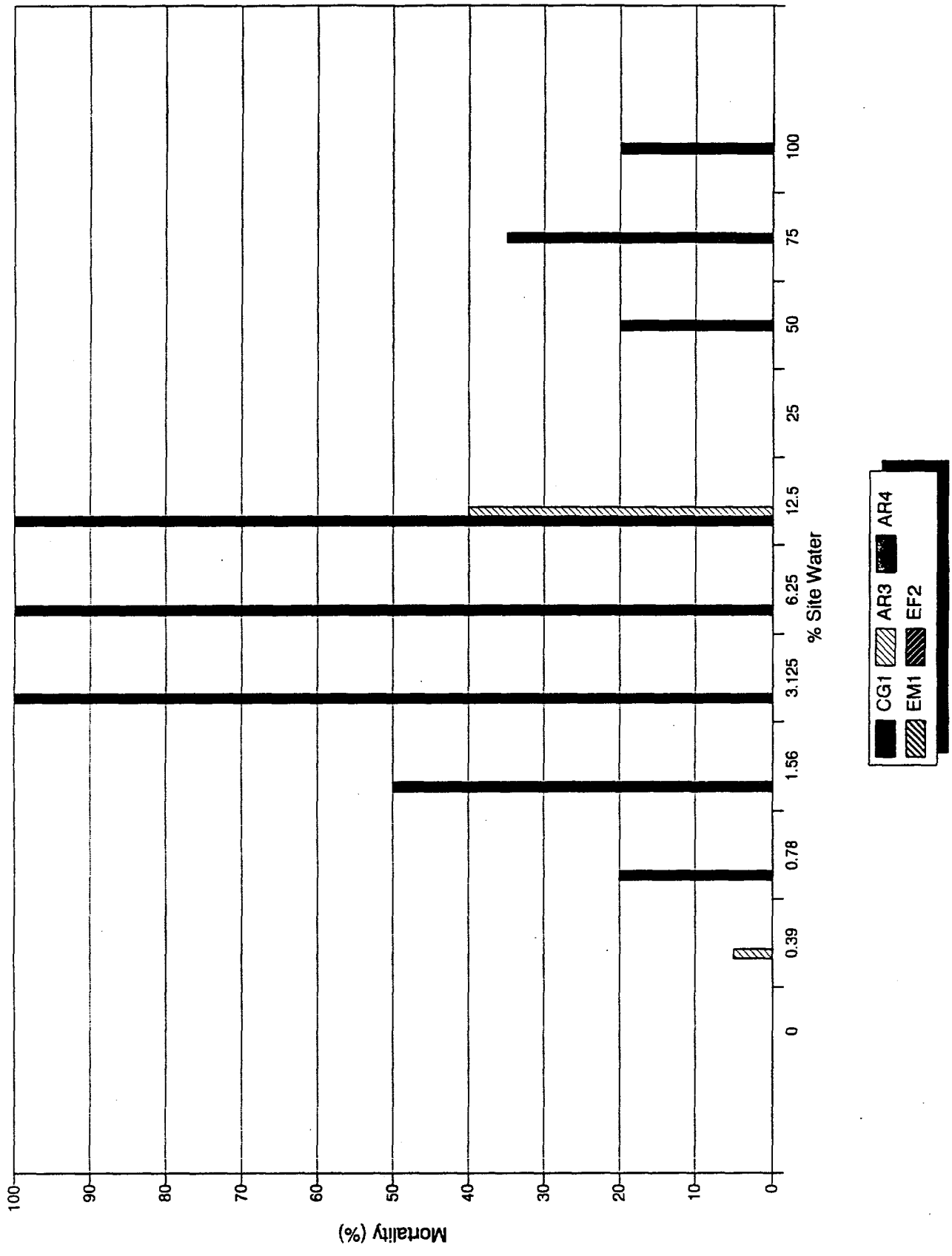

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Table 3-1

LC_{50} Values and Lower and Upper 95% Confidence Limits for Aquatic Invertebrates and Fish

Station	Invertebrate LC_{50} Values (% Site Water)		Fish LC_{50} Values (% Site Water)	
	24-hr Exposure	48-hr Exposure	48-hr Exposure	96-hr Exposure
CG1	1.4 (1.1 - 1.7)	1.0 (0.8 - 1.6)	1.7 (1.3 - 2.1)	1.3 (1.1 - 1.6)
AR3	6.0 (3.125 - 12.5)	5.3 (3.125 - 6.25)	> 12.5	> 12.5
AR4	37.0 (25 - 50)	37.0 (25 - 50)	> 100	> 100
EM1	> 100	> 100	> 100	> 100
EF2	> 100	> 100	> 100	> 100

Source: ENSR, 1991.

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Gulch (EM2, EM3, EM4) (Figure 1-1). Evans Gulch was dry during both fall and spring sampling periods, thus biological data were not obtained from this stream. IG1 was dry during the spring (ice-off) period and was not sampled during this period. Descriptions of the study site habitats were reported in Aquatics Associates (1991), and are summarized in Table 3-2.

Whether or not California Gulch and the Arkansas River have good productive capabilities (i.e., serve as functional habitat) is an assessment endpoint for the ecological risk assessment (Table 1-2). This assessment endpoint is evaluated by measurements such as diversity, density, presence, absence, and changes in community structure. The measurement endpoints for which ASARCO (1993) provided data include altered community structure, number of taxa, species diversity, and density. The specific mechanism of toxicity (e.g., mortality or decreased reproduction) causing the effects can not be determined from these data. However, the population parameters can be linked to site-related disturbances by evaluating apparent trends in the data for on-site compared to off-site areas. For the study sites evaluated by Aquatics Associates for ASARCO, on-site areas are considered to be streams within the California Gulch site boundary.

3.2.3.1 Periphyton

Periphyton are the communities of attached algae and diatoms that include microscopic and larger species (Hynes, 1970). During the fall sampling period of 1991, periphyton sampling was performed by Aquatics Associates at all study stations except EM2 and EM3. Table 3-3 presents a summary of the periphyton data collected by Aquatics Associates (1991). During field activities, sampling locations were located on the left (west), middle, and right (east) side of the river or stream. The sampling locations were similar in depth, substrate type, and sunlight exposure (ASARCO, 1993). The sampling transects were perpendicular to stream channels except at CG6, where the samples were collected from the middle of the channel because the stream was small (ASARCO, 1993). Further details regarding periphyton sampling procedures are found in ASARCO's Aquatic Ecosystem Characterization Report (1993).

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Table 3-2

Summary of Habitats at the Sampling Locations

Location	Substrate	Depth	Turbidity	Habitat
AR1	Small-large cobble, sand and silt where low flow.	Deep	Clear	Riffle, run, undercut banks, gravel, sand, boulders; overhanging willows.
AR2	Cobble, sand, gravel.	Deep 2-4 ft	Clear	Pool, undercut banks; overhanging vegetation.
AR3	Cobble, gravel, sand.		Clear, left (west) Turbid, right (east)	Pool, run, undercut banks; overhanging willow.
AR4	Cobble, gravel, sand; silt and sand deposits where flow reduced.	Deep 4-5 ft	Clear	Long pool-riffle, undercut banks, grazed.
AR5	Small-large cobble.	Deep	Clear	Long riffle-run, few pools, bank erosion, aquatic moss present.
EF1	Small-medium cobble, gravel, sand.		Clear	Pool-riffle-run, overhanging willow, undercut banks, occasional woody debris and boulders.
EF2	Cobble, gravel, sand.		Clear	Short pool-riffle-run, overhanging trees, undercut banks, roots, woody debris.
TC1	Cobble, gravel, sand.		Clear	Short pool-riffle-run, overhanging willow, undercut banks, boulder pools.
LF1	Cobble covered by aquatic vegetation, silt.		Clear	Riffle-run, small pools and undercuts, bank erosion.
HC1	Small cobble, gravel, sand; heavy algal growth.		Clear	Riffle-run, low summer flow, overhanging willows, undercut banks, iron seeps.
CG6	Cobble with bright green algae.	Shallow 6 ft. Dry at ice-off	Clear	Riffle-run, minimal trout cover.
EM4	Small cobble, gravel sand, silt in runs where velocity reduced.	Shallow 6-8 ft.	Clear	Riffle, run.

Source: Aquatics Associates, 1991.

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Table 3-3

Periphyton Data and Summary Statistics for the Fall Sampling Period, 1991

Station	Number of Taxa					Density (cells/sq. mm)				
	Left	Middle	Right	Mean	SD	Left	Middle	Right	Mean	SD
CG6	7	5	6	6.0	1.0	2.65E+05	6.06E+05	4.28E+04	3.05E+05	2.84E+05
AR1	41	34	41	38.7	4.0	2.75E+04	1.82E+04	5.78E+04	3.45E+04	2.07E+04
AR2	31	34	22	29.0	6.2	2.37E+03	2.55E+03	2.86E+03	2.59E+03	2.52E+02
AR3	25	26	12	21.0	7.8	7.47E+03	3.71E+03	5.71E+04	2.28E+04	2.98E+04
AR4	24	31	39	31.3	7.5	1.46E+05	3.06E+04	8.25E+03	6.17E+04	7.41E+04
AR5	32	16	32	26.7	9.2	3.07E+04	9.42E+03	4.40E+03	1.48E+04	1.40E+04
EF1	46	32	45	41.0	7.8	5.40E+04	1.19E+05	5.87E+04	7.73E+04	3.64E+04
EF2	21	27	25	24.3	3.1	1.50E+04	2.43E+04	2.01E+04	1.98E+04	4.65E+03
TC1	47	39	38	41.3	4.9	4.63E+04	1.27E+05	6.54E+04	7.95E+04	4.22E+04
LF1	40	45	53	46.0	6.6	8.58E+04	2.27E+05	2.39E+05	1.84E+05	8.51E+04
HC1	25	19	24	22.7	3.2	6.18E+04	4.28E+04	9.91E+04	6.79E+04	2.86E+04
IG1	29	19	16	21.3	6.8	1.11E+04	8.28E+04	8.67E+04	6.02E+04	4.26E+04
EM4	33	34	27	31.3	3.8	3.44E+04	7.76E+05	1.61E+05	3.24E+05	3.97E+05

SD - standard deviation

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Data that were collected include the number of taxa observed and cell density (cells/sq. mm) for each position in the stream (left or west, middle, and right or east). A mean and sample standard deviation are reported for periphyton data as well. Species diversity was calculated with the Shannon-Weiner formula by Aquatics Associates (1991).

The total and average number of periphyton taxa observed at each stream position and sampling location are presented in Figure 3-6. At AR3, the first sampling location on the Arkansas River downstream of California Gulch, the total number of taxa appears lower on the right (east) side where the California Gulch drainage first mixes with Arkansas River water than at the middle or left stream positions. Total number of taxa also were lower at AR3 than at upgradient locations, especially when the right bank is compared to upstream sampling locations. Total taxa number were also lower for CG6 than at any other sampling location. When total number of periphyton taxa were averaged across stream position, both CG6 and AR3 had lower numbers of taxa than upgradient locations AR1 and AR2 (Figure 3-6). AR4 and AR5 both had lower average numbers of taxa than did AR1. Periphyton density (Figure 3-7) was highest at EM4 and CG6, and generally low at all locations in the Arkansas River.

Species diversity was measured by the Shannon-Weiner diversity index, a standard index where a low value indicates lower diversity than a higher index value. Diversity was calculated by Aquatics Associates (1991) and is presented in Figure 3-8. Species diversity was lower at CG6, AR3, AR4,and AR5 than at AR1 or AR2.

The species present at CG6 reflect that the major stress is elevated metal concentrations, as opposed to organic enrichment from the Leadville Sewage Treatment Plant (ASARCO, 1993). *Euglena* sp., a genus tolerant to metals, was the major contributor to algal density at CG6 (ASARCO, 1993). The diatoms abundant at CG6, which were metal tolerant, also dominated the community at AR3E, where California Gulch empties into the Arkansas River. Other species present were apparently unable to establish a healthy population (ASARCO, 1993). Increases in algal diversity and number of taxa downstream at AR4 and AR5 suggest an improvement in water quality with distance from California Gulch (ASARCO, 1993).

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Figure 3-6

Number of Periphyton Taxa by Location

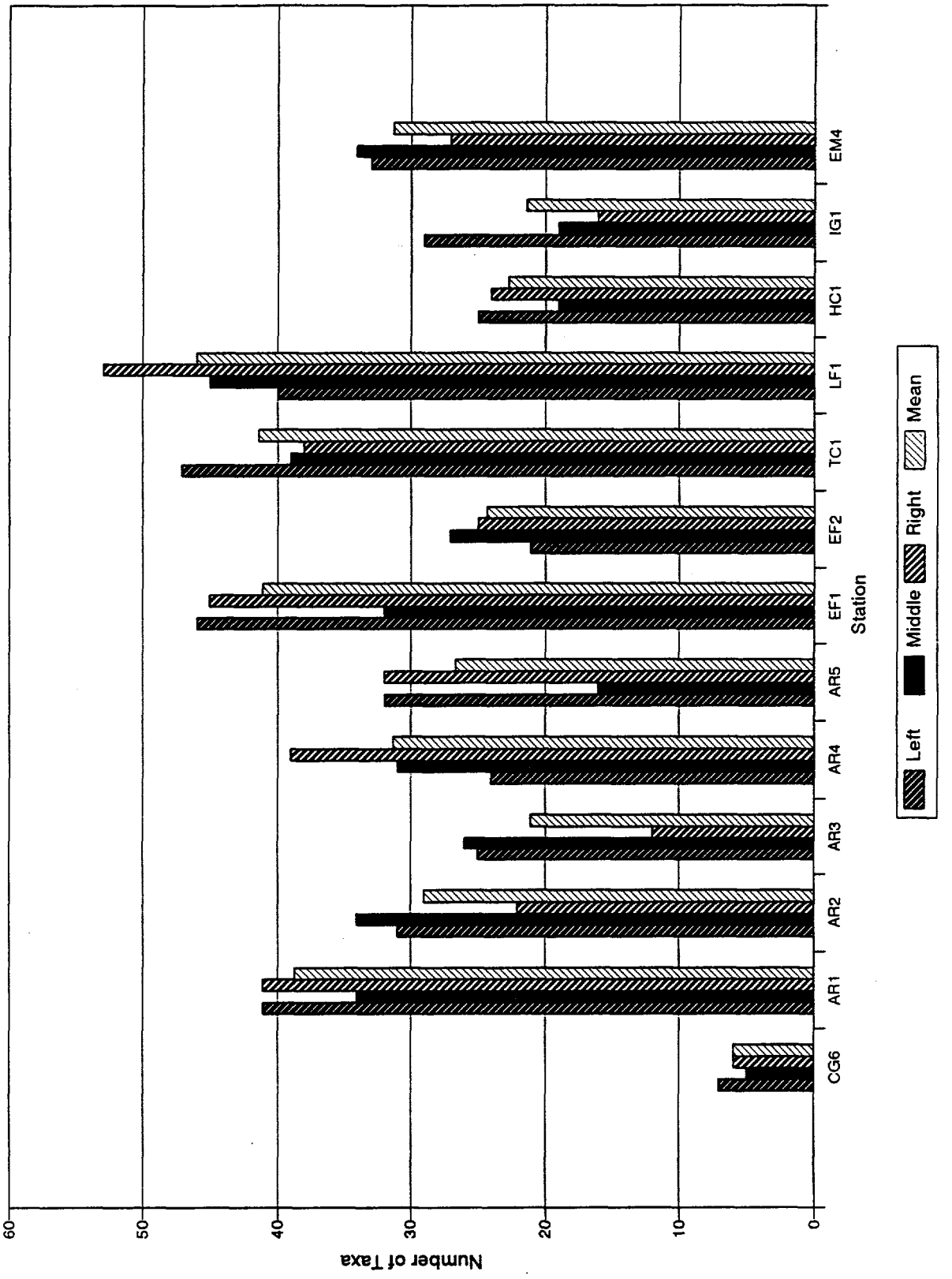

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Figure 3-7

Periphyton Density by Location

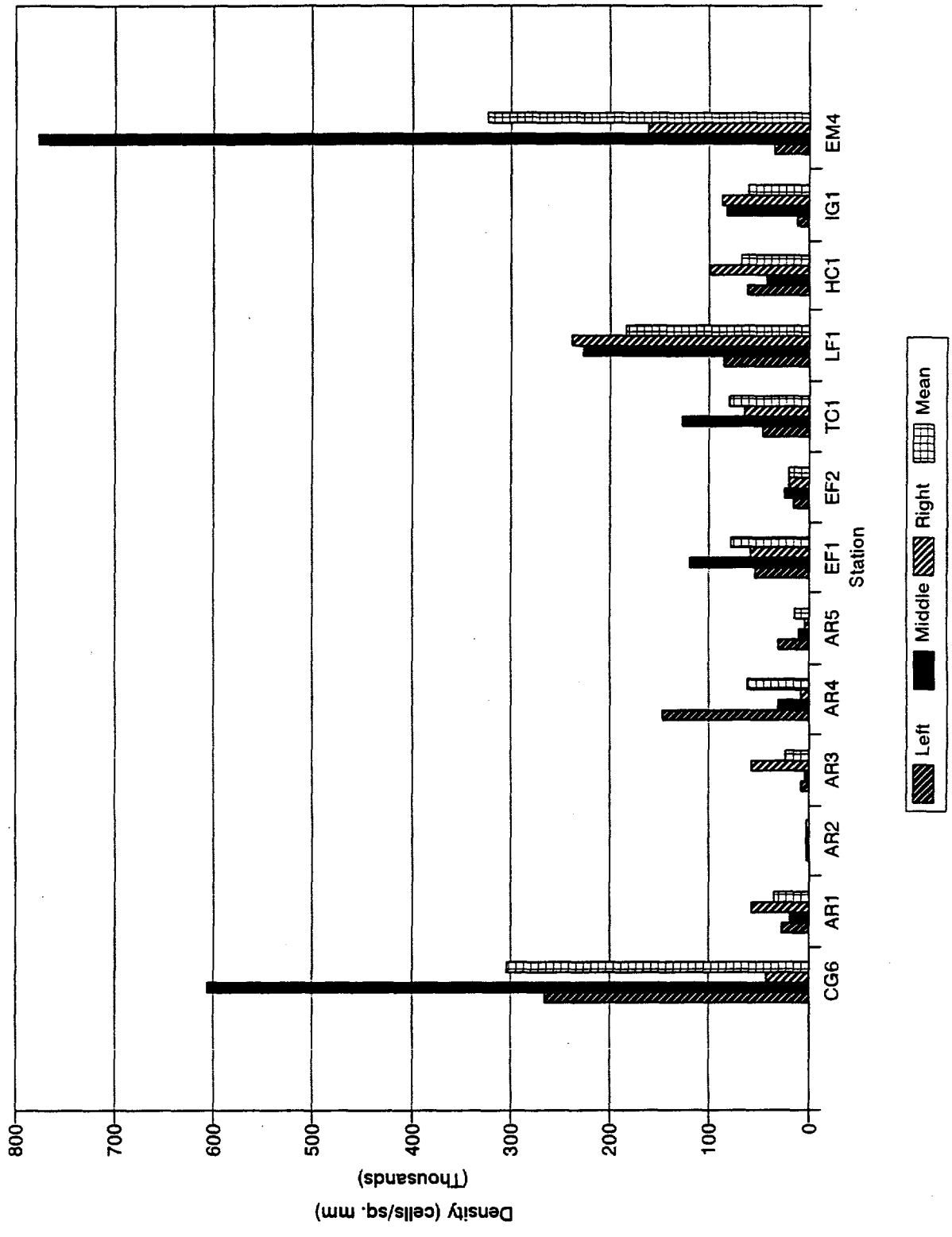

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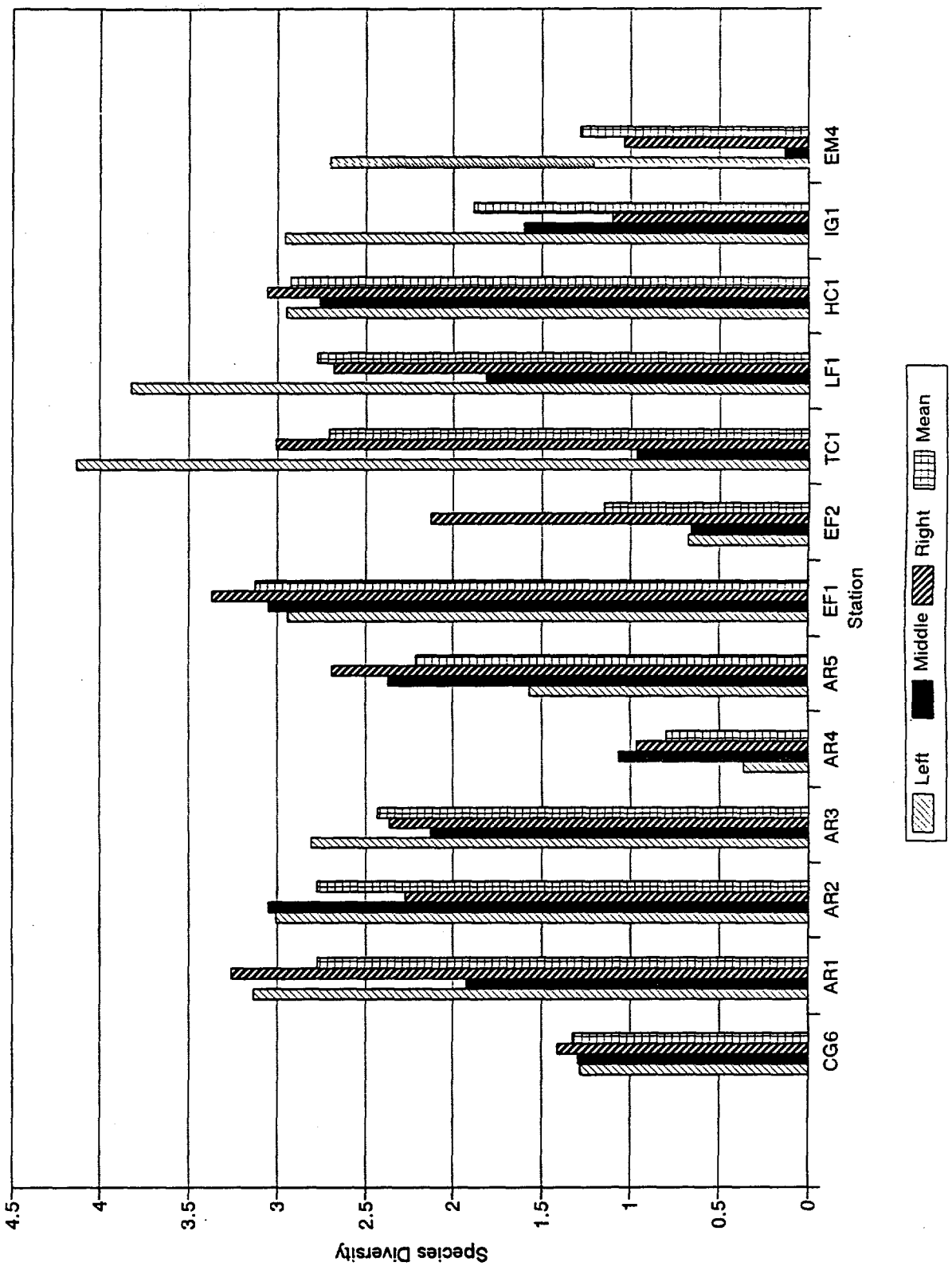
Figure 3-8

Periphyton Diversity by Location



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Periphyton communities are expected to be variable due to natural causes such as substrate and water quality; however, periphyton communities are also sensitive to metal concentrations in water. Percent composition by taxa is a method for evaluating changes in community structure resulting from contaminant-related excessive mortality or other toxic effects. Certain species that are intolerant of excessive metals concentrations may not be able to survive or reproduce, thus allowing changes in community structure reflective of more tolerant species.

ASARCO (1993) calculated Community Loss and Jaccard Coefficient of Community Indices, which are values that indicate similarity in community composition between sites. The Community Loss Index measures the loss of species between treated and reference areas; higher values of the index indicate greater differences between treated and reference areas. The Jaccard Coefficient measures the degree of similarity; higher values of this index indicate that the communities are more similar. A comparison of AR1 to other stations on the Arkansas indicated greatest dissimilarity with AR3, especially with AR3E, for both indices. AR4 and AR5 were not as similar to AR1 as AR2, although the communities were not as different as those at AR3. These indices indicate that changes in the benthic community as a result of inflow from California Gulch are occurring. The greatest impacts to the community are in the vicinity of California Gulch, and that the impacts to the community decrease with distance downstream.

Analysis of Variance (ANOVA) indicated that the differences in number of taxa, but not density, were statistically significant ($p < 0.05$) by location. Duncan's Multiple Range Test indicated that CG6, AR3, and AR5 had significantly fewer taxa at the 95% confidence level than AR1. Some of the assumptions necessary to perform ANOVA were violated. The variances did not appear homogeneous. Therefore, nonparametric statistics may be more appropriate. A nonparametric test (Kruskal-Wallis test), which makes no assumptions regarding the underlying data distribution, was also used to assess the data. The Kruskal-Wallis test indicated CG6, AR3, AR4, and AR5 had fewer periphyton taxa than AR1. These results are similar to those obtained by application of parametric statistics. CG6, AR3, and AR5 had fewer taxa than AR2 as well. The Kruskal-Wallis test indicated CG6 had a higher density of periphyton than any stations in the Arkansas River.

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3.2.3.2 Benthic Macroinvertebrates

Benthic macroinvertebrate communities may be affected by concentrations of inorganics as well as by physical disturbance related to increased sediment load from run-off. Additionally, invertebrate communities are influenced by natural factors such as substrate, current, temperature, food, oxygen, vegetation, acidity, hardness, shade, and drought (Hynes, 1970). The availability of suitable habitat may also influence the benthic community because of migration between habitats and drift of individuals from upstream habitats into downgradient areas. The sampling locations were selected to minimize variance in the data due to habitat-related variables, and to emphasize differences between stations due to inflow from California Gulch.

Invertebrates are an important component of stream systems and serve as a major food source for fish. There are families in certain insect orders that are normally associated with streams, and sometimes whole orders are associated with lotic ecosystems. These taxa include members of the orders *Diptera*, *Ephemeroptera*, *Trichoptera*, and *Plecoptera*.

Benthic macroinvertebrates were collected in the spring during ice-off, and during the fall sampling period of 1991 (Aquatics Associates, 1991). Benthic macroinvertebrates were collected at all study locations except EM2 and EM3. Table 3-4 presents the data from the benthic macroinvertebrate sampling conducted by Aquatics Associates (1991) for ASARCO (1993). The number of taxa and density (number of individuals/sample) observed at the left (west), middle, and right (east) sampling positions at each sampling location are reported (Table 3-4). The mean and sample standard deviation for both number of taxa and density are also presented in Table 3-4. Relative abundance in percent for five commonly observed taxa (*Ephemeroptera*, *Diptera*, *Trichoptera*, *Oligochaeta*, and *Plecoptera*) and a category containing other taxa (i.e., *Coleoptera*, *Cladocera*) are provided as well. Further information may be obtained by referring to ASARCO's Aquatic Ecosystem Characterization Report (1993). The mean number of taxa by sampling location for ice-off and fall sampling periods is presented in Figure 3-9. The fewest number of taxa were observed at CG6. Density of benthic macroinvertebrates was lowest at CG6 as well (Figure 3-10). The Community Loss Index was reported in ASARCO (1993),

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Table 3-4

Benthic Macroinvertebrate Data and Summary Statistics

Ice-off and Fall Sampling Periods, 1991

ICE-OFF

Station	Number of Taxa						Density				
	Left	Middle	Right	Total	Mean	SD	Left	Middle	Right	Mean	SD
CG6	3	0	0	3	1.0	1.7	12	0	0	4.0	6.9
AR1	17	17	15	21	16.3	1.2	70	66	77	71.0	5.6
AR2	12	10	8	18	10.0	2.0	93	81	67	80.3	13.0
AR3	14	15	8	22	12.3	3.8	86	93	16	65.0	42.6
AR4	16	10	14	20	13.3	3.1	202	666	179	349.0	274.8
AR5	16	20	12	24	16.0	4.0	386	232	127	248.3	130.3
EF1	13	20	16	23	16.3	3.5	90	88	104	94.0	8.7
EF2	17	12	12	21	13.7	2.9	73	75	87	78.3	7.6
TC1	15	14	13	23	14.0	1.0	124	144	66	111.3	40.5
LF1	20	17	14	21	17.0	3.0	1415	2614	1095	1708.0	800.8
HC1	3	16	15	21	11.3	7.2	16	200	59	91.7	96.3
IG1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
EM4	8	13	9	19	10.0	2.6	37	100	60	65.7	31.9

FALL

Station	Number of Taxa						Density				
	Left	Middle	Right	Total	Mean	SD	Left	Middle	Right	Mean	SD
CG6	1	1	1	1	1.0	0	40	58	44	47.3	9.5
AR1	14	11	16	20	13.7	2.5166	133	186	133	150.7	30.6
AR2	15	10	10	19	11.7	2.8868	199	70	108	125.7	66.3
AR3	19	12	15	24	15.3	3.5119	294	58	145	165.7	119.3
AR4	19	19	17	23	18.3	1.1547	201	182	202	195.0	11.3
AR5	15	15	12	20	14.0	1.7321	189	246	116	183.7	65.2
EF1	19	13	12	23	14.7	3.7859	187	191	153	177.0	20.9
EF2	12	13	10	15	11.7	1.5275	217	169	117	167.7	50.0
TC1	10	11	10	16	10.3	0.5774	135	114	103	117.3	16.3
LF1	14	11	14	15	13.0	1.7321	195	223	133	183.7	46.1
HC1	10	9	14	17	11.0	2.6458	61	71	100	77.3	20.3
IG1	7	8	13	15	9.3	3.2146	130	105	92	109.0	19.3
EM4	15	13	9	23	12.3	3.0551	57	83	197	112.3	74.5

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Figure 3-9

Benthic Macroinvertebrate Mean Number of Taxa by Location

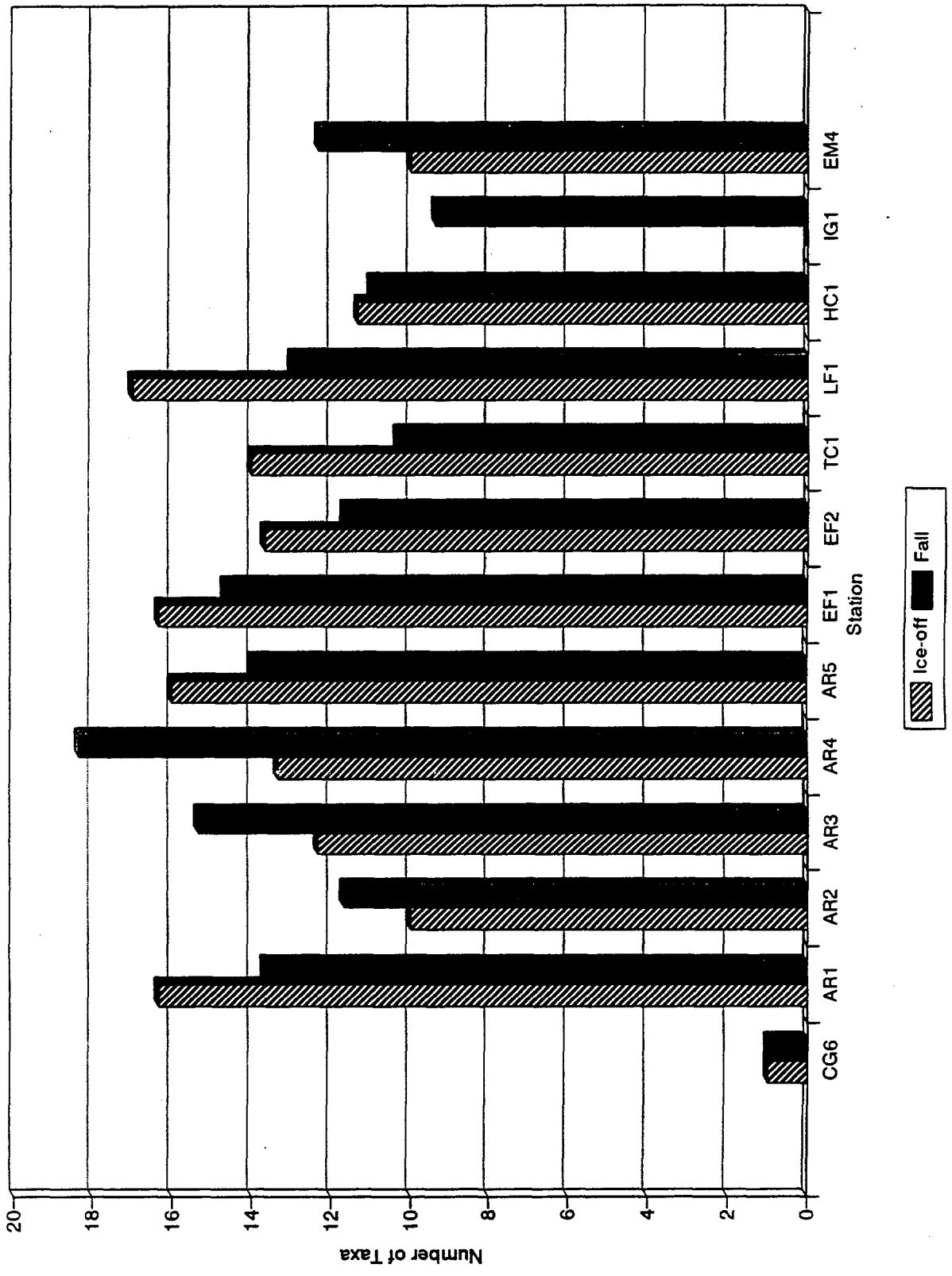

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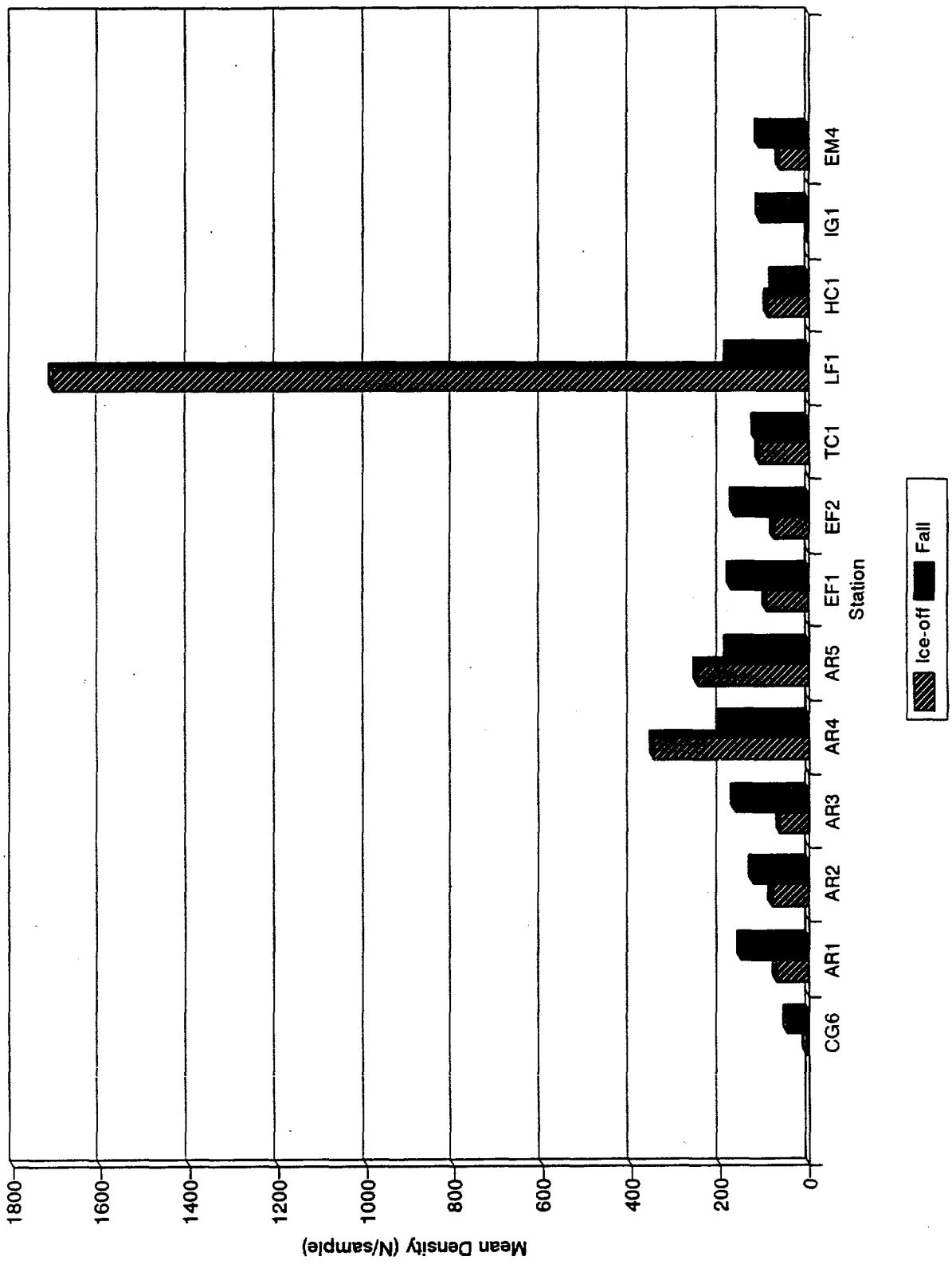
Figure 3-10

Benthic Macroinvertebrate Average Density by Location



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along with other indices that reflect community health, including the Rapid Bioassessment Protocol (RBP III) results. The RBP III procedure weighs the results of various indicators of community health, and relates the results to reference areas. The Community Loss Index measures loss of taxa between reference areas and on-site areas. Values of the Community Loss Index increase with the degree of dissimilarity between reference and treated sites. The Community Loss Index was highest at AR3 for both the ice-off and the fall sampling events. Results of the RBP III assessment indicated that AR3 was moderately impaired, AR4 was slightly impaired, and AR5 was not impaired relative to AR1. The fluctuations in community structure are also evident in Figures 3-11 and 3-12, which show the relative abundance of the dominant taxa. CG6 is obviously different than the other stations.

The California Gulch (CG6) sampling location was markedly different for percent composition, number of taxa, and diversity. The organisms present in the CG6 sample may have originated upstream as suggested by CDOW (1990) and Aquatics Associates (1991). If so, this indicates that California Gulch is devoid of invertebrate life. The benthic invertebrate community in the Arkansas River downstream of the confluence of California Gulch also was impacted as indicated by reduced number of taxa and diversity. The right side of the Arkansas River receives a turbid plume from California Gulch that probably affects the benthic community. IG1 was dry during the ice-off and was not sampled.

ANOVA indicated significant differences in number of taxa by location ($p < 0.05$), but not by season or stream position. Duncan's Multiple Range Test indicated that CG6 had statistically significantly fewer taxa than AR1 at the 95% confidence level. AR1 is upgradient of California Gulch, and CG6 is the station on California Gulch nearest the Arkansas River. Density of invertebrates was also significant by location, but not by season or stream position. Density of aquatic invertebrates was lower at all stations in the Arkansas River, and all other stations examined, as compared to LF1, which is a reference stream originating off-site.

The Kruskal-Wallis test indicated density was significantly different between stations, and that density at CG6 was lower than at AR1. Density was statistically significantly higher at AR4 and AR5 than at AR1, AR2, and AR3. The Kruskal-Wallis test for differences in number of taxa

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Figure 3-11

Benthic Macroinvertebrate Abundance by Location Ice-Off

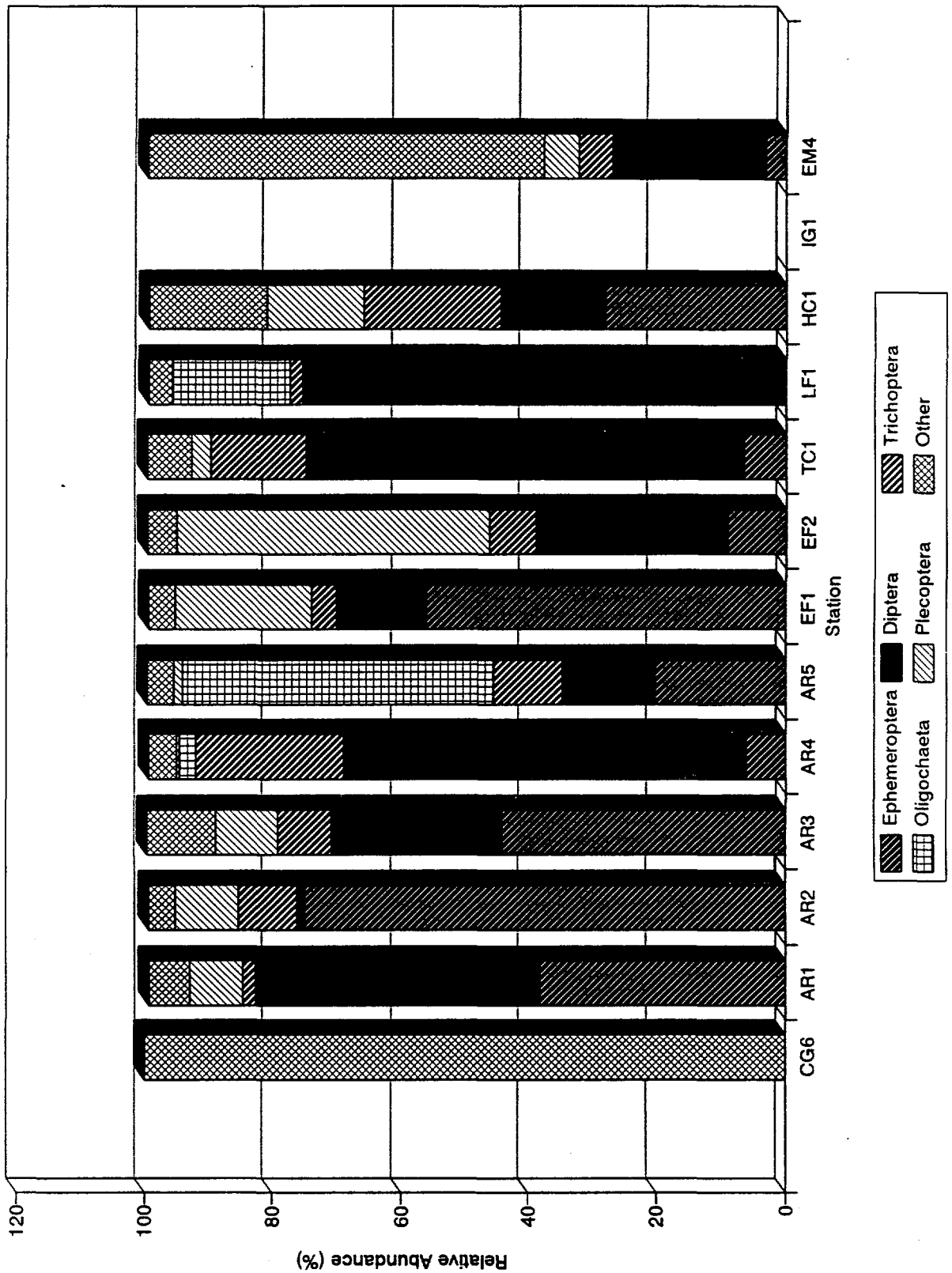

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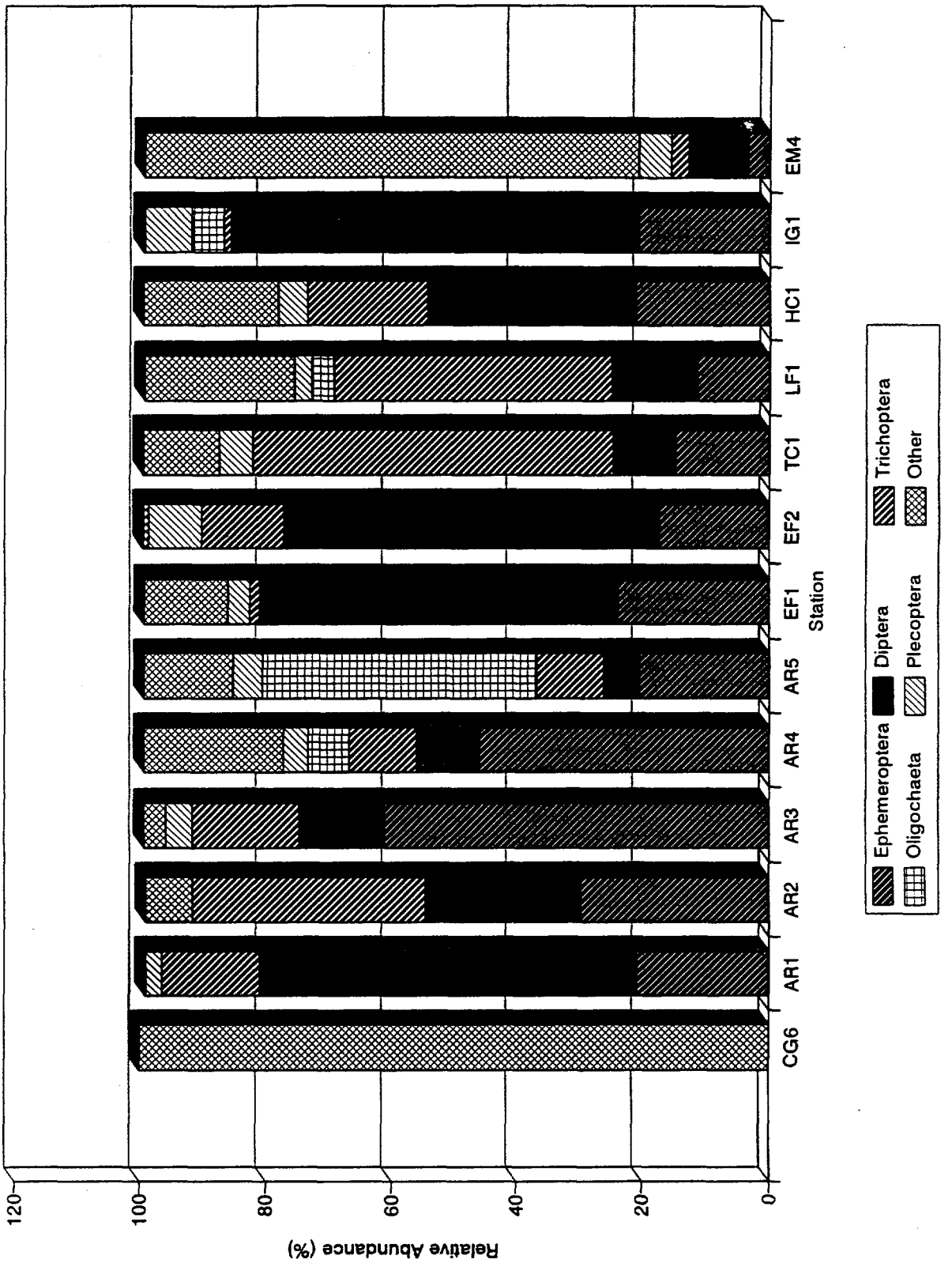
Figure 3-12

Benthic Macroinvertebrate Abundance by Location Fall



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was also significant, and CG6 had fewer taxa than AR1. Number of taxa was similar at AR1, AR4 and AR5, although the rank for number of taxa at AR3 was lower than at the other Arkansas River stations. These results indicate CG6 is the only station that is statistically significantly affected with respect to the benthic invertebrate density and number of taxa based on ASARCO's data.

Even when the data for both seasons are combined, there are only six samples at each station (two samples from each stream position); these sample sizes are unlikely to provide good statistical power (i.e., the ability to detect a difference that exists). A stronger sampling design for both periphyton and benthic invertebrates would include replicate samples at each stream position, in addition to sampling at the three stream positions during different seasons or for multiple years. It is possible that other stations could be affected, but differences are undetectable due to low statistical power and limited sampling design.

3.2.3.3 Fish

Fish data were collected and reported by ASARCO (1993). Fish population data may be affected by the stocking of cutthroat and rainbow trout by CDOW during 1991. Metal concentrations observed in fish (Section 2) may also be influenced by this, as fish sampled may not have been in the ecosystem very long.

Average body weight, body length, density or number of individuals per hectare (N/ha), and biomass or kilograms of fish per hectare (kg/ha) are summarized from the Aquatics Associates data. Length–weight ratios were determined and used with the population estimates to predict biomass and fish condition or well-being. Fish were sampled in August 1991 during low-flow (ASARCO, 1993) in areas representative of the stream reach that included, where possible, a complete habitat sequence (pool-run-riffle). Two of the fish collected from each station were analyzed for metal concentrations in fillets (see Section 2). Salmonids (trout) were the only species collected from the upper Arkansas River in 1991 (ASARCO, 1993).

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Data for the Arkansas River indicated trends in variation for body weight and length (Figure 3-13). Average body weight was lowest at station AR3. Average fish length was higher at AR1, AR2, AR4, and AR5 than at AR3 (Figure 3-13).

The density (N/ha) decreased sharply at AR2 and AR3 as compared to AR1. The number of individuals gradually recovered at AR4 and AR5, although density remained lower than at AR1 or Empire Gulch, East Fork, or Lake Fork (Figure 3-14). The number of fish was also low at Iowa Gulch above the Arkansas River. Too few individuals were collected to allow calculation of biomass or density at AR2, AR3, or IG1, in part due to access limitations at AR2 and AR3. Density (N/ha) for AR4 and AR5 was lower than AR1 (Figure 3-14). A similar pattern is seen for biomass (Figure 3-15). When the data are evaluated for all sites for all trout species combined, AR4 and AR5 appear different than AR1 as well as other tributaries. AR4 and AR5 also had lower biomass than the other locations.

The fish data were also examined by species and size class collected (Figures 3-16 and 3-17). Nearly all trout collected at AR1 were brown trout (90%). The majority were >15 cm in length. The brown trout are self-sustaining, whereas the rainbow and cutthroat trout are stocked (ASARCO, 1993). Age classes ranged from young of the year to 4-year-old trout. Brown trout were also the dominant species collected at AR4 and AR5 (85 and 87%, respectively). Although nearly all age classes were collected, the population was skewed toward the older age classes (ASARCO, 1993).

Fish population data for relative abundance (number of individual fish per species) and number of species were examined for possible contaminant-related effects. As indicated above, the number of individuals decreased at AR2, AR3, and IG1 (Aquatics Associates, 1991). At AR2 and AR3, only brown trout were observed. Upgradient at AR1, cutthroat trout were also observed. Downgradient at AR4 and AR5, brook and rainbow trout were observed in addition to brown trout. In Lake Fork and East Fork, all four trout species were observed. Both brook and brown trout were observed at Empire Gulch. While habitat variables can have a significant impact on fish populations, it appears that most areas upgradient of California Gulch, or streams

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Figure 3-13

Weight and Length Data for Trout

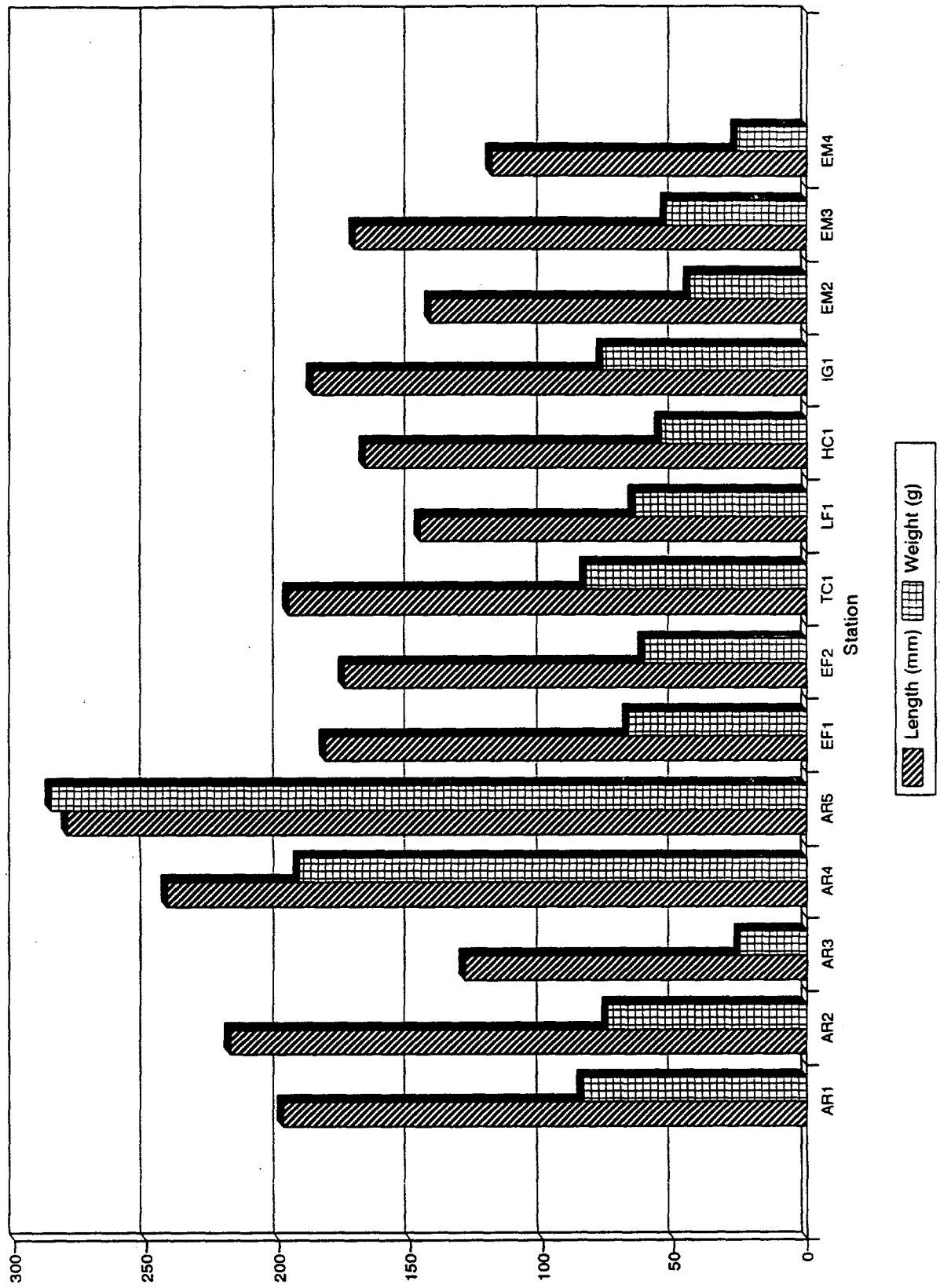

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Figure 3-14

Density of Trout at All Locations

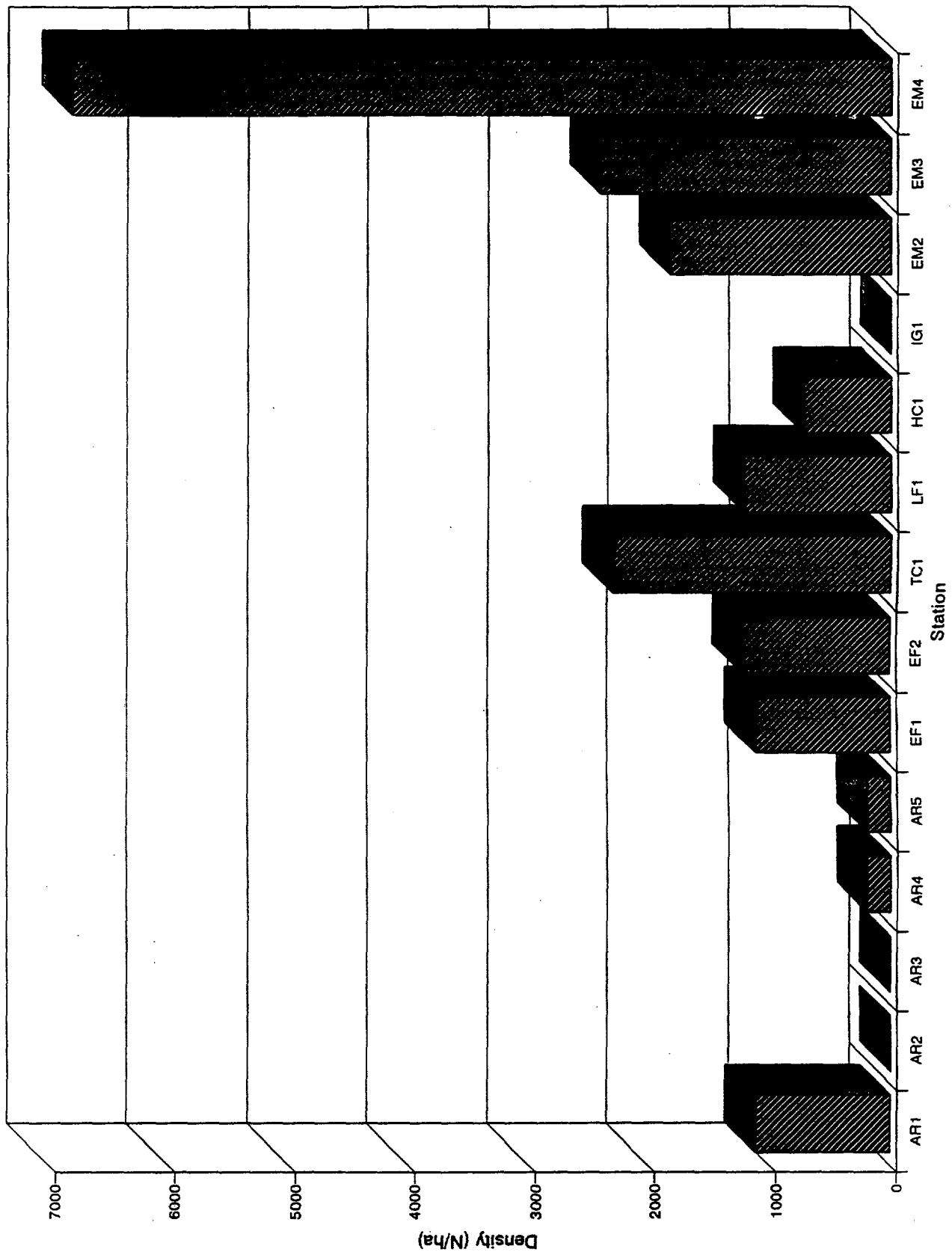

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Figure 3-15

Biomass of Trout at All Locations

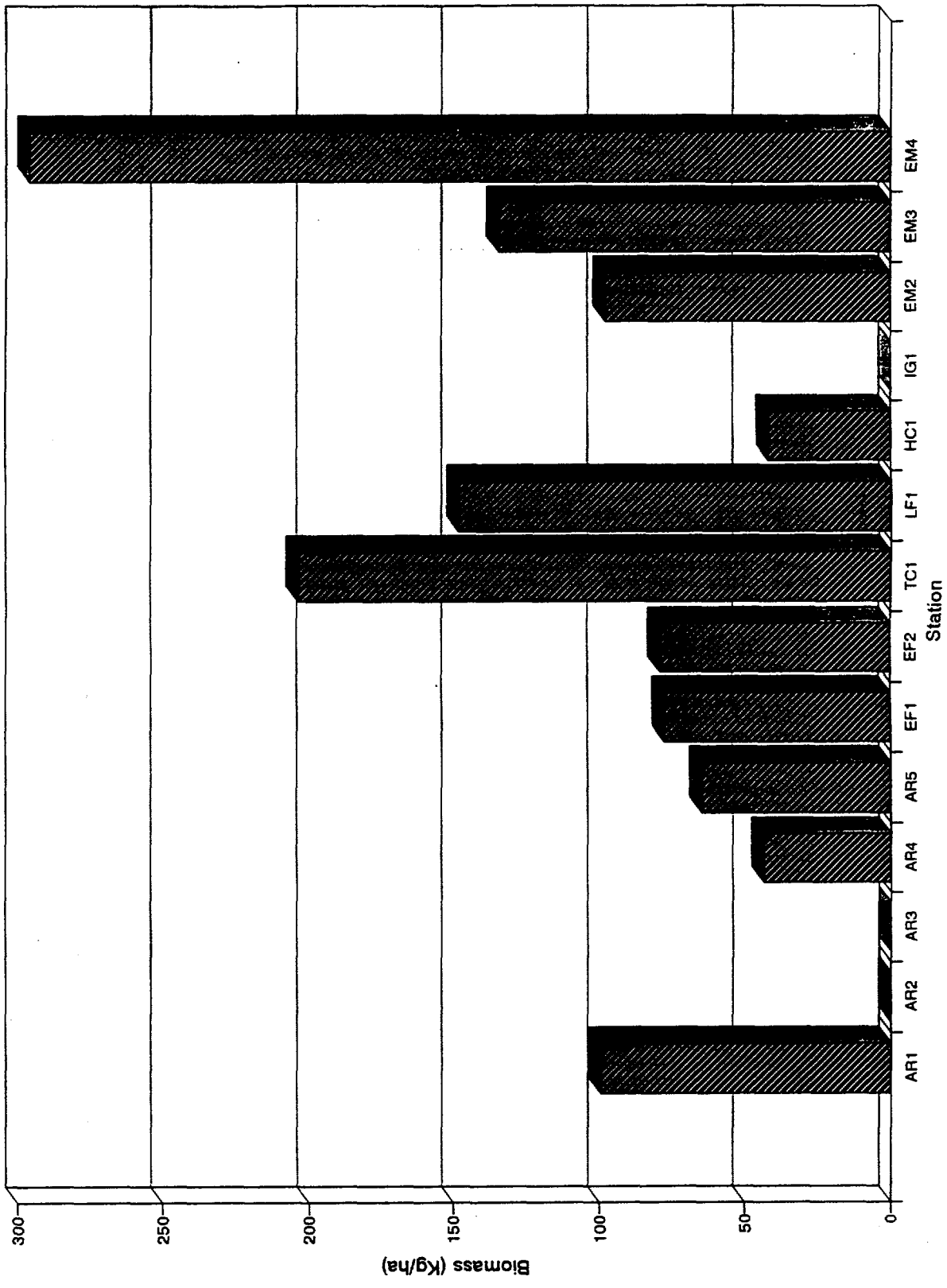

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Figure 3-16

Trout Species Density by Size Class

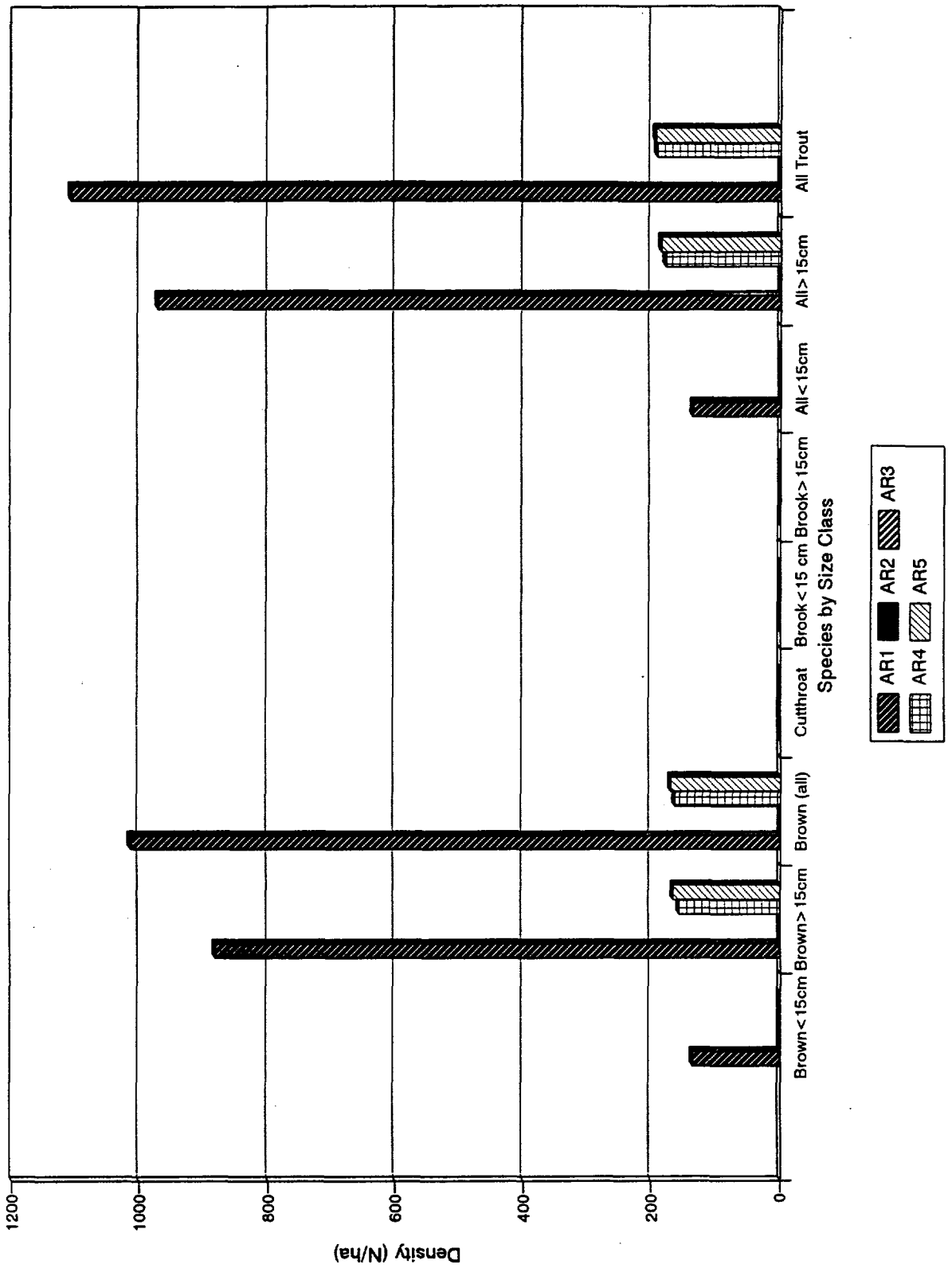

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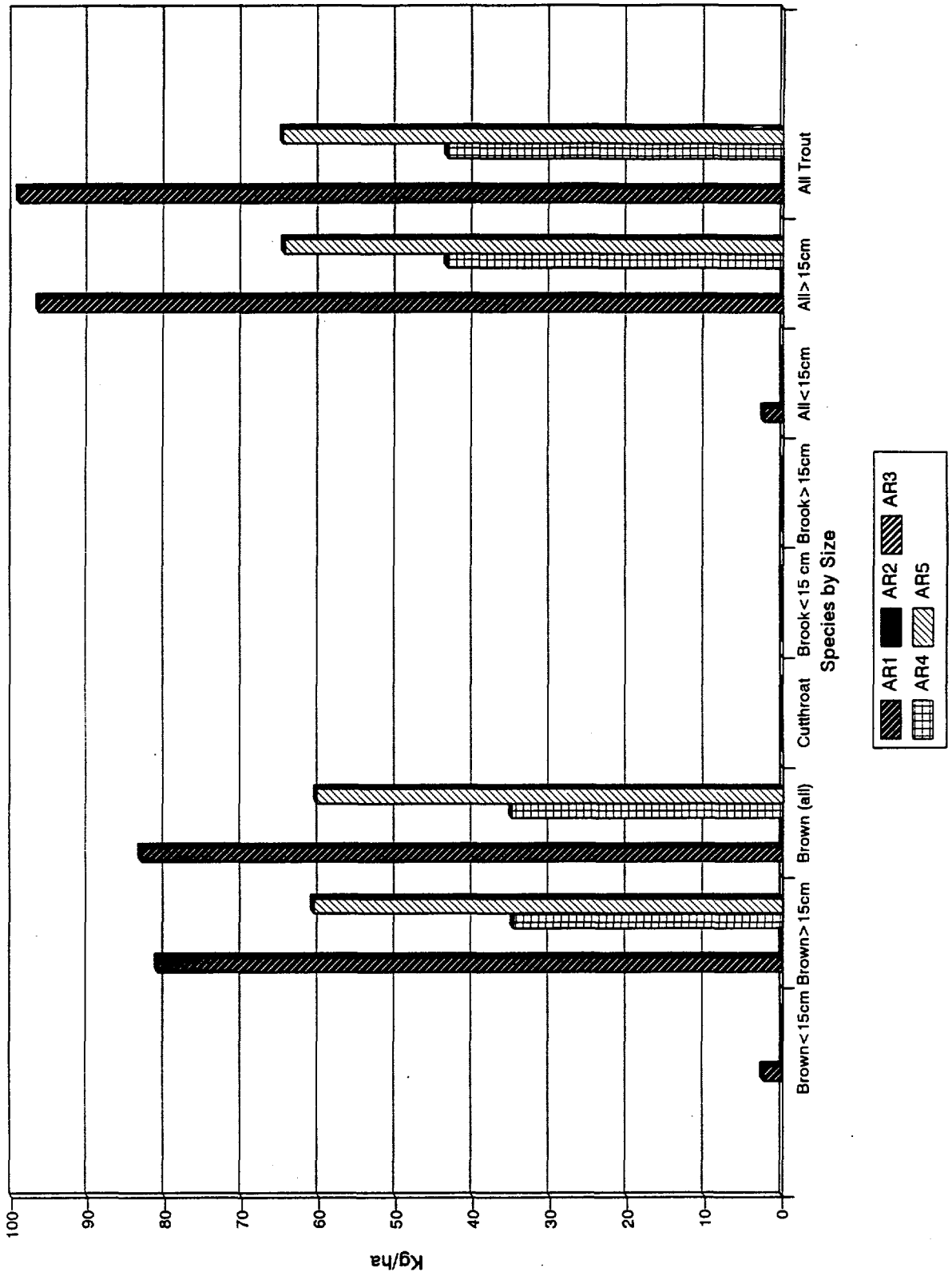
Figure 3-17

Trout Species Biomass by Size Class



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that do not flow across the site (i.e., East Fork, Lake Fork, AR1, Empire Gulch) have both greater numbers and more species present.

3.2.4 Data Collected by Researchers at Colorado State University

Some taxa are notably more sensitive to water quality and metal-related disturbances than others (Clements, 1992). Macroinvertebrate communities in Virginia exposed to copper indicated that reference streams were dominated by *Ephemeroptera* and *Diptera (Tanytarsini chironomids),* whereas contaminated streams were dominated by *Trichoptera (Hydropsychidae)* and *Diptera (Orthocladiini chironomids)* (Clements et al., 1988). Both taxa and numbers were decreased by contaminants. At the California Gulch Site, ASARCO (1993) data indicated that *Orthocladiini chironomids* were more abundant at metal-enriched locations, and *Tanytarsini chironomids* were less abundant, which is consistent with Clements et al. (1988).

Data were collected for benthic invertebrates on the Arkansas River (Clements, in press; Clements and Kiffney, 1994; Kiffney and Clements, 1993). These data and the results of these studies are summarized in Table 3-5. Clements and Kiffney (1994) concluded that there were elevated metal concentrations in benthic invertebrates and altered community structure downgradient of the California Gulch inflow to the Arkansas River. At AR5, the community was impacted by metals. There was also a strong seasonal component to metal concentrations in water. Kiffney and Clements (1993) concluded that metal concentrations were elevated in the benthic community at AR3, AR5, and AR8 relative to AR1. Concentrations of cadmium, copper, and zinc were higher in spring.

Clements (in press) concluded that benthic invertebrate communities at reference areas were dominated by *Ephemeroptera* (mayflies), whereas at the contaminated areas *Orthocladiini chironomids* and *Trichoptera* (caddisflies) dominated the community. Invertebrate communities in the Arkansas River were influenced by metal tolerance of the taxa, environmental conditions, food availability, and ability to recolonize. Clements (in press) observed that the greatest effects on the benthic community in terms of composition and abundance of sensitive taxa were in the

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Table 3-5
Summary of Studies Conducted by Colorado State University Researchers

Reference	Study Endpoints	Results	Conclusions
Kiffney and Clements, 1993	Concentrations of cadmium, copper, and zinc in biota and water.	Metals in water higher in spring. Metals in biota higher at AR3 and AR5 than AR1 or EF1 ($p<0.05$).	Metal concentrations in biota are a better indicator of metal bioavailability than metal concentrations in water
Clements and Kiffney, 1994	Metal concentrations (cadmium, copper, and zinc) in water and biota, chronic laboratory toxicity tests, and benthic community structure.	Laboratory toxicity tests indicated reduced water quality at AR1, AR5, and AR8 relative to upstream reference area. Metals were elevated in periphyton and invertebrates from AR5 and AR8. Community structure altered with station and season; mayflies were reduced at AR5 in fall but recovered at AR8. *Orthocladiini chironomids* dominated at AR5. In spring, mayflies reduced at AR8 as well as AR5, and *Orthocladiini chironomids* dominated at AR8 and AR5.	An integrated assessment provides a more complete analysis of adverse effects than consideration of any endpoint alone. Sensitive species were replaced by metal tolerant ones, making species richness a poor indicator for identifying contaminant-related effects.
Clements, in press.	Community structure.	Polluted sites dominated by *Orthocladiini chironomids* and caddisflies. Reference sites dominated by mayflies.	Greatest effects on community composition observed when metal concentrations in water were the highest. Evaluation of effects due to water quality must also consider natural sources of variation.

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spring, when concentrations of metals were highest. Some of the changes in community structure were dependent more on longitudinal changes (i.e., downstream gradient from headwaters) as predicted by the River Continuum Concept (Vanote et al., 1980), thus making the interpretation of ecological data from upgradient reference areas compared to contaminated areas downstream more difficult (Clements, in press).

3.2.5 Data Collected by the Colorado Division of Wildlife and the USFWS

Woodling (CDOW, 1990) collected three samples at each sampling location, and limited his survey to riffles. Stream width and depth were measured and reported for each station, and riparian vegetation as well as substrate siltation was evaluated. Descriptions of each location were provided. In 1989, California Gulch was devoid of macroinvertebrate life except for taxa that may have originated in the sewage treatment plant. No fish were present. Ferric hydroxide deposits cemented the substrate, riparian vegetation was sparse, and at certain places slag lined the bank. Tailings, waste rock, and slag were located along the gulch where Highway 24 intersects the stream. In the lower reaches, California Gulch was narrower and deeper, and the habitat consisted of riffle-run. Substrates were gravel and cobble up to 8 inches in diameter. Cementing was less, although ferric hydroxides and silt coated the substrate. Dense riparian vegetation leaned over the stream, and the area was grazed.

Woodling applied the Rapid Bioassessment Protocol (Plafkin et al., 1989) to the data. For the Arkansas River, unimpacted areas averaged 13 taxa (range 7-17), and density was 49 organisms/sq. foot (range 26-67). There were a total of 22 taxa, 14 of which were considered "sensitive." At downgradient, impacted areas, abundance was higher, and 25 taxa were identified. However, the species composition changed slightly due to an increase in chironomids. *Orthocladius frigidus* made up 42% of the total organisms collected. The spring data differed from the fall data in that more individuals and more taxa were collected. The number of taxa were decreased at downstream locations as compared to locations upstream of California Gulch. The eight species of chironomids collected made up more than 60% of the total collected. The fauna 0.5 miles downstream was similar to that in the mixing zone. The conclusion that Woodling drew from the Rapid Bioassessment Protocol was that

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macroinvertebrate communities downstream of the confluence of the Arkansas River and California Gulch were moderately impacted compared to unimpacted areas.

Woodling (CDOW, 1990) investigated fish populations in the Arkansas River during the spring and fall of 1990. The fall data determined an estimate of 1,005 brown trout/acre for areas upstream of the California Gulch confluence with the Arkansas River. There were also three brook and one cutthroat trout collected. The biomass was 122.9 pound brown trout/acre. In the mixing zone with California Gulch, brown, brook, and rainbow trout were collected. Population estimates of 263 brown trout/acre and 32 brook trout/acre were made. The number of rainbow trout collected was too low to allow population estimates. The biomass was 22.7 brown trout/acre and 3.6 pound brook trout/acre. Total trout biomass was 26.3 pounds/acre. Fish were therefore 3.4 times more abundant upstream than in the mixing zone. Spring data differed from fall in that fewer brown trout were collected at the location upgradient of California Gulch; however, more trout were collected upstream than in the mixing zone. There were 759 brown trout/acre, and a biomass of 152 pounds/acre upstream, whereas in the mixing zone there were 153 brown trout/acre, and a biomass of 37.6 pounds/acre. At 0.5 miles downstream from the confluence of the Arkansas River and California Gulch, there were even fewer fish. Only 49 brown trout/acre were collected, and biomass was only 22 pounds/acre. Woodling concluded that for the multiple seasons, fish were 4.9 times more abundant upstream than in the mixing zone, and 23 times more abundant than 0.5 miles downstream.

The U.S. Fish and Wildlife compiled a summary of information on the Upper Arkansas River Basin (USFWS, 1993). This report documents the many factors that can influence the survival of healthy trout populations, including ambient metal concentrations in water, biomass of benthic invertebrates, presence of forage fish to serve as prey for larger trout, and physical factors relating to aquatic habitat. The data indicate poor survival of brown trout in excess of 14 cm or 4 years of age. The data contained within this report were collected prior to initiation of water treatment at the Yak Tunnel.

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3.3 CONCLUSIONS

Nearly all of the biological data were collected prior to the operation of water treatment facilities at the Yak Tunnel. The results may therefore not reflect current conditions. Only the EPA acute laboratory toxicity tests were performed with water collected following remediation. Historically, aquatic life were adversely affected by the inflow of water from California Gulch to the Arkansas River.

Water collected from the site was acutely lethal to laboratory species. The results of the laboratory bioassays indicate that adverse effects due to site-related contamination could be evident in the field in the form of reduced populations of aquatic biota. Data collected by ASARCO, CDOW, and researchers at Colorado State University indicated reduced populations of aquatic life and altered community structures for periphyton, invertebrates and fish in the Arkansas River in the vicinity of California Gulch (AR3). Aquatic life in downstream areas (AR4, AR5) also appears affected, but to a lesser extent. Alterations in the aquatic community are evident as far downstream as AR8 during certain seasons.

There were statistically significant differences observed between sampling locations for benthic invertebrates and periphyton populations for the ASARCO 1993 data. The interpretation of the results is complicated by the strong seasonal variation in metal concentrations. Stream position as well as season affects aquatic communities; however, in order to perform statistical analyses on the data, the samples at each stream position had to be considered as replicate samples for each station. In fact, there is a sample size of one at each stream position at each station, considering that the populations vary with season and stream position. Therefore, important differences could have been missed due to the lack of replicates. That statistical differences were observed given the limited nature of the data indicate a strong difference between locations with respect to measurement endpoints of density and abundance.

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SECTION 4
TOXICITY ASSESSMENT

The toxicity of the COCs at California Gulch was reviewed for aquatic and terrestrial species; the results of this review are reported in Appendix A and are arranged by analyte in alphabetical order. A brief summary of the toxicity of the COCs follows here. A summary of bioconcentration factors, and toxicity values such as Lowest Observed Adverse Effect Levels (LOAELS), No Effect Levels (NELS), and No Observed Adverse Effect Levels (NOAELS) is included.

4.1 SUMMARY OF ADVERSE EFFECTS

A number of metals are essential in small amounts for animal nutrition. These include, but are not limited to, chromium, copper, iron, magnesium, manganese, and zinc. Animals have developed a variety of homeostatic mechanisms for metabolism of the essential metals, so they are less likely to produce toxic effects than are the nonessential elements such as barium, cadmium, and lead. Nevertheless, physiological control mechanisms can be overwhelmed or circumvented and essential elements can produce toxic effects when exposure occurs to concentrations greater than the optimum levels.

The toxicity of many elements is influenced by the chemical speciation in which they occur. Concentrations in diet and water are often the most important, although soil ingestion may be a significant route of exposure for some animals. Because potentially toxic trace elements occur in many different chemical forms that have varying toxicity, total concentration of some elements in the exposure media may not be a good predictor of toxicity. This section attempts to identify both highly toxic and environmentally predominant forms in abiotic media in order to provide insight into potential health risks.

The toxicity of inorganic elements varies so widely that it is not practical to list specific concentrations that cause adverse effects under conditions of acute or chronic exposure for all species. The Toxicity Assessment identified potentially toxic levels and adverse health effects

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for species similar to those that may be found at California Gulch. When information was unavailable for closely related species, data for distantly related species in the same class were cited. Lack of toxicity information increases uncertainty in assessing ecological risk. General toxicity information is summarized below.

Both antimony and arsenic can have multiple valance states and are metalloids. In general, inorganic arsenic compounds are more toxic than organic compounds. Organic arsenicals are used as feed additives in agriculture, and their biological fate and toxicity differs from the inorganic forms. Trivalent inorganic arsenicals (arsenites) are often more toxic than pentavalent compounds (arsenates). Arsenic is a teratogen and carcinogen that may cause death and malformations in mammals. Arsenic is toxic to aquatic life as well as terrestrial species. Arsenic does not tend to bioaccumulate or biomagnify as it is readily metabolized and excreted. Antimony has not been studied to the extent that arsenic has. Antimony has been reported to produce toxicity in several species of aquatic life.

Cadmium is nonessential, but it may be stimulatory to some species at very low doses. Cadmium is bioaccumulative and is highly toxic at relatively low levels to most species. Cadmium is not controlled by homeostasis. Cadmium is retained in tissues, and body burdens may increase with age and exposure duration. Freshwater biota tend to be sensitive to cadmium.

Silver is highly toxic to aquatic life. There is relatively little toxicity information for terrestrial wildlife exposed to silver.

Mercury is a nonessential element for animals. Mercury occurs in inorganic and organic forms. Organic mercury is more bioavailable and more toxic than inorganic forms. Inorganic mercury is methylated in biotic and abiotic media. Like cadmium, mercury is bioaccumulative and retained in tissues.

There is no evidence that aluminum is an essential element in animal nutrition. Intestinal absorption is generally very poor, and toxicity is low in comparison to many other metals.

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Lead is a nonessential metal. Lead compounds are readily absorbed from the digestive tract. Organic lead compounds are more toxic than inorganic salts because they are more readily bioavailable. Organic lead compounds have greater lipid solubility, higher stability in biological fluids, and greater assimilation into target tissues such as central nervous system and brain than inorganic lead. However, inorganic species are the primary forms of lead in the California Gulch environment. Inorganic lead may be toxic to aquatic and terrestrial biota.

Magnesium and calcium are essential elements for animals. Barium is stimulatory but not essential. All three are considered relatively nontoxic at physiological concentrations, but barium is toxic at higher levels. Homeostatic mechanisms maintain normal levels of magnesium and calcium, and to a certain extent barium.

Zinc is an essential trace metal that is relatively nontoxic because of efficient homeostatic mechanisms that maintain a proper balance within the body. High concentrations in water may adversely affect aquatic life.

Copper is an essential trace metal that stimulates growth when moderately high levels are fed to mammals, but it is highly toxic to aquatic organisms. Copper is a metal that is likely to be present at potentially adverse levels, at least in surface water, at the Site.

Chromium is an essential trace metal for mammals. Chromium is toxic at high doses, and certain chemical species are highly toxic. Hexavalent chromium is the most biologically active form, although little data are available for the toxicological properties of organic chromium compounds, water-soluble chemical species, or interactions of different chromium compounds in complex mixtures. Hexavalent chromium is chemically reduced in the acid fluid of the mammalian stomach. Chromium concentrations are usually highest at the lowest trophic levels, which may be in more direct contact with the abiotic source media. Biomagnification has not been observed in food chains.

Selenium is also an essential trace element that is toxic at greater than optimum doses. Selenium is bioconcentrated to some extent by both aquatic and terrestrial species. Plants can

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concentrate selenium to levels that are toxic to mammals. Selenium is toxic to aquatic and terrestrial life, and has been indicated as a teratogen in waterfowl.

Manganese salts appear to be among the least toxic of the essential metals. An efficient homeostatic mechanism prevents manganese from accumulating in tissues. Toxic or adverse effects due to exposure to manganese are not common.

Iron is an essential metal. It is generally not considered to be toxic to animals; however, high doses may be toxic to mammals. Little information is available for aquatic species regarding toxicity of iron.

4.2 SUMMARY OF LOAELS AND NOAELS

The LOAEL or the highest NOAEL reported for each inorganic is listed in Table 4-1. These toxicity values represent those concentrations assumed to be protective of the ecological receptors at California Gulch. When data were available, toxicity values for wildlife species likely to be found at California Gulch were used. However, in most cases, toxicity values are for laboratory animals or wildlife species commonly used in toxicity testing (i.e., quail).

Where possible, data from short-term studies (i.e., single dose or less than at least a week) and dose levels or dietary intakes that resulted in mortality were avoided. Many values were reported as dietary concentrations (i.e., mg/kg diet). Dietary concentrations were converted to intakes (mg/kg body weight/day) using dietary ingestion rates. These toxicity values were used in the risk characterization as the denominator of the hazard quotient. The conversion factors applied to each toxicity value were described in Table 4-1.

4.3 SUMMARY OF BIOCONCENTRATION FACTORS

Bioconcentration factors (the ratio of the tissue concentration compared to the water concentration) were compiled from the literature summarized in Appendix A. These bioconcentration factors (BCF) are summarized in Table 4-2. The BCFs are on a wet-weight

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Table 4-1

Toxicity Values for Birds and Mammals

Analyte	Toxicity Benchmark Value (mg/kg bw/day)	Test Species	Ecological Receptor	Effect	Source
Arsenic	14	Mallard	Passerine Wader Waterfowl Raptor	NOAEL @ 100 ppm in diet for behavior (LOAEL was 300 ppm for behavior and growth). Converted with 0.14 kg diet/kg bw from Camardese et al., 1990.	Camardese et al., 1990 Whitworth et al., 1991
	3.8	Rat	Small omnivore Small herbivore Large omnivore	NOAEL (LOAEL was 22.5 mg/kg bw/day for growth, liver lesions).	Schroeder et al., 1968
	2.0	Grazers	Large herbivore	Maximum tolerated in diet 50 ppm, dwb (convert with 0.04 kg diet/kg bw from Sax, 1984).	Bodek et al., 1988
Barium	1,000	Chicken	Passerine Wader Waterfowl Raptor	NOAEL	Johnson et al., 1988
	5.1	Rat	Small omnivore Small herbivore Large omnivore Large herbivore	NOAEL	Perry et al., 1983
Cadmium	0.2	Mallard	Passerine Wader Waterfowl Raptor	NOAEL for adults; kidney lesions in ducklings. UF of 10 applied.	Cain et al., 1983 White et al., 1978
	2.5	Rat	Small omnivore Small herbivore Large omnivore	NOAEL for behavior, condition, body weight, food consumption (30 ppm).	Groten et al., 1991
	0.002	Grazers	Large herbivore	Maximum tolerated 0.05 ppm dwb (converted with 0.04 kg diet/kg bw, [Sax, 1984]).	Bodek et al., 1988

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Table 4-1 (Continued)

Toxicity Values for Birds and Mammals

Analyte	Toxicity Benchmark Value (mg/kg bw/day)	Test Species	Ecological Receptor	Effect	Source
Copper	No bird values				
	3.6	Rat	Small omnivore Small herbivore Large omnivore	LOAEL for decreased hemoglobin and increased SGOT was 36 mg/kg bw/day for 49 day exposure. UF of 10 applied.	Suttle and Mills, 1966
	0.4	Sheep	Large herbivore	Toxic. Maximum chronic intake tolerated for grazers is 25-300 ppm in diet, dwb. Daily intake calculated with 0.04 kg diet/kg bw for cow (Sax, 1984).	Doherty et al., 1969 Bodek et al., 1988
Iron	No bird values				
	100	Rat	Small omnivore Small herbivore Large omnivore Large herbivore	Estimated from dose response curve with minimum lethal 5.3 and maximum lethal 500, most likely lethal dose 256 mg/kg. UF of 2.5 applied.	Venugopal and Luckey, 1978
Lead	14.5	Kestrel	Passerine Wader Waterfowl Raptor	NOAEL (for survival, histopathology, organ weight, and reproduction) from diet of 50 ppm converted with 0.11 kg diet/kg bw for raptors (Section 2).	Franson et al., 1983 Pattee, 1984
	3.0	Mice and rats	Small omnivore Small herbivore Large omnivore	LOAEL of 25 mg/kg diet lead salts. Caused impaired reproduction. Converted with 0.12 kg diet/kg bw (Sax, 1984). UF of 10 applied.	Venugopal and Luckey, 1978
	1.2	Grazer	Large herbivore	Maximum tolerated in diet 30 ppm, dwb. Convert with 0.04 kg diet/kg bw (Sax, 1984).	Bodek et al., 1988

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Table 4-1 (Continued)

Toxicity Values for Birds and Mammals

Analyte	Toxicity Benchmark Value (mg/kg bw/day)	Test Species	Ecological Receptor	Effect	Source
Manganese	No bird values				
	140	Rat	Small omnivore Small herbivore Large omnivore	290 is a NOAEL for mortality for chronic exposure; 930 mg/kg/day the LOAEL for mortality. Levels from 140 to 600 mg/kg/day cause biochemical and behavior changes.	Hejtmancik et al., 1987a,b
	80	Grazer	Large herbivore	Maximum chronic tolerated dietary level is 400 - 2000 ppm dwb (converted with 0.04 kg diet/kg bw (Sax, 1984))	Bodek et al., 1988
Mercury	0.018	Chicken, starling	Passerine Wader Waterfowl Raptor	Reproductive effects, kidney lesions. UF of 10 applied.	Cogburn et al., 1973 Nicholson and Osborn, 1984
	0.18	Mouse	Small herbivore Small omnivore Large herbivore	Renal tumors. UF of 10 applied.	Mitsumori et al., 1981
	0.75	Mink	Large omnivore	NOAEL	Aulerich et al., 1974
Selenium	0.088	Chicken	Passerine Wader Waterfowl Raptor	Reproductive effects. UF of 10 applied.	Ort and Latshaw, 1978
	0.08	Grazer	Small herbivore Small omnivore Large herbivore Large omnivore	Maximum tolerated in diet is 2 ppm dwb (convert with 0.04 kg diet/kg bw (Sax, 1984)). Alkali disease occurs at 0.12 mg/kg bw/day.	Bodek et al., 1988 EPA, 1984d

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Table 4-1 (Continued)

Toxicity Values for Birds and Mammals

Analyte	Toxicity Benchmark Value (mg/kg bw/day)	Test Species	Ecological Receptor	Effect	Source
Silver	65	Rat	Small herbivore Small omnivore	NOAEL	Walker, 1971
	52	Pig	Large herbivore Large omnivore	NOAEL	Van Vleet, 1976
Zinc	No bird values				
	75	Rat, pig	Small herbivore Small omnivore Large omnivore	NOAEL	Sutton and Nelson, 1937 Lewis et al., 1957
	40	Grazer	Large herbivore	Maximum tolerated in diet 300-1000 ppm, dwb.	Bodek et al., 1988

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Table 4-2

Bioconcentration of Inorganics by Aquatic Organisms

Analyte	Taxon	Tissue	Duration (days)	BCF	Studies	Reference
Antimony						
SbO_3	Fish	WB	28	<1	1	EPA, 1980a
Barium	Plants	Vegetative	—	0.15 DWB 0.03 WWB	1	Baes et al., 1984
Cadmium						
$CdSO_4$	Invertebrates	WB	2-8	184-320	2	EPA, 1980b
$CdCl_2$	Plants	Whole	364	1200	1	EPA, 1980b
	Plants	Whole	—	600-1200	—	EPA, 1986
	Fish	WB	—	3-2213	—	EPA, 1980b
	Invertebrates	WB	28-364	373-4190	6	EPA, 1986
	Invertebrates	WB	—	164-4190	—	EPA, 1980b
	Fish	WB	70-140	33-540	2	EPA, 1980b
	Fish	Muscle	93-490	3-10	2	EPA, 1986
$CdNO_3$	Plants	Whole	21	603	1	EPA, 1980b
Nickel						
	Algae	Cells	6	9.8	1	EPA, 1980c
	Invertebrates	WB	2-4	100	1	EPA, 1980c
$NiCl_2$	Fish	WB	30	61	1	EPA, 1980c
Silver						
	Plants	Vegetative	—	0.4 DWB 0.08 WWB	1	Baes et al., 1984
	Invertebrates	WB	3-10	15-240 @ 31-65 mg/ℓ $CaCO_3$	5	EPA, 1980d
	Fish	WB	28	<1	1	EPA, 1980d

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Table 4-2 (Continued)

Bioconcentration of Inorganics by Aquatic Organisms

Analyte	Taxon	Tissue	Duration (days)	BCF	Studies	Reference
Lead						
	Algae	—	28	26000-92000	1	Vighi, 1981
	Invertebrates	—	28	1000-9000	1	Demayo et al., 1982
	Fish	WB	7	726	1	Wong et al., 1981
	Fish	WB	*in situ*	45	1	EPA, 1980f
$PbNO_3$	Invertebrates	WB	28	499-1120	2	EPA, 1980f
	Fish	WB	140	42	1	EPA, 1980f
Manganese						
	Plants	Vegetative	—	0.25 DWB 0.05 WWB	1	Baes et al., 1984
Chromium						
VI	Fish	Muscle	22	<1	1	EPA, 1980g
	Fish	WB	30	1	1	EPA, 1980g
	Fish	—	—	1-3	—	EPA, 1984a
Zinc						
$ZnSO_4$	Invertebrates	WB	14	107-1130 @ 30-70 mg/ℓ $CaCO_3$	2	EPA, 1980h
	Fish	WB	80	51 @ 12-24 mg/ℓ $CaCO_3$ 30-70 mg/ℓ	1	EPA, 1980h
	Plants	Vegetative	—	15 DWB 0.03 WWB	1	Baes et al., 1984

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Table 4-2 (Continued)

Bioconcentration of Inorganics by Aquatic Organisms

Analyte	Taxon	Tissue	Duration (days)	BCF	Studies	Reference
Copper						
	Algae	—	<1	2000	1	EPA, 1980i
	Invertebrates	—	14	203	1	EPA, 1980i
	Fish	—	30-660	0-290	2	EPA, 1980i
Selenium						
	Fish	WB	48-351	8-78	4	EPA, 1980j
	Fish	Muscle	48	15	1	EPA, 1980j
	Fish	Various	lifetime	180-5691	1	Kaiser et al., 1979
Mercury						
MeHg	Fish	WB	273-756	12000-63000	3	EPA, 1980k
	Fish	WB	30	36000	1	Boudou and Ribeyre, 1984
Hg	Fish	WB	30	7000	1	Boudou and Ribeyre, 1984
Hg	Invertebrates	WB	7	2500	1	EPA, 1984b
Hg	Plants (algae)	—	0.29	29000	1	EPA, 1984b
Arsenic						
III	Invertebrates	WB	21-28	0-10	3	EPA, 1984c
V	Fish	WB	28-30	0-3	2	EPA, 1984c
Total	Plants			97	1	Wagemann et al., 1978

WB - whole body
DWB - dry weight basis; WWB - wet weight basis

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or fresh-weight basis. The BCF values were considered only to determine which inorganics were most likely to accumulate in fish or invertebrate tissues. Site-specific data were used in the Exposure Assessment.

4.4 AMBIENT WATER QUALITY CRITERIA

The EPA Ambient Water Quality Criteria for the Protection of Freshwater Aquatic Life and Their Uses were used as toxicity values in the Risk Characterization (Table 4-3). The State of Colorado Standards were lower than the EPA AWQC for chromium, copper, cyanide, nickel, and zinc. Acute criteria (also termed Criterion Maximum Concentrations or CMC) are for a 1-hour exposure duration. Chronic criteria (also termed Criterion Continuous Concentrations or CCC) are for a 4-day exposure duration.

Toxicity to aquatic life are more closely linked to dissolved metal concentrations in water than total metal concentrations. The EPA AWQC are based on exposure to total metals in water. The EPA AWQC were adjusted to reflect dissolved metal concentrations instead of total concentrations by multiplying by recommended factors (EPA, 1993a). This decreased the criterion concentrations, but allowed the criteria to be compared to dissolved metal concentrations in water from the site, which also tend to be lower than total metals concentrations. The correction factors applied and example adjusted criteria are reported in Table 4-4. Correction factors were unavailable for selenium.

Although the present arsenic criteria may be adequate for some of the less toxic forms of arsenic in water, data from studies by Paladino (1976) indicated 100% mortality of larval fish exposed to As (III) at concentrations of 50 $\mu g/\ell$. Volatilization of potentially toxic methyl arsines from water may occur; thus, the present criteria may not be protective from these more toxic forms of arsenic (AFS, 1979).

The criteria that are hardness dependent are reported by station in Table 4-5. The water hardness criteria vary by station because the concentrations of calcium and magnesium are different. Use of the hardness equations allows for site-specific criterion concentrations.

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Table 4-3

EPA Ambient Water Quality Criteria for the Protection of Freshwater Aquatic Life and Their Uses

Analyte	EPA Acute (μg/ℓ)	EPA Chronic (μg/ℓ)	Colorado Acute[a] (μg/ℓ)	Colorado Chronic[b] (μg/ℓ)
Aluminum	NA	NA	750	87
Antimony	NA	NA	NA	NA
Arsenic (total)[c]	NA	NA	360	150
Arsenic (III)	360	190	NA	NA
Arsenic (V)	NA	NA	NA	NA
Barium	NA[d]	NA	NA	NA
Beryllium	NA	NA	NA	NA
Cadmium[e]	3.9	1.1	3.9[f]	1.1
Chromium (total)	NA	NA	NA	NA
Chromium (III)[e]	1700	210	1700	210
Chromium (VI)	16	11	16	11
Copper[e]	18	12	18	12
Cyanide	22	5.2	5	NA
Iron	NA	NA	NA	1000[g]
Lead[e]	82	3.2	95.8	3.9
Manganese	NA	NA	NA	1000
Mercury	2.4	0.012	2.4	0.1
Nickel[e]	1400	160	922.2	95.6
Selenium	20	5	135	17
Silver	4.1	NA	2.0	0.08[f]
Thallium	NA	NA	NA	15
Zinc[e]	120	110	120	110

Source: EPA, 1992; AFS, 1979; Water Quality Commission, 1991[h].

[a] The chronic standard is not to be exceeded by the concentration for either a single representative sample or calculated as an average of all samples collected during a 30 day period.

[b] The acute standard is not to be exceeded by the concentration in a single sample or calculated as an average of all samples collected during a 1 day period.

[c] The recommended criterion of 50 μg/ℓ for domestic water was not considered to be satisfactorily safe for the protection of aquatic life (AFS, 1979). Refer to text.

[d] The 1000 μg/ℓ criterion for drinking water may be adequate for the protection of aquatic life (AFS, 1979). Exposure duration (acute or chronic) was not indicated.

[e] Criteria are hardness dependent. Values in table reflect hardness of 100 mg/ℓ $CaCO_3$.

[f] Value specifically designated for trout species.

[g] Total recoverable value, the proportion of a water and suspended sediment sample measured by the total recoverable procedure described in "Methods for chemical analysis of water and wastes (EPA, 1979).

[h] The water quality criteria are stated as dissolved unless otherwise specified; dissolved value is the proportion of a water and suspended sediment sample which passed through a 0.40 or 0.45 um filter.

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Table 4-4

EPA Ambient Water Quality Criteria (CFR 12/22/92; EPA, 1993[a,b])

Element	CMC Acute (μg/ℓ)	Correction Factor	Dissolved Acute (μg/ℓ)	CCC Chronic (μg/ℓ)	Correction Factor	Dissolved Chronic (μg/ℓ)
Arsenic	360.00	0.95	342.00	190.00	0.95	180.50
Cadmium	1.79	0.85	1.53	0.66	0.85	0.56
Copper	9.22	0.85	7,84	6.54	0.85	5.56
Lead	33.78	0.50	16.89	1.32	0.25	0.33
Nickel	789.00	0.85	670.65	87.71	0.85	74.56
Selenium	20.00	NA	NA	5.00	NA	NA
Silver	1.23	0.85	1.05	NA	NA	NA
Zinc	65.04	0.85	55.29	58.91	0.85	50.08

Note: Representative hardness 50 mg/ℓ for cadmium, copper, lead, nickel, silver, and zinc.

NA = Not available

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Table 4-5

Ambient Water Quality Criteria

Hardness Dependent Criteria - 1991 Data

Station	Average AWQC (ug/l)										
	Hardness	Cd acute	Cd chronic	Cu acute	Cu chronic	Pb acute	Pb chronic	Ni acute	Ni chronic	Zn acute	Zn chronic
AG01	9.11	0.22	0.15	1.58	1.30	1.93	0.04	158.83	17.66	13.06	11.83
AR01	76.04	2.45	0.78	11.64	7.95	28.80	0.56	956.11	106.29	78.86	71.43
AR02	81.62	2.65	0.82	12.44	8.45	31.52	0.61	1015.19	112.86	83.74	75.85
AR03	82.06	2.67	0.83	12.51	8.49	31.74	0.62	1019.81	113.37	84.12	76.20
AR03A	92.42	3.05	0.91	13.99	9.40	36.92	0.72	1127.71	125.37	93.04	84.27
AR03E	103.21	3.45	0.99	15.52	10.33	42.50	0.83	1238.21	137.65	102.17	92.54
AR03W	87.42	2.86	0.87	13.27	8.96	34.40	0.67	1075.93	119.61	88.76	80.40
AR04	149.33	5.24	1.32	21.98	14.16	68.01	1.33	1692.35	188.14	139.71	126.54
AR05	81.43	2.64	0.82	12.42	8.43	31.43	0.61	1013.24	112.64	83.58	75.70
CG01	156.32	5.52	1.37	22.95	14.72	72.09	1.40	1759.20	195.57	145.24	131.55
CG02	686.64	29.29	4.38	92.55	52.14	474.30	9.24	6152.35	683.95	508.91	460.95
CG03	608.38	25.56	3.98	82.58	47.02	406.59	7.92	5553.68	617.40	459.32	416.03
CG04	738.97	31.82	4.64	99.18	55.52	520.79	10.15	6546.81	727.80	541.60	490.55
CG05	586.30	24.51	3.87	79.75	45.55	387.90	7.56	5382.67	598.39	445.16	403.20
CG06	351.70	13.77	2.59	49.27	29.44	202.39	3.94	3493.28	388.35	288.71	261.50
EF01	81.50	2.65	0.82	12.42	8.44	31.46	0.61	1013.89	112.71	83.64	75.75
EF02	130.38	4.50	1.19	19.34	12.61	57.22	1.11	1508.78	167.73	124.53	112.80
EG01	91.10	3.00	0.90	13.80	9.28	36.25	0.71	1114.09	123.85	91.91	83.25
EG02	75.38	2.42	0.77	11.54	7.89	28.49	0.56	949.10	105.51	78.28	70.90
EM01	66.02	2.09	0.70	10.19	7.05	24.06	0.47	848.43	94.32	69.97	63.37
EM02	94.66	3.13	0.92	14.31	9.59	38.07	0.74	1150.82	127.94	94.95	86.00
EM03	103.80	3.48	0.99	15.61	10.38	42.81	0.83	1244.19	138.32	102.67	92.99
EM04	114.38	3.88	1.07	17.10	11.27	48.44	0.94	1350.65	150.15	111.46	100.96
GG01	14.54	0.38	0.21	2.45	1.93	3.51	0.07	235.93	26.23	19.42	17.59
HC01	34.60	1.01	0.42	5.54	4.06	10.57	0.21	491.18	54.60	40.47	36.66
IG01	316.87	12.24	2.38	44.66	26.93	177.22	3.45	3198.24	355.55	264.29	239.38
LF01	31.51	0.91	0.39	5.08	3.75	9.39	0.18	453.85	50.45	37.39	33.87
MG01	20.28	0.55	0.28	3.35	2.57	5.35	0.10	312.55	34.75	25.74	23.31
OG01	3885.94	206.96	17.07	473.84	229.30	4308.53	83.95	26661.44	2963.94	2210.38	2002.03
PG01	33.20	0.96	0.41	5.33	3.92	10.03	0.20	474.32	52.73	39.08	35.40
SD01	218.44	8.05	1.78	31.46	19.59	110.37	2.15	2334.76	259.55	192.84	174.66
SG01	487.94	19.93	3.35	67.08	38.94	307.04	5.98	4608.18	512.29	381.01	345.10
SPR01	889.41	39.22	5.36	118.10	65.04	659.33	12.85	7657.87	851.32	633.66	573.94
SPR02	1210.05	55.51	6.83	157.84	84.61	975.68	19.01	9936.23	1104.60	822.52	744.99
SPR03	1288.96	59.61	7.18	167.52	89.31	1057.39	20.60	10481.68	1165.24	867.74	785.95
SPR04	241.99	9.03	1.93	34.64	21.39	125.74	2.45	2546.03	283.04	210.32	190.50
SPR05	876.81	38.60	5.30	116.52	64.25	647.47	12.62	7566.01	841.11	626.05	567.04
SPR07	1669.33	79.79	8.79	213.74	111.39	1469.59	28.63	13044.87	1450.19	1080.30	978.48
SPR08	1445.36	67.83	7.85	186.61	98.49	1223.35	23.84	11548.09	1283.79	956.17	866.04
SPR09	740.27	31.89	4.64	99.35	55.60	521.96	10.17	6556.52	728.88	542.40	491.28
SPR10	174.85	6.26	1.50	25.51	16.20	83.14	1.62	1934.00	215.00	159.69	144.64
SPR11	236.86	8.82	1.90	33.95	21.00	122.36	2.38	2500.29	277.96	206.53	187.07
SPR12	137.39	4.77	1.24	20.32	13.18	61.17	1.19	1577.18	175.33	130.19	117.92
SPR13	416.56	16.67	2.96	57.79	34.02	251.05	4.89	4031.06	448.13	333.23	301.82
SPR14	121.06	4.14	1.12	18.04	11.83	52.07	1.01	1417.10	157.54	116.96	105.93
SPR15	87.86	2.88	0.87	13.34	9.00	34.62	0.67	1080.50	120.12	89.14	80.74
SPR16	66.27	2.10	0.70	10.22	7.07	24.18	0.47	851.11	94.62	70.19	63.57
SPR17	125.18	4.29	1.15	18.62	12.18	54.33	1.06	1457.75	162.06	120.32	108.98
TC01-1	27.99	0.79	0.35	4.54	3.39	8.07	0.16	410.50	45.64	33.82	30.63
TC01-2	32.32	0.93	0.40	5.20	3.83	9.69	0.19	463.69	51.55	38.20	34.60
TP01	64.65	2.04	0.68	9.99	6.92	23.43	0.46	833.49	92.66	68.73	62.25
TP02	78.01	2.52	0.79	11.92	8.13	29.76	0.58	977.09	108.62	80.60	73.00
TP03	98.59	3.28	0.95	14.87	9.93	40.09	0.78	1191.12	132.42	98.28	89.02

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Average hardness (as mg/ℓ $CaCO_3$) in surface water at each station was used to calculate criterion values.

4.5 SEDIMENT TOXICITY VALUES

There are no sediment criteria equivalent to the AWQC values. NOAA (1991) suggests Effects Range Low (ERL) and Effects Range Medium (ERM) values (Table 4-6). These are not standards or criteria, but adverse effects are generally not observed at ERL concentrations. These values are intended for marine sediments. Effects may be observed at ERM concentrations.

Other sediment toxicity data are available. A summary of sediment criteria and guidelines was compiled by Bennett and Cubbage (1991). The ranges of criteria (mg/kg, dry weight) reported in this document are:

Arsenic	<3 to 33
Barium	<20 to 500
Cadmium	0.6 to 10
Chromium	<25 to 120
Copper	16 to 110
Iron (%)	1 to 5.9
Lead	31 to 500
Manganese	<300 to 1200
Mercury	0.1 to 0.6
Nickel	16 to 100
Selenium	1 to 2
Silver	0.5
Zinc	<90 to 820

These guidelines and criteria represent a wide variety of ecotoxicological endpoints, including no effects, lowest effect, and severe effect levels. The summary includes disposal criteria as well as guidelines for sediment quality. For the purposes of the risk assessment, the lower end of the range is more appropriate because these values represent no effect or minimal effect levels.

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Table 4-6

NOAA Sediment Values (mg/kg)

Analyte	ERL	ERM
Cadmium	5	9
Copper	70	390
Lead	35	110
Zinc	120	270

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SECTION 5
RISK CHARACTERIZATION

The risk characterization section compares the EPA AWQC from Section 4 to the exposure intakes estimated in Section 2. A hazard quotient (HQ) approach was used, where the exposure point concentration for each metal at each station was divided by the AWQC. When the HQ exceeds 1, risk to aquatic ecological receptors is indicated. This allows identification of areas that are a source of risk to the aquatic ecosystem. It also allows identification of the inorganics that are causing the risk. The HQs were summed for each station to obtain a hazard index, which predicts risk to ecological receptors due to exposure to all inorganics measured at each station.

The areas upgradient of California Gulch were considered reference areas. Hazard indices for the reference areas may also exceed 1 because of natural background concentrations of inorganics, or because disturbances unrelated to the California Gulch site may have elevated the concentrations of inorganics in the aquatic ecosystem. The hazard indices at the reference areas were used as a baseline to define excess ecological risk attributable to the California Gulch site.

The uncertainty inherent in the risk assessment is also described in this section.

5.1 UNCERTAINTY ANALYSIS

The uncertainty analysis qualitatively addresses the primary sources of uncertainty identified in the risk assessment. The likelihood that the uncertainty will result in the under- or overestimation of risk is stated when it is possible to determine or predict this.

5.1.1 Sampling Process and Analytical Data

There is uncertainty in any sampling methodology. Collection of field duplicates and repeated sampling events help to minimize this uncertainty, but there is always the possibility that the samples were not fully representative of site conditions. Sample data were averaged over

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multiple sampling periods in order to more accurately estimate exposure concentrations for long durations. Field duplicates were averaged with the corresponding sample to more adequately represent between-sample variability.

The upper 95th percent confidence interval on the sample mean (UCL95) was used to represent the upper bounds on exposure. The UCL95 was compared to the maximum observed concentration at a station, and the lower of the two values used to represent the RME. The RME value is considered to represent periods of high flow and storm events, where concentrations of metals exceed average concentrations.

There is some error inherent in the analytical measurements performed on any samples. This error includes both instrument error and method error. The quality assurance and quality control procedures followed as part of the EPA Contract Laboratory Program minimize inherent error. In addition, stringent data validation procedures applied to the risk assessment data minimize analytical and reporting errors. Analytical errors may under- or overestimate the potential for risk for any given data point, but are unlikely to bias or affect the results of the risk assessment in either direction.

Many of the data values were below contract reporting limits (CRL). These data were utilized in the quantitative exposure assessment by dividing the CRL in half. The actual concentration in these samples is unknown, and ranges from zero to the CRL. Use of one-half the CRL may thus under- or overestimate any given data value. For data sets with few values below detection, this is unlikely to influence either the mean or the UCL95. However, for data sets where most or all of the values are less than the CRL, the resulting interpretation is likely to be highly uncertain. Locations where the data were less than the CRL are indicated in Appendix B by the "U" qualifier.

Some of the CRL values exceed the EPA AWQC. Mercury CRLs regularly exceed at least the chronic AWQC. The mercury chronic AWQC is 0.012 micrograms per liter ($\mu g/\ell$), whereas the CRL is 0.1 $\mu g/\ell$. For mercury, the CRL was not low enough to detect low concentrations that may be toxic to aquatic life. This makes interpretation of HQs as well as prediction of

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adverse effects difficult since the concentrations are not low enough to be toxicologically relevant. If mercury were analyzed quantitatively (i.e., use of mercury concentrations compared to the EPA AWQC), all nondetects would cause exceedances of the HQ because even one-half of the CRL exceeds the chronic AWQC. For this reason, mercury does not appear on the surface water exposure point concentration tables. Total site risk may be underestimated because mercury exceedances were not included in the hazard index.

The available data are reported as dissolved metals. Speciation of the inorganics is not available, and this introduces uncertainty to the risk assessment. Metal speciation can change rapidly with changing water quality parameters. Because metal speciation may affect bioavailability and toxicity of a given inorganic, it is likely that lack of this information may lead to under- or overestimation of risk for some metals at some stations.

Small sample sizes and uneven sample numbers at each station make the risk analyses more uncertain, although the use of the UCL95 or maximum concentration to represent RME levels encompasses the increased uncertainty. The UCL95 concentrations tend to decrease with increasing sample sizes due to the effect of sample size on estimates of variance; therefore, small sample sizes and large variances result in a wider confidence interval. In addition, use of the CDOW data set and the storm water sampling events to verify that the RME concentrations were conservative enough to ensure adequate protection of aquatic life tends to minimize this uncertainty.

There are often less than four data points at each station for the 1991 data, and each of these data points represents a different sampling event. Thus, there are no replicates for a given event except for the field duplicates. For some stations, the exposure point concentration is represented by a single value. This occurred in cases where the stream was not sampled again because of low flow, and in instances where other data were rejected for analytical or data validation reasons. Sample sizes of one were observed in the 1991 data at all of the springs sampled by Resurrection (SPR locations), as well as at Malta Gulch, Georgia Gulch, Pawnee Gulch, and Airport Gulch. In the 1992 to 1993 data set, sample sizes of one were observed in the springs sampled by Resurrection, as well as at several stations on the Arkansas River and

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California Gulch. Sample size can be determined for any site and data set by referring to Appendix B.

5.1.2 Ecological Receptors

A comprehensive species list was considered during preparation of the risk assessment. However, there is the possibility that some species were overlooked and not considered during the risk assessment. This would underestimate the exposure for these species, and underestimate the potential for risk because these species were not considered.

Some receptors may be more sensitive to the toxic effects of the COCs than others. Lack of toxicological information regarding the inherent toxicity of each of the COCs for each of the ecological receptors makes this uncertainty difficult to quantify. The AWQC are based on toxicity tests with cold-water as well as warm-water species. Trout species are a major component of many of the tests; therefore, the AWQC should be sufficiently conservative to predict risk to the trout species occurring at California Gulch. Many of the EPA criteria are the same as the State of Colorado criteria. Some of the State criteria are specific to trout species, and are likely to be more protective.

5.1.3 Bioavailability

Bioavailability of the contaminants in media is another source of uncertainty when quantifying exposure intakes. Inorganics are influenced by physical properties of the environment, such as the cation exchange capacity of soil sediment or water pH, or water hardness. The AWQC incorporate corrections for water hardness, which helps to minimize this uncertainty. However, contaminants may be more or less bioavailable on a site by site basis due to other reasons, thus resulting in under- or overestimating risk.

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5.1.4 Contaminant Interactions

The Ecological Risk Assessment addresses risk due to exposure to each contaminant individually by use of the HQ approach. However, ecological receptors are continuously exposed to a mixture of contaminants, which is represented by the hazard index. The relative concentrations of contaminants in this mixture may change on an hourly, daily, or seasonal basis. Inorganics may have either an antagonistic, synergistic, or no effect on the toxicity of other inorganics in the mixture. For example, the presence of zinc may ameliorate the toxicity of cadmium at certain concentrations for some species. However, toxicological data are insufficient to model or predict the toxicity of mixtures of inorganics at this time. This may result in either the under- or overestimation of risk to any given receptor.

5.1.5 Site Variability

In addition to any variability that can be introduced as a result of sampling, data analysis, or the risk assessment process, there is the inherent variability of the abiotic and biotic media of the site. The California Gulch site has a large area, with many separate source areas, and many different historical mining activities that resulted in metals contamination of the environment. In addition, hydrologic and physical parameters can affect the results of the risk assessment. For instance, the storm water runoff contains higher metals concentrations, as does the runoff during the spring sampling periods. A risk estimate is made for every station; however, many of the tributaries to the Arkansas River and California Gulch are intermittent. This means that a tributary such as Airport Gulch could present a risk to aquatic life, but most of the year the stream does not flow. Some tributaries that were dry during most sampling events were only sampled once, so there are no data to reflect varying concentrations of metals for different days.

Data collected daily by CDOW (Woodling, 1994) indicate that during the spring, metal concentrations can be over an order of magnitude higher than indicated by maximum observed concentrations from samples collected by ASARCO. In addition, these high concentrations are maintained for several days to over a week, which is sufficient in duration to adversely affect aquatic life. Woodling (1994) suggests that the risk assessment is not conservative enough to

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protect aquatic life in the Arkansas River based on the discrepancies between the data collected by CDOW and by ASARCO.

5.2 RISK TO AQUATIC ECOSYSTEMS

The risk to aquatic ecosystems was evaluated by considering risk to the aquatic receptors (i.e., aquatic plants, invertebrates, and fish). Risk to aquatic receptors was addressed by comparing average and RME dissolved metal concentrations in sediment pore water and surface water to the EPA AWQC. Prior to the comparison, the EPA AWQC were adjusted to reflect dissolved metals concentrations as the criteria (EPA, 1993). This adjustment involved multiplying a factor for the proportion of dissolved metal in a sample by the EPA AWQC, which were based on total metal concentrations (EPA, 1993). Table 5-1 presents the adjusted criteria, and the factors used to perform the adjustment. While this caused the criteria to decrease (i.e., become more stringent), it is technically more defensible from a toxicological perspective. In addition, concentrations of dissolved inorganics tend to be lower than concentrations of total inorganics.

The HQ approach was used to predict risk to aquatic ecosystems. For aquatic life, the exposure point concentration was divided by both the acute and chronic EPA AWQC. When the HQ was less than one, no risk to the aquatic ecosystem was predicted. When the HQ exceeded one, risk to the ecological receptors was indicated. All of the HQs for each station were summed to obtain a hazard index (HI). The assumption behind the HI calculation is that the toxicity due to exposure to the different metals is additive. A weight of evidence approach was also used, which compared the results of the Stress Response Assessment, as well as the HI values, in order to predict a risk to aquatic life at each station.

5.2.1 Ecological Risk Based on 1991 Data

Risk to aquatic life (aquatic plants, invertebrates, and fish) was evaluated for exposure to sediments and surface water dissolved metals for the 1991 data. These data were collected prior to the initiation of water treatment at the Yak Tunnel.

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Table 5-1

EPA Ambient Water Quality Criteria

Element	CMC Acute (ug/l)	Correction factor	Dissolved Acute (ug/l)	CCC Chronic (ug/l)	Correction factor	Dissolved Chronic (ug/l)
Arsenic	360.00	0.95	342.00	190.00	0.95	180.50
Cadmium	3.92	0.85	3.33	1.13	0.85	0.96
Copper	17.73	0.85	15.07	11.82	0.85	10.05
Lead	81.65	0.50	40.82	3.18	0.25	0.80
Nickel	1418.24	0.85	1205.51	157.67	0.85	134.02
Selenium	20.00	NA	NA	5.00	NA	NA
Silver	4.06	0.85	3.45	NA	NA	NA
Zinc	117.02	0.85	99.47	105.99	0.85	90.09

Note: representative hardness 100 mg/l for Cd, Cu, Pb, Ni, Ag, and Zn (CFR 12/22/92; EPA, 1993)

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5.2.1.1 Sediment

Sediment concentrations were converted to pore water concentrations (Medine, 1994) prior to analyzing exposure. Pore water concentrations were estimated by the MINTEQA2 model (Medine, 1994). Average, minimum, and maximum pore water concentrations were predicted. The average and maximum values were used to predict ecological risk.

Average and maximum sediment pore water exposure point concentrations were compared to the acute and chronic EPA AWQC (Tables 5-2 and 5-3). For some of the analytes, exceedances of acute and chronic AWQC occurred at every station. For cadmium and zinc, exceedances of both the acute and chronic AWQC occurred at all stations (Table 5-2 and 5-3). The magnitude of the exceedances was higher at California Gulch stations, particularly CG04, CG05 and CG06. Lead and copper caused exceedances of acute criteria primarily at stations in lower California Gulch (i.e., CG04, CG05, and CG06). Lead produced exceedances of the chronic criteria in the Arkansas River as well as California Gulch.

The season has a strong influence on the concentrations of metals in sediment pore water. This would affect benthic community responses, and may influence the observed effects on populations reported in Section 3.

Figures 5-1 through 5-4 show the hazard indices based on estimated pore water concentrations. Stations in lower California Gulch had the highest exceedances based on both acute and chronic criteria. Differences in the magnitude of the exceedance by season are over an order of magnitude (i.e., CG03), indicating that concentrations of dissolved metals can fluctuate greatly by season. These results indicate that sediments are likely to be toxic to benthic invertebrates, thereby impacting the benthic community and ultimately affecting the prey available for trout.

HQs were calculated for measured dissolved metals in sediment pore water (Table 5-4) (Resurrection, 1994). Most of the HQs were less than one. Cadmium produced an exceedance of the chronic criterion at AR03 and AR05, and of both the acute and chronic criteria at all California Gulch sampling stations. Lead produced an exceedance of the chronic criterion as

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Table 5-2

Stations Where Estimated Average Pore Water Concentrations Exceed the EPA AWQC

		HQs Based on Acute Criteria					HQs Based on Chronic Criteria				
Station	**Date**	**Cadmium**	**Copper**	**Lead**	**Zinc**	**HI**	**Cadmium**	**Copper**	**Lead**	**Zinc**	**HI**
AR03	4/30/91	24.02	2.32	1.22	26.14	53.71	83.33	3.48	62.50	28.86	178.18
AR03A	9/16/91	12.91	0.13	0.24	24.13	37.42	44.79	0.20	12.50	26.64	84.13
AR03A	7/24/91	31.53	0.33	0.24	21.41	53.52	109.38	0.50	12.50	23.64	146.02
AR04	9/16/91	70.87	0.27	2.99	172.92	247.04	245.83	0.40	152.50	190.92	589.65
AR04	4/29/91	231.83	0.20	0.56	186.99	419.59	804.17	0.30	28.75	206.46	1039.68
AR05	9/16/91	31.53	0.07	0.17	37.20	68.97	109.38	0.10	8.75	41.07	159.29
AR05	4/29/91	54.35	0.13	0.20	44.03	98.72	188.54	0.20	10.00	48.62	247.36
CG03	3/24/92	6.01	0.60	0.20	6.89	13.69	20.83	0.90	10.00	7.60	39.33
CG03	9/17/91	89.49	1.33	1.03	227.20	319.05	310.42	1.99	52.50	250.86	615.77
CG03	7/24/91	123.72	0.86	0.49	119.13	244.21	429.17	1.29	25.00	131.54	587.00
CG04	3/24/92	62.16	0.27	0.73	96.32	159.48	215.63	0.40	37.50	106.35	359.87
CG04	7/24/91	113.81	1.00	0.88	163.87	279.56	394.79	1.49	45.00	180.93	622.21
CG05	3/24/92	3.60	0.46	0.71	61.22	66.00	12.50	0.70	36.25	67.60	117.05
CG05	9/17/91	148.95	0.93	3.21	278.48	431.56	516.67	1.39	163.75	307.47	989.28
CG05	7/24/91	298.20	1.46	2.16	715.79	1017.61	1034.38	2.19	110.00	790.32	1936.88
CG06	3/24/92	39.94	3.78	2.82	49.36	95.90	138.54	5.67	143.75	54.50	342.46
CG06	9/17/91	14.41	3.92	0.71	29.15	48.19	50.00	5.87	36.25	32.19	124.31
CG06	7/24/91	148.05	2.06	2.87	322.71	475.68	513.54	3.08	146.25	356.31	1019.19

Note: AWQC based on an assumed hardness of 100 mg/l.

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Table 5-3

Stations Where Estimated Maximum Pore Water Concentrations Exceed the EPA AWQC

		HQs Based on Acute Criteria					HQs Based on Chronic Criteria				
Station	**Date**	**Cadmium**	**Copper**	**Lead**	**Zinc**	**HI**	**Cadmium**	**Copper**	**Lead**	**Zinc**	**HI**
AR03	4/30/91	97.90	9.75	5.39	110.59	223.63	339.58	14.63	275.00	122.10	751.31
AR03A	9/16/91	23.72	0.20	0.49	44.54	68.95	82.29	0.30	25.00	49.17	156.76
AR03A	7/24/91	84.68	0.86	0.64	52.88	139.06	293.75	1.29	32.50	58.39	385.93
AR04	9/16/91	140.54	0.60	6.49	359.91	507.54	487.50	0.90	331.25	397.38	1217.03
AR04	4/29/91	410.21	0.40	1.18	352.87	764.65	1422.92	0.60	60.00	389.61	1873.12
AR05	9/16/91	52.55	0.07	0.29	61.83	114.74	182.29	0.10	15.00	68.27	265.66
AR05	4/29/91	110.51	0.20	0.44	89.57	200.73	383.33	0.30	22.50	98.90	505.03
CG03	3/24/92	12.91	1.19	0.47	14.07	28.65	44.79	1.79	23.75	15.54	85.87
CG03	9/17/91	137.54	2.59	2.28	319.69	462.10	477.08	3.88	116.25	352.98	950.19
CG03	7/24/91	222.52	1.53	1.00	205.69	430.74	771.88	2.29	51.25	227.11	1052.52
CG04	3/24/92	130.63	0.60	1.69	203.48	336.40	453.13	0.90	86.25	224.66	764.93
CG04	7/24/91	226.73	1.86	1.81	309.64	540.04	786.46	2.79	92.50	341.88	1223.62
CG05	3/24/92	81.98	1.13	1.84	144.77	229.71	284.38	1.69	93.75	159.84	539.66
CG05	9/17/91	275.98	1.59	6.10	489.59	773.26	957.29	2.39	311.25	540.57	1811.50
CG05	7/24/91	569.97	2.39	4.09	489.59	1066.04	1977.08	3.58	208.75	540.57	2729.99
CG06	3/24/92	224.32	23.16	16.66	286.52	550.66	778.13	34.73	850.00	316.35	1979.20
CG06	9/17/91	75.98	18.78	3.31	138.74	236.80	263.54	28.16	168.75	153.18	613.63
CG06	7/24/91	602.10	8.63	12.03	1367.25	1990.00	2088.54	12.94	613.75	1509.60	4224.83

Note: AWQC based on an assumed hardness of 100 mg/l.

Figure 5-1

Hazard Indices for Exposure to Average Sediment Pore Water Concentrations - Acute Criteria

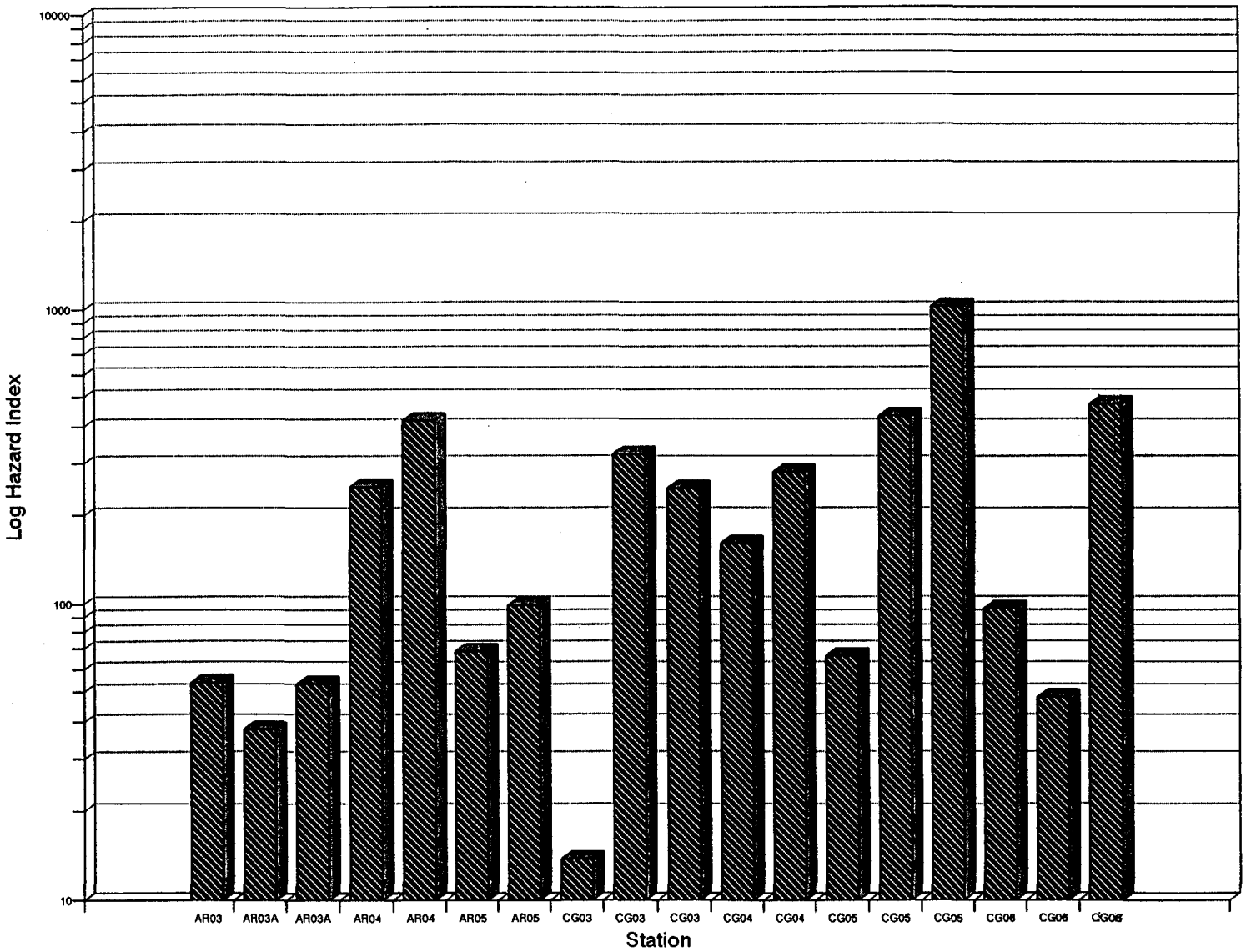

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Figure 5-2

Hazard Indices for Exposure to Average Sediment Pore Water Concentrations - Chronic Criteria

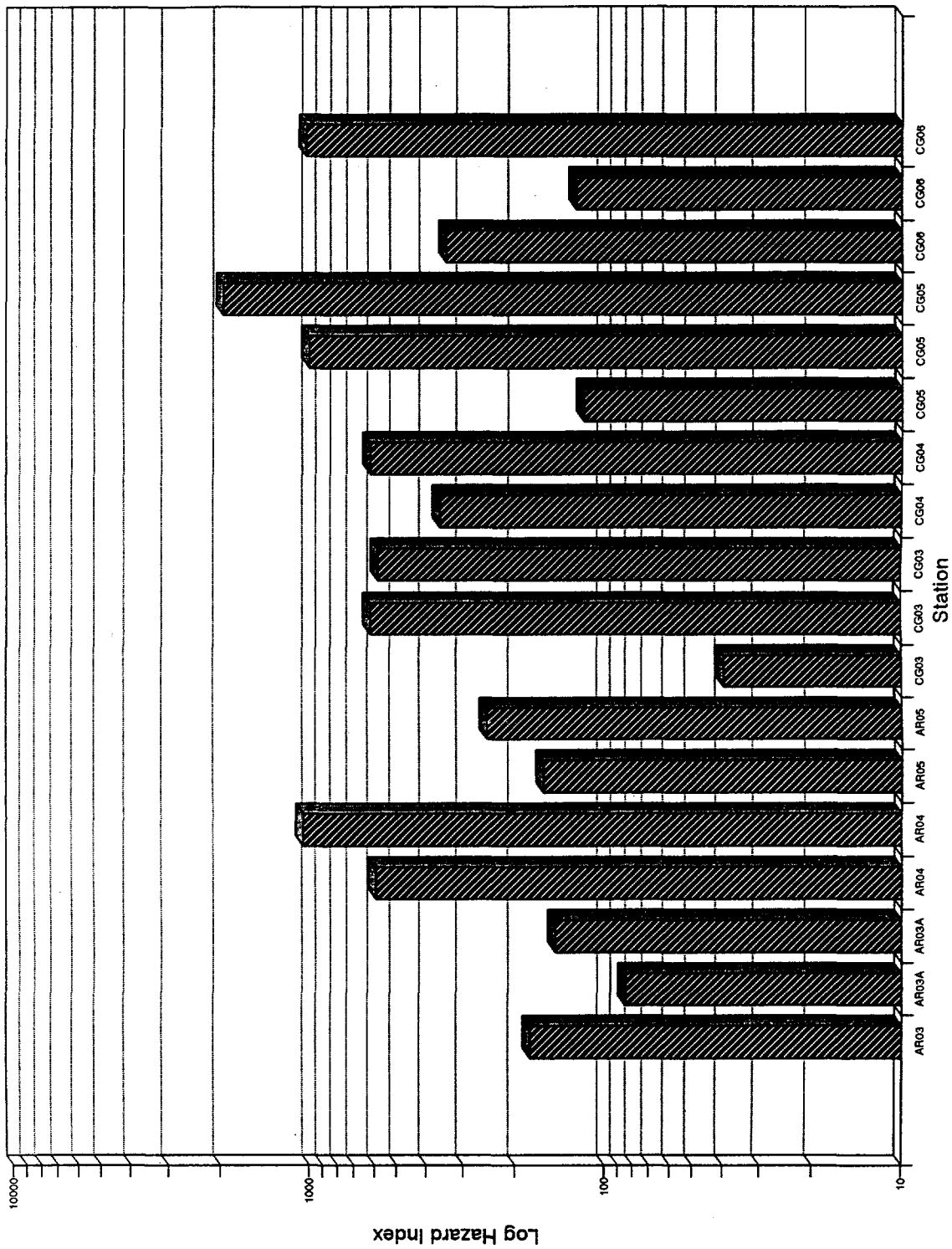

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Figure 5-3

Hazard Indices for Exposure to Maximum Sediment Pore Water Concentrations - Acute Criteria

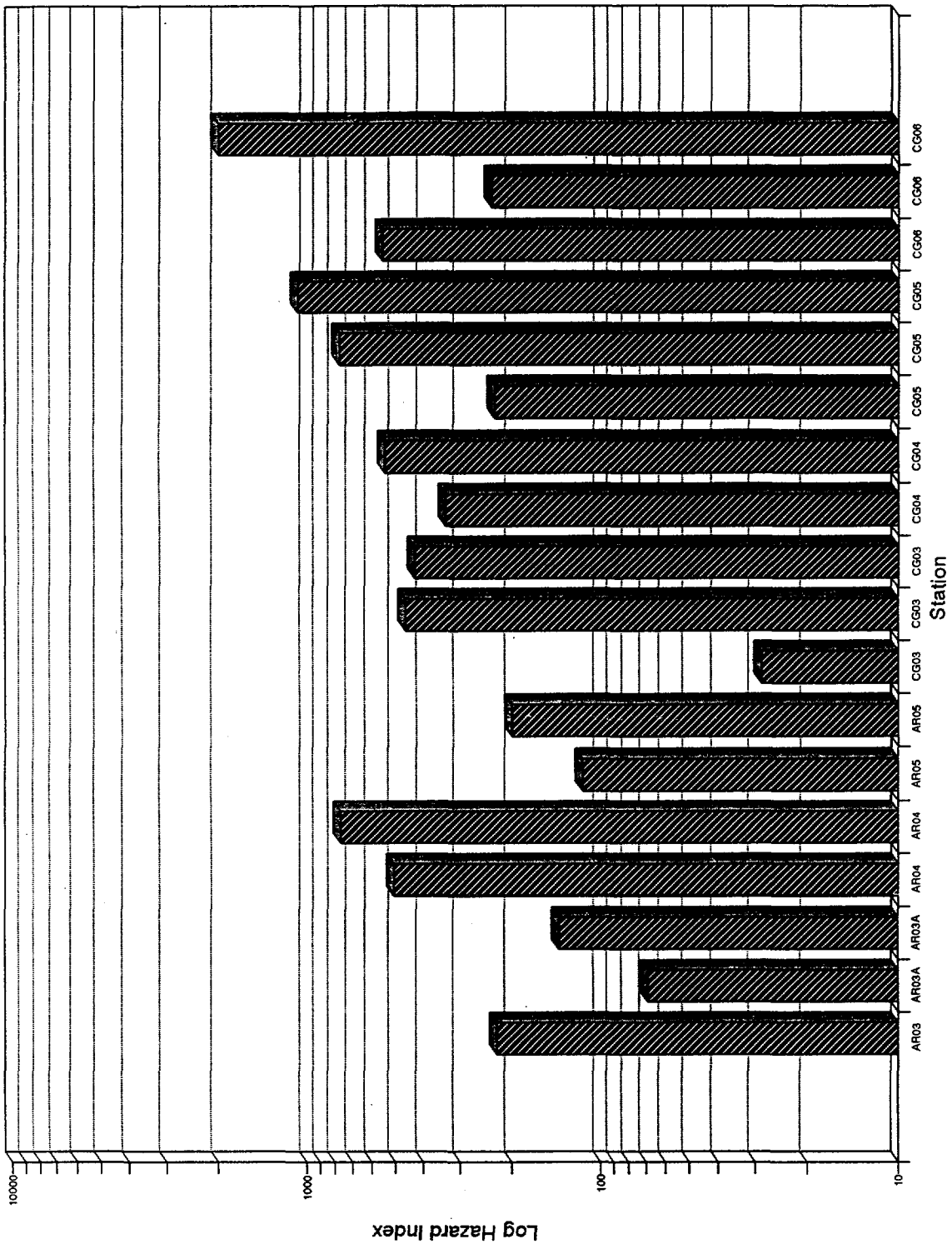

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Figure 5-4

Hazard Indices for Exposure to Maximum Sediment Pore Water Concentrations - Chronic Criteria

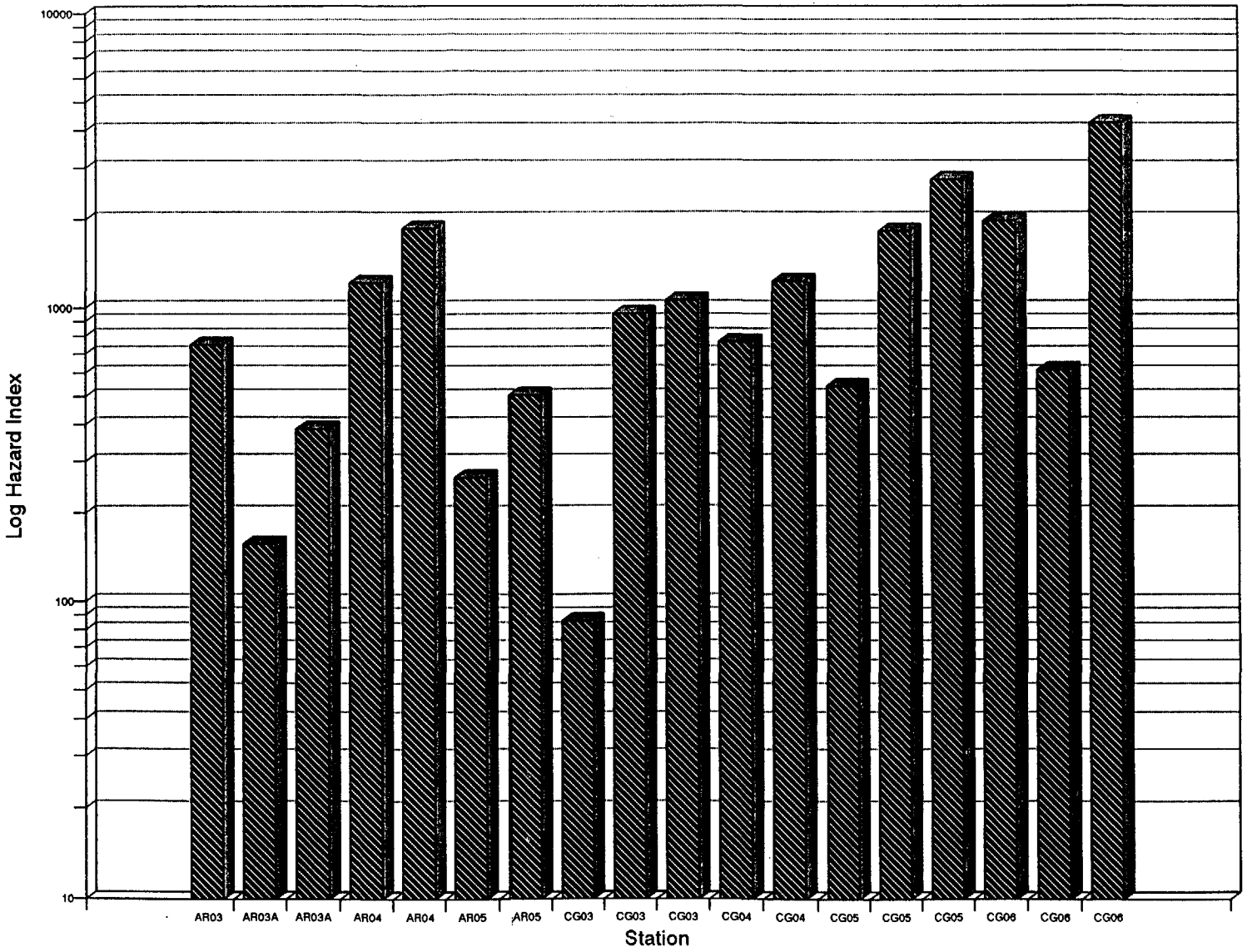

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Table 5-4

Hazard Quotients for the Measured Pore Water Concentrations

Based on EPA Acute and Chronic AWQC

	Arsenic		Cadmium				Copper				Iron	Lead				Zinc			
Station	Acute HQ	Chronic HQ	Acute AWQC	Chroni AWQC	Acute HQ	Chronic HQ	Acute AWQC	Chroni AWQC	Acut HQ	Chronic HQ	Acute HQ	Acute AWQC	Chronic AWQC	Acute HQ	Chronic HQ	Acute AWQC	Chroni AWQC	Acute HQ	Chronic HQ
AR01	NA	NA	2.04	0.68	0.15	0.44	10.0	6.9	0.50	0.72	NA	23.4	0.9	0.04	1.09	70.1	63.5	1.00	1.10
AR02	NA	NA	6.19	1.48	0.06	0.27	25.3	16.1	0.20	0.31	NA	82.1	3.2	0.01	0.16	161.5	146.3	0.43	0.48
AR03	NA	NA	8.95	1.92	0.74	3.44	34.4	21.2	0.15	0.24	NA	124.4	4.8	0.02	0.41	213.0	192.9	9.06	10.00
AR04	NA	NA	5.87	1.43	0.27	1.12	24.2	15.4	0.21	0.32	NA	77.2	3.0	0.01	0.17	155.1	140.5	2.13	2.35
AR05	NA	NA	2.15	0.43	0.56	2.77	10.5	7.2	0.48	0.69	NA	24.9	1.0	0.60	15.46	73.0	66.1	4.24	4.69
CG04	NA	NA	3.33	0.96	9.30	32.15	15.1	10.1	NA	NA	0.14	40.8	1.6	0.24	6.29	101.5	91.9	109.38	120.77
CG05	NA	NA	3.33	0.96	NA	NA	15.1	10.1	NA	NA	NA	40.8	1.6	NA	NA	101.5	91.9	NA	NA
CG06	0.00	0.00	3.33	0.96	10.35	35.78	15.1	10.1	0.33	0.50	0.27	40.8	1.6	NA	NA	101.5	91.9	119.24	131.65
EF02	NA	NA	6.19	1.48	0.10	0.40	25.3	16.1	0.20	0.31	NA	82.1	3.2	0.01	0.31	161.5	146.3	0.62	0.68
LF01	NA	NA	5.81	1.42	0.02	0.07	24.0	15.3	0.21	0.33	NA	76.5	3.0	0.01	0.17	154.1	139.6	0.45	0.50

Note: Hardness dependent AWQC are presented in the table by station. All criteria adjusted to reflect dissolved metal data.
For stations lacking hardness values, an assumed water hardness of 100 was used to obtain the AWQC for hardness dependent metals (Cd, Cu, Pb, and Zn).
No AWQC for Iron (Chronic), Manganese (Acute & Chronic)

Data are from Resurrection, 1994

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AR05 and C604. Lead concentrations at AR01 just barely caused an exceedance of the chronic criteria. Zinc concentrations produced exceedances at AR03, AR04, and AR05, and all California Gulch locations. Copper concentrations do not appear to pose a risk to aquatic life in sediments.

In general, potential risk to aquatic life is greatest in California Gulch. This statement is based upon the modeled and measured sediment pore water concentrations. Risk to benthic communities in the Arkansas River is also predicted by HQs exceeding one at AR03, AR04, and AR05. In contrast, HQs were below one at EF0, LF01, AR01, and AR02, with the exception of lead and zinc at AR01, which were 1.1. Therefore, the California Gulch site continues to elevate metals above background, and above levels acceptable to benthic invertebrate communities. Because trout rely on health benthic communities for a prey base, adverse effects on trout populations may occur.

5.2.1.2 1991 Surface Water Dissolved Metals Data

Average and RME surface water exposure point concentrations were compared to the acute and chronic EPA AWQC (Tables 5-5 and 5-6). The 1991 data are discussed separately from the 1992 and later data in order to determine current risk to aquatic ecosystems following initiation of water treatment at the Yak Tunnel Water Treatment Facility.

Nickel acute and chronic criteria were not exceeded by the average concentrations (Table 5-2). Arsenic criteria were exceeded very infrequently; Stray Horse Gulch was the only station that had an exceedance for arsenic, both the acute and chronic criterion for arsenic were exceeded and this station. Acute and chronic criteria for cadmium and copper were exceeded at Airport Gulch, California Gulch, Malta Gulch, Oregon Gulch, Pawnee Gulch, Strayhorse Gulch, Georgia Gulch, and Starr Ditch. Chronic criteria for cadmium were exceeded in the Arkansas River at AR03, AR04, and AR05 (Table 5-2). Chronic criteria for cadmium were also exceeded at stations not associated with California Gulch; this included AR02, Lake Fork, and Tennessee Creek. Lead acute and chronic criteria were exceeded at Airport Gulch, California Gulch, Malta Gulch, Pawnee Gulch, Strayhorse Gulch, Georgia Gulch, and Starr Ditch. There were

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Table 5-5

Hazard Quotients for Exceedances of the EPA AWQC by Average Surface Water Concentrations - 1991 Data

Station	As		Cd		Cu		Fe		Pb		Ni		Zn	
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
AG01	0.02	0.00	23.27	35.39	24.36	29.60	No AWQ	0.12	28.86	1481.09	0.06	0.57	13.09	14.45
AR01	0.00	0.00	0.12	0.38	0.25	0.37		0.15	0.02	0.89	0.01	0.09	0.28	0.30
AR02	0.00	0.00	0.46	1.49	0.32	0.47		0.15	0.02	1.02	0.01	0.09	0.70	0.78
AR03	0.00	0.00	0.52	1.70	0.42	0.62		0.20	0.02	0.81	0.01	0.09	3.07	3.39
AR03A	0.00	0.00	0.33	1.10	0.10	0.15		0.04	0.01	0.69	0.01	0.08	0.71	0.79
AR03E	0.00	0.00	0.24	0.85	0.21	0.32		0.06	0.01	0.60	0.01	0.07	0.89	0.98
AR03W	0.00	0.00	0.08	0.27	0.24	0.35		0.07	0.01	0.75	0.01	0.08	0.09	0.10
AR04	0.00	0.00	0.26	1.04	0.32	0.49		0.14	0.01	0.51	0.01	0.05	0.62	0.68
AR05	0.02	0.01	0.85	2.74	1.18	1.74		0.12	0.31	15.84	0.01	0.09	1.16	1.28
CG01	0.00	0.00	31.44	126.71	16.21	25.27		1.27	11.68	599.42	0.01	0.10	147.69	163.06
CG02	0.00	0.00	7.17	47.98	1.98	3.51		4.95	0.56	28.52	0.00	0.03	51.09	56.41
CG03	0.00	0.00	4.81	30.86	2.67	4.69		0.70	0.03	1.65	0.00	0.03	72.39	79.93
CG04	0.00	0.00	4.87	33.45	4.65	8.30		28.19	1.25	64.32	0.01	0.08	82.68	91.29
CG05	0.00	0.00	4.87	30.90	2.93	5.14		6.96	0.47	23.98	0.01	0.05	72.32	79.84
CG06	0.00	0.00	3.57	19.01	0.24	0.40		0.01	0.01	0.29	0.00	0.04	39.27	43.36
EF01	0.00	0.00	0.05	0.16	0.13	0.20		0.03	0.02	0.82	0.01	0.09	0.06	0.07
EF02	0.00	0.00	0.10	0.39	0.12	0.18		0.02	0.01	0.45	0.01	0.06	0.08	0.09
EG01	0.00	0.00	0.13	0.44	0.15	0.22		0.01	0.01	0.71	0.01	0.08	0.05	0.06
EG02	0.00	0.00	0.22	0.70	0.35	0.51		0.01	0.02	0.90	0.01	0.09	0.06	0.07
EM01	0.00	0.00	0.02	0.07	0.17	0.24		0.04	0.02	1.07	0.01	0.11	0.07	0.08
EM02	0.00	0.00	0.02	0.07	0.09	0.14		0.05	0.02	1.25	0.01	0.08	0.05	0.06
EM03	0.00	0.00	0.02	0.07	0.11	0.16		0.08	0.01	0.60	0.01	0.07	0.05	0.05
EM04	0.00	0.00	0.11	0.40	0.42	0.63		0.05	0.01	0.53	0.01	0.07	0.04	0.05
GG01	0.03	0.00	5.02	8.96	3.96	5.01		0.06	4.42	226.82	0.04	0.38	1.75	1.93
HC01	0.00	0.00	0.07	0.16	1.34	1.84		0.03	0.05	2.43	0.02	0.18	0.12	0.14
IG01	0.00	0.00	0.01	0.06	0.07	0.11		0.02	0.01	0.36	0.00	0.03	0.02	0.02
LF01	0.00	0.00	0.88	2.05	0.51	0.69		0.25	0.05	2.73	0.03	0.25	0.18	0.20
MG01	0.00	0.00	114.67	229.46	12.95	16.88		0.05	42.21	2166.11	0.03	0.29	127.45	140.72
OG01	0.41	0.00	2.14	25.95	8.15	16.84		2392.67	0.01	0.28	0.02	0.18	171.46	189.31
PG01	0.01	0.00	3.02	7.15	2.48	3.37		0.11	1.29	66.00	0.02	0.19	4.53	5.00
SD01	0.00	0.00	22.81	103.09	9.85	15.81		3.60	2.74	140.75	0.01	0.05	101.49	112.06
SG01	1.08	0.01	24.96	148.60	43.33	74.65		365.88	2.11	108.15	0.01	0.09	140.68	155.32
SPR01	0.00	0.00	0.69	5.04	0.42	0.77		17.30	0.10	5.21	0.01	0.09	74.65	82.41
SPR02	0.03	0.00	1.42	11.55	0.32	0.59		0.12	0.03	1.39	0.01	0.07	49.85	55.03
SPR03	0.07	0.00	1.29	10.72	1.01	1.90		28.20	0.96	49.02	0.02	0.17	88.97	98.22
SPR04	0.03	0.00	5.57	26.07	0.72	1.17		0.05	0.05	2.45	0.03	0.28	12.41	13.70
SPR05	0.03	0.00	1.37	9.94	0.21	0.39		0.05	0.01	0.48	0.01	0.10	28.27	31.21
SPR07	0.07	0.00	0.31	2.84	0.29	0.56		326.00	0.02	0.90	0.01	0.09	92.57	102.20
SPR08	0.06	0.00	0.74	6.43	0.54	1.03		168.00	0.70	36.08	0.02	0.17	92.14	101.73
SPR09	0.03	0.00	4.67	32.09	0.50	0.90		5.44	0.19	9.78	0.01	0.11	101.40	111.95
SPR10	0.03	0.00	1.60	6.69	0.98	1.54		57.00	0.12	6.30	0.04	0.37	2.70	2.98
SPR11	0.03	0.00	0.57	2.64	0.74	1.19		0.10	0.02	1.26	0.02	0.14	0.00	0.00
SPR12	0.03	0.00	1.05	4.04	1.85	2.84		0.10	0.05	2.52	0.03	0.23	0.31	0.34
SPR13	0.03	0.00	0.60	3.38	0.43	0.73		0.10	0.01	0.61	0.02	0.18	0.21	0.23
SPR14	0.03	0.00	1.21	4.46	2.77	4.23		0.10	0.06	2.96	0.03	0.25	0.17	0.19
SPR15	0.03	0.00	9.72	32.15	27.67	41.01		0.87	1.46	75.01	0.07	0.67	69.89	77.16
SPR16	0.03	0.00	13.88	41.69	181.92	263.05		17.10	0.12	6.37	0.09	0.85	19.23	21.24
SPR17	0.03	0.00	8.31	31.04	8.70	13.30		1.31	0.44	22.77	0.05	0.49	66.32	73.23
TC01-1	0.00	0.00	0.61	1.37	1.62	2.17		0.42	0.06	3.18	0.02	0.22	1.73	1.91
TC01-2	0.00	0.00	0.05	0.13	1.48	2.01		0.17	0.17	8.47	0.02	0.19	0.28	0.31
TP01	0.03	0.00	4.91	14.61	33.74	48.68		7.24	0.13	6.57	0.10	0.86	7.35	8.11
TP02	0.03	0.00	2.30	7.31	4.19	6.15		0.10	0.30	15.18	0.08	0.74	17.25	19.04
TP03	0.03	0.00	1.52	5.24	1.68	2.52		0.05	0.04	1.92	0.03	0.30	0.41	0.45

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Table 5-6

Hazard Quotients for Exceedances of the EPA AWQC by RME Surface Water Concentrations - 1991 Data

Station	As		Cd		Cu		Fe		Pb		Ni		Sb		Zn		Hl	
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
AG01	0.02	0.05	23.27	35.39	24.36	29.60	No AWQ	0.12	28.86	1481.09	0.06	0.57	No AWQ	No AWQ	13.09	14.45	89.66	1561.26
AR01	0.00	0.00	0.16	0.49	0.39	0.57		0.38	0.02	0.89	0.01	0.09			0.58	0.64	1.15	3.06
AR02	0.00	0.01	1.47	4.74	0.66	0.97		0.34	0.03	1.63	0.01	0.09			1.87	2.07	4.05	9.85
AR03	0.00	0.00	0.60	1.94	0.44	0.65		0.31	0.02	0.81	0.01	0.09			4.47	4.93	5.54	8.73
AR03A	0.00	0.00	0.33	1.10	0.17	0.26		0.05	0.01	0.69	0.01	0.08			1.31	1.45	1.83	3.63
AR03E	0.00	0.01	0.35	1.21	0.34	0.50		0.08	0.01	0.60	0.01	0.07			1.63	1.80	2.34	4.28
AR03W	0.00	0.00	0.11	0.37	0.29	0.42		0.07	0.01	0.75	0.01	0.08			0.13	0.14	0.55	1.84
AR04	0.00	0.00	0.29	1.14	0.60	0.93		0.25	0.02	0.91	0.01	0.05			1.45	1.60	2.35	4.87
AR05	0.07	0.13	1.02	3.29	2.69	3.96		0.20	0.59	30.37	0.01	0.09			2.26	2.50	6.64	40.54
CG01	0.00	0.00	40.77	164.32	31.06	48.43		2.34	13.83	709.77	0.02	0.15			210.00	231.85	295.68	1156.87
CG02	0.00	0.00	8.77	58.72	3.84	6.81		8.70	1.04	53.24	0.00	0.04			67.01	73.98	80.66	201.49
CG03	0.00	0.01	7.99	51.33	10.58	18.59		2.32	0.09	4.77	0.01	0.06			151.42	167.17	170.10	244.25
CG04	0.00	0.01	7.43	51.02	8.95	16.00		90.60	1.66	85.33	0.01	0.13			132.76	146.57	150.82	389.66
CG05	0.00	0.00	8.20	51.98	7.27	12.73		22.90	1.10	56.55	0.01	0.07			133.44	147.32	150.02	291.56
CG06	0.00	0.00	9.49	50.49	0.42	0.70		0.02	0.01	0.63	0.01	0.07			115.69	127.73	125.62	179.65
EF01	0.00	0.00	0.11	0.37	0.21	0.31		0.04	0.02	0.82	0.01	0.09			0.06	0.07	0.41	1.69
EF02	0.00	0.00	0.20	0.74	0.20	0.30		0.04	0.01	0.45	0.01	0.06			0.16	0.18	0.57	1.77
EG01	0.00	0.00	0.15	0.51	0.26	0.39		0.01	0.01	0.71	0.01	0.08			0.05	0.06	0.49	1.76
EG02	0.00	0.00	0.22	0.70	0.35	0.51		0.01	0.02	0.90	0.01	0.09			0.06	0.07	0.66	2.28
EM01	0.00	0.00	0.02	0.07	0.31	0.45		0.05	0.02	1.07	0.01	0.11			0.07	0.08	0.44	1.82
EM02	0.00	0.00	0.04	0.13	0.20	0.30		0.10	0.06	2.97	0.01	0.08			0.05	0.06	0.36	3.64
EM03	0.00	0.01	0.03	0.12	0.19	0.29		0.13	0.01	0.60	0.01	0.07			0.05	0.05	0.30	1.27
EM04	0.00	0.01	0.39	1.40	1.46	2.22		0.08	0.01	0.53	0.01	0.07			0.04	0.05	1.92	4.35
GG01	0.03	0.06	5.02	8.96	3.96	5.01		0.06	4.42	226.82	0.04	0.38			1.75	1.93	15.22	243.23
HC01	0.00	0.00	0.10	0.24	4.29	5.86		0.06	0.05	2.43	0.02	0.18			0.12	0.14	4.59	8.92
IG01	0.00	0.00	0.02	0.08	0.17	0.28		0.06	0.02	1.01	0.00	0.03			0.02	0.02	0.23	1.49
LF01	0.00	0.00	1.66	3.85	0.89	1.20		0.29	0.05	2.73	0.04	0.40			0.27	0.30	2.91	8.77
MG01	0.00	0.01	114.67	229.46	12.95	16.88		0.05	42.21	2166.11	0.03	0.29			127.45	140.72	297.32	2553.52
OG01	0.76	1.45	2.60	31.46	10.02	20.70		2635.17	0.01	0.56	0.02	0.20			194.69	214.95	208.10	2904.50
PG01	0.01	0.02	3.02	7.15	2.48	3.37		0.11	1.29	66.00	0.02	0.19			4.53	5.00	11.34	81.84

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Table 5-6 (continued)

Hazard Quotients for Exceedances of the EPA AWQC

by RME Surface Water Concentrations - 1991 Data

Station	As		Cd		Cu		Fe		Pb		Ni		Sb		Zn		Hl	
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
SD01	0.00	0.01	34.50	155.92	28.93	46.44		10.50	4.97	255.24	0.01	0.08			271.73	300.00	340.14	768.19
SG01	1.37	2.59	29.68	176.70	56.54	97.40		608.39	2.91	149.30	0.02	0.20			248.81	274.70	339.33	1309.29
SPR01	0.00	0.00	0.69	5.04	0.42	0.77		17.30	0.10	5.21	0.01	0.09			74.65	82.41	75.87	110.82
SPR02	0.03	0.06	1.42	11.55	0.32	0.59		0.12	0.03	1.39	0.01	0.07			49.85	55.03	51.65	68.80
SPR03	0.07	0.14	1.29	10.72	1.01	1.90		28.20	0.96	49.02	0.02	0.17			88.97	98.22	92.32	188.38
SPR04	0.03	0.06	5.57	26.07	0.72	1.17		0.05	0.05	2.45	0.03	0.28			12.41	13.70	18.81	43.77
SPR05	0.03	0.06	1.37	9.94	0.21	0.39		0.05	0.01	0.48	0.01	0.10			28.27	31.21	29.90	42.22
SPR07	0.07	0.14	0.31	2.84	0.29	0.56		326.00	0.02	0.90	0.01	0.09			92.57	102.20	93.27	432.73
SPR08	0.06	0.11	0.74	6.43	0.54	1.03		168.00	0.70	36.08	0.02	0.17			92.14	101.73	94.21	313.55
SPR09	0.03	0.06	4.67	32.09	0.50	0.90		5.44	0.19	9.78	0.01	0.11			101.40	111.95	106.81	160.33
SPR10	0.03	0.06	1.60	6.69	0.98	1.54		57.00	0.12	6.30	0.04	0.37			2.70	2.98	5.47	74.94
SPR11	0.03	0.06	0.57	2.64	0.74	1.19		0.10	0.02	1.26	0.02	0.14			0.00	0.00	1.37	5.38
SPR12	0.03	0.06	1.05	4.04	1.85	2.84		0.10	0.05	2.52	0.03	0.23			0.31	0.34	3.30	10.13
SPR13	0.03	0.06	0.60	3.38	0.43	0.73		0.10	0.01	0.61	0.02	0.18			0.21	0.23	1.31	5.30
SPR14	0.03	0.06	1.21	4.46	2.77	4.23		0.10	0.06	2.96	0.03	0.25			0.17	0.19	4.27	12.24
SPR15	0.03	0.06	9.72	32.15	27.67	41.01		0.87	1.46	75.01	0.07	0.67			69.89	77.16	108.84	226.92
SPR16	0.03	0.06	13.88	41.69	181.92	263.05		17.10	0.12	6.37	0.09	0.85			19.23	21.24	215.28	350.35
SPR17	0.03	0.06	8.31	31.04	8.70	13.30		1.31	0.44	22.77	0.05	0.49			66.32	73.23	83.87	142.20
TC01-1	0.00	0.00	0.71	1.58	1.89	2.54		0.72	0.06	3.18	0.02	0.22			1.92	2.12	4.61	10.36
TC01-2	0.00	0.00	0.05	0.13	1.48	2.01		0.17	0.17	8.47	0.02	0.19			0.28	0.31	2.00	11.28
TP01	0.03	0.06	4.91	14.61	33.74	48.68		7.24	0.13	6.57	0.10	0.86			7.35	8.11	46.24	86.13
TP02	0.03	0.06	2.30	7.31	4.19	6.15		0.10	0.30	15.18	0.08	0.74			17.25	19.04	24.15	48.57
TP03	0.03	0.06	1.52	5.24	1.68	2.52		0.05	0.04	1.92	0.03	0.30			0.41	0.45	3.71	10.54

Note: No AWQC for Barium Acute and Chronic; Manganese Acute and Chronic; Antimony Acute and Chronic

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exceedances of the chronic criterion for lead from stations not associated with California Gulch, these stations included AR02, Empire Gulch, Halfmoon Creek, Lake Fork, and Tennessee Creek (Table 5-1).

There were intermittent exceedances for various elements in samples from the springs. These data were evaluated, but may not be biologically relevant if the springs are at any distance from a receiving water body.

Table 5-6 presents the exceedances based on RME exposure point concentrations. Nickel produced no exceedances, and arsenic few exceedances, for either the acute or chronic AWQC. The other inorganics produced exceedances at many stations. At AR01, EF01, and EF02, there were no exceedances for any inorganic; this station is upgradient of California Gulch. Exceedances were few and minimal at AR02 and TC01 as well. In contrast, stations within California Gulch, such as MG01, had exceedances for most metals.

Figures 5-5 and 5-6 presents the HI values for exceedances of the acute AWQC by average water concentrations at each station. It is clear from this figure that streams associated with California Gulch pose a greater risk to the aquatic environment than do streams not associated with the site. Hazard indices are high for Airport Gulch, California Gulch, Malta Gulch, Oregon Gulch, Starr Ditch, Stray Horse Gulch, and the springs.

Hazard Indices based on average exposure point concentrations relative to chronic criteria (Figures 5-7 and 5-8) are higher than 1,000 at some stations associated with the site. Adverse effects may be evident at AR03 and AR05 based on average stream concentrations of inorganics. Hazard Indices based on RME exposure point concentrations relative to the acute and chronic AWQC (Figure 5-9 and Figure 5-10) also indicate that surface water originating on the site has a great potential to cause adverse effects in the aquatic ecosystem. Hazard Indices for the Arkansas River below California Gulch are generally higher than for stations above the confluence, particularly at AR05. The apparent toxicity of surface water at AR05 is likely due to decreased water hardness with increasing distance from California Gulch; AWQC hardness

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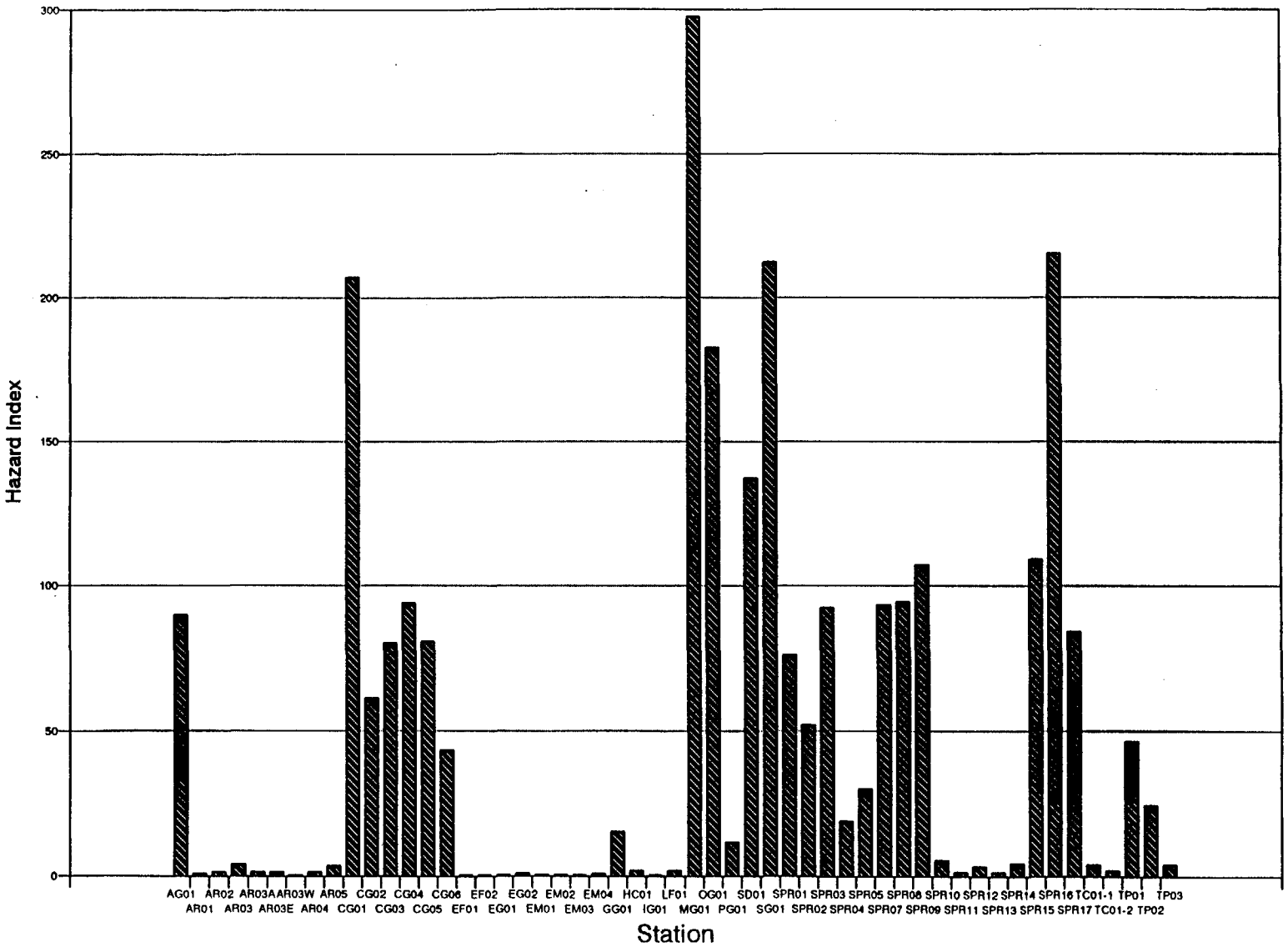
Figure 5-5

Hazard Indices Based on Acute AWQC

and Exposure to Average Concentrations in Surface Water - 1991 Data



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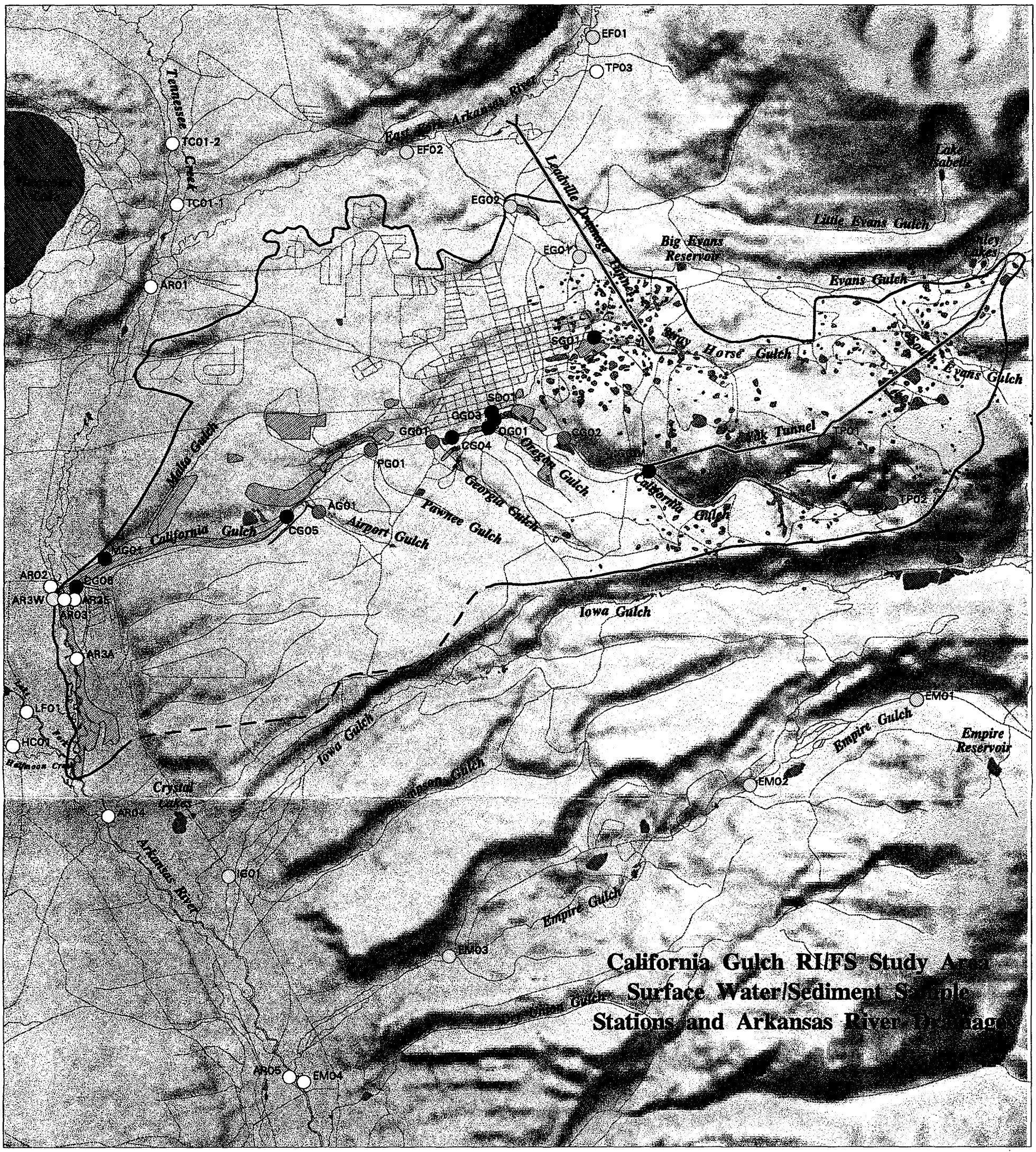

Legend

- Transportation
- Hydrography
- RI/FS Study Area Boundary
- Boundary Pending RI Completion
- Tailing Impoundments
- Waste Rock
- Slag Piles
- Areas Containing Fluvial Tailing

Station Hazard Index:
- < 1
- >= 1 and < 10
- >= 10 and < 100
- >= 100

SCALE 1:60000

1 0 1 2

MILES

Elevation Key - feet

9200 9600 10000 10400 10800 11200 11600 12000 12400 12800 13200 13600

N

S

References

Hydrography from IntraSearch, Inc., 1992 and Roy F. Weston, Inc. Transportation From USGS 1:100,000 DLG data. Topography from USGS 1:24,000 DEM data. Study Area boundary defined by EPA. Tailing Impoundments, Waste Rock Piles, and Slag Piles compiled by Roy F. Weston, Inc., from Steffan,Robertson and Kirsten (SRK), 1989; and Walsh and Associates 1993. Fluvial areas by Woodward – Clyde Consultants, 1993. Surface Water Sampling and Hazard Index data compiled by Roy F. Weston, Inc. and Terra Technologies, Inc. This map may be revised following the resolution of outstanding data issues and completion of remedial investigations.

Exceedances of the Acute AWQC Dissolved Metal Concentrations in Surface Water
Pre - Yak Tunnel Acute Hazard Index

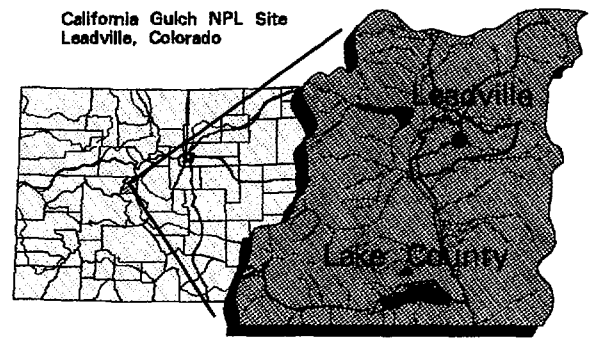

Figure 5-6

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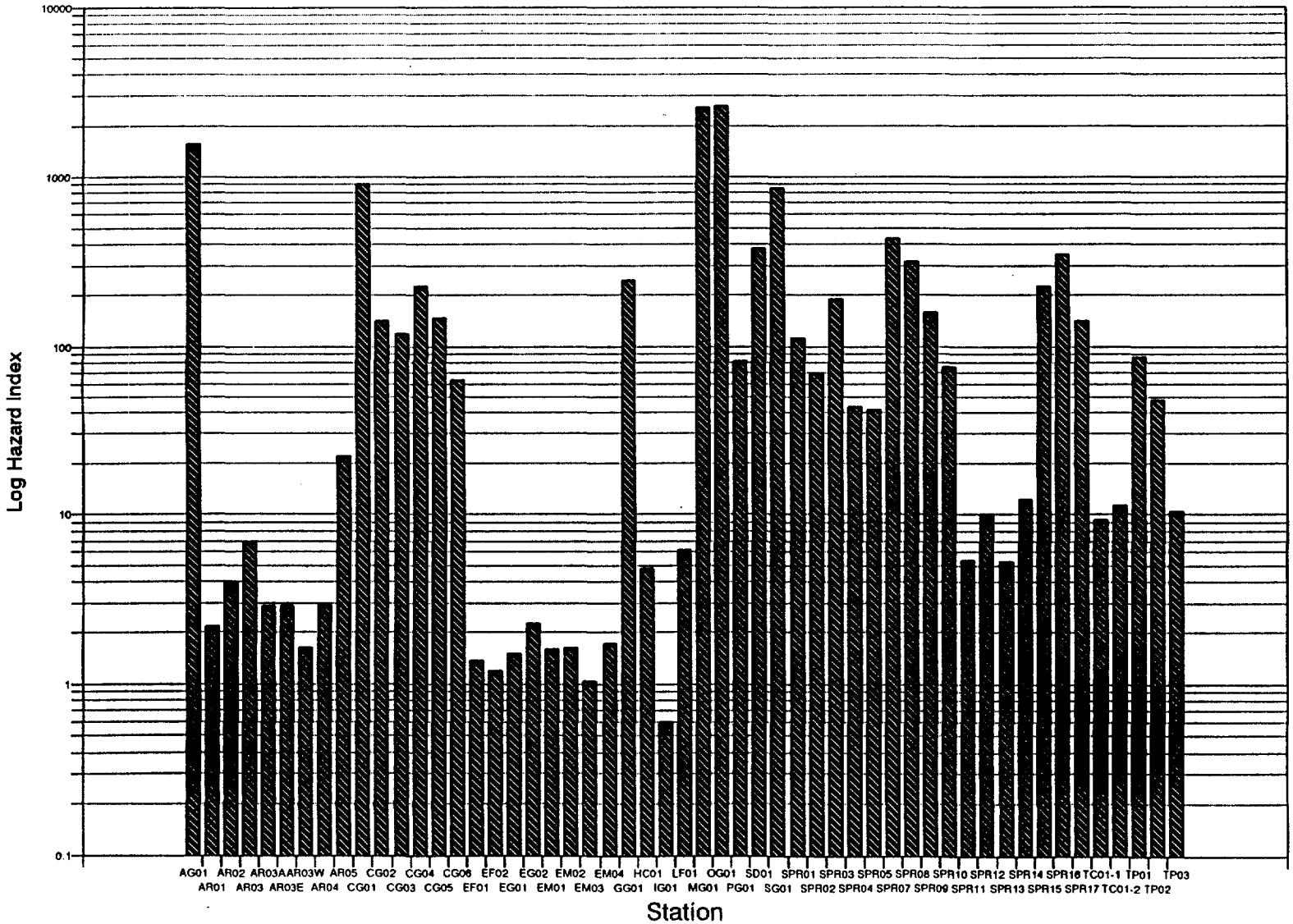
DCN: 4800-31-M002

Revision Date: 27-OCT-1994

Figure 5-7

Hazard Indices Based on Chronic AWQC and Exposure to Average Concentrations in Surface Water - 1991 Data



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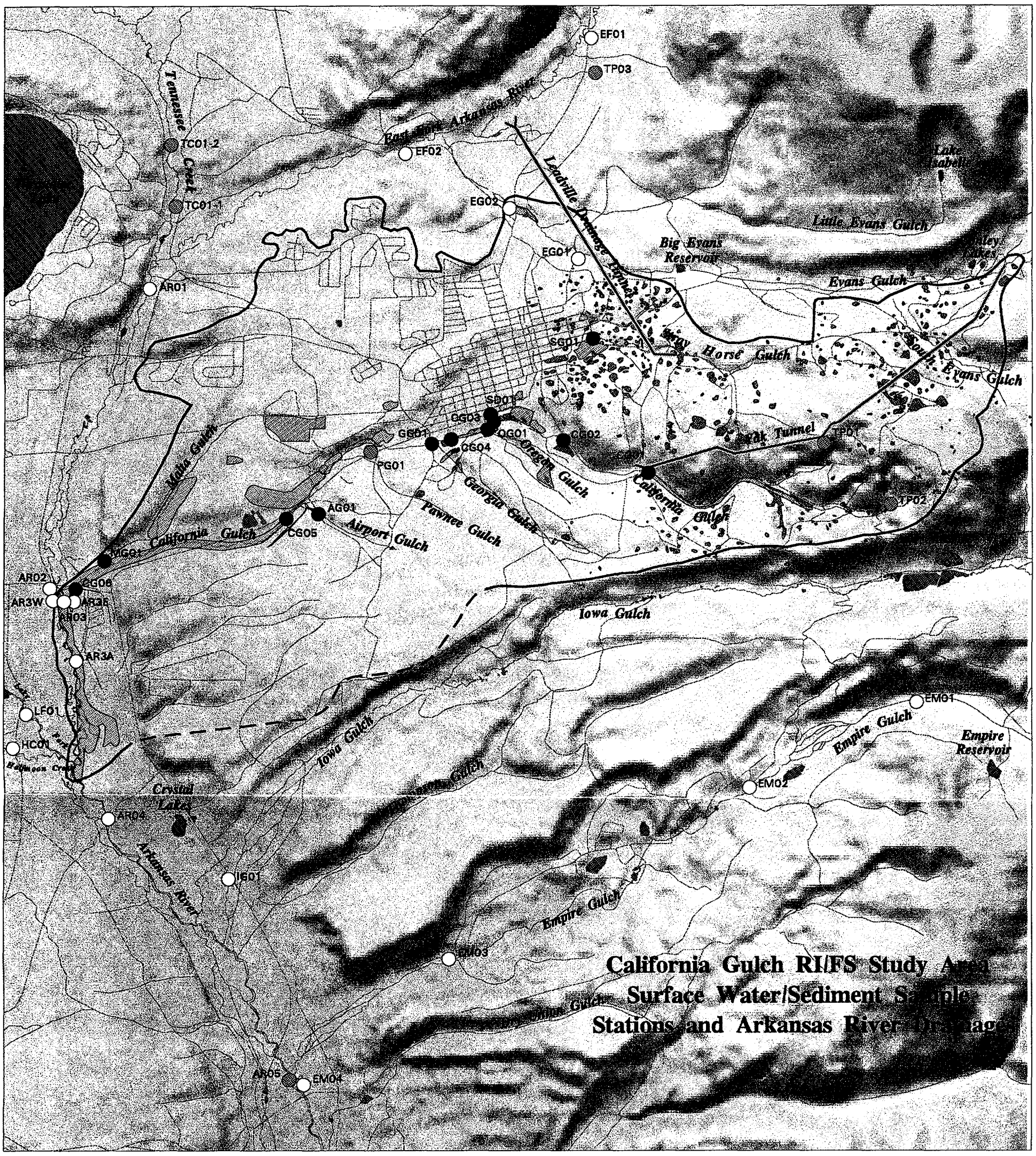

California Gulch RI/FS Study Area
Surface Water/Sediment Sample
Stations and Arkansas River Drainage

Legend

Transportation
Hydrography
RI/FS Study Area Boundary
Boundary Pending RI Completion
Tailing Impoundments
Waste Rock
Slag Piles
Areas Containing Fluvial Tailing

Station Hazard Index:
< 1
>= 1 and < 10
>= 10 and < 100
>= 100

SCALE 1:50000
1 0 1 2
MILES

Elevation Key - feet
9200 9600 10000 10400 10800 11200 11600 12000 12400 12800 13200 13600

N
S

Exceedances of the Chronic AWQC Dissolved Metal Concentrations in Surface Water
Pre - Yak Tunnel Chronic Hazard Index

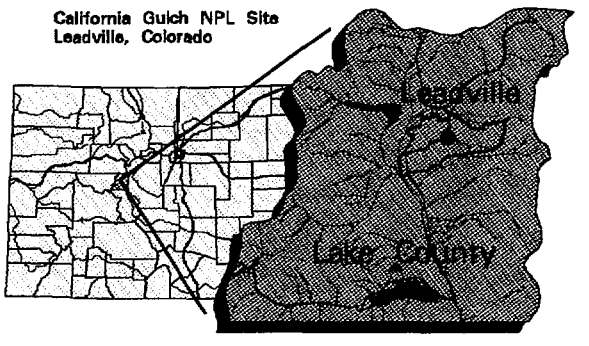

Figure 5-8

EPA

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DCN: 4800-31-M003
Work Assignment No. 38-8L29

Revision Date: 27-OCT-1994
Contract No. 68-W8-0016

References

Hydrography from IntraSearch, Inc., 1992 and Roy F. Weston, Inc. Transportation From USGS 1:100,000 DLG data. Topography from USGS 1:24,000 DEM data. Study Area boundary defined by EPA. Tailing Impoundments, Waste Rock Piles, and Slag Piles compiled by Roy F. Weston, Inc., from Steffan,Robertson and Kirsten (SRK), 1989; and Walsh and Associates 1993. Fluvial areas by Woodward – Clyde Consultants, 1993. Surface Water Sampling and Hazard Index data compiled by Roy F. Weston, Inc. and Terra Technologies, Inc. This map may be revised following the resolution of outstanding data issues and completion of remedial investigations.

Figure 5-9

Hazard Indices Based on Acute AWQC and Exposure to

RME Concentrations in Surface Water - 1991 Data

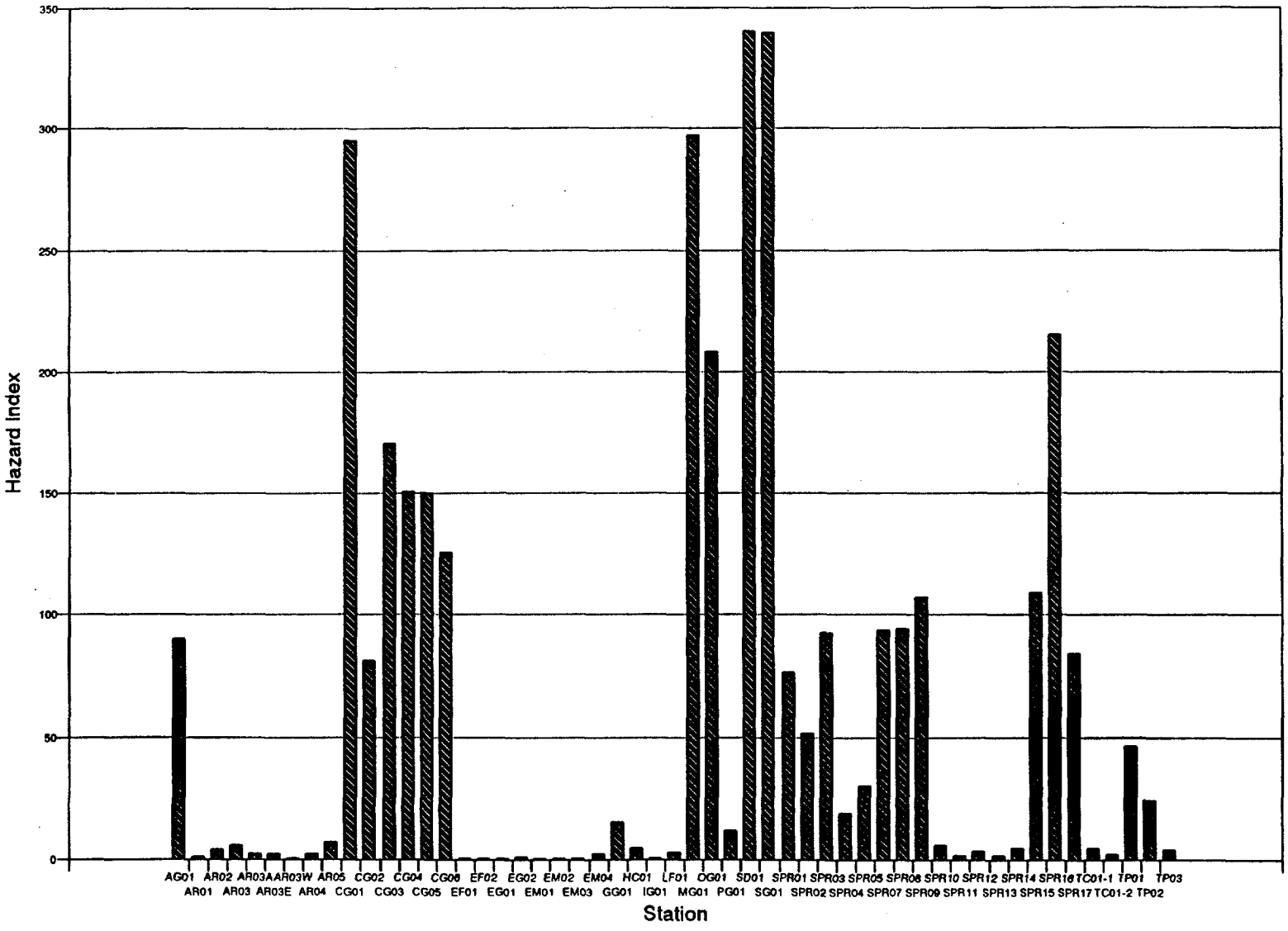

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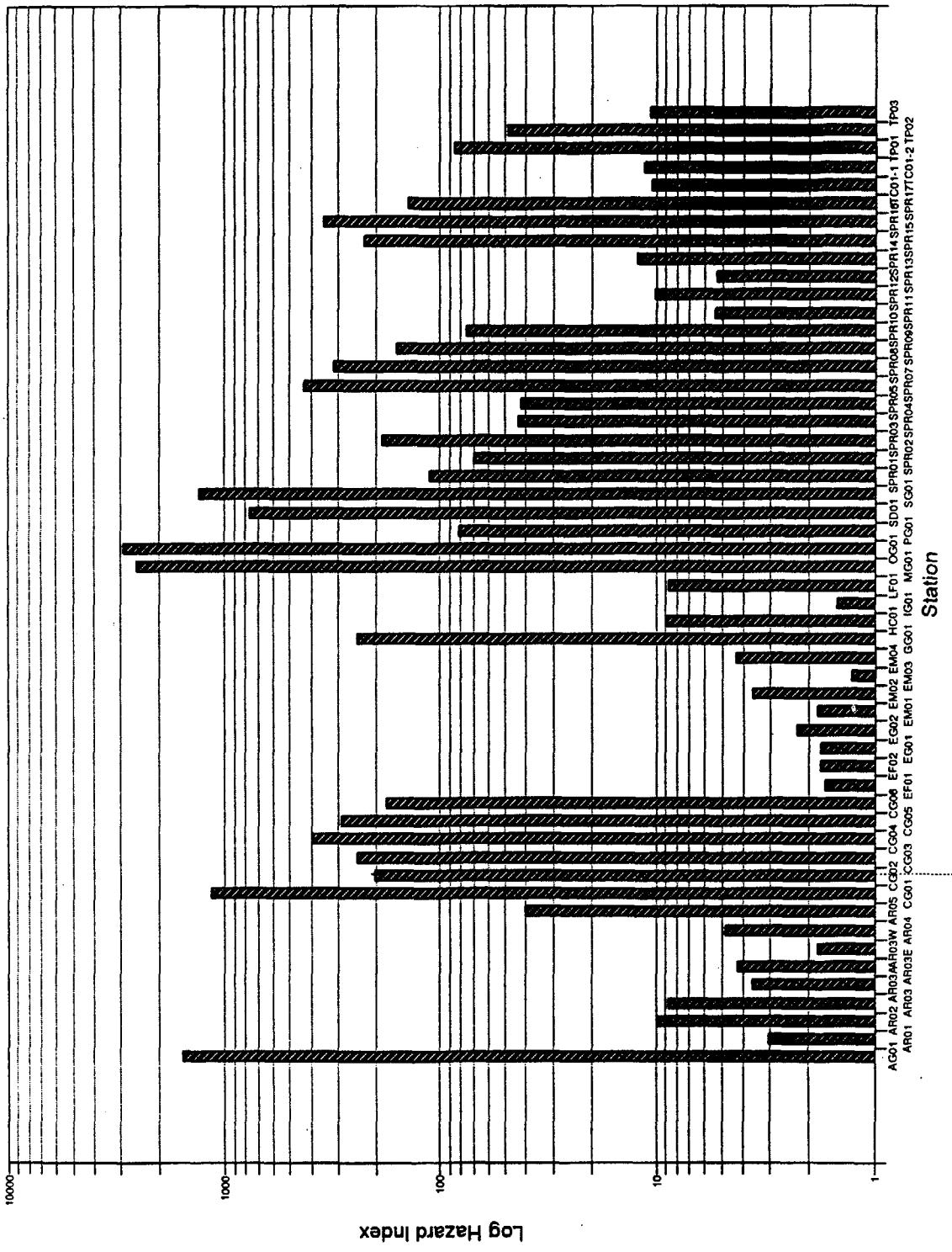
Figure 5-10

Hazard Indices Based on Chronic AWQC and Exposure to RME Concentrations in Surface Water - 1991 Data



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dependent criteria increase with increasing hardness, and California Gulch surface waters have higher hardness, and therefore corresponding higher criteria, than stations in the Arkansas River.

The magnitude and frequency of exceedances of HQs at upgradient stations indicates that concentrations of inorganics are elevated in the Leadville area, but that site concentrations are higher than what would be expected based upon background. Concentrations of inorganics in samples from on-site areas exceeded the EPA AWQC by several orders of magnitude, as indicated by the high hazard indices in Figures 5-5 and 5-7. This indicates that the aquatic ecosystem was exposed to elevated concentrations of inorganics due to site-related inputs, and that there was increased risk of adverse effects to the aquatic ecosystem directly linked to site-related activities as compared to upgradient areas.

5.2.1.3 1992 - 1993 Surface Water Dissolved Metals Data

The 1992 - 1993 data were collected following installation of the treatment plant at the Yak Tunnel. These data reflect the current conditions in California Gulch and the Arkansas River (Table 5-7 and 5-8). The acute and chronic criteria for zinc, and the chronic criterion for lead, were exceeded by average surface water dissolved metals concentrations at AR03A, but not at AR03W (Table 5-7). AR03A is the area receiving discharge from California Gulch. The chronic criterion for cadmium was also exceeded at AR03A, but not at AR02 or AR03W. Zinc concentrations in California Gulch produced exceedances of both the acute and the chronic criterion for this inorganic.

Table 5-8 indicates a pattern similar to exposure to the average concentrations. However, the magnitude of the exceedances is increased slightly. It is important to note that there were no exceedances for AR02, the reference station, whereas multiple exceedances were observed for California Gulch and the Arkansas River downgradient of the confluence with California Gulch.

The HI values were calculated by summing the HQs for each element by station (Figure 5-2). The HIs indicate that despite the water treatment plant, there is risk to aquatic life as a result of drainage from California Gulch into the Arkansas River (Figure 5-11 through 5-14). These

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Table 5-7

Hazard Quotients based on Average Concentrations in Surface Water

July 1992 through August 1993 Data

	Cadmium		Copper		Iron	Lead		Nickel		Silver	Zinc	
Station	**Acute**	**Chronic**	**Acute**	**Chronic**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Acute**	**Chronic**
AR02	0.04	0.13	0.33	0.49	0.04	0.01	0.69	0.00	0.01	0.02	0.10	0.11
AR03E	0.27	0.93	0.28	0.42	0.02	0.01	0.60	0.00	0.01	0.02	0.13	0.14
AR03W	0.08	0.25	0.58	0.85	0.03	0.02	0.82	0.00	0.01	0.02	0.06	0.07
AR3A	0.29	1.07	0.41	0.62	0.09	0.02	1.03	0.00	0.01	0.02	3.21	3.54
CG03	0.29	1.66	0.11	0.19	0.03	0.00	0.09	0.00	0.01	0.02	4.72	5.21
CG04	0.58	3.53	0.17	0.29	1.37	0.00	0.08	0.00	0.03	0.02	19.98	22.06
CG05	0.83	5.00	0.14	0.24	0.70	0.00	0.08	0.00	0.03	0.02	23.00	25.39
CG06	1.70	8.76	1.35	2.25	2.68	0.18	9.34	0.00	0.03	0.02	24.25	26.77

No AWQC for:

Antimony (Acute & Chronic)
Barium (Acute & Chronic)
Iron (Acute)
Manganese (Acute & Chronic)
Silver (Chronic)

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Table 5-8

Hazard Quotients based on RME Concentrations in Surface Water

July 1992 through August 1993 Data

	Cadmium		Copper		Iron	Lead		Nickel		Silver	Zinc	
Station	**Acute**	**Chronic**	**Acute**	**Chronic**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Acute**	**Chronic**
AR02	0.08	0.26	0.54	0.81	0.13	0.01	0.69	0.00	0.02	0.02	0.43	0.48
AR03E	0.43	1.52	0.41	0.62	0.06	0.01	0.60	0.00	0.01	0.02	0.24	0.27
AR03W	0.13	0.43	1.04	1.52	0.09	0.02	0.82	0.00	0.02	0.02	0.06	0.07
AR3A	1.29	4.69	0.88	1.34	0.63	0.08	4.24	0.00	0.01	0.02	12.62	13.93
CG03	0.50	2.89	0.20	0.35	0.08	0.00	0.09	0.00	0.02	0.02	8.08	8.93
CG04	0.93	5.70	0.19	0.33	2.82	0.00	0.08	0.00	0.03	0.02	27.78	30.67
CG05	1.20	7.27	0.33	0.58	1.71	0.00	0.08	0.00	0.03	0.02	31.06	34.30
CG06	2.26	11.67	1.92	3.19	4.46	0.27	13.97	0.00	0.03	0.02	31.94	35.27

No AWQC for:

Antimony (Acute & Chronic)
Barium (Acute & Chronic)
Iron (Acute)
Manganese (Acute & Chronic)
Silver (Chronic)

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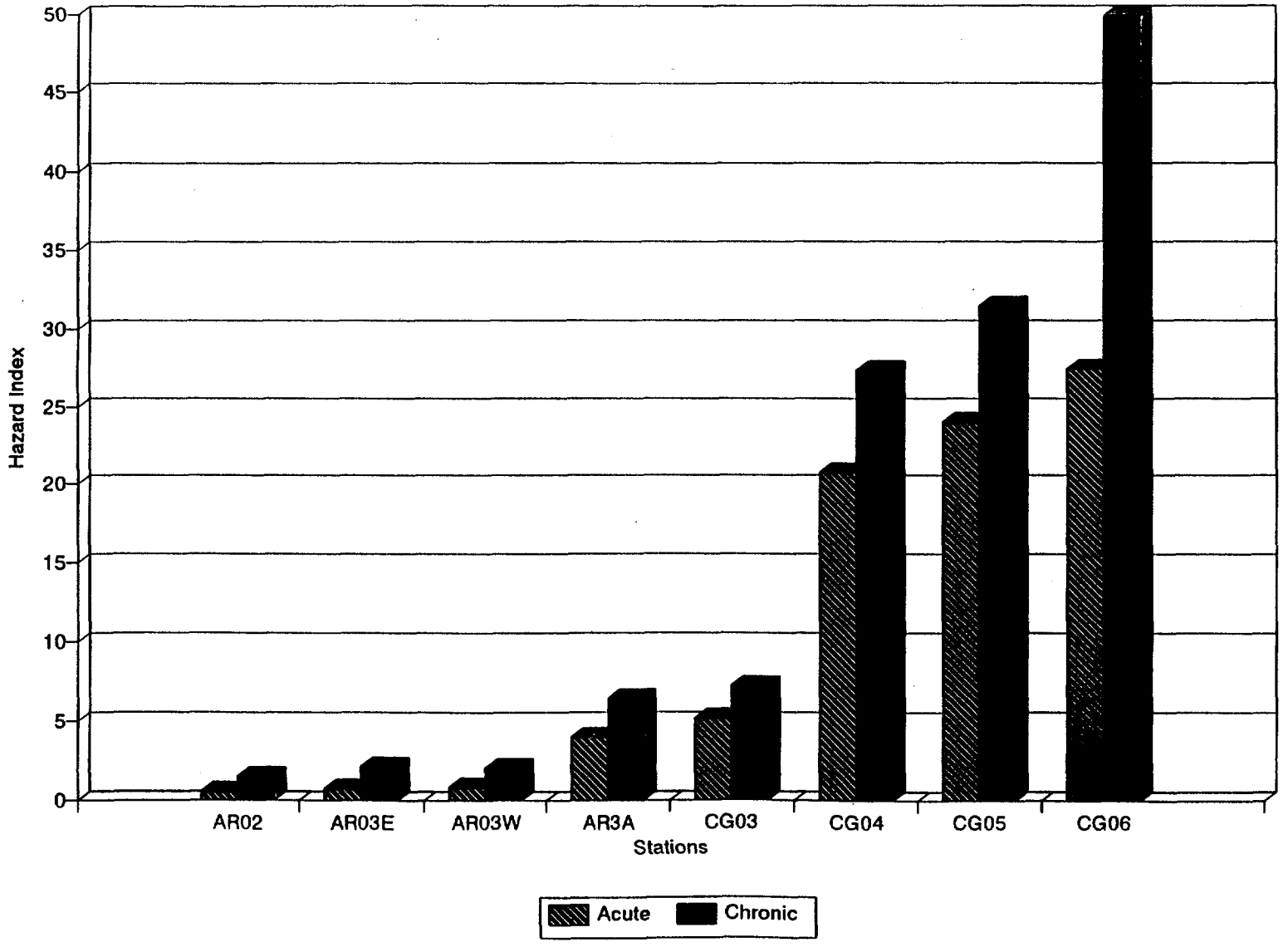
Figure 5-11

Hazard Indices Based on Average Concentrations in Surface Water

July 1992 through August 1993 Data



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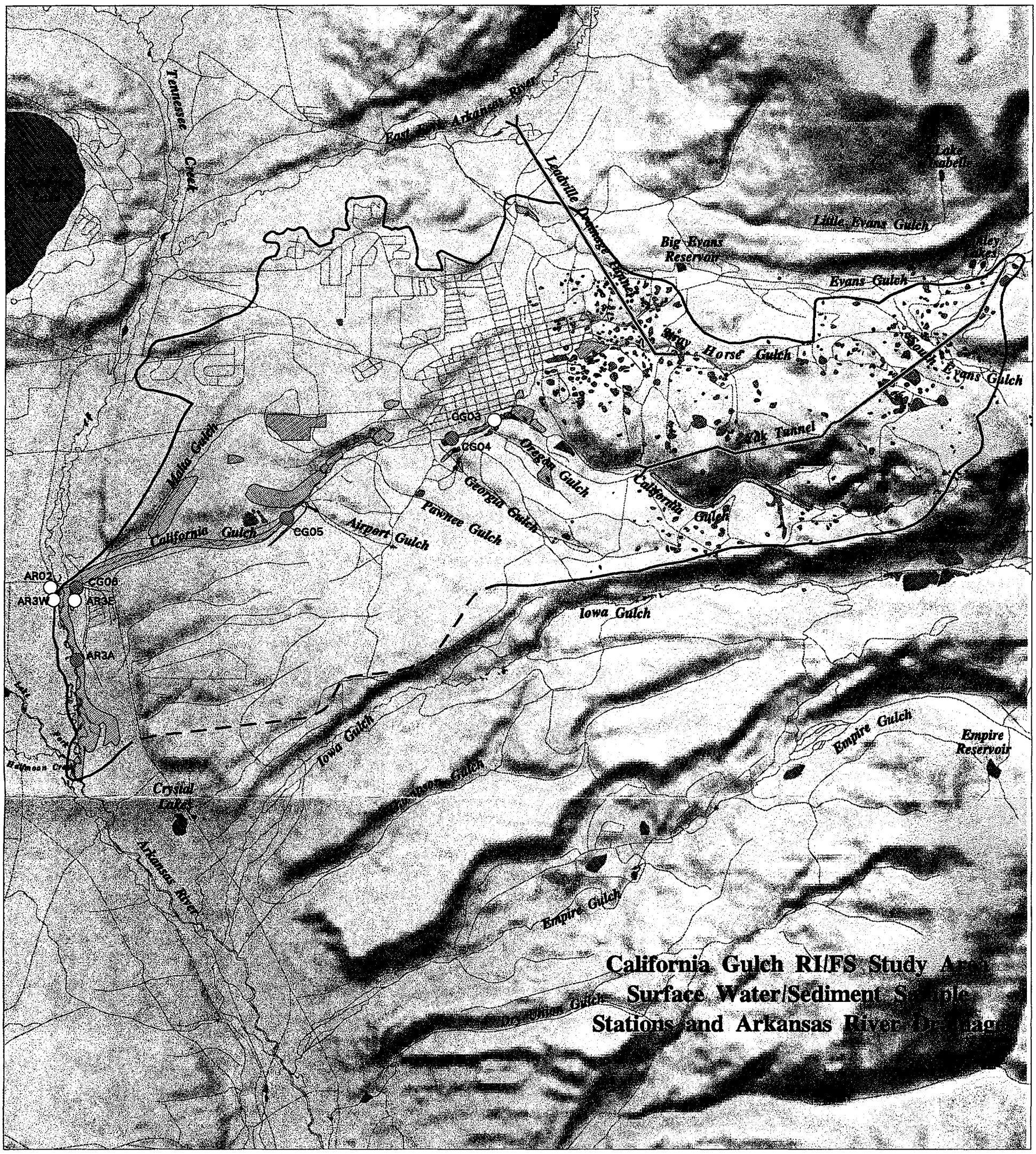


Legend

- Transportation
- Hydrography
- RI/FS Study Area Boundary
- Boundary Pending RI Completion
- Tailing Impoundments
- Waste Rock
- Slag Piles
- Areas Containing Fluvial Tailing

Station Hazard Index:
- < 1
- >= 1 and < 10
- >= 10 and < 100
- >= 100

SCALE 1:50000

1 0 1 2

MILES

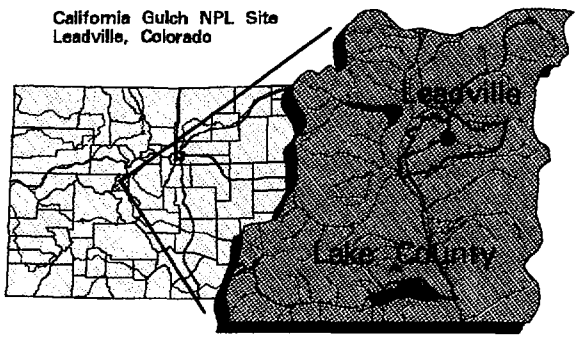
Elevation Key - feet

9200 9600 10000 10400 10800 11200 11600 12000 12400 12800 13200 13600

N

S

Exceedances of the Acute AWQC Dissolved Metal Concentrations in Surface Water Post - Yak Tunnel Acute Hazard Index



References

Hydrography from IntraSearch, Inc., 1992 and Roy F. Weston, Inc. Transportation From USGS 1:100,000 DLG data. Topography from USGS 1:24,000 DEM data. Study Area boundary defined by EPA. Tailing Impoundments, Waste Rock Piles, and Slag Piles compiled by Roy F. Weston, Inc., from Steffan,Robertson and Kirsten (SRK), 1989; and Walsh and Associates 1993. Fluvial areas by Woodward – Clyde Consultants, 1993. Surface Water Sampling and Hazard Index data compiled by Roy F. Weston, Inc. and Terra Technologies, Inc. This map may be revised following the resolution of outstanding data issues and completion of remedial investigations.

Figure 5-12

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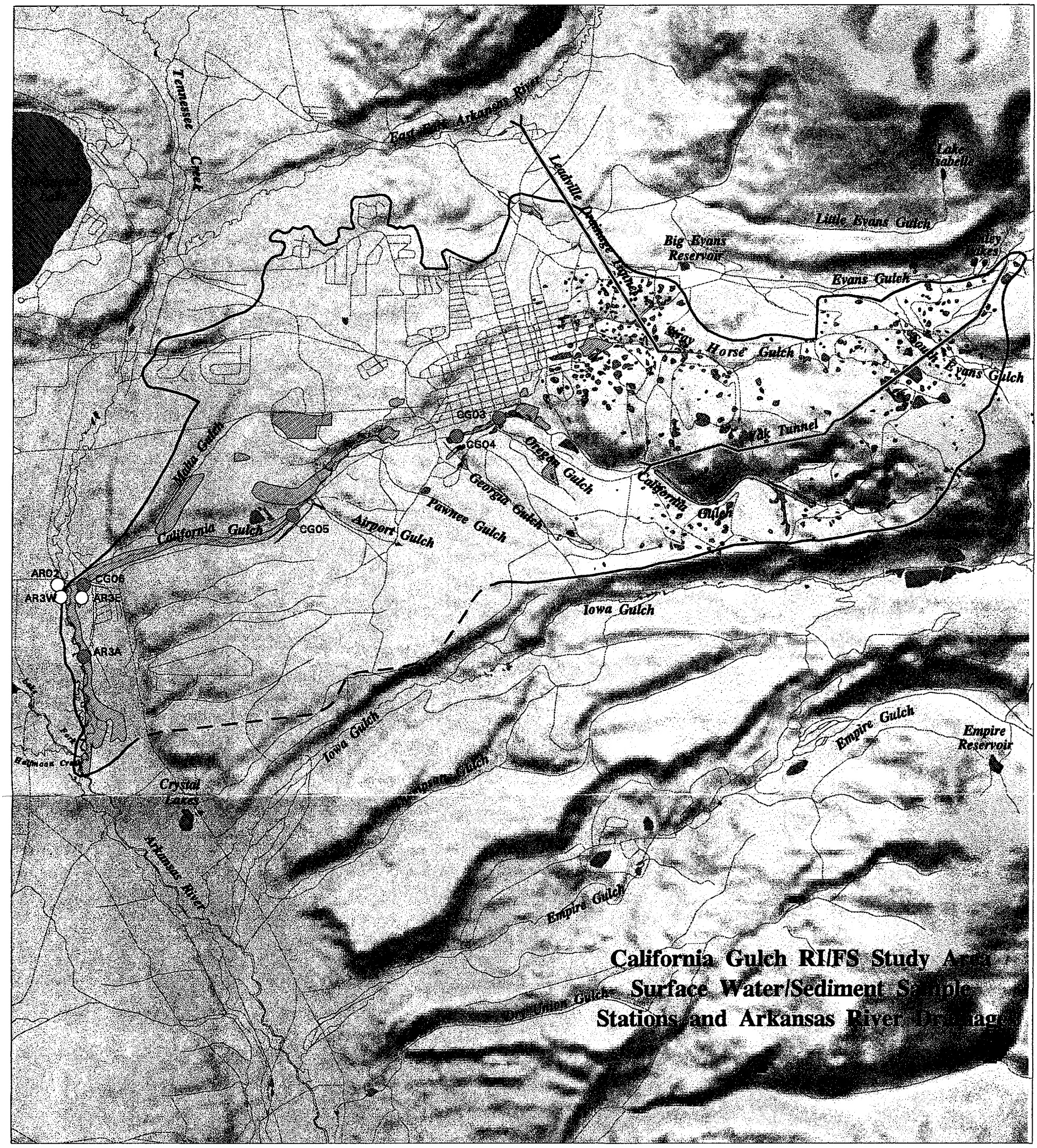
Work Assignment No. 38-8L29

WESTON

Revision Date: 27-OCT-1994

Contract No. 68-W8-0015



Legend

- Transportation
- Hydrography
- RI/FS Study Area Boundary
- Boundary Pending RI Completion
- Tailing Impoundments
- Waste Rock
- Slag Piles
- Areas Containing Fluvial Tailing

Station Hazard Index:

- < 1
- >= 1 and < 10
- >= 10 and < 100
- >= 100

SCALE 1:50000

1 0 1 2

MILES

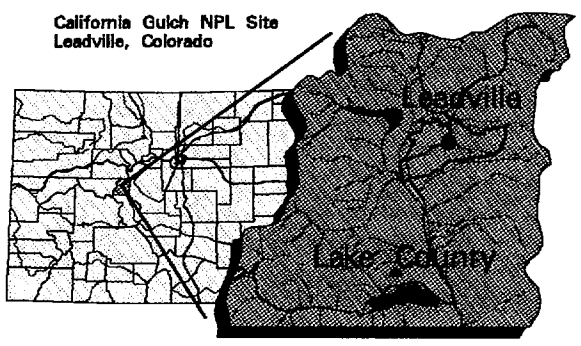
Elevation Key - feet

9200 9600 10000 10400 10800 11200 11600 12000 12400 12800 13200 13600

N

S

Exceedances of the Chronic AWQC Dissolved Metal Concentrations in Surface Water Post - Yak Tunnel Chronic Hazard Index



References

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Figure 5-13

EPA

WESTON

DCN: 4800-31-M005

Revision Date: 27-OCT-1994

APPENDIX A

TOXICITY ASSESSMENTS FOR THE CONTAMINANTS OF CONCERN

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LIST OF ACRONYMS

Ag	silver
Al	aluminum
ALAD	aminolevolinic acid dehydratase
As	arsenic
ATP	adenosine triphosphate
AWQC	Ambient Water Quality Criteria
Ba	barium
BCF	biocentration factor
Bi	bismuth
°C	degrees Celsius
CAS	Chemical Abstracts System
Cd	cadmium
cm	centimeter
Cu	copper
EC_{50}	median effect concentration
Eh	reduction—oxidation potential
Fe	iron
g	gram
g/cm^3	grams per cubic centimeters
Hg	mercury
kg	kilogram
LC_{50}	median lethal concentration
LOAEL	lowest observed adverse effect level
m^2	meter squared
MATC	maximum acceptable toxicant concentration
mg	milligram
mℓ	milliliter
mm	millimeter
Mn	manganese
mol/ℓ	mole per liter
NOAEL	no observed adverse effect level
Pb	lead
ppb	parts per billion
ppm	parts per million
redox	oxidation/reduction
Sb	Antimony
Se	Selenium
Te	tellurium
Tl	thallium
μg/ℓ	micrograms per liter
μg/g	micrograms per gram
w/w	wet weight
Zn	zinc

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Table 2-9 Summary of CDOW Surface Water Data (Woodling, 1994)

River	Location	Date Range		pH	Temp C	Total Alkalinity	Total Hardness	Dissolved Oxygen mg/l
Arkansas	Above Cal Gulch	05/06/91	Max	8.16	18.00	78.00	100.00	8.00
		10/15/91	Min	7.46	4.00	30.00	40.00	8.00
			Mean	7.87	10.75	44.85	54.46	8.00
			Count	25	26	26	26	1
	Below Cal Gulch	05/06/91	Max	7.97	18.00	73.00	150.00	8.00
		10/15/91	Min	7.03	4.00	30.00	46.00	8.00
			Mean	7.54	11.45	46.65	75.70	8.00
			Count	24	26	26	27	1
	Above Cal Gulch	06/01/92	Max	8.20	18.50	74.00	162.00	9.00
		11/23/93	Min	7.31	3.00	17.00	38.00	9.00
			Mean	7.73	10.61	43.81	67.58	9.00
			Count	78	70	78	78	1
	Below Cal Gulch	06/01/92	Max	8.27	18.50	78.00	414.00	11.00
		12/03/93	Min	6.65	0.50	8.00	18.00	7.00
			Mean	7.60	9.60	46.71	109.58	9.00
			Count	93	87	93	95	2

River	Location	Date Range		Cadmium Dissolved	Cadmium Total	Copper Dissolved	Copper Total	Iron Dissolved	Iron Total	Manganes Dissolved	Manganes Total	Lead Dissolved	Lead Total	Zinc Dissolved	Zinc Total
Arkansas	Above Cal Gulch	05/06/91	Max	1.31	2.11	4.90	6.10	283.00	877.00	88.00	146.00	27.70	2.70	187.00	209.00
		10/15/91	Min	0.10	0.11	1.20	2.40	111.00	132.00	20.00	37.00	27.70	2.70	50.00	62.00
			Mean	0.48	0.61	2.74	4.28	199.25	410.00	47.36	70.36	27.70	2.70	100.27	124.00
			Count	24	27	9	4	8	10	11	11	1	1	11	11
	Below Cal Gulch	05/06/91	Max	4.83	9.99	8.10	34.70	329.00	869.00	871.00	892.00	0.00	29.30	1441.00	1612.00
		10/15/91	Min	0.84	1.29	1.90	1.80	100.00	320.00	182.00	251.00	0.00	0.60	270.00	448.00
			Mean	2.48	3.96	3.93	12.96	180.67	593.40	482.30	506.40	0.00	13.85	705.10	911.60
			Count	23	27	10	10	6	10	10	10	0	6	10	10
	Above Cal Gulch	06/01/92	Max	1.54	3.17	16.70	19.40	640.00	4342.00	373.00	588.00	0.00	34.70	558.00	624.00
		11/23/93	Min	0.10	0.10	1.00	1.00	101.00	107.00	10.00	10.00	0.00	25.70	16.00	18.00
			Mean	0.31	0.50	2.51	3.03	200.22	381.45	26.85	57.61	0.00	30.20	55.11	78.85
			Count	53	60	59	64	32	73	66	77	0	2	75	78
	Below Cal Gulch	06/01/92	Max	30.00	36.00	68.00	156.00	7240.00	14110.00	7770.00	7220.00	594.00	347.00	7800.00	7450.00
		12/03/93	Min	0.14	0.21	1.00	1.10	100.00	127.00	13.00	28.00	12.60	10.30	47.00	55.00
			Mean	2.37	3.92	6.47	16.41	647.62	1217.46	508.16	623.37	85.03	74.12	642.70	885.37
			Count	89	93	78	87	47	91	89	93	10	33	89	93

Note: all metal concentrations in ug/l

APPENDIX A
TOXICITY ASSESSMENTS

A.1 ALUMINUM

A.1.1 Environmental Chemistry

A.1.1.1 Physical Properties

Aluminum (Al) occurs naturally in the soil and composes approximately 8% of the earth's surface. Higher concentrations may exist in soil surrounding waste sites associated with certain industries such as coal combustion and aluminum mining and smelting.

Aluminum is approximately the third most abundant element in the earth's crust (Brusewitz, 1984). Aluminum does not occur naturally in the metallic, elemental state, but is widely distributed in the earth's crust in combination with oxygen, fluorine, silicon, and other constituents (NRC, 1982).

Because of its reactivity, aluminum is not found as a free metal in nature. Aluminum has only one oxidation state (+3), thus its behavior or fate and transport in the environment depends upon its coordination chemistry and the characteristics of the local environmental system. Aluminum partitions between solid and liquid phases by reacting and complexing with water molecules and electron-rich anions such as chloride, fluoride, sulfate, nitrate, phosphate, and negatively charged functional groups on humic material and clay (Bodek et al., 1988).

A.1.1.2 Chemical Properties

At a pH greater than 5.5, naturally occurring aluminum compounds exist predominantly in an undissolved form such as gibbsite, Al(OH)3. The presence of high amounts of dissolved organic material or fulvic acid binds aluminum and can cause increased dissolved aluminum concentrations in streams and lakes (Brusewitz, 1984). In general, decreasing pH results in an increase in mobility for monomeric forms of aluminum, which is a concern with respect to the occurrence of acid rain and the release of acid mine drainage. In soils, the most soluble form

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of aluminum under acidic conditions is nonsilicate, organically bound aluminum (Mulder et al., 1989), whereas the most soluble form of hydroxy aluminum is amorphous Al(OH)3 (Bodek et al., 1988).

In groundwater or surface water systems, an equilibrium with a solid phase or form is established that largely controls the extent of aluminum dissolution that can occur. The mineral gibbsite, Al(OH)3, is a common control on aluminum solubility in many environments (Bodek et al., 1988). In a Colorado alpine watershed soil, the chemical equilibria of aluminum in interstitial water at a pH range of 4.4 to 7.2 are controlled by amorphous aluminosilicate rather than gibbsite (Litaor, 1987).

The adsorption of aluminum on to clay surfaces can be a significant factor in controlling aluminum mobility in the environment, and these adsorption reactions, measured in one study at pH 3.0 to 4.1, have been observed to be very rapid (Walker et al., 1988). However, clays may act either as a sink or a source for soluble aluminum depending on the degree of aluminum saturation on the clay surface (Walker et al., 1988). The presence of high levels of suspended solids in stream surface water during storm episodes resulted in higher concentrations of adsorbed aluminum than observed during the absence of suspended solids (Goenaga and Williams, 1988). The increased adsorption was not strictly linear.

Aluminum occurs ubiquitously in natural waters. Aluminum levels in surface waters can be increased directly or indirectly by human activity through surface run-off, tributary inflow, groundwater seepage, industrial and municipal effluents, and wet and dry atmospheric deposition (Eisenreich, 1980). Weathering of sulfide ores exposed to the atmosphere in inactive mines and tailings dumps releases large quantities of sulfuric acid and metals such as aluminum (Filipek et al., 1987). Increasingly acid environments caused by acid mine drainage or by acid rain will subsequently cause an increase in the dissolved aluminum content of the surrounding waters (Brusewitz, 1984).

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A.1.1.3 Biological Fate

In subalpine ecosystems, the large root biomass of the Douglas fir, *Abies amabilix,* takes up aluminum and immobilizes it, preventing large accumulation in aboveground tissue (Vogt et al., 1987). It is unclear to what extent aluminum is taken up into root food crops and leafy vegetables. An uptake factor (concentration of aluminum in the plant/concentration of aluminum in soil) of 0.004 was observed for leafy vegetables.

The potential for accumulation of aluminum has been studied in several aquatic species. Bioconcentration of aluminum in fish is a function of the water quality (i.e., pH and total organic carbon) (Cleveland et al., 1989). Brook trout have been shown to accumulate slightly more aluminum (measured as whole body residues) at pH 5.6 to 5.7 than at pH 6.5 to 6.6 (Cleveland et al., 1989). Aluminum concentrations were higher and more variable in gill tissue of smallmouth bass than in other tissues (Brunbaugh and Kane, 1985). Aluminum concentrations in rainbow trout from an aluminum treated lake, an untreated lake and a hatchery were highest in gill tissue and lowest in muscle (Buergel and Soltero, 1983). Aluminum residue analyses in brook trout have shown that whole-body aluminum content decreases as the fish advance from larvae to juveniles (Cleveland et al., 1989).

These results imply that the aging larvae begin either to decrease their rate of aluminum uptake or to eliminate aluminum at a rate that exceeds uptake. The decline in whole-body aluminum residues in juvenile brook trout may be related to growth and dilution by edible muscle tissue that accumulate less aluminum than did the other tissues (Cleveland et al., 1989). No information was found on the biomagnification of aluminum in the food chains in the literature reviewed.

Measured bioconcentration factors (BCFs) were unavailable; however, Chapman et al. (1968) estimated a BCF of 10 for edible portions of freshwater fish.

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A.1.2 Toxicity

A1.2.1 Aquatic Organisms

In acute toxicity tests, the most sensitive invertebrate is the cladoceran *Ceriodaphnia sp.* Under static test conditions using organisms less than 24 hours old, Call (1984) reported a median effect concentration (EC_{50}) of 3,690 micrograms per liter ($\mu g/\ell$). Studies using *Daphnia magna* reported EC_{50} values under static test conditions ranging from 3,900 $\mu g/\ell$ (Biesinger and Christensen, 1972) to 38,200 $\mu g/\ell$. The most tolerant invertebrate species is the midge *(Tanytarsus dissimilis),* which has a larval EC_{50} of 79,900 $\mu g/\ell$ (Lamb and Bailey, 1981).

In a chronic life cycle test using *Daphnia magna,* a maximum acceptable toxicant concentration (MATC) of 1,388 $\mu g/\ell$ was reported. Using the data reported in EPA (1986) an acute–chronic ratio of 28 was established for *Daphnia magna.* Anderson (1944) reported incipient immobilization in *Daphnia magna* following 16 hours of exposure to 21,450 to 21,530 $\mu g/\ell$ aluminum. Biesinger and Christensen (1972) reported a 21-day median lethal concentration (LC_{50}) of 1,400 $\mu g/\ell$ for *Daphnia magna;* 16% reproductive impairment was reported in *Daphnia magna* at a concentration of 320 $\mu g/\ell$.

Most studies regarding toxicity of aluminum to vertebrates were conducted using juvenile fish. The most sensitive vertebrate species is the brook trout *(Salvelinus fontinalis)* with a 96-hour LC_{50} of 3,600 $\mu g/\ell$ under flow-through conditions (Decker and Menenez, 1974). The green sunfish *Lepomis cyanellus* is the most tolerant species, with a 96-hour LC_{50} of greater than 50,000 $\mu g/\ell$ under static test conditions (Call, 1984).

In an early-life stage test using the fathead minnow *(Pimephales promelas)* an MATC of 5,777 $\mu g/\ell$ was reported (EPA, 1986). An acute–chronic ratio of 6.1 was established for the fathead minnow (EPA, 1986). Birge et al., (1978) reported an 8-day EC_{50} of 170 $\mu g/\ell$ based on lethality and deformity in an early-life-stage test using largemouth bass *(Micropterus salmoides),* a 7-day EC_{50} of 150 $\mu g/\ell$ for the goldfish *(Carassius auratus),* and a 28-day EC_{50} of 560 $\mu g/\ell$ for rainbow trout *(Salmo gairderi).* After exposure for 3, 7.5, 32, and 39 days, EC_{50} values based on mortality were all within the range of 5,140 $\mu g/\ell$ to 5,230 $\mu g/\ell$. Following a 44-day

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exposure period, an EC_{50} of 513 $\mu g/\ell$ was reported. In a later study, Everhart and Freeman (1973) reported that a concentration of 5,200 $\mu g/\ell$ had no effect on fertility of rainbow trout embryos exposed from fertilization to hatch.

Birge et al., (1978; 1979) have conducted early-life-stage tests using amphibians. They report a 7-day EC_{50} of 50 $\mu g/\ell$ based on mortality and deformity for the narrow-mouthed toad *Gastrophryne carolinensis,* and an 8-day EC_{50} of 2,280 $\mu g/\ell$ based on mortality and deformity for the marbled salamander *Ambystoma opacum.*

A.1.2.2 Terrestrial and Riparian Organisms

It is well established that plant roots are not entirely selective in extracting substances from the soil. The roots of a plant extract not only essential nutrients, but non-essential and potentially phytotoxic substances as well. Physical and chemical characteristics of soil, including pH, temperature and waterlogging, as well as the specific characteristics of a particular metal, play a significant role in determining uptake. For example, it is possible for plants to take up non-nutrient metals if they are physically similar to a nutrient (i.e., substituting cadmium for zinc). Soil pH strongly affects the processes of adsorption, precipitation, and chelation, which govern the mobility of heavy metals. Soil pH is often the most important factor affecting the transfer of metals to plants; metal solubility generally increases by a factor of 10 as pH decreases by 1 unit (Forstner, 1988). Metals with strong pH-dependent solubility include aluminum, manganese, copper, nickel, lead, and zinc (Chaney, 1985; Merry, 1986).

Chapman (1966) observed a 61 to 68% reduction in alfalfa crop yield in response to 1.9 to 0.56 part per million (ppm) soluble aluminum in soil. A suggested toxicity threshold of 0.5 ppm soluble aluminum was suggested by the author for "sensitive crop species" (Chapman, 1966). Results of an investigation by Chapman (1966) indicate that concentrations of 0.1 to 6.8 ppm aluminum in nutrient solution culture cause a reduction in crop yield, while concentrations of 2 to 80 ppm aluminum in nutrient solution cause a reduction in grass yield.

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Studies investigating aluminum effects on soil invertebrates, indicate that concentrations between 2,500 to 6,100 ppm cause various toxic effects to exposed wood louse *(Porcellio scaber)* (Beyer and Anderson, 1985).

Aluminum is fatal to laboratory animals only at very high doses. For the nitrate form, the LD50 is 261 milligrams (mg) aluminum/kilogram (kg) for rats (Llobet et al., 1987), and for the chloride form the LD50 is 770 mg aluminum/kg for mice (Ondreicka et al., 1966). Aluminum chloride increased mortality in rat pups when given to dams during gestation at 155 mg aluminum/kg (Bernuzzi et al., 1986). In a study of dogs exposed to 75 mg aluminum/kg/day, no fatalities were observed (Pettersen et al., 1990).

A no observed adverse effect level (NOAEL) of 50 mg Al/kg body weight/day for rats, and 60 mg Al/kg body weight/day for dogs was derived for aluminum in oral exposure studies (McCormack et al., 1979; Katz et al., 1984). A NOAEL of 19 mg Al/kg body weight/day was established for mice in a chronic oral ingestion study (Ondreicka et al., 1966).

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A.2 ANTIMONY

A.2.1 Environmental Chemistry

Antimony (Sb) is a naturally occurring element found as various salts in surface water, soils, and sediments. Its terrestrial abundance is in the order of 0.7 micrograms per gram (μg/g) (Brannon and Patrick, 1985). Antimony concentrations in the earth's crust range from 0.2 to 10.0 mg/kg (Hazardous Substances Databank, 1988), with an average concentration of 1 mg/kg (Bowen 1979). Surface water median concentrations of antimony are 0.2 μg/ℓ; the range reported was 0.01 to 5 μg/ℓ (Bowen, 1979).

More than half of the naturally occurring antimony in sediments is bound to extractable iron and aluminum (Crecelius et al., 1975). Brannon and Patrick (1985) determined that antimony-amended sediment releases volatile antimony compounds during anaerobic incubation and antimony evolution rates of 8 μg/meter squared (m^2)/week were detected. Observations during this investigation revealed that release of volatile antimony compounds from anaerobic sediments containing recent deposits of soluble antimony can occur regardless of the oxygen status of the overlying water.

Various forms of antimony found in the environment from both natural and anthropogenic sources undergo a complex cycle of chemical interconversion and transfer between media. Antimony in water may undergo either oxidation or reduction, depending on the pH and presence of other ions. Soluble forms tend to be quite mobile in water, while less mobile species adsorb to clay or soil particles (Callahan et al., 1979). Antimony(III) and antimony(V) forms and methyl antimony compounds have been shown to exist in natural waters (Andreae et al., 1981; Parris and Brinkman 1975; 1976)

A.2.1.1 Physical Properties

Antimony (atomic No. 51) has an atomic weight of 121.75, a boiling point of 1,750 degrees Celsius (°C), a melting point of 630.74°C, and a specific gravity of 6.68 grams per cubic centimeters (g/cm^3) (Adriano, 1986). Antimony is classified as both a metal and a metalloid.

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The principal oxidation states include III and V (EPA, 1980). The Chemical Abstracts System (CAS) registry number is 7440-36-0.

A.2.1.2 Chemical Properties

The chemical behavior of antimony is similar to arsenic in that it exists in a variety of chemical forms. It is found in any of four valence states (-III, 0, III, or V). In the environment, stibnite (Sb_2S_3) is the most common naturally occurring form of antimony, although it is also found as the native metal, as antimonides of heavy metals, and as antimony oxides (Adriano, 1986).

Antimony is present as the soluble oxide of antimonite(III) salt in most natural waters. In reducing environments, volatile stibine (SbH_3) may be formed. Stibine is a gas at room temperature and very water soluble. The formation of stibine in sediments, which usually provide a reducing environment, may offer a mechanism for remobilization of antimony previously removed from solution (EPA, 1980).

The extent to which sorption reduces the aqueous transport of antimony is unknown, but it is clear that sorption to clays and minerals is an important mechanism that results in the removal of antimony from solution. Most species of antimony in natural waters are soluble and quite mobile and are eventually transported in solution (EPA, 1980).

A.2.1.3 Biological Fate

Biomethylation processes resulting in the formation of volatile stibine derivatives may also cause some remobilization of antimony (EPA, 1980).

Bioaccumulation appears to be only a minor fate process for antimony (EPA, 1980). Bioconcentration of antimony by freshwater algae has not been observed (EPA, 1980). A BCF of 1 is reported by EPA (1990).

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A.2.2 Toxicity

A.2.2.1 Aquatic Organisms

Little information is available regarding toxicity of antimony to aquatic life. In general, antimony appears to be more toxic to aquatic plants than to fish or invertebrate species (EPA, 1980). Available toxicity data are summarized in Table A.2-1.

Antimony is only slightly bioaccumulated and has been little studied in aquatic organisms. Leatherland et al. (1973) found low levels of antimony in various fish and invertebrates collected off the northwest coast of Africa; antimony was generally present in higher concentrations in invertebrates than in fish.

The LC_{50} and EC_{50} values for cladoceran (*Daphnia magna*) and fathead minnow (*Pimephales promelas*) range from 9,000 to 21,900 $\mu g/\ell$. Chronic values for the fathead minnow and *Daphnia magna* are 1,600 and 5,400 $\mu g/\ell$, respectively. The freshwater algae (*Selenastrum capricornutum*) is more sensitive than the animal species tested. The 96-hour EC_{50} for freshwater algae was 610 $\mu g/\ell$; adverse effects observed were the inhibition of chlorophyll-a synthesis (EPA, 1980).

Summary of Available Criteria

Available data are not adequate for establishing criteria. Neither federal guidance nor state standards have been promulgated; however, EPA reports the lowest values known to be toxic to freshwater aquatic organisms (the acute value is 9,000 $\mu g/\ell$, and the chronic value is 1,600 $\mu g/\ell$) (EPA, 1980).

A.2.2.2 Terrestrial and Riparian Organisms

Oral acute and chronic exposure of antimony potassium tartrate causes vomiting and diarrhea in dogs and cats at concentrations of 10 mg/kg body weight. Whereas antimony trioxide and

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Table A.2-1

Toxic Effects of Antimony to Freshwater Aquatic Life

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Plants					
Algae	*Selenastrum capricornutum*	Antimony trioxide	0.610	EC_{50} (96-hours) chlorophyll-a inhibition	EPA, 1978
Algae	*Selenastrum capricornutum*	Antimony trioxide	0.630	EC_{50} (96-hours) cell number reduction	EPA, 1978
Invertebrates					
Cladoceran	*Daphnia magna*	Antimony trioxide	19.80	EC_{50} (48-hours)	EPA, 1978
Cladoceran	*Daphnia magna*	Antimony trioxide	<530.00	EC_{50} (48-hours)	EPA, 1978
Vertebrates					
Bluegill	*Lepomis macrochirus*	Antimony trioxide	>530.00	LC_{50} (96-hours)	EPA, 1978

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pentoxide could be given to the same animals in doses above 100 mg/kg body weight for several months with not observable effects (Flury, 1927; Bradley and Frederick, 1941).

The acute oral LD_{50} values for potassium antimony tartrate (tartar emetic) in mice and rats range from 115 to 600 mg Sb/kg (Bradley and Fedrick, 1941; Hazardous Substances Databank, 1988), whereas an oral LD_{50} of 15 mg Sb/kg has been reported for rabbits (Hazardous Substances Databank, 1988).

Flury (1927) fed high doses of five antimony compounds to rats (one per chemical) for 9 days. Each rat received daily doses increasing from 100 mg to 2 or 3 g. Doses up to 2 g/day of antimony trioxide or antimony pentoxide or up to 3 g/day of sodium meta-antimonate caused no adverse effects. Potassium antimony tartrate was found to be toxic, causing death after the daily dose was increased to 500 mg (about 1,000 mg/kg Sb/kg) on day 7. Potassium antimonate produced adverse effects at dose levels of 2 g/day, but recovery was rapid when dosing was ceased.

Pribyl (1927) investigated the toxicity and effect on nitrogen metabolism in four rabbits given 15 mg potassium antimony tartrate/kg/day (given in a milk plus sugar solution) over a 7- to 22-day period. This corresponds to a dose of 5.6 mg Sb/kg/day. As a result, elevated ammonia nitrogen was interpreted as evidence of increased protein catabolism. This study suggested a lowest observed adverse effect level (LOAEL) of 5.6 mg/kg/day based on minimal histological injury in tissues.

In a 91-day oral toxicity study in male and female Wistar rats, Bomhard et al. (1982) reported that two antimony-containing pigments produced no effects on behavior, food consumption, growth, mortality, hematological and clinical data, or organ weights. Concentrations administered ranged up to 10,000 ppm (corresponding to daily doses of 36 and 22 mg/kg/day respectively).

Schroeder et al. (1968) administered potassium antimony tartrate (0 or 5 ppm Sb) in drinking water to male and female Charles River CD mice from the time of weaning until death.

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Exposure resulted in weight loss in males after 18 months and decreased weight gain in females at 12 and 18 months. This study suggest a LOAEL of 0.5 mg Sb/kg/day based on minimal liver fatty degeneration and decreases in weight and longevity. An average mouse weight of 40 g, and 4 milliliters (mℓ) of water consumption/day containing 5 ppm antimony was assumed.

Browning (1969) reported effects of antimony exposure to glucose and cholesterol metabolism in rats exposed to 5 mg/ℓ in drinking water, but no signs of adverse effects were noted in rats receiving doses of up to 100 mg/kg/day. Substantially higher doses of antimony trioxide were tolerated in rat studies by Sunagawa (1981) suggesting a NOAEL of 500 mg/kg.

In a similar study (Kanisawa and Schroeder, 1969), mice were given 0 to 5 mg/ℓ (5 ppm) for 540 days (18 months) in drinking water. Lifespans were significantly reduced in both females and males, but the degree of toxicity was less severe in mice than rats.

Schroeder et al. (1970) administered potassium antimony tartrate (0 or 5 mg Sb/ℓ) in drinking water to groups of at least 50 male and 50 female Long-Evans rats from the time of weaning until death. This corresponds to an average daily dose of 0.35 mg Sb/kg/day, based on the authors' calculations. Antimony accumulated in the soft tissues with age (from 279 to 1,070 days); pooled samples showed a tendency to increase in concentration with a correlation coefficient of 0.525. This study identified a LOAEL of 0.35 mg Sb/kg/day based on decreased longevity, and altered blood glucose and serum cholesterol levels.

Reported oral LD_{50} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) to laboratory animals include:

- 100 for rats exposed to antimony metal.
- >20,000 for rats exposed to antimony trioxide.
- 675 for rats exposed to antimony trichloride.
- 600 for mice exposed to potassium antimonyl tartrate.

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- 115 for rats exposed to potassium antimonyl tartrate.
- 115 for rabbits exposed to potassium antimony tartrate.

A.2.3 References

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A.3 ARSENIC

A.3.1 Environmental Chemistry

Arsenic (As) is a naturally occurring metalloid that can be present in a number of different valence states and as a constituent in both inorganic and organic compounds. Elemental arsenic is used in industry as an alloying agent; both inorganic and organic arsenical compounds have been used as pesticides (OSHA, 1983).

Arsenic is ubiquitous in nature and is found in detectable concentrations in all soils and nearly all other environmental matrices. The concentration of arsenic in the continental crust of the earth generally ranges from 1.5 to 2 ppm (NAS, 1977). An additional source reports the range of soil and plant background levels are 0.1 to 50, and 0.01 to 1.7 ppm respectively (Bodek et al., 1988). Arsenic ranks as the 52nd most abundant element in the earths crust (NAS, 1977).

Arsenic levels in uncontaminated, nontreated soils seldom exceed 10 ppm (Adriano, 1986); however, agricultural uses (pesticides and fertilizers) can cause accumulation of 600 ppm or more. A survey of soils in the United States indicates that arsenic levels for normal soils ranged from 0.1 to 50 ppm (Bodek et al., 1988). Shacklette and Boerngen (1984) reported an average concentration of 7.2 ppm (the range was less than 0.1 to 97 ppm) for arsenic in United States surface soil.

Average concentrations of arsenic in Colorado soil collected from control soil locations and orchard field soils treated with an arsenic pesticide or defoliant ranged from 1.3 to 2.3 (total arsenic in ppm) and 13 to 69 ppm, respectively (Adriano, 1986).

Arsenic has variable chemical behavior, and is widely distributed in the environment, including in plant and animal tissues. It forms a variety of inorganic and organic compounds of different toxicity to aquatic and terrestrial organisms (EPA, 1984).

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A.3.1.1 Physical Properties

Arsenic (atomic no. 56) has an atomic weight of 74.922, a density of 5.73 g/cm^3, a melting point of 817°C, and a sublimation point of 613°C (Adriano, 1986; Eisler, 1988). Arsenic is a steel-gray, brittle, crystalline metalloid with three allotropic forms that are yellow, black, and gray. It tarnishes in air and when heated it is rapidly oxidized to arsenous oxide (AS_2O_3) with the odor of garlic.

Arsenic forms molecular compounds with most non-metals and metals and forms stable organic compounds in both its trivalent and pentavalent states. The most common oxidation states are -III, 0, III, and V. The most important compounds are white arsenic (As_2O_3), the sulfide Paris Green [3 $Cu(AsO_2).Cu(C_2H_3O_2)_2$], calcium arsenate ($As_2Ca_3O_8$, and lead arsenate ($PbHAsO_4$). The last three compounds are used as agricultural pesticides and poisons (Adriano, 1986).

A.3.1.2 Chemical Properties

The chemical speciation of an element in soil, water, and sediment is important for understanding the environmental chemistry of the element. The speciation affects mobility, bioavailability, and potential toxicity of the metal. Factors that influence speciation of metals in water include solubility of compounds, oxidation state of the element, availability of complexing agents, complex formation, ion-pair formation, adsorption or desorption onto particulate material, and redox and pH conditions of the environment and biochemical processes (Fergusson, 1990; Hood, 1985). Arsenic can undergo a variety of chemical reactions. This behavior is demonstrated by its acid behavior as As(III) and base behavior in compounds such as R_3As, where R is an alkyl or aryl group (Moore and Ramamoorthy, 1983). Arsenic is assumed to react similarly to phosphorous in soils, since both belong to the same chemical group, and have comparable dissociation constants for their acids and solubility products for their salts (Adriano, 1986).

The availability of arsenic in soil is related to soil properties such as pH and reduction–oxidation potential (Eh). In general, as the soil pH increases, the amount of available

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arsenic will decrease. In contrast, the amount of available arsenic will increase as soil Eh decreases (Fergusson, 1990).

In strongly reducing environments, elemental arsenic and As(III) can exist, but arsenate(V) is the stable oxidation state in aerobic environments. Under moderately reducing conditions, such as flooded soils, arsenite(III) may be the dominant form (Adriano, 1986). The reduced state of As(III) has been reported to be 4 to 10 times more soluble in soils than the oxidized state (V), thereby suggesting that the oxidation state of arsenic influences its soil sorption capacity.

Arsenic occurs in soils mainly as arsenate (AsO_4^{3-}) under oxygenated conditions. This species can be irreversibly bound onto clays, iron, manganese oxides/hydroxides, and organic matter. The extent of sorption is dependent upon concentration, time, and manganese content of the soil. In acidic soils, the main forms of arsenic are aluminum and iron arsenates ($AlAsO_4$ and $FeAsO_4$), respectively. Removal of iron or aluminum amorphous components from soils markedly reduces the arsenate sorption ability. The predominant form of arsenic in calcareous soils is $Ca_3(AsO_4)_2$ (Adriano, 1986; Ferguson, 1990; Fergusson and Gavis, 1972).

Arsenic in surface soils is fairly immobile; however, it has been shown that appreciable arsenic concentrations can leach downward with water, especially in coarse textured soils. An average half-life of 6.5 $\pm$ 0.4 years was observed for arsenic in the upper 20 centimeters (cm) of soil in the Netherlands (Adriano, 1986). In general, arsenic will leach in sandy soils and will persist in soils of low sorption capacity.

The soluble (extractable) arsenic fraction may provide some indication of its phytoavailability and mobility in soils. In general, the total concentration of arsenic in soil does not accurately reflect its phytoavailability. The extractable portion of arsenic is correlated with effects on plant growth, whereas the total concentration of arsenic is not (Adriano, 1986).

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In general, the most prevalent forms of arsenate compounds in soil occur in the following sequence:

Iron (Fe)-As > > Calcium (Ca)-As > Al-As> water-soluble arsenic

Aluminum (Al)-As and Fe-As are the dominant forms of arsenic in acid soils (Adriano, 1986).

Arsenic can undergo eight electron reductions from the V to the -III state and occurs in the +V, +III, 0, and -III states in aquatic systems. The +V, +III, and methylated species of arsenic occur in natural waters. Arsenic generally occurs in four oxidation states (V, III, 0, and -III) in water depending upon the pH and Eh of the system. The trivalent state (+III) occurs at lower Eh values and under mild reducing conditions, whereas oxidation states 0 and -III occur only under strongly reducing conditions and are rare in the natural water environment. The dominant species in water tends to be arsenate (AS(V) as $HAsO_4^{2-}$). This species can be removed by the formation of insoluble arsenates of iron(III), chromium(III), and barium (Moore and Ramamoorthy, 1983).

Arsenic mobility in water increases in the trivalent oxidation state and arsenate tends to predominate over arsenite. The production of organoarsenic species in aqueous environments occurs (Ferguson and Gavis, 1972).

Arsenic levels are often correlated to iron levels in sediments and can occur in mineral forms of arsenic such as FeAsS and As_2S_3 in sediments (Aggett and O'Brien, 1985). Arsenic in solution is removed from water mainly by sorption on the hydrous iron oxides and released again when reduction of Fe(III) to Fe(II) occurs. Association with organic material appears to be insignificant for arsenic as compared to other metals. As(V) is more strongly sorbed onto sediments, but interconversion between As(III) and As(V) occurs depending on the anoxic/oxic conditions of the sediment (Aggett and O'Brien, 1985).

Concentrations of arsenic in sediment are often positively correlated with decreasing particle size, whereas organic matter content of sediments is less important in the sorption of arsenic

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than for some other metals (Forstner and Wittmann, 1983). Desorption of arsenic from contaminated sediments can be correlated to the reduction of Fe(III) to Fe(II). Anaerobic conditions can produce higher concentrations of As(III) as compared to aerobic conditions (Moore and Ramamoorthy, 1983).

A.3.1.3 Biological Fate

Both oxidative and reductive transformations of methanearsonates occur in soil; the mobility of arsenic is increased by the alkylation process. Bacteria accelerate the oxidation of arsenite to arsenate (Fergusson, 1990). Microorganisms in soils treated with inorganic and methylated forms of arsenic produce volatile arsenicals by a reductive and/or reductive and demethylative pathway. Approximately 14 to 15% of arsenic applied to soil can be lost through volatilization of alkyl arsines each year (Adriano, 1986). Methylation of arsenic by fungi, soil microorganisms, and bacteria can produce dimethyl and trimethylarsine (Hood, 1985).

In reduced environments, methanogenic bacteria can reduce arsenate(V) to arsenite(III) and methylated to methylarsinic acid(III) or dimethylarsinic (cacodylic) acid. These compounds may be further methylated (trimethylarsine,II-III) or reduced (dimethylarsine, -III) and may volatilize to the atmosphere with the formation of cacodylic acid through oxidation reactions (Adriano, 1986).

Food is generally the main source of arsenic for invertebrates, but most species also derive considerable amounts of arsenic directly from water. During initial exposure, uptake from water increases linearly with increasing water concentration (EPA, 1984). Thereafter, the rate of sorption decreases followed by a period of equilibrium. Uptake is also directly dependent on the metabolic rate of the invertebrate; therefore, low temperatures and the administration of metabolic inhibitors (i.e., cyanide) cause a reduction in tissue residues. Rate of uptake of different arsenic forms decreases in the order As(V), As(III), and disodium methyl arsenate (EPA, 1984).

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Arsenic does not usually accumulate in either freshwater or marine fish species (Moore and Ramamoorthy, 1983). Toxic inorganic arsenicals are rapidly transformed to organic arsenic in fish. Ingestion of arsenic by rainbow trout (*Onchorhynchus mykiss*) resulted in high levels of inorganic arsenic in various tissues within 6 hours of exposure. This was followed by a gradual increase in the proportion of organic arsenicals and a corresponding decrease in the amount of inorganic forms (Moore and Ramamoorthy, 1983).

Although arsenic may be transferred to invertebrates from food and water, there is no evidence for high rates of bioaccumulation in the majority of species (Moore and Ramamoorthy, 1983). EPA (1984) reported whole body bioconcentration factors of 3 and 17 for As(III), and 3 and 6 for As(V) were reported for two species of snail. *Daphnia* whole body bioconcentration factors of 10 and 4 were observed.

Arsenic is often found in plant and animal tissues. This is due to the varying physico-chemical properties reported for As(III) and As(V), respectively, following a 21-day exposure. It appears that the stable, soluble inorganic arsenites and arsenates are readily absorbed by the intestinal tract and muscle tissue (EPA, 1984).

Arsenate is excreted faster than arsenite, mostly through urine, because of its poor affinity for thiol groups. Thus arsenate is less toxic than arsenite and does not inhibit enzyme systems. However, arsenate inhibits adenosine triphosphate (ATP) synthesis by uncoupling oxidative phosphorylation and replacing the stable phosphoryl group, whereas arsenite inhibits thiol-dependent enzymes, and is retained in the body tissue proteins such as keratin disulfides in hair, nails, and skin (Moore and Ramamoorthy, 1983; Forstner and Wittmann, 1983).

A.3.2 Toxicity

In general, the toxicity of various arsenic species and compounds can be summarized by the following:

- Inorganic arsenicals are more toxic than organic forms.

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- Trivalent forms are more toxic than pentavalent forms.
- Early developmental stages of organisms are the most sensitive to arsenic.
- Inorganic arsenic can move across placental barriers.
- Biomethylation is the preferred detoxification mechanism of inorganic arsenic.
- Arsenic is bioconcentrated, but not biomagnified up the food chain.
- A decrease in crop yields is observed at concentrations ranging from 3 to 28 mg of water soluble soil arsenic/ℓ (or 25 to 85 mg total arsenic/kg soil).
- Sensitive bird species died following single oral doses of 17.4 to 47.6 mg arsenic/kg body weight.
- Adverse effects in mammals were noted for single oral doses of 2.5 to 33 mg As/kg body weight for acute exposure, whereas oral doses of 1 to 10 mg As/kg body weight caused effects for chronic exposures (EPA, 1984; Eisler, 1988).

The chemical form of arsenic is important in regard to its toxicity. The toxicity of the element decreases in the order As(III) is greater than As(V) is greater than organoarsenic (Fergusson, 1990).

Competition between metals is common and likely to occur. Competition may either be antagonistic or synergistic. Antagonism results in the combined adverse effect of the two elements being less than the sum of their separate effects. Synergism results in a combined effect that is greater than the sum of the two elements individual effects. The more common interaction is antagonism. Arsenic is antagonistic to phosphorous, manganese, and zinc in plants (Fergusson, 1990). Selenium and arsenic are antagonistic in several animal species (rats, dogs, swine, cattle, poultry) (NAS, 1977; Thatcher et al., 1985).

A.3.2.1 Aquatic Organisms

Toxicity to aquatic organisms varies with the valence state (Table A.3-1). Under similar conditions, arsenite is more toxic than arsenate. Although the exact mechanisms of arsenic toxicity are not known, the main pathological effect appears to be enzyme inhibition.

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Table A.3-1

Toxic Effects of Arsenic to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Plants					
Algae	Various	AS^{+3}	1.7	Toxic	NRCC, 1978
Algae	Various	AS^{+3}	2.3	95-100% mortality	EPA, 1980; EPA, 1984
Algae	Selenastrum capricornutum	AS^{+5}	0.69	EC_{50}	EPA, 1984
Invertebrates					
Cladoceran	*Daphnia magna*	AS^{+3}	0.63 - 1.32	MATC	EPA, 1984
Cladoceran	*Daphnia magna*	AS^{+5}	7.4	LC_{50} (96 hours)	EPA, 1980
Cladoceran	*Daphnia magna*	Total As	1.4	50% reproductive impairment	NRCC, 1978
Cladoceran	*Daphnia magna*	Total As	2.8	LC_{50}	NRCC, 1978
Stonefly	*Pteronarcys californica*	As^{+3}	38	LC_{50}	Johnson and Finely, 1980
Stonefly	*Pteronarcys dorsata*	As^{+5}	0.97	LC_{20}	Spehar et al., 1980
Zooplankton	*Various*	As^{+3}	0.4	No effect	NRCC, 1978
Zooplankton	*Various*	As^{+3}	1.2	Population reduction	NRCC, 1978

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Table A.3-1 (Continued)

Toxic Effects of Arsenic to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Vertebrates					
Marbled salamander	*Ambystoma opacum*	AS^{+3}	4.5	EC_{50}	EPA, 1984
Narrow -mouthed toad	*Gastrophyrne carolnensis*	As^{+3}	0.04	50% mortality of embryos	EPA, 1984
Rainbow trout	*Onchorynchus mykiss* (embryos)	As^{+3}	0.13	EC_{10}	EPA, 1984
Rainbow trout	*Onchorynchus mykiss* (adults)	As^{+3}	0.54	LC_{50}	EPA, 1980
Rainbow trout	*Onchorynchus mykiss* (adults)	As^{+5}	23 - 26.6	LC_{50}	Spehar et al., 1980
Brook trout	*Salvelinus 22fontinalis*	As^{+3}	15	LC_{50}	EPA, 1984

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Although data are limited, excretion through gills appears to be the main route of arsenic elimination in rainbow trout and presumably other species (Moore and Rammoorthy, 1983)

The LC_{50} of sodium arsenite and arsenic trioxide range from 0.05 to 59 mg/ℓ depending on age, species, and test conditions. While the LC_{50} of arsenate falls within a similar range (5 to 15 mg/ℓ), arsenic trisulfide and methylated arsenicals appear to be less hazardous to fish (Spehar et al., 1980). In general, total arsenic is acutely toxic at 1 to 50 mg/ℓ. Because the toxicity of arsenicals varies significantly with the oxidation state, such general data are of limited value. Toxicity of most organic arsenicals to aquatic organisms has not been extensively evaluated (EPA, 1984).

Rainbow trout appear to be the most sensitive fish species tested to both forms of arsenic. In a 96-hour static test using 2-month-old trout, acute effects were reported at 10,800 μg/ℓ of As(V) (Hale, 1977). The LC_{50} for adult rainbow trout exposed to As(III) under static conditions was 13,340 μg/ℓ (EPA, 1984).

Reported 48-hour EC_{50} values for invertebrates exposed to trivalent arsenic for static test conditions range from 812 μg/ℓ for a cladoceran (*Simocephalus serrulatus*) to 97,000 μg/ℓ for a midge (*Tantytarsus dissimils*). Toxicity data for pentavalent arsenic are limited to cladocerans. The lowest reported result of an EC_{50} was 840 μg/ℓ in static conditions for *Bosmina longirostris*. In a life-cycle exposure to As(III), a MATC of 914 μg/ℓ was reported for cladocerans. Cowell (1965) reported a significant population reduction in cladocerans and copepods when exposed to 2,320 μg/ℓ of As(III) for 1 week. Certain mayflies can tolerate 3 to 14 mg/ℓ of arsenic in water without adverse effects, while 10 to 20 mg/ℓ arsenic causes no adverse effects to certain dragon and damsel flies. In contrast, concentrations of 4.3 to 7.5 mg/ℓ will immobilize *Daphnia magna* (Buchanan, 1962; U.S.D.H., 1962; Lambou and Lim, 1970).

Toxicity can be modified by chemical properties of the surrounding media. Toxicity to aquatic plants and invertebrates generally decreases with an increase in the pH, reflecting the formation of higher arsenic oxidation states. The addition of phosphate also produces an

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antagonistic effect on toxicity to aquatic plants. This is due to the fact that phosphate inhibits biotransformations of arsenicals to more toxic states (Moore and Ramamoorthy, 1983). Exposure to As(III) resulted in 100% mortality in three species of algae and a submerged plant at a concentration of 2,320 $\mu g/\ell$ (Cowell, 1965).

Summary of Available Criteria

The Ambient Water Quality Criteria document (AWQC) (EPA, 1984) recommended that the 4-day average concentration of dissolved As(III) should not exceed 190 $\mu g/\ell$ more than once every 3 years on the average, and the 1-hour average concentration should not exceed 360 $\mu g/\ell$ more than once every 3 years on the average. Colorado State standards (CDH, 1991) indicate an acute criterion of 360 $\mu g/\ell$ and a chronic criterion of 150 $\mu g/\ell$.

No aquatic life criteria for As(V) have been established due to insufficient data.

A.3.2.2 Terrestrial and Riparian Organisms

There is no evidence that arsenic is essential for plant growth, although stimulation of root growth with small amounts of arsenic in solution culture was observed (Adriano, 1986). Arsenic may slightly stimulate growth of plants at low concentrations, but higher concentrations are phytotoxic. Results of foliar toxicity investigations indicate heavy metal toxicity to plants is As(III) = mercury (Hg) > cadmium (Cd) > thallium (Ti) > selenium (Se)(IV)]) > tellurium (Te)(IV) > lead (Pb) > bismuth (Bi) = antimony (Sb). The order is dependent upon soil properties and plant species. In addition, small yield increases have been observed at low levels of arsenic for tolerant crops such as corn, potatoes, and wheat (Adriano, 1986). Growth stimulation does not always occur, and arsenic exposure may result in the reduction of top growth. Uptake and translocation of arsenic by cotton plants grown in culture solution were influenced by the source of arsenic. Arsenic as As_2O_3 was readily taken up by the roots, but not translocated to the shoots; however, when cacodylic acid was applied, arsenic was translocated to the shoot and reproductive tissues (Adriano, 1986).

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Crops have differing degrees of tolerance to soil arsenic. Members of the bean family, rice, and most of the legumes are fairly sensitive to arsenic (Adriano, 1986). The sequence for crop tolerance to arsenic was snap beans < sweet corn < peas < potatoes (Adriano, 1986).

Arsenic uptake in plants appears to be passive with water. Arsenic is translocated to many parts of plants where most is concentrated in old leaves and roots (Bodek et al., 1988). Levels of arsenic considered toxic to plants range from 3 to 20 ppm in soil (Bodek et al., 1988). The lethal concentration of arsenic in solution to tomato plants is 120 to 150 mg/ℓ or 100 mg/ℓ in the plant; however, exposure to lower concentrations of arsenic resulted in smaller plants (McKee and Wolf, 1963).

Arsenic concentrations less than 0.02 mg/ℓ are considered to be minimally toxic to plants; concentrations ranging between 0.02 to 7.5 mg/ℓ are considered toxic (Kabata-Pendias and Pendias, 1984). Residue concentrations ranging from 1 to 1.7 μg/g in leaves were non-toxic, whereas leaf concentrations ranging from 5 to 20 μg/g resulted in toxicity (Kabata-Pendias and Pendias, 1984).

Visible signs of arsenic toxicity to plants include red-brown necrotic spots on old leaves, yellow-brown color of roots and growth reduction (Fergusson, 1990). Phytotoxicity symptoms include wilting of new-cycle leaves, followed by retardation of root and top growth of the plant. This process is often accompanied by root discoloration and necrosis of leaf tips and margins. Biochemical processes that are influenced by exposure to arsenic include competition for essential metabolites, occupation of sites for essential groups such as PO_4^{3-} and NO_3^-, and transpiration (Peterson and Girling, 1980).

Two studies reporting of arsenic effects on amphibians have been reported (Table A-3.1). For the narrow-mouthed toad (*Gastrophryne carolinensis*) in an early life stage test, an EC_{50} of 40 μg/ℓ As(III) was reported following a 7-day exposure (Birge, 1978).

The bioaccumulation and bioconcentration of arsenic in mammals, fish, birds, mosses, lichens, and algae are moderate, while it is high or very high to in molluscs, crustacea, and higher

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plants (Bodek et al., 1988). Biominification has been reported in terrestrial plants, carnivores, aquatic herbivores, and carnivores. Biomagnification has been reported at least once in terrestrial herbivores, aquatic herbivores, and aquatic carnivores (Bodek et al., 1988).

Data on the effects of arsenic to soil biota and insects are limited. In general, soil microorganisms are capable of tolerating and metabolizing relatively high concentrations (Eisler, 1988). Tolerant soil microbiota can withstand soil concentrations of up to 1,600 mg/kg; however, growth and metabolism were reduced at concentrations of 375 mg As/kg and at concentrations of 150 to 165 mg As/kg earthworms and protozoans were absent (NAS, 1977).

Arsenic affects the central nervous system and causes physiological distress in mammals and waterfowl. The sublethal effects of arsenic on the behavior of mallard ducklings was investigated. Day-old ducks received an untreated diet or a diet containing 10, 100, or 300 ppm arsenic as sodium arsenite. Ducks in the highest exposure group had significant effects on activity schedules, including increased time at rest and a decrease in alert behavior. Arsenic had no effect on feeding behavior (Whitworth et al., 1991).

Camardese et al. (1990) investigated the effects of arsenate on the growth and physiology of mallard ducklings. Mallards receiving 30, 100, and 300 ppm arsenic added as sodium arsenate in the diet demonstrated brain and liver accumulation as well as decreased brain adenosine triphosphate (ATP) in the 300 ppm exposure group. Ducklings exposed to the 300 ppm exposure group also demonstrated overall decreased growth (weight gain). These findings, in combination with observed altered behavior (increased resting time), suggest that concentrations of arsenic that have been found in aquatic plants (up to 430 ppm dry weight) could adversely affect normal duckling development. In another study with mallards, sodium arsenite in the diet had an established LD_{50} of 323 mg/kg body weight (Tucker and Crabtree, 1970).

Rats tolerate As(III) at 50 ppm in their diets, but at 300 ppm significant growth depression, chronic hepatitis, and cirrhosis of the liver occurred (Schroeder et al., 1968). Pentavalent

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arsenic is slightly less toxic than the trivalent form and inhibits enzyme systems such as alpha-glycerophosphate dehydrogenase and cytochrome oxidase. It also inhibits ATP synthesis by uncoupling oxidative phosphorylation in the liver mitochondria because of its ability to replace the stable phosphoryl group (Venugopal and Luckey, 1978). The maximum chronic tolerated level of arsenic in a grazing animal diet is 50 ppm (Bodek et al., 1988).

Reported oral LD_{50} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) in the literature include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Arsenic Trioxide	Mouse	43
Arsenic Trioxide	Rat	143
Arsenic Pentoxide	Rat	8
Potassium Arsenite	Rat	14
Calcium Arsenate	Mouse	794

Other reported oral toxicity values (Venugopal and Luckey, 1978) for arsenic exposure to laboratory animals include air LD_{100} of 50 mg/kg body weight for dogs exposed to arsenic trioxide, and mild toxicity for rabbits exposed to 15 mg/kg body weight arsenic trioxide.

While an association between arsenic inhalation exposure by humans and the incidence of cancer mortality has been observed (Axelson et al., 1978), there has not been consistent demonstration of arsenic carcinogenicity in test animals (IARC, 1980). There are some data to indicate that arsenic may produce tumors if retention time in the lung is increased (Pershagan et al., 1982; 1984).

Toxic effects of arsenic are different with various animal species and even between strains of the same species. Solutions of arsenic trioxide are many times more toxic than the dry powder. High purity compound was less of an irritant to the gastro-intestinal tract than a

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crude sample, possibly due to the presence of impurities such as antimony (Ming-Dean Luh and Henley, 1973).

In rats and mice, the LC_{50} over a 96-hour test period varied from 15.1 to 214 mg/kg of orally administered arsenic trioxide. Disturbances in muscular coordination, red blood cell count, and kidney function were reported in rabbits administered 2.5 ppm in the diet (Lambou and Lim, 1970).

Daily consumption of 6.48 grams (g) of lead arsenate for an unspecified period was not harmful to the cow. Doses of 1.3 to 56.7 g/day lead arsenate killed 18 of 31 chickens tested in 24 hours (Lambou and Lim, 1970). The lethal dose of arsenic in animals is estimated to be 20 mg/kg body weight. The reported toxic dose of arsenic (g) (Ming-Dean Luh and Henley, 1973) for various animals includes:

- Fowl — 0.05 to 0.10 (approximately 62 mg/kg body weight).
- Dog — 0.10 to 0.20 (approximately 5 mg/kg body weight).
- Swine — 0.50 to 1.0 (approximately 8 mg/kg body weight).
- Sheep, goat, and horse — 10.0 to 15.0 (approximately 167 mg/kg body weight).
- Cow — 15.0 to 30.0 (approximately 30 mg/kg body weight).

The above values in grams were converted to mg/kg body weight by dividing by an average body weight for these animals (Sax, 1984).

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A.4 BARIUM

A.4.1 Environmental Chemistry

Barium (Ba) is a reactive metal that is not found free in nature but exists as a number of salts. Barium occurs in nature chiefly as the mineral barite witherite ($BaCO_3$). The mineral forms are relatively insoluble in water, have high melting and boiling points, and have very low vapor pressures (Preisman, 1964; Kirkpatrick, 1978). Barium compounds occur in most geologic materials at levels between 300 and 500 ppm. Barium occurs at low levels in most surface waters and groundwaters with reported concentrations of less than 340 $\mu g/\ell$ (EPA, 1987). Barium ranks as the 14th most abundant element in the earth's crust, which has an average concentration of 500 ppm. Surficial soils in the United States have an average concentration of 440 ppm and a range of 100 to 3,000 ppm (Adriano, 1986). In freshwater, median barium concentrations are 10 $\mu g/\ell$ (range <3 to 150 $\mu g/\ell$) (Bowen, 1979).

A.4.1.1 Physical Properties

Barium (atomic no. 56) has an atomic weight of 137.3, a boiling point of 1,640°C, a melting point of 725°C and a specific gravity of 3.5 g/cm^3. Barium is a silvery-white metal that belongs to the alkaline earth group of elements and geochemically resembles calcium and strontium (Adriano, 1986; Kirkpatrick, 1978).

A.4.1.2 Chemical Properties

Barium is extremely reactive, decomposes in water, and readily forms insoluble carbonate and sulfate salts. Barium is generally present in solution in surface or groundwater only in trace amounts (Kirkpatrick, 1978). Large amounts will not dissolve because natural waters usually contain sulfate, and the solubility of barium sulfate is generally low. Barium is not soluble at more than a few parts per billion (ppb) in water that contains sulfate at more than a few parts per million (Sax, 1975).

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In natural waters, barium forms insoluble carbonate or sulfate salts and is usually present at concentrations of less than 1 mg/ℓ. Soluble forms of barium are not very toxic, but soluble barium salts are highly toxic following acute exposure (Adriano, 1986).

A.4.1.3 Biological Fate

Data were unavailable in the literature reviewed.

A.4.2 Toxicity

A.4.2.1 Aquatic Organisms

Data were unavailable in the literature reviewed.

Summary of Available Criteria

There are insufficient data for the development of federal or state criteria for the protection of freshwater aquatic life.

A.4.2.2 Terrestrial and Riparian Organisms

Barium has a strong, prolonged stimulant action on all forms of muscle, including that of the heart, where it causes irregular contraction followed by arrest in systole; on the gastro-intestinal tract, where it causes vomiting and diarrhea; and on the bladder. Its action on the spinal cord and medulla causes violent tonic and clonic spasms, and in large quantities, paralysis (Browning, 1969; Integrated Risk Information System, 1992).

Median lethal doses (in mg/kg body weight) for ingested barium chloride were reported (Schroeder et al., 1972; Syed and Hoseain, 1972; Tardiff et al., 1980) as:

- Mice — 7 to 29.
- Dogs — 90.

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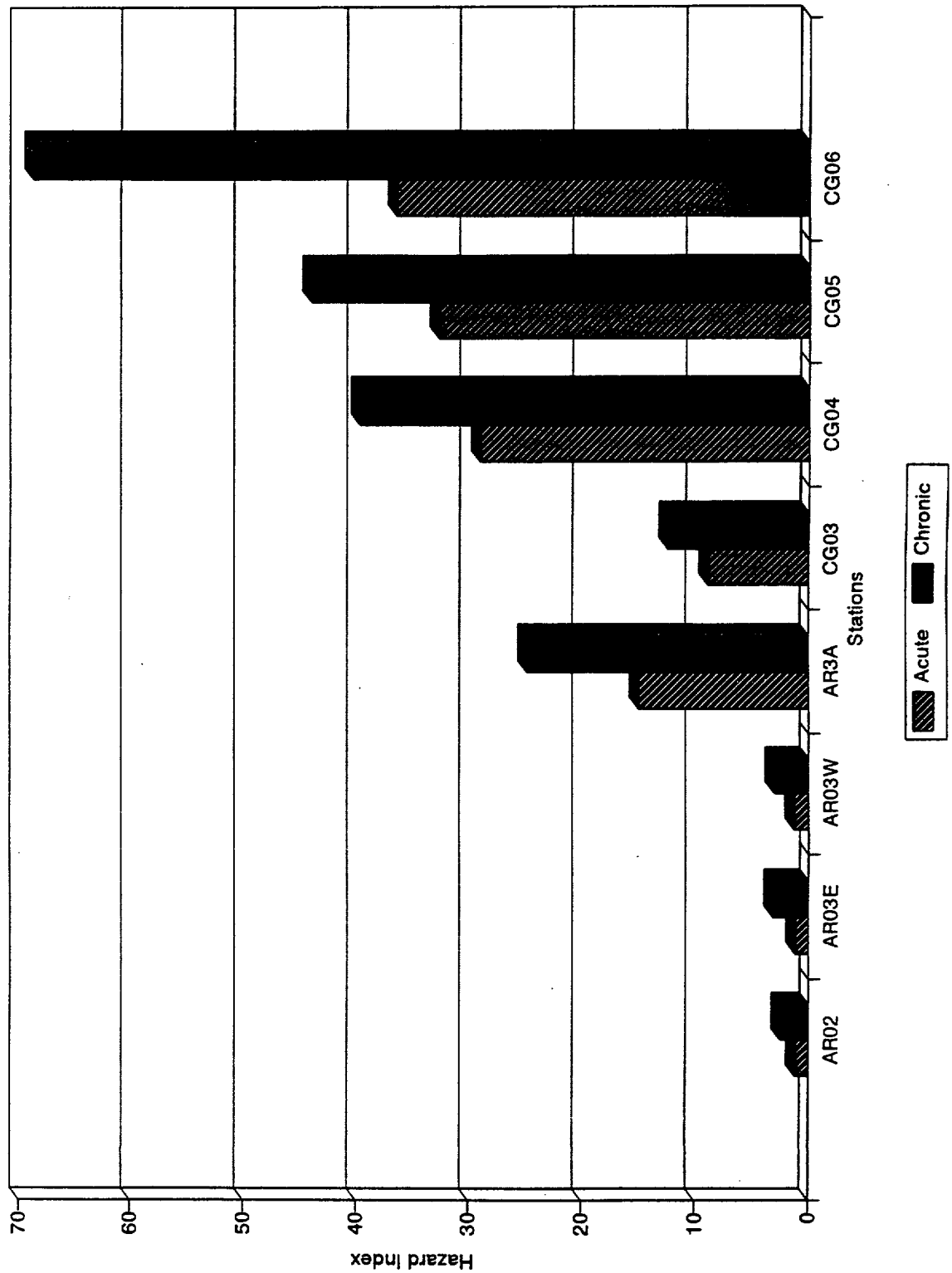
Figure 5-14

Hazard Indices Based on RME Concentrations in Surface Water

July 1992 through August 1993 Data



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results indicate that California Gulch remains a threat to the aquatic ecosystem in the Arkansas River, and that California Gulch is incapable of supporting a healthy aquatic ecosystem as would be expected in a tributary of its size. Data for the other tributaries (i.e., Lake Fork, Halfmoon Creek) were not collected.

5.3 WEIGHT OF EVIDENCE FOR THE AQUATIC RISK CHARACTERIZATION

There were two types of data utilized in the risk assessment:

- Abiotic data.
- Biological data.

The abiotic data were compared to EPA AWQC to obtain the following endpoints:

- HQs exceeding 1, or exceeding those observed at reference areas.
- Hazard indices exceeding 1, or exceeding those observed at reference areas.

The biological data were compared to reference areas to determine if there was a significant difference or detectable trend. The biological data were collected for the measurement endpoints in Table 1-2.

Tables 5-9 through 5-12 summarizes the findings of the risk assessment, and indicate whether or not there is an ecological risk based on the data available for each station.

The results indicate that the California Gulch site does present a hazard to the aquatic ecosystem. Despite the Yak Tunnel treatment facility, high metals concentrations continue to flow into the Arkansas River. The data indicate metal concentrations in water increase downgradient from the Yak Tunnel discharge, although the risk assessment did not determine if the cause of the metal pollution was from existing sediments or inputs from terrestrial sources.

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Table 5-9

Assessment Endpoint: Productive Capability of California Gulch

Station	Invertebrate Community Status	Invertebrate Density	Invertebrate Diversity	Periphyton Density	Periphyton Diversity	Periphyton Community Status	Surface Water Bioassay	Sediment Toxicity Bioassay	HI Sediment Pore Water (Avg and Max)	HI 1991 Surface Water (Avg and RME)	HI 1992-1993 Surface Water (Avg and RME)	Site Presents Ecological Risk
CG01	NA	NA	NA	NA	NA	NA	<AR2	NA	NA	a: HI AR01	NA	Possibly
CG02	NA	NA	NA	NA	NA	NA	NA	NA	NA	a: HI AR01	NA	Possibly
CG03	NA	NA	NA	NA	NA	NA	NA	NA	a: HI >1 c: HI >1	a: HI >AR01 c: HI >AR01	a: HI >AR02 c: HI >AR02	Yes
CG04	NA	NA	NA	NA	NA	NA	NA	NA	a: HI >1 c: HI >1	a: HI >AR01 c: HI >AR01	a: HI >AR02 c: HI >AR02	Yes
CG05	NA	NA	NA	NA	NA	NA	NA	NA	a: HI >1 c: HI >1	a: HI >AR01 c: HI >AR01	a: HI >AR02 c: HI >AR02	Yes
CG06	Severely impaired	<AR01	<AR01	>AR01	<AR01	Taxa different than AR01	100% Mortality >AR02	Mortality >AR02	a: HI >1 c: HI >1	a: HI >AR01 c: HI >AR01	a: HI >AR02 c: HI >AR02	Yes

a = acute
c = chronic
NA = Data Not Available

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Table 5-10

Assessment Endpoint: Extinction of Trout Species in the Arkansas River

Station	Trout Density Size Class	Trout Density	Prey Availability	Fish Bioassay	Invertebrate Bioassay	Surface Water 1991		1992-1993 HI Surface Water	Sediment Toxicity Bioassay	Presents Ecological Risk
						Avg HI	RME HI			
AR01	Some <15 CM Most >15 CM	—	—	NA	NA	—	—	—	NA	Control
AR02	NA	<AR01	Similar to AR01	No mortality	≤10% mortality	—	—	—	—	Control
AR03	NA	<AR01	Density and number <AR01	≤20% mortality	100% mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01	HI >AR02	Mortality >AR02	Yes
AR04	All >15 cm No recruitment	<AR01	Similar to AR01	No mortality	100% mortality	a: ≅AR01 c: <AR01	a: ≅AR01 c: >AR01	NA	NS	Yes
AR05	All >15 cm No recruitment	<AR01	Similar to AR01	No mortality	30-100% mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01	NA	NS	Yes

a = acute
c = chronic
NA = Data not available
NS = No significant difference
— = Not applicable

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Table 5-11

Assessment Endpoint: Abundance of Fish in the Arkansas River and its Tributaries

Station	Trout Density by Size Class	Trout Density	Prey Availability	Fish Bioassay	Surface Water 1991		Surface Water 1992-1993		Site Presents Ecological Risk
					Avg HI	RME HI	Avg HI	RME HI	
AR01	Some <15 CM Most >15 CM	—	—	NA	—	—			Control
AR02	NA	<AR01	Similar to AR01	No mortality	—	—			Control
AR03	NA	<AR01	Density and Number <AR01	≤20% mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01			Yes
AR04	All >15 cm No recruitment	<AR01	Similar to AR01	No mortality	a: ≅AR01 c: <AR01	a: ≅AR01 c: >AR01			Yes
AR05	All >15 cm No recruitment	<AR01	Similar to AR01	No mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01			Yes
CG06	No trout	No trout	No prey	NA	a: >AR01 c: >AR01	a: >AR01 c: >AR01	a: >AR01 c: >AR01	a: >AR01 c: >AR01	Yes

a = acute
c = chronic
NA = Data not available
— = Not applicable

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Table 5-12

Assessment Endpoints; Productive Capability of Arkansas River

Station	Invertebrate Community	Invertebrate Density/ No. Taxa	Fish Biomass	Periphyton Diversity	Periphyton Density	Periphyton Community Status	Sediment Toxicity Test	Invertebrate Surface Water Bioassay	Fish Bioassay	Surface Water 1991		Surface Water 1992-1993		Sediment		Site Presents Ecological Risk
										Avg HI	RME HI	Avg HI	RME HI	Avg HI	Max HI	
AR01	—	—	—	—	—	—	—	NA	NA	—	—	—	—	NA	NA	Control
AR02	—	—	<AR01	—	—	—	—	≤10% mortality	0% mortality	—	—	—	—	NA	NA	Control
AR03	Moderately Impaired	• Taxa <AR01 • Density <AR01	<AR01	<AR01	>AR02 <AR01	• Taxa <AR01 • Community dissimilar to AR01	Toxic (100%) mortality	100% mortality	≤20% mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01	a: >AR02 c: >AR02	a: >AR02 c: >AR02	a: >10 c: >100	a: >10 c: >100	Yes
AR04	Slightly impaired	• Taxa ≅AR01 • Density >AR01	<AR01	<AR01	>AR02 <AR01	• Taxa <AR01 • Community dissimilar to AR01	NS	100% mortality	0% mortality	a: ≅AR01 c: <AR01	a: ≅AR1 c: >AR01	NA	NA	a: >100 c: >1000	a: >100 c: >1000	Yes
AR05	Not impaired	• Taxa ≅AR01 • Density >AR01	<AR01	<AR01	>AR02 <AR01	• Taxa <AR01 • Community dissimilar to AR01	NS	30-100% mortality	0% mortality	a: >AR01 c: >AR01	a: >AR01 c: >AR01	NA	NA	a: >10 c: >100	a: >10 c: >1000	Possibly

A = Exposure point concentration exceeds acute AWQC
C = Chronic AWQC
NA = Not Available
NS = No significant difference
— = Not applicable

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- Rabbits — 170.
- Rats — 300 to 500.
- Horses — 800 to 1,200.

For the less soluble barium carbonate, the median lethal doses (mg/kg body weight) derived from the literature (Schroeder et al., 1972; Syed and Hoseain, 1972; Tardiff et al., 1980) were:

- Cats — 35 to 56.
- Pigs — 104 to 139.
- Mice — 200.
- Rabbits — 418 to 557.
- Chickens — 623 to 800.
- Rats — 1,480 to 1,500.

Perry et al. (1983) exposed rats to 1, 10, or 100 ppm barium in drinking water for up to 16 months (average daily barium doses of 0.051, 0.51, and 5.1 mg/kg respectively). There were no signs of toxicity at any of the barium dose levels.

Schroeder and Mitchener (1975a and b) exposed rats and mice to 5 mg/ℓ barium in drinking water for a lifetime (approximately 0.25 mg/kg/day for rats and 0.825 mg/kg/day for mice). No adverse effects were observed.

Tardiff et al. (1980) exposed rats to barium at 1, 10, 50, or 250 ppm barium in drinking water. Although the barium body burden increased with increasing dosage, no signs of toxicity were observed.

Reported LD_{50} values (mg/kg body weight) (Venugopal and Luckey, 1978) for various laboratory animals exposed orally to barium include:

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Compound	Test Animal	LD_{50} (mg/kg bw)
Barium Chloride	Mice	54
Barium Chloride	Rat	150
Calcium Carbonate	Rat	630
Barium Silicofluoride	Rat	175

Other reported (Venugopal and Luckey, 1978) LD_{100} toxicity values (mg/kg body weight) for laboratory animals exposed orally to barium salts include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Barium Fluoride	Guinea Pigs	350
Barium Carbonate	Guinea Pigs	121
Barium Acetate	Rabbits	236 to 815
Barium Chloride	Dogs	90

Chickens were found to tolerate 1 g Ba(II)/kg in their diet, whereas 2 g/kg caused slight depression of growth, and 4 g/kg caused substantial depression of growth but no increase in mortality. An estimate of 8 g/kg was the approximate LD_{50} in 4 weeks (Johnson et al., 1960).

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The risk assessment indicates that:

- California Gulch cannot function as an aquatic habitat for aquatic biota typically found in a stream of its size (Table 5-9).

- California Gulch is adversely affecting trout populations in the Arkansas River, especially at AR03, which is in close proximity to the confluence of California Gulch with the Arkansas River (Table 5-10 and 5-11). By AR05, the impacts to the Arkansas are reduced; acute toxicity of the water and sediments, and availability of prey for trout are similar to upgradient reference areas.

- Trout biomass, density, and reproduction (as indicated by young trout in the population) are affected from the confluence of California Gulch downstream beyond AR05. However, the data do indicate that impacts are becoming less by AR05 (Table 5-10 and 5-11).

- The productive capability of the Arkansas river is reduced due to inflows from California Gulch (Table 5-12). This is indicated by biological data collected from the site, laboratory bioassays, and the abiotic data relative to EPA AWQC. The impacts are most severe at AR03, and become less severe with increasing distance downgradient. By AR05, some endpoints indicate continuing stress on the aquatic ecosystem, but some indicate that the river is returning to the productive capability of AR01.

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A.5 CADMIUM

A.5.1 Environmental Chemistry

Cadmium (Cd) is an element of the transitional metal series that occurs widely in nature, usually in lead or zinc ores at levels below 1 ppm. Cadmium is commercially obtained as a by-product during the production of zinc. Commercial uses of cadmium and its compounds include metal plating, electronics, paints, and pigments (EPA, 1987).

A.5.1.1 Physical Properties

Cadmium (atomic no. 48) is a group IIB metal and has an atomic weight of 112.41, a boiling point of 765°C, a melting point of 321°C, and a specific gravity of 8.642 g/cm^3. Cadmium is a soft, bluish-white metal that is obtained as a by-product from the treatment of the ores of copper, lead, and iron. Cadmium has a valence of II and has properties similar to those of zinc. Cadmium forms both organic and inorganic compounds (Eisler, 1985).

Naturally occurring concentrations of cadmium in surface water and groundwater normally range from 1 to 10 $\mu g/\ell$ (EPA, 1987).

A.5.1.2 Chemical Properties

Cadmium is present in a variety of chemical forms in the environment. Some forms are insoluble in water, but in general, cadmium is relatively mobile in the aquatic environment. It is removed from aqueous media by complexing with organic materials and subsequent adsorption to sediment (Forstner and Wittmann, 1983).

The inorganic speciation of cadmium in water is similar to that of lead except that the free cation (hydrated) exists to relatively high pH. The hydroxy species only persists down to a pH of 8 to 9, below which the cadmium (Cd(II)) ion occurs. The cadmium carbonate ($CdCO_3$) form usually constitutes 4 to 21% of the cadmium species in freshwater, depending upon the

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pH and alkalinity. The higher the alkalinity, the greater the amount of $CdCO_3$ expected (Fergusson, 1990).

Under oxidizing conditions, cadmium is mobile and present as the hydrated Cd(II) ion. In contaminated environments, different cadmium species may exist such as cadmium sulfate ($CdSO_4$) and cadmium organic-ligand complexes. The amount of cadmium organic complex formation depends upon the amount of competing ions. The estimated percentage of total cadmium in the organic ligand complexes is between 10 to 80%, and unless the organic content of the water is high, cadmium organic complexation is of less importance than the free ion and other inorganic species (Fergusson, 1990).

The forms of cadmium that are commonly found in water and are not measured by the total recoverable procedure, such as the cadmium that is a part of minerals, clays, and sand, probably are forms that are less bioavailable and less toxic to aquatic life and will not be converted to the more toxic forms very readily under natural conditions (Eisler, 1985).

The availability of an element in soil is related to soil properties such as pH and Eh. In general, as the soil pH and Eh increases, the amount of available cadmium will increase (Fergusson, 1990). Cadmium moves relatively slowly through soil, but only limited information on soil transport is available.

The range of background cadmium concentrations in soil and plants are 0.01 to 1.0, and 0.1 to 0.8 ppm respectively (Bodek et al., 1988).

Cadmium tends to accumulate in the upper soil surface. The presence of zinc, copper, and selenium tend to inhibit cadmium uptake from soil. Similarly, soils that are moderately alkaline ($pH > 7$) inhibit cadmium bioavailability while low soil pH promotes cadmium biomobility. Cadmium generally assumes the Cd(II) valence and forms complex ions and organic chelates. Cadmium is also quickly adsorbed to clays and organic matter (Bodek et al., 1988).

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Cadmium tends to increase in concentration with decreasing sediment particle size and density. The species of cadmium in sediments is highly variable. In general, the most common feature is that organic material is a significant site for cadmium accumulation. In addition, the sorption capacity of cadmium increases with increasing pH and the release of cadmium from sediments is affected by acidity, Eh conditions, and complexing agents in the water. Sediments characterized by extremely high or low pH, and a moderate reducing capacity can immobilize cadmium. It has been found that the affinity of materials for cadmium is Mn > $Fe_{amorphous}$ > chlorite > $Fe_{crystalline}$ = illite = humic= > kaolinite > silica (Laxen, 1983).

Acid volatile sulfides in sediments bind cadmium and may be a predominant determinant of toxicity to benthic organisms because of its affect on bioavailability (DiToro et al., 1990). Iron and manganese oxide may also affect sediment toxicity.

A.5.1.3 Biological Fate

Cadmium is readily concentrated into plants; however, concentration effects may be confounded by phytotoxic effects. Uptake by aquatic weeds is variable, with BCFs ranging between 600 to 1,200 (EPA, 1980). BCFs for cadmium in freshwater aquatic life range from 164 to 4,190 for invertebrates and from 3 to 2,213 for fishes (EPA, 1980). Whole body bioconcentration factors for cadmium range from 33 for rainbow trout to 2,213 for mosquito fish (*Gambusia affinis*) (Eisler, 1985).

Cadmium bioaccumulates in mammals and other terrestrial and riparian organisms, particularly in the kidney and liver (EPA, 1981). Raccoons from an impacted area had 1 mg/kg cadmium (w/w) in liver (Herbert and Peterle, 1990). Bullfrogs have been observed to contain 5 mg/kg cadmium (Niethammer et al., 1985). At a smelter site, cadmium was highest in carrion insects, and high observed concentrations occurred in fungi and shrews (Beyer et al., 1985). Renal tissue of deer contained 372 mg/kg (Beyer et al., 1985).

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A.5.2 Toxicity

Zinc and cadmium display capacity for combining with sulfhydryl (SH) and imadizole-containing ligands; the stability of such complexes increases in the order of zinc is less than cadmium is less than mercury. Cadmium and mercury compete with and displace zinc in a number of zinc-containing metalloenzymes by irreversibly binding to active sites, thereby destroying normal metabolism (Forstner and Wittmann, 1983).

A.5.2.1 Aquatic Organisms

Cadmium is a cumulative poison to aquatic organisms because there is almost no excretion of this metal. Studies reporting cadmium toxicity to aquatic organisms demonstrate an inverse relationship between cadmium toxicity and water hardness (EPA, 1980). Cadmium toxicity to aquatic organisms is summarized in Table A.5-1.

Among invertebrates, cladocerans were the most sensitive species, and mayflies and stoneflies were the most resistant. However, insects and other invertebrates are sensitive during molting (Eisler, 1985). For fish species, salmonids appear to be the most sensitive to cadmium exposure (Eisler, 1985).

The acute LC_{50} for freshwater fish and invertebrates generally ranged from 100 to 1,000 $\mu g/\ell$; salmonids in general are much more sensitive than other fish (EPA, 1980). In both acute and chronic assays, the most sensitive invertebrate species was *Daphnia*. The *Daphnia* 48-hour EC_{50} was 9.9 $\mu g/\ell$ and the MATC was 0.15 $\mu g/\ell$.

Fish can detect and respond to cadmium in water at 0.2 $\mu g/\ell$ (McNicol and Scherer, 1991). Parameters associated with locomotor behavior or spatial selection were quantified for cadmium at concentrations of 0.2 to 256 $\mu g/\ell$. The responses varied between individuals, as some were attracted while others were repelled. The behavioral responses of lake whitefish (*Coregonus clupeaformis*) to contact with cadmium solutions of 0.2 to 256 $\mu g/\ell$ were monitored for preference and avoidance. Most fish demonstrated avoidance behaviors;

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Table A.5-1

Toxic Effects of Cadmium to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration ($\mu g/\ell$)	Effect	Reference
Invertebrates					
Midge	*Tanytarsus dissimils*	Cadmium chloride	3.8 (hardness 47 mg/ℓ $CaCO_3$)	LC_{50} (10 days)	Anderson et al., 1980
Mayfly	*Ephermra sp.*	Cadmium chloride	<3 (hardness 44-48 mg/ℓ $CaCO_3$)	LC_{50} (28 days)	Spehar et al., 1978
Cladoceran	*Daphnia magna*	Cadmium chloride	0.21 (hardness 103 mg/ℓ $CaCO_3$)	Chronic value	Chapman, 1978
Cladoceran	*Daphnia magna*	Cadmium chloride	0.44 (hardness 209 mg/ℓ CaCO3)	Chronic value	Chapman, 1978
Amphibians					
Toad	*Gastrophyrne carolinensis*	Cadmium chloride	40 (hardness 195 mg/ℓ $CaCO_3$)	LC_{50}	Birge, 1978
Vertebrates					
Rainbow trout	*Onchorhynchus mykiss*	Cadmium chloride	0.8 (hardness 23 mg/ℓ $CaCO_3$)	LC_{10} (200 hours)	Chapman, 1978
Rainbow trout	*Onchorhynchus mykiss*	Cadmium chloride	1.8 (hardness 31 mg/ℓ $CaCO_3$)	LC_{50}	Davies, 1976
Brook trout	*Salvelinus fontinalis*	Cadmium chloride	9.2 (hardness 187 mg/ℓ $CaCO_3$)	Chronic value	EPA, 1980
Mallard	*Anas platy-rhynchos*	Cadmium chloride	200 mg/kg diet	Reproductive impairment	White and Finley, 1978

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however, some showed attraction behaviors (McNicol and Scherer, 1991). Field studies show fish distribution, migration, and spawning may be influenced by metal gradients.

Summary of Available Criteria

Freshwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration of cadmium ($\mu g/\ell$) does not exceed the numerical value given by $e^{(0.7852[\ln(hardness)]-3.490)}$ more than once every 3 years on the average, and if the 1-hour average concentration ($\mu g/\ell$) does not exceed the numerical value given by $e^{(1.128[\ln(hardness)]-3.828)}$ more than once every 3 years on the average (EPA, 1980; EPA, 1985). The state standards are similar to EPA for chronic criteria. The acute EPA criterion corresponds to an acute state standard specific for trout, while a different acute equation exists for the state ($e^{(1.128[\ln(hardness)]-2.905)}$) for acute toxicity for other aquatic life (CDH, 1991).

A.5.2.2 Terrestrial and Riparian Organisms

Cadmium and some of its compounds are known to be carcinogenic in experimental animals exposed by injection or inhalation, but the carcinogenic effects appear to be absent when cadmium is administered orally (EPA, 1984). Cadmium has not been demonstrated to be a systemic carcinogen, but it has been shown to be a highly potent pulmonary carcinogen (EPA, 1984). Cadmium tends to accumulate in the kidneys.

Concentrations of cadmium less than 0.05 mg/ℓ in nutrient solution are considered non-toxic whereas concentrations between 0.2 and 9 mg/ℓ have produced toxic effects to plants. Concentrations of cadmium in leaves between 0.05 and 0.2 μg/g are non-toxic, whereas concentrations ranging from 5 to 30 μg/g resulted in toxic effects (Kabata-Pendias and Pendias, 1984). Visible signs of cadmium toxicity in plants includes browning of leaf margins, chlorosis, curled leaves, brown stunted roots, reddish veins and petioles, and reduction in growth (Fergusson, 1990). Biochemical processes within plants that are affected by cadmium include cell membrane permeability, thiol and -SH group bonding, phosphate group affinity, enzyme inhibition, respiration, photosynthesis, stomatal opening and transpiration (Jastow and

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Koeppe, 1980). Concentrations of cadmium in soil ranging from 5 to 700 ppm are considered toxic to plants (Bodek et al., 1988). Cadmium uptake by plant roots is passive and cadmium is highly mobile within plant tissues. The highest concentrations of cadmium are found in leafy portions, whereas concentrations are low in seeds (Bodek et al., 1988).

Soil arthropods differ widely in cadmium uptake and seasonal burden (Janssen and Bergema, 1991) due to effects of temperature and metabolic activity. Assimilation increased with temperature in both arthropod species tested (*Platynothrus peltifer, Orcheseila cineta*). Excretion increased in *Orcheseila cineta*, keeping the equilibrium concentration constant. In *Platynotrus peltifer*, there was no temperature effect on excretion, thus there was an increase in storage. Cadmium kinetics varies within soil faunal community and makes extrapolation between species difficult. Assimilation efficiency from diet ranged from 7.3 to 100% depending on species and temperature.

The bioaccumulation and bioconcentration of cadmium is high to very high in terrestrial and aquatic organisms (Bodek et al., 1988). Biomagnification has been recorded at least two times in terrestrial plants, aquatic herbivores and aquatic plants, and terrestrial herbivores and carnivores. Biominification has been reported in aquatic carnivores (Bodek et al., 1988). Marine and terrestrial animals have been shown to be abundant in the presence of a marine sewer outfall (Brown et al., 1977), although these animals were contaminated with high levels of cadmium, copper, and zinc (Brown et al., 1977). Health effects were not reported, nor were changes in ecosystem or community structure.

Birds are comparatively resistant to the biocidal properties of cadmium. Adult drake mallards (*Anas platyrhynchos*) fed up to 200 ppm cadmium in the diet for 90 days survived with no body weight loss (White and Finley, 1978). Laying hens fed 200 ppm in the diet exhibited no weight loss but decreased egg production was observed (White and Finley, 1978). Mallard ducklings exposed to 20 ppm cadmium in the diet exhibited an increase in glutamic pyruvictransaminase, decreased hemoglobin and decreased packed cell volumes (Cain et al., 1983). Cadmium has been shown to affect behavior in black ducks at 4 ppm in the diet (Heinz et al., 1983), resulting in hyperresponsiveness.

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In another experiment conducted by White et al. (1978), adult mallards were experimentally dosed with various levels of cadmium. No effects were observed at dietary levels of 20 ppm for 90 days; however, 200 ppm caused renal tubular necrosis and testicular atrophy by 60 days. Mallard ducklings fed 20 ppm dietary cadmium exhibited mild to severe kidney lesions by 8 to 12 weeks (Cain et al., 1983). Cadmium chloride in the diets of quail, pheasants, and ducks resulted in LC_{50} values of 2584, 767, and 5,000 ppm, respectively (Hill et al., 1975). Significant adverse effects were observed in ducks exposed to 200 ppm of cadmium in the diet for 90 days (Eisler, 1985). From the preceeding data, and using a dietary intake rate of 0.175 kg diet/kg body weight for chickens (Sax, 1984), the LC_{50} values for quail and pheasants are approximately 452 and 134 mg/kg body weight. An intake rate of 0.10 kg diet/kg body weight for ducks (Sax, 1984) results in an LC_{50} of 500 mg/kg body weight. The 200 ppm in diet of ducks converts to 20 mg/kg body weight. Using a directory intake rate of 0.1 kg diet/kg body weight (Sax, 1984) 20 ppm in diet was converted to approximately 2 mg/kg body weight.

Sublethal effects in birds are similar to those in other species and include growth retardation, anemia, and testicular damage (Hammons et al., 1978; Scheuhammer, 1987). Japanese quail fed 75 ppm cadmium in the diet developed bone marrow hypoplasia, anemia, and hypertrophy of both heart ventricles at 6 weeks (Richardson et al., 1974). Cain et al. (1983) investigated dietary exposure of 200 ppm cadmium for 12 weeks in mallards. Results indicated altered blood chemistry, and severe to mild kidney lesions.

The degree of cadmium toxicity to mammals varies widely and is influenced by external factors. Cadmium exposure can cause derangement in carbohydrate and mineral metabolism, in renal hepatic, testicular and prostate functions, and disturbs the integrity of the central nervous system (Venugopal and Luckey, 1978). The lowest oral dose (producing death), was 250 mg/kg body weight of cadmium (as cadmium fluoride) in rats; and 150 mg/kg body weight of cadmium (as cadmium fluoride) in guinea pigs (EPA, 1980). Laboratory rats chronically exposed to low cadmium concentrations had weight gain effects and reproductive effects (decreased litter size, survival of young and size of young [Herman et al., 1982]). *Microtus sp.* have been shown to have behavioral effects in the field when exposed to sludge-amended

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soil containing cadmium. Locomotor and exploratory behavior were affected, and females indicated hyperactivity and inability to habituate to new environments (Hall et al., 1990).

Chronic feeding of cadmium at low levels to rats, rabbits, lambs, pigs, calves, and poultry causes diminished growth and feed consumption. Rats tolerate 10 mg/ℓ cadmium (as cadmium chloride) in their drinking water, and show no visible symptoms of toxicity, but their longevity is reduced (Schroeder et al., 1963). From a water intake rate of 0.125 ℓ/kg body weight (Sax, 1984), the value of 10 ppm in drinking water for rats was converted to 1.2 mg/kg body weight. Loss of weight was observed in rabbits that were fed 40 to 44 mg cadmium/kg as cadmium chloride (Nomiyama et al., 1973). This is equivalent to approximately 1.2 mg/kg body weight based on dietary intake rates for rabbits of 0.03 kg diet/kg body weight (Sax, 1984). Growth retardation was observed in lambs fed 60 ppm cadmium (Doyle et al., 1974), in pigs fed 450 ppm cadmium (Cousins et al., 1973), and in calves fed 640 ppm cadmium (Powell et al., 1964). The maximum chronic tolerated level of cadmium in grazing animal diet is 0.05 mg/kg (Bodek et al., 1988).

Various toxicity values reported (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory animals exposed orally to cadmium salts include:

- LD_{50} of 88 for rats exposed to cadmium chloride.
- LD_{50} of 150 for guinea pigs exposed to cadmium fluoride.
- LD_{50} of 660 for rats exposed to cadmium succinate.
- LD_{100} of 70 for rabbits exposed to cadmium chloride.
- LD_{100} of 27 for dogs exposed to cadmium sulfate.

A.5.3 References

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A.6 COPPER

A.6.1 Environmental Chemistry

Estimates for the average crustal abundance of copper (Cu) in the earth range from 24 to 55 ppm; the United States has an average soil content of 25 ppm copper (Adriano, 1986). Copper occurs in natural waters primarily as the divalent cupric ion, in free and in complex forms. Concentrations of 1 to 10 $\mu g/\ell$ are usually reported for unpolluted surface waters in the United States (Boyle, 1979).

The background concentration of copper in soils and plants has been reported to range from 2 to 100 ppm for soils, and 4 to 30 ppm for plants (Bodek et al., 1988).

A.6.1.1 Physical Properties

Copper (atomic no. 29) has an atomic weight of 63.546, a melting point of 1,083°C, and a specific gravity of 8.96 g/cm^3 (Adriano, 1986). Copper is reddish colored with a bright metallic luster and is malleable, ductile, and a good conductor to heat and electricity.

Copper naturally occurs in the I and II oxidation states and is found in minerals such as cuprite, malachite, azurite, chalocopyrite, and bornite. Copper naturally forms sulfides, sulfates, sulfosalts, and carbonates.

A.6.1.2 Chemical Properties

Copper has two oxidation states, I (cuprous) and II (cupric). The amount of copper compounds and complexes that exist depend on pH, temperature, alkalinity, and concentrations of other chemical species. Generally, ionic copper is more soluble in low pH waters and less soluble in high pH waters. Cuprous copper is unstable in aerated water over the pH range of most natural waters (6 to 8) and oxidizes to the cupric state (Adriano, 1986).

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Several processes determine the fate of copper in the aquatic environment, including formation of complexes, especially with humic substances; sorption of hydrous metal oxides, clays, and organic materials; and bioaccumulation (Adriano, 1986). Copper has a strong affinity for hydrous iron and manganese oxides, clays, carbonate minerals, and organic matter. Sorption to these materials both suspended in the water column and in the sediment results in the reduction of dissolved levels.

In waters polluted with soluble organic material, complexation with organic ligands can occur, thus favoring the prolonged dispersion of copper in solution. The presence of organic acids also can lead to the mobilization of copper from the sediments to solution (Boyle, 1979). Sorption processes are efficient in scavenging dissolved copper and in controlling its mobility in natural unpolluted streams.

Applied or deposited copper will persist in soil due to its strong fixation by organic matter, oxides of iron, aluminum, manganese, and clay minerals. It is therefore one of the least mobile of the trace elements. Jones (1971) observed virtually no downward movement of copper on silty and clayey soils and only slight movement (1 to 3 cm) in sandy soils with a low cation exchange capacity. Several investigators have observed that copper is the most extensively complexed metal by humic materials (Adriano, 1986).

Copper tends to accumulate in the litter overlying the soil profile or within the top few centimeters of the soil (Adriano, 1986). The solubility, mobility, and availability of copper to plants is dependent upon soil pH. At pH levels above 7, copper availability is drastically reduced (Adiano, 1984). Copper in the oxide-bound and residual forms is also relatively unavailable to plants. The majority of available copper resides in the organically bound fraction.

Copper acts antagonistically with nitrogen, phosphorous, and zinc in soils for plant uptake (Adriano, 1986).

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A.6.1.3 Biological Fate

Since copper is an essential nutrient, the significance of copper residues in tissues has not been established, and few tests have been completed for the purpose of determining BCFs. Results of several investigations reveal a broad range of BCFs from zero for the bluegill *(Lepomis macrochirus)* to 2,000 for the alga *(Chlorella regularis)*. The highest BCFs for copper are those for bivalve mollusks. Shuster and Pringle (1969) found that the eastern oyster *(Crassostrea virginica)* could concentrate copper by factors of 28,200 times during a 140-day continuous exposure to 50 $\mu g/\ell$.

In general, copper is accumulated by plants and animals, although apparently it is not generally biomagnified. Since copper is strongly bioaccumulated, biological activity is a major factor in determining the distribution and occurrence of copper in the ecosystem (EPA, 1980).

A.6.2 Toxicity

A.6.2.1 Aquatic Organisms

Copper is a minor nutrient for both plants and animals at low concentrations, but causes toxicity to aquatic life at slightly higher concentrations (EPA, 1980). In general, the acute toxicity to several families of freshwater fish is inversely related to water hardness (EPA, 1980). The toxicity of copper to aquatic life has been shown to be related primarily to activity of the cupric (Cu(II)) ion, and possibly to some of the hydroxy complexes (EPA, 1980). The proportion of copper present as the free cupric ion is generally low and may be less than 1% in eutrophic waters where complexation can occur. Various copper complexes and precipitates appear to be largely non-toxic and tend to reduce toxicity attributable to copper (Andrew, 1976).

Most of the available toxicity information has been derived using four salmonid species (*Oncorhynchus mykiss, Salvelinus fontinalis, Salmo clarkii, and Oncorhynchus kisutch*), fathead and bluntnose minnows, and the bluegill sunfish. The acute values range from 7.24 $\mu g/\ell$ for *Daphnia puligaria* in soft water to 10,200 $\mu g/\ell$ for the bluegill in hard water. Summarized

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results from many of these studies that were determined applicable for the purposed of this ecological evaluation are presented in Table A.6-1.

In acute studies, the most sensitive species was *Daphnia magna*; the EC_{50} was 6.5 $\mu g/\ell$. Acute data for aquatic life are reported in Table A.6-1. The lowest MATC for brook trout (*Salvelinus fontinalis*) was 3.873 $\mu g/\ell$ (EPA, 1980).

An application factor of 0.04 reported in a chronic study (Benoit, 1975; Sandheinrich and Atchison, 1989), and an LC_{50} of 3,750 $\mu g/\ell$ were used to derive a MATC of 150 $\mu g/\ell$ (Sandheinrich and Atchison, 1989). Bluegill were exposed to four concentrations of copper (5 [control], 31, 180, and 1,710 $\mu g/\ell$) at 157 mg/ℓ calcium carbonate ($CaCO_3$) and behavioral responses were tested (Sanheinrich and Atchison, 1989). In general, control fish consumed more prey than treated fish. The amount of handling time of each prey item increased with copper exposure, and was the most sensitive behavioral response measured in this study. In the Sanheinrich and Atchison (1989) study, behavior was affected at 31 $\mu g/\ell$, suggesting feeding and growth may be adversely impacted by chronic exposure to concentrations at the AWQC level (Sanheinrich and Atchison, 1989). Growth, reproduction, and survival were reduced at 162 $\mu g/\ell$ at 45 mg/ℓ $CaCO_3$ in another chronic study (Benoit, 1975), leading to the development of a MATC of 21 to 40 $\mu g/\ell$ for these endpoints.

The primary mechanism of copper toxicity in aquatic organisms is osmoregulatory disruption and failure (Rand and Petrocelli, 1985). Other effects of copper include damage to corneas of larval striped bass (*Morone saxatilis*) (Bodammer, 1985); and to gills, olfactory tissue and lateral line systems (Klima and Applehans, 1990; Baker, 1969; Gardner and LaRoche, 1973).

Chronic copper toxicity has been tested on a wide range of invertebrate and plant species. Results of these investigations reveal that most invertebrate and plant species seem to be equally sensitive to the chronic toxicity of copper. It was determined that the protection of animal species would provide adequate protection of plant species (EPA, 1980).

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Table A.6-1

Toxic Effects of Copper to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration ($\mu g/\ell$)	Effect	Reference
Invertebrates					
Cladoceran	*Daphnia magna*	Copper chloride	30 (hardness 99 mg/ℓ $CaCO_3$)	LC_{50}	Anderson, 1948
Cladoceran	*Daphnia magna*	Copper chloride	12.7 (hardness 120 mg/ℓ $CaCO_3$)	LC_{50}	Anderson, 1948
Cladoceran	*Daphnia magna*	Copper chloride	69 (hardness 207 mg/ℓ $CaCO_3$)	LC_{50}	Chapman et al., Manuscript
Vertebrates					
Rainbow trout	*Onchorhynchus mykiss*	Copper sulfate	176 (hardness 101 mg/ℓ $CaCO_3$)	LC_{50}	Howarth & Sprague, 1978
Rainbow trout	*Onchorhynchus mykiss*	Copper sulfate	33.1 (hardness 99 mg/ℓ $CaCO_3$)	LC_{50}	Howarth & Sprague, 1978
Rainbow trout	*Onchorhynchus mykiss*	Copper sulfate	48.1 (hardness 100 mg/ℓ $CaCO_3$)	LC_{50}	Howarth & Sprague, 1978
Rainbow trout	*Onchorhynchus mykiss*	Copper sulfate	232 (hardness 370 mg/ℓ $CaCO_3$)	LC_{50}	Howarth & Sprague, 1978
Rainbow trout	*Onchorhynchus mykiss*	Copper sulfate	70 (hardness 366 mg/ℓ $CaCO_3$)	LC_{50}	Howarth & Sprague, 1978
Rainbow trout	*Onchorhynchus mykiss*	Copper chloride	221 (hardness 194 mg/ℓ $CaCO_3$)	LC_{50}	Chakoumakos et al., 1979
Rainbow trout	*Onchorhynchus mykiss*	Copper chloride	165 (hardness 194 mg/ℓ $CaCO_3$)	LC_{50}	Chakoumakos et al., 1979
Cutthroat trout	*Salmo clarki*	Copper chloride	232 (hardness 204 mg/ℓ $CaCO_3$)	LC_{50}	Chakoumakos et al., 1979
Cutthroat trout	*Salmo clarki*	Copper chloride	91 (hardness 160 mg/ℓ $CaCO_3$)	LC_{50}	Chakoumakos et al., 1979

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Field studies have indicated predator–prey interaction effects due to copper (Clements et al., 1989). Two caddisflies (*Chimarra sp.* and *Hydropsyche morosa*) were more likely to succumb to predation by the stonefly (*Paragnetina media*) at a nominal concentration of 6 $\mu g/\ell$ (observed concentration 5.5 $\mu g/\ell$) than in control streams. The number of caddisflies in stonefly guts were significantly greater than in control streams. Stoneflies tend to be more resistant to the toxicity of heavy metals than their prey (mayflies, chironomids, and caddisflies), thus potentially resulting in greater impacts on predator–prey relationships in metal impacted systems.

Summary of Available Criteria

EPA (1980) recommends that the 4-day average concentration of copper ($\mu g/\ell$) should not exceed the value given by $e^{(0.8545[\ln(hardness)]-1.465)}$, and the 1-hour average concentration should not exceed the value given by $e^{(0.942[\ln(hardness)]-1.464)}$. At a water hardness of 165 mg/ℓ $CaCO_3$ this equates to a 4-day average of 18 and 28 $\mu g/\ell$ more than once every 3 years, respectively (Sandheinrich and Atchison, 1989). The State of Colorado chronic standard is the same as the EPA 4-day average concentration. The State acute standard is $0.5e^{(0.942[\ln (hardness)]-0.7703)}$ (CDH, 1991).

A.6.2.2 Terrestrial and Riparian Organisms

Copper is one of the micronutrients that include zinc, copper, manganese, iron, boronmolybdenum, and chlorine essential for normal plant nutrition. Approximately 5 to 20 ppm in plant tissue is adequate for normal growth while less than 4 ppm is considered deficient and less than 20 ppm is toxic (Jones, 1972).

Copper is actively and passively taken up by plant roots. Concentrations less than 2 to 5 ppm in soil are considered deficient for plant survival, while concentrations ranging from 20 to 50 ppm are considered toxic (Bodek et al., 1988).

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The bioaccumulation and the bioconcentration potential for copper is high to very high for certain aquatic and terrestrial organisms. Copper has been reported to biomagnify in terrestrial herbivores and aquatic plants, whereas biominification has been observed in terrestrial plants, and aquatic plants, herbivores, and carnivores (Bodek et al., 1988).

Rats and pigs can tolerate 200 ppm copper in their diet. This is approximately 15 mg/kg body weight using a dietary intake rate of 0.075 kg diet/kg body weight (Sax, 1984). Pigs reveal no signs of copper toxicity when fed diets containing 250 ppm copper (Suttle and Mills, 1966). However, without substantial zinc and iron in the diet, 250 ppm dietary copper may be harmful to pigs (Gipp et al., 1974).

Sheep are very susceptible to copper toxicosis, and poisoning may be acute or chronic. Acute poisoning is caused by direct action of copper salts on the gastrointestinal tract, resulting in gastroenteritis, shock, and death. The acute toxic dose is about 200 mg/kg and is usually obtained through an accidental overdose. Cattle are more resistant to acute effects of copper in the diet than sheep or swine (Jones, 1972). In sheep, a sustained copper dietary level of 10 to 15 ppm can lead to hepatic copper accumulation and sudden hemolytic crisis. The toxic effects of chronic copper poisoning in sheep and calves are comparable. Calves fed 20 to 125 ppm copper develop jaundice after 20 weeks (Doherty et al., 1969). The maximum chronic tolerated level in grazing animal diet for copper is 25 to 300 mg/kg (Bodek et al., 1988).

Observed LD_{50} values (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory rats include:

Compound	LD_{50} (mg/kg bw)
Copper Chloride	140
Cupric Perchlorate	29
Cupric Nitrate	940
Copper Sulfate	960

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Observed lethal values for laboratory mice include 33 mg/kg body weight, while an oral LD_{100} of 420 mg/kg body weight for goats was reported (Venugopal and Luckey, 1978).

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A.7 IRON

A.7.1 Environmental Chemistry

Iron (Fe) occurs at 4.6% by weight in igneous rock, and 4.4% by weight in sedimentary rock (Cerundolo et al., 1988). Primary minerals release iron that precipitates as ferric oxides and hydroxides, which are major sources of iron-containing minerals in soil and natural water. Throughout the pH range of most natural waters, the ferric hydroxide complexes are expected to dominate the soluble iron fraction (Cerundolo et al., 1988). Iron occurs at a median concentration of 500 $\mu g/\ell$ in most freshwater. The range from eight sources was 10 to 1,400 $\mu g/\ell$ (Bowen, 1979). The median concentration of iron in soil is 40,000 mg/kg; the range from 11 sources was 2,000 to 550,000 (Bowen, 1979).

A.7.1.1 Physical Properties

Iron is a group VIII metal. In the environment, it is commonly found in soluble forms such as soluble organic fractions. Factors affecting iron uptake by plants include acidic, waterlogged environments and the presence of organic complexes and chelates which all promote uptake.

Iron will readily precipitate as oxides and hydroxides, and can be chelated where organic matter is high. Iron activity can be inhibited by antagonistic activity of other metals and phosphorous (Bodek et al., 1988)

A.7.1.2 Chemical Properties

The inorganic chemistry of iron is dominated by compounds in the ferrous(II) and ferric(III) oxidation states. In aquatic media, iron can undergo chemical reactions including precipitation, speciation, redox, chelation, and microbial interactions resulting in oxidation and photochemical reactions (Adriano, 1986). The chemical reactions in water depend upon the pH and redox potential of the water. In most bodies of water, iron is expected to be present largely in the form of suspended particles and sediments, although some small amounts of

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dissolved iron may occur as Fe(II) and Fe(III). Small quantities of iron also exist in colloidal form, generally as ferric oxyhydroxides. The residence time of iron in aquatic media has been estimated to be greater than 140 years (EPA, 1981).

Iron is present primarily in the Fe(III) state in most soils, although Fe(II) may be predominant in oxygen-deficient soils (flooded soils and soils rich in organic matter). The fate of iron compounds in soils is primarily determined by chemical and microbiological reactions in soils and the capacity of soils to sorb iron-organic complexes. In most soils, iron is not mobile (Adriano, 1986). Both biological and chemical reactions may cause precipitation of iron in soils; however, small amounts of iron are transported through soil in the form of colloidal ferric oxyhydroxides and in solution as iron-organic chelate (EPA, 1981). Soil pH is one of the most important regulators of iron mobility, and lower pH favors mobility.

In aqueous environments, ferric and ferrous iron hydrolyze to give various hydrolysis species. The reactions are highly dependent on pH. In natural waters, iron can be present as several species, forming complexes with both inorganic and organic materials. With organic substances, ferric iron forms stronger complexes than does ferrous iron and also forms complexes with chloride, fluoride, sulfate, and phosphate (Bodek et al., 1988).

The main variables affecting the solubility of iron include pH and concentrations of the dissolved carbon dioxide and sulfur species. In moderate oxidizing conditions above a pH of 5 ferric hydroxide forms. In strong alkaline and oxidizing conditions, Fe(VI) is the most stable species of iron (Bodek et al., 1988).

The background concentration for iron in soil and plants has been reported to range from 7,000 to 550,000 mg/kg in soil, and 3 to 300 mg/kg in plants (Bodek et al., 1988).

A.7.1.3 Biological Fate

Data were unavailable in the literature reviewed.

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A.7.2 Toxicity

A.7.2.1 Aquatic Organisms

Data were unavailable in the literature reviewed.

Summary of Available Criteria

There are insufficient data for the development of federal criteria for the protection of freshwater aquatic life. The State of Colorado has promulgated a chronic criterion for iron of 1,000 (total recoverable iron) (CDH, 1991).

A.7.2.2 Terrestrial and Riparian Organisms

Iron is an essential nutrient in plants and animals and iron deficiency in plants is common. There has been no reported toxic level of iron in soil to plants (Bodek et al., 1988).

Reported oral LD_{50} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) for iron exposure to laboratory animals include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Ferrous Chloride	Rats	984
Ferric Chloride	Rats	900
Ferric Nitrate	Rats	3,250
Ferrous Sulfate	Mice	1,170
Ferrous Sulfate	Rats	1,480
Ferrous Ammonium Sulfate	Rats	3,250
Ferrous Gluconate	Rats	4,600

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Other oral LD_{100} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) established for the exposure of laboratory animals to iron compounds include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Ferrous Chloride	Rabbits	890
Ferrous Sulfate	Rats	2,130
Ferrous Sulfate	Rats	2,780
Iron Carbonyl	Guinea Pigs	36
Iron Carbonyl	Rabbits	18

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A.8 LEAD

A.8.1 Environmental Chemistry

Lead (Pb) is the most abundant metal in the earth's crust; the concentration ranges from 13 to 16 ppm. Lead occurs in rocks as a discrete mineral or it can replace potassium, strontium, barium, cadmium, and sodium and become fixed in a mineral lattice (Adriano, 1986).

Lead concentrations in agricultural soil have been reported to range from 2 to 200 ppm; a range of 5 to 25 ppm is typical for soils remote from human activity. Much higher values can be expected in areas near densely populated centers, near industrial facilities, and in areas of geochemical deposits. A mean value of 19 ppm for the United States has been reported; a similar mean Canadian average was reported of 20 ppm. Lead is present in all soils at levels that range from less than 1.0 ppm in normal soils to well over 10% in ore materials (Adriano, 1986).

Other reported background concentrations of lead in soils and plants range from 2 to 200 ppm in soil, and 0.1 to 10 ppm in plants (Bodek et al., 1988).

A.8.1.1 Physical Properties

Lead (atomic no. 82) has an atomic weight of 207.19, a melting point of 327.5°C, and a specific gravity of 11.35 g/cm^3. Lead has two oxidation states, II or IV (Adriano, 1986). Lead is a bluish-white metal with a bright luster and is soft, malleable, ductile and a poor conductor of electricity.

A.8.1.2 Chemical Properties

Lead resembles the divalent alkaline earth group metals in chemical behavior more than its own Group IVA metals. Lead compounds produced industrially are quite soluble, but the lead compounds found in the environment are not usually mobile in normal groundwater and surface water because the lead leached from ores either becomes adsorbed to oxides or

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combines with carbonate or sulfate ions to form insoluble compounds (EPA, 1980). In the aquatic environment, lead exists mainly as the divalent cation and becomes adsorbed onto particulate phases; however, in polluted waters, organic complexation with contaminants is important (EPA, 1980). Mechanisms by which lead adsorbs to inorganic solids, organic materials, and hydrous iron and manganese oxides, usually control the mobility of lead, resulting in a strong partitioning of lead to bed sediments in aquatic systems (EPA, 1980). Benthic microbes can methylate lead to form tetramethyl lead, which is volatile and more toxic than inorganic lead (EPA, 1980).

The dominant form of lead in natural water is the divalent cation, but its concentration is usually low because of precipitation by hydroxide and carbonate and sorption by organic matter (Callahan et al., 1979). Some of the more common species of lead in freshwater include $PbCO_3$, $Pb(CO_3)_2^{2-}$, $PbCl^+$,$PbCl_2$, $PbOH^+$ and $Pb(OH)_2$. Only in acid conditions is the Pb(II) ion likely to occur. In freshwater, the dominant species are the lead carbonates that comprise approximately 90% of the available lead. Organic ligand complexes from decayed material can readily form metal complexes (Fergusson, 1990). Lead salts such as lead hydroxides, sulfates halides, and phosphates are poorly soluble.

The availability of an element in soil is related to soil properties such as pH and Eh. In general, as the soil pH and Eh increases, the amount of available lead will decrease (Fergusson, 1990). Lead is largely immobile in soils due to soil clays, phosphates, sulfates, carbonates, hydroxides, sesquioxides, and organic matter that quickly assimilate it. Lead tends to accumulate in surface soils with a soil half-life of 700 to 6,000 years. High soil concentrations may inhibit microbial processes and reduce natural decomposition rates (Bodek et al., 1988).

Lead mobility and uptake in soil are positively influenced by flooding, other anaerobic conditions, low pH, low organic content soils, and phosphate deficient soils. Lead uptake is negatively influenced by soils with a pH above 6, lime, phosphate, and organic matter (Bodek et al., 1988).

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The species of lead available in sediments are highly variable and depend on ambient site conditions. Lead species are strongly sorbed to iron/manganese oxides, which are more important than association with organic material. Lead concentrations correlate with iron when manganese levels are low, and with manganese when iron levels are low. Lead concentrations are also affected by particle size, density, and organic matter content. Lead concentrations generally increase with decreasing size, increasing density, and organic content (Salim and Cooksby, 1980).

A.8.1.3 Biological Fate

Metabolism of lead involves deposition in and mobilization from bone. Since lead can remain immobilized for years metabolic disturbance can remain undetected. Under normal conditions, more than 90% of the lead retained in the body is in the skeleton. Although lead is a nonessential element it is present in all tissues and organs of mammals.

The results of investigations with four species of invertebrates indicate that BCF values ranged from 499 to 1,700. BCFs for brook trout and bluegill sunfish were 42 and 45, respectively (EPA, 1980).

Birdsall et al. (1986) reported elevated lead concentrations of up to 270 mg/kg dry weight in tadpoles collected near heavily traveled highways. These concentrations may affect lead levels in wildlife that eat tadpoles. Fish tend to accumulate little lead in edible tissues; however, invertebrates can accumulate lead in high levels.

A.8.2 Toxicity

Most forms of lead are toxic and can be incorporated into the body through inhalation, ingestion, dermal absorption, and placental transfer to the fetus. In general, organo-lead compounds are more toxic than inorganic lead compounds (U.S. Fish and Wildlife Service, 1988). Biomagnification up the food chain tends to be negligible, and young organisms are more susceptible than mature organisms (U.S. Fish and Wildlife Service, 1988). Lead is not

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essential for plant growth, and excessive amounts can inhibit growth and reduce photosynthesis, mitosis, and water absorption (U.S. Fish and Wildlife, 1988).

The large affinity of Pb(II) for thiol- and phosphate-containing ligands inhibits the biosynthesis of heme and thereby affects membrane permeability of kidney, liver, and brain cells. This results in either reduced functioning or complete breakdown of these tissues since lead is a cumulative poison (Forstner and Wittmann, 1983).

Lead has a high to very high potential to bioaccumulate and bioconcentrate in aquatic and terrestrial organisms. Bioaccumulation has been reported several times in the literature to occur in terrestrial plants, terrestrial herbivores and aquatic plants. However, biominification of lead has been observed in terrestrial plants, herbivores, carnivores, and aquatic carnivores (Bodek et al., 1988).

Lead toxicity is manifested in different areas of physiologic functions. Chronic subtoxic levels also cause subtle changes in the permeability of cells of the testis, kidneys, liver, and brain, leading to their reduced function or complete breakdown. Toxic effects of lead are seen as reduced growth and longevity, impaired renal function and reproduction, and reduced functioning of the hematopoietic system and the central and peripheral nervous systems (Venugopal and Luckey, 1978).

A.8.2.1 Aquatic Organisms

The acute and chronic toxicity of lead to freshwater animals has been shown to decrease as the hardness of water increases. At a hardness of 50 mg/ℓ ($CaCo_3$) the acute toxicity of lead to nine invertebrate families ranged from 148.9 μg/ℓ for scuds, to 236,000 μg/ℓ for a midge (EPA, 1980). Table A.8-1 summarizes toxicity information for aquatic life.

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Table A.8-1

Toxic Effects of Lead to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
PLANTS					
Alga	*Chlamydomonas reinhardii*	lead	1,000 μg/ℓ (3 hours)	50% inhibition of photosynthesis	Irmer et al., 1986
Alga	*Chlamydomonas reinhardii*	lead	4,140 μg/ℓ (24 hours)	lethal	Irmer et al., 1986
Alga	*Microcystis aeroginosa*	lead	450 μg/ℓ (8 days)	immobilization	EPA, 1985
INVERTEBRATES					
Cladoceran	*Daphnia magna*	lead	1 μg/ℓ (19 days)	Repro. impairment 10%	Berglind et al., 1985
Cladoceran	*Daphnia magna*	lead	10 μg/ℓ (19 days)	Repro. impairment 50%	Berglind et at., 1985
Cladoceran	*Daphnia magna*	lead	78-181 μg/ℓ (hardness 1-2 mg/ℓ $CaCO_3$)	lifetime MATC	EPA, 1985
Cladoceran	*Daphnia magna*	lead	85-193 μg/ℓ (hardness 151 mg/ℓ $CaCO_3$)	lifetime MATC	EPA, 1985
Midge	*Tanytarsus dissimilis*	lead	258 μg/ℓ (10 days)	LC_{50}	EPA, 1985
VERTEBRATES					
Rainbow Trout	*Onchorhynchus mykiss*	tetramethyl lead	3.5 μg/ℓ (72 hours)	LC_{50}	Wong et al., 1981
Rainbow trout	*Onchorhynchus mykiss*	Pb^{2+}	13 μg/ℓ (32 weeks)	Reduced ALAD activity	EPA, 1985
Rainbow trout	*[eyed eggs]*	Pb^{2+}	10,000 μg/ℓ (56 hours)	LC_{50}	Rombaugh, 1985
Rainbow trout	*Onchorhynchus mykiss*	dissolved Pb	4.1-7.6 μg/ℓ (hardness 28 mg/ℓ $CaCO_3$)	lifetime MATC	Davies et al., 1976
Brook trout	*Salvelinus fontinalis*	total Pb	58-119 μg/ℓ (hardness 44 mg/ℓ $CaCO_3$)	MATC	EPA, 1985
Brook trout	*Salvelinus fontinalis*	dissolved Pb	39-84 μg/ℓ (hardness 44 mg/ℓ $CaCO_3$)	MATC	EPA, 1985

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The toxicity of lead to aquatic species depends on the form of lead to which they are exposed and depends on both water pH and hardness. When in the presence of minerals, clays, or sand, lead is converted to a form less toxic to aquatic life and is not likely to be converted to more toxic forms under natural conditions. Free ion forms such as hydroxide, carbonate, and sulfate salts are more toxic to aquatic life or can be converted to the more toxic forms under natural conditions (EPA, 1985).

Laboratory studies performed by Everard and Denny (1985) on freshwater angiosperms, mosses, and benthic algae showed that aquatic mosses are extremely efficient at sorbing lead in solutions containing less than 1.0 mg/ℓ of lead. In an algal/angiosperm association, algae takes up most of the lead, thus decreasing the concentration of lead reaching the angiosperm (Everard and Denny, 1985). Because algae are continually grazed and replaced by new growth, lead is easily transferred into the aquatic food web.

Some studies have shown that uptake of lead by submerged angiosperms, bryophytes, and algae is mainly passive and occurs when sediments disturbed by turbulence release lead (Welsh and Denny, 1980). Behan et al. (1979), found that the roots of rooted aquatic plants contain more lead than shoots.

Lead in the form of free ions of hydroxide, carbonate, and sulfate salts, which are the forms commonly found in waterbodies, are more toxic to aquatic life, or can be converted to more toxic forms under natural conditions (EPA, 1980). Water hardness and pH also affect lead toxicity. Data compiled by EPA (1980) indicate that lead is more toxic to rainbow trout, fathead minnow, bluegill, and water flea (*Daphnia magna*) in soft water than in hard water. Different species exhibit different sensitivities to lead; amphipods were reported to be more sensitive to lead than any other freshwater animals in acute and chronic tests (EPA, 1980). Acute LC_{50} and EC_{50} toxicity values of lead to freshwater invertebrate species range from 124 μg/ℓ for an amphipod (*Gammarus pseudolimnaeus*) to 224,900 μg/ℓ for a midge. The LC_{50} values for fish range from 300 μg/ℓ to 56,000,000 μg/ℓ.

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EPA (1980) reported that in soft water (99 mg/ℓ as $CaCO_3$) some marbled salamanders *(Ambystoma opacum)* exposed to 1.4 mg/ℓ of lead died within 8 days.

Borgmann et al. (1978) conducted a chronic bioassay test to observe the effects of lead on rates of mortality, growth, and biomass production of snails (*Lymnaea palustris*) when exposed to low levels of lead throughout their life cycle. Concentrations of lead as low as 19 μg/ℓ significantly decreased survival but not growth or reproduction. The NOEL for survival was reported at 12 μg/ℓ, and almost complete mortality was observed at 54 μg/ℓ.

Demayo et al. (1982) report that concentrations of waterborne lead greater than 10 μg/ℓ are expected to produce long-term effects on fish and fisheries.

In acute bioassays invertebrate species are more sensitive than vertebrate species. The lowest MATC from chronic studies is 19 μg/ℓ for the rainbow trout at a water hardness of 128 mg/ℓ $CaCO_3$ (EPA, 1980).

The 30-day LC_{50} for the leopard frog (*Rana pipiens*) was 105 mg Pb/ℓ, but some deaths and elevated liver residues were noted at water concentrations as low as 25 mg/ℓ (Kaplan et al., 1967). Marbled salamanders exposed to 1.4 mg Pb/ℓ died within 8 days of exposure (EPA, 1985).

Summary of Available Criteria

The EPA has established a 4-day average concentration criterion for lead not to exceed the value given by $e^{(1.273[\ln(hardness)]-4.705)}$ and a 1-hour average concentration criterion not to exceed the value given by $e^{(1.273[\ln(hardness)]-1.460)}$ more than once every 3 years (EPA, 1990). The State of Colorado chronic and acute criteria are calculated with the formula $e^{(1.417[\ln(hardness)]-5.167}$ and $0.5e^{(1.6148[\ln(hardness)]-2.1805)}$ (CDH, 1991).

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A.8.2.2 Terrestrial and Riparian Organisms

The phytotoxicity of lead is relatively low compared with other trace elements. Lead concentrations below 3 mg/ℓ in nutrient solution have no adverse effects, while levels ranging from 3 to 20 mg/ℓ are toxic. Concentrations of 5 to 10 μg/g in the leaves are correlated to NOAELs, while concentrations from 30 to 300 μg/g are considered toxic (Kabata-Pendias and Pendias, 1984).

Visible signs of lead exposure in plants includes darkening of leaves, stunted foliage, and increased amounts of shoots (Fergusson, 1990). Biochemical processes that are affected by lead exposure include cell membrane permeability, thiol and -SH group bonding, affinity for phosphate groups, inhibition of enzymes, respiration, photosynthesis, stomatal opening, and transpiration (Koeppe, 1981).

The phytotoxicity of lead to plants is low compared with other trace elements, such as zinc and copper (Adriano, 1986). Lead uptake in plants is limited by the low bioavailability of lead from the soil. National Research Council of Canada (1973) and Boggess and Wixson (1977) found that low soil pH and reduced amounts of organic matter, inorganic colloids, iron oxide, and phosphorus enhance the bioavailability of lead.

Phytotoxicity results from interference with leaf stomatal diffusion, mitochondrial respiration, photosynthesis, and ion uptake and translocation (Adriano, 1986). Plant mortality was reported at a soil solution concentration of 50 mg/ℓ; plant toxicity was reported at 25 mg/ℓ at soil concentrations of 400 to 500 mg/kg (Adriano, 1986). EPA (1985) reports that a tolerable level of 250 mg/kg for total soil lead is based on "no effects" concentrations for alfalfa, oats, and rye grass at this level; one exception was corn seedlings, which evidenced stunted seedlings at 125 mg/kg. From this information, a total soil lead concentration of 125 mg/kg is recommended as protective of vegetation.

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Several studies have been conducted on lead accumulation in agricultural crops. Sadiz (1985) found that corn grown in soils containing 786 mg/kg of lead accumulated 17 mg/kg, and corn grown in soils containing 924 mg/kg of lead accumulated 30 mg/kg. Soil levels of 12 mg/kg reduced reproduction in corn (Krishnayya and Bedi, 1986).

Lead absorbs and concentrates in plant roots. Little translocation of lead occurs to other parts of the plant. The tendency of inorganic lead to form highly insoluble salts and complexes with various anions, together with its tight binding to soils, drastically reduces its availability to terrestrial plants via the roots. As a result, in most experimental studies on lead toxicity, high lead concentrations in the range of 100 to 1,000 mg/kg soil are needed to cause visible toxic effects on photosynthesis, growth, or other parameters (World Health Organization, 1989). Other studies indicate levels of 30 to 300 ppm lead in soil are considered toxic to plants (Bodek et al., 1988).

Bazzaz et al. (1974) grew sunflowers (*Helianthus annus*) in vermiculite in a controlled environment. After 3 to 5 weeks, when the plants were 45 to 60 cm tall, the top 15 cm of each plant was excised and placed in a solution of lead salts at concentrations of 2, 20, 100 or 200 mg/ℓ for 5 days. All doses caused a reduction in net photosynthesis and respiration over the exposure period.

In a study by Broyer et al. (1972), researchers found no effect on the yield of commercial beans, barley, or tomato plants exposed to lead nitrate through a hydroponic culture solution at lead concentrations of up to 50 μg/ℓ.

Barker (1972) exposed cauliflower, lettuce stem, carrot root, and potato tubers to lead acetate at concentrations between 0.005 and 50 mg/ℓ over 20 days. There was a significant reduction in mean fresh weight of lettuce and carrot after exposure to lead concentrations of 0.005 mg/ℓ or more. Cauliflower and potato, both slower growing, showed significant reductions in yield only at 0.5 mg/ℓ or more.

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Davis and Barnes (1973) dosed growing seedlings of loblolly pine (*Pinus taeda*) and red maple (*Acer rubrum*) with solutions of lead chloride between 2 x 10^{-4} and 5 x 10^{-3} mole per liter (mol/ℓ) twice weekly for 2.5 months. Following exposure to 10^{-3} mol/ℓ or more, they observed a significant reduction in height and root dry weight for both species. There was a significant reduction in pine stem dry weight at 5 x 10^{-3} mol/ℓ.

Beyer and Anderson (1985) found that survival and reproduction were reduced in woodlice (*Porcellio scaber*) fed soil litter treated with 12,800 mg/kg of lead oxide for 64 weeks, or two generations. Lead at similar concentrations can eliminate or reduce populations of bacteria and fungi on leaf surfaces and in soil. Many of these microorganisms are important decomposers (U.S. Fish and Wildlife Service, 1988).

Exposure of green frog (*Rana clamitans*) tadpoles to 750 μg/ℓ as lead nitrate for 6 days revealed that sublethal exposures adversely affected acquisition learning and memory (Stickler-Shaw and Taylor, 1990). Kaplan et al. (1967) exposed tree frogs (*Rana pipiens*) for 30 days to solutions of lead nitrate at concentrations from 25 to 300 mg lead/ℓ. They observed sloughing of the integument, loss of postural tone, and sluggishness at all concentrations tested. The estimated LC_{50} was 105 mg/ℓ.

Khangarot et al. (1985) reported LC_{50} values for tadpoles of *Rana hexadactyla* of 100, 66.7, 41.3, and 33.3 mg/ℓ after 24, 48, 72 and 96 hours of exposure, respectively. Birge et al., (1979) exposed narrow-mouth toad (*Gastrophryne carolinensis*) eggs to inorganic lead in a continuous-flow bioassay from fertilization through to 4 days post hatch (7-days exposure). The results estimated an LC_{50} of 0.04 mg/ℓ.

Reports of lead poisoning in wild animals usually involve waterfowl. Single oral doses of lead shot (200 to 1,400 mg) can cause acute or chronic effects in mallard ducks. In studies with mourning doves (*Zenaida macrocura*) conducted by Buerger et al. (1986), an oral dose of 72 mg produced 24% mortality within 1 month as well as significantly reduced egg hatching.

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Exposure of waterfowl to lead at concentrations of 75 to 150 mg/Pb(II)/kg body weight, or 28 mg alkyl lead/kg body weight caused reduced survival, while dietary levels of 50 mg Pb(II)/kg caused reproduction impairment (Eisler, 1988). Concentrations as low as 2.8 mg alkyl lead/kg body weight produced signs of poisoning in sensitive species of birds (Eisler, 1988).

Oral exposure of mallards of 25 mg Pb/kg body weight for 12 weeks resulted in no deaths, no pathology, and no significant accumulation of lead in liver, kidney, or bone. However, a single oral dose of tetraethyl lead resulted in an LC_{50} derivation of 107 mg/kg body weight. Birds exhibited signs of hypoactivity, muscle incoordination, and loss of appetite within minutes of exposure (Hudson et al., 1984). American kestrels (*Falco sparverius*) fed mallard homogenate containing 16 to 87 mg Pb/kg fresh weight for 60-days accumulated lead resulting in tissue residues of 0.4 mg/kg fresh weight in the liver, and 7.6 mg/kg dry weight in the bone (Stendell, 1980). American kestrels exposed to 50 mg Pb/kg diet suffered from no adverse effects to survival, egg laying, fertility, or eggshell thickness. Elevated residues were observed in the humerus, tibia, and liver (Franson et al., 1983).

Lead fed to 1-day-old kestrels inhibited their growth rates by the third day of exposure. Residues of 2 ppm wet weight (w/w) in the liver and 6 ppm in the kidney were indicative of impacts on growth rates. Cockerels fed 1,850 ppm lead in the diet for 4 weeks weighed 47% as much as control birds and both hemoglobin and the packed cell volumes were significantly reduced (Franson et al., 1983).

Chickens (*Gallus sp.*) fed diets of 1,850 mg Pb/kg as lead acetate for 4 weeks resulted in no deaths or severe clinical hematological effects. Growth of the chickens was suppressed by 47% and blood residues of 3.2 to 8.4 mg/ℓ were detected (Franson and Custer, 1982).

European starlings (*Sturnus vulgaris*) fed capsules of triethyl lead chloride at 2,000 μg daily exhibited 100% mortality by day 6 of the exposure. Starlings fed 200 μg daily (2.8 mg/kg body weight) demonstrated no mortality but exhibited a reduction in food consumption (Kendall and Scanlon, 1981). The veterinary medical literature on lead toxicity is abundant for domestic animals but is lacking for wildlife species. Cases of lead poisoning have been

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reported for a variety of domestic animals including cattle, horses, dogs, and cats. Relatively low levels of exposure in food can cause fatalities when organisms forage in contaminated areas.

Lead salts are only toxic to birds at a high dietary dosage (100 mg/kg or more). Almost all of the experimental work is on chickens and other gallinaceous birds. Exposure of quail from hatching and up to reproductive age resulted in effects on egg production at dietary lead levels of 10 mg/kg (World Health Organization, 1989).

In two separate studies, Damron et al. (1969) dosed 4-week old broiler chickens with dietary lead acetate at levels between 10 and 2,000 mg lead/kg for 4 weeks. They reported that at dietary lead levels of 100 mg/kg or less, there was no significant effect on body weight gain or on food consumption. At dosing levels of 1,000 and 2,000 mg/kg, there was a significant depression of body weight gain and food consumption.

In a study conducted by Damron and Wilson (1975), bobwhite quail (*Colinus virginianus*) were exposed to various forms of lead acetate in the diet at dose rates of up to 2,000 mg/kg. Organisms exposed to doses below 1,500 mg/kg showed no effect, while organisms exposed to 2,000 mg/kg demonstrated depressed feeding, and exposure groups at 3,000 mg/kg demonstrated significant depression in growth rate and an increase in mortality.

Metallic lead is not toxic to birds except at very high dosage when administered in the form of powder. It is highly toxic to birds when given as lead shot; ingestion of a single pellet of lead shot can be fatal for some birds (World Health Organization, 1989).

There is little information on the effects of organolead compounds. Trialkyllead compounds produced adverse effects in starlings dosed at 0.2 mg/day. Birds dosed at 2 mg/day died within the period of exposure (World Health Organization, 1989).

Wild starlings (*Sturnus vulgaris*) whose crop contents contained 80 to 100 ppm lead exhibited depressed haematocrits (Grue et al., 1986). Feeding adult kestrels (*Falco sparverius*) 448 ppm

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biologically incorporated lead in the form of lead-contaminated cockerels did not affect survival, body weight or haematocrit, whereas nestling kestrels fed comparable levels of lead revealed decreased growth rates and increased mortality rates compared to controls (Custer et al., 1984; Hoffman et al., 1985).

Growing quail fed diets containing 500 or 1,000 ppm lead for 6 weeks and cockerels fed 1,850 ppm for 4 weeks had depressed haematocrits (anaemia) and slower growth rates compared with controls, but mortality was not increased (Morgan et al., 1975; Franson and Custer, 1982).

Levels of lead below 100 ppm in the diet usually cause few significant reproductive effects in birds. For example, 50 ppm lead fed to breeding kestrels resulted in no adverse effects with respect to egg laying, incubation, fertility, or eggshell thickness (Pattee, 1984). This is approximately 8.8 mg/kg body weight, assuming a dietary ingestion rate equivalent to that of a laboratory chicken (Sax, 1984).

Japanese quail may be especially sensitive to the reproductive effects of lead since it has been reported that as little as 10 ppm lead in the diet of quail hens was sufficient to cause a significant reduction in egg production, whereas over 200 ppm dietary lead was required to produce similar effects in chickens (Edens and Garlich, 1983).

Among sensitive species of mammals, survival was reduced at acute exposure to oral doses of 5 mg/kg body weight in the rat, at chronic oral doses of 0.3 mg/kg body weight in the dog, and at dietary levels of 1.7 mg Pb/kg body weight in the horse. A reduced level of aminolevolinic acid dehydratase (ALAD) enzyme activity was noted in rabbits exposed orally to 0.005 mg Pb/kg body weight, and in mice given 0.05 mg Pb/kg body weight (Eisler, 1988).

Impaired reproductive capacity was observed in rats and mice fed 25 ppm diet lead salt (Schroeder and Mitchner, 1971). Male rats suffered from prostatic hyperplasia after a month of feeding 100 μg/Pb per kg diet (Fahim et al., 1972). Rats and mice fed 25 ppm diet lead as soluble lead salt over a chronic period suffered early mortality and weight loss. About 200 to 400 mg Pb/kg body weight caused mortality in calves, but in older cattle the toxic doses were

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higher at 600 to 800 mg/kg body weight. Horses are more susceptible to lead toxicity from chronic ingestion than cattle, since an estimated daily intake of 1 mg lead/kg body weight produced toxicity symptoms in horses (Ammerman et al., 1973). The reported maximum chronic tolerated level in grazing animal diet for lead is 30 mg/kg (Bodek et al., 1988).

Reported oral LD_{50} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory animals includes:

Compound	Test Animal	LD_{50} (mg/kg bw)
Lead Chloride	Guinea Pigs	2,000
Lead Arsenate	Rats	100
Lead Arsenate	Rabbits	125
Lead Lactate	Guinea Pigs	3,000

Other reported oral toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory animals include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Lead Difluoride	Guinea Pigs	4,000
Lead Nitrate	Guinea Pigs	1,330
Lead Carbonate	Guinea Pigs	1,000
Lead Fluosilicate	Rats	250

Results of laboratory investigations reveal that soluble lead salts are more toxic than insoluble salts, and rabbits and guinea pigs are more susceptible to lead poisoning than rats and mice (Venugopal and Luckey, 1978).

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There are too few reports to draw conclusions about the effects of lead on non-laboratory mammals. However, wild rats showed similar effects to their laboratory counterparts (World Health Organization, 1989).

Calves (*Bos sp.*) exposed orally to 5 mg Pb/kg body weight for 10 to 20 days exhibited blindness and a 16% mortality rate. Calves given 5 mg Pb/kg body weight as lead acetate for 7 days in a milk diet showed signs of poisoning and mortality (Clark, 1979). Dogs (*Canis familiaris*) fed 0.32 mg Pb/kg body weight daily were considered to be the chronic toxic level. Dogs fed 3 mg Pb/kg body weight daily as lead carbonate exhibited signs of anorexia and convulsions after 180 days of exposure (Demayo et al., 1982).

Rabbits (*Lepus sp.*) exposed to 2.46 μg Pb/m^3 in air for a lifetime duration showed no effects, whereas rabbits fed less than 5 μg Pb/kg bodyweight daily had reduced blood ALAD activity (Barth et al., 1973).

Table A.8-2 summarizes potential lead criteria that are relevant to the protection of terrestrial or riparian species or ecological health.

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Table A.8-2

Proposed Lead Criteria Relevant for the Protection of Natural Resources

Resource	Criterion	Reference
Crops (mg/ℓ)		
• Neutral and alkaline soils	< 10 mg/ℓ	Demayo et al., 1982
• Acidic soil	< 5 mg/ℓ	Demayo et al., 1982
• Chronic use	< 5 mg/ℓ	Abbasi and Soni, 1986
Livestock		
• Drinking water	< 100 μg/ℓ	Demayo et al., 1982
• Horse	< 500 μg/ℓ	Demayo et al., 1982
• Forage (horse)	< 80 mg/kg fresh weight	Edwards and Clay, 1977
• Forage (cattle)	200 mg/kg	Edwards and Clay, 1977
Mouse		
• Total intake	> 0.05 (mg/kg body weight)	Schlick et al., 1983
Mule Deer		
• Total intake	> 3 mg/day	Harrison and Dyer, 1984
Canvasback *(Aythya ralisineria)*		
• Blood	> 0.2 mg/ℓ*	Dieter el al., 1976
American Kestrel *(Falco sparverius*		
• Liver	> 2 mg/kg* fresh weight	Hoffman et al., 1985
• Kidney	> 6 mg/kg fresh weight	Pattee et al., 1981
Bald Eagle *(Haliaeetus leucocephalus*		
• Liver	> 10 mg/kg* dry weight	Friend, 1985
• Kidney	> 6 mg/kg* dry weight	
Waterfowl		
• Liver	> 2 mg/kg* fresh weight	Friend, 1985
• Blood	>0.2 mg/kg fresh weight	

* Concentrations correspond to levels considered elevated as compared to control/background levels.

Source: Eisler 1988. Lead Hazards to Fish, Wildlife and Invertebrates: A Synoptic Review. U.S. Fish Wildlife Service Biol. Rep. 85(1.14) 134 pp.

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A.9 MANGANESE

A.9.1 Environmental Chemistry

Elemental manganese (Mn) is a highly reactive metal that resembles iron and is often used as an alloy to impart hardness (EPA, 1984). Normal soils contain an average manganese concentration of 850 ppm (range of 100 to 4,000 ppm [Adriano, 1986]).

A.9.1.1 Physical Properties

Manganese (atomic no. 25) has an atomic weight of 54.938, a melting point of 1,244°C, and a specific gravity of 7.21 to 7.22 g/cm^3 depending on the allotropic form. Manganese is a whitish-gray metal that is quite brittle and readily oxidizes and rusts. Manganese is a ubiquitous metal in the earth's crust, and is the 121th most abundant element. It composes about 0.10% of the earth's crust and is a principal metallic component of nodules deposited on the ocean floor. The most abundant manganese minerals include the oxides, carbonates, and silicates (Adriano, 1986).

Manganese ore deposits are widespread throughout the tropical, subtropical, and warmer temperate zones of the earth (NAS, 1973). The average content of manganese in the lithosphere is about 1,000 ppm (NAS, 1973). Manganese is widely distributed in metamorphic, sedimentary, and igneous rocks. One reason for its wide distribution in different types of rocks is due to its similar ionic size to magnesium and calcium which enables it to replace these two elements in silicate structures (Adriano, 1986).

Background concentrations of manganese in soils and plants have been reported to range from 100 to 4,000 ppm in soil, and 15 to 300 ppm in plants (Bodek et al., 1988).

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A.9.1.2 Chemical Properties

Manganese is similar to iron in chemical behavior. Manganese can exist in the oxidation states of I, II, III, IV, VI, and VII. Its most stable salts are formed from the oxidation states of II, IV, VI, and VII.

The transport and partitioning of manganese in water is controlled by the solubility of the specific chemical form present. The specific form of manganese present is controlled by the pH and Eh of the aqueous system. The metal may exist in water in four possible oxidation states II, III, IV, or VII. Divalent manganese (Mn(II)) predominates in most waters with neutral to acidic pH (pH 4 to 7), but may become oxidized at pH greater than 8 or 9 (EPA, 1984).

Manganese is often transported in lotic systems as suspended sediments (Malm et al., 1988)

Manganese occurs most commonly in the II and IV oxidation states in aquatic systems. Its solubility depends upon the pH, dissolved oxygen and presence of complexing agents. In freshwater, manganese can occur as the soluble ion, complex organic ions, or as colloidal suspensions.

Although manganese can exist in all the valence states from -III to VII (Cotton and Wilkinson, 1980), the inorganic chemistry of manganese is dominated by compounds in the II, IV, and VII valence states. The fate of manganese in aquatic systems may be determined by its ability to undergo chemical and microbiological reactions. In most natural aquatic systems, manganese is expected to be present predominantly in the suspended particulates and sediments as MnO_2, Mn_3O_4, or both (EPA, 1982). Although manganese may undergo speciation through chemical and microbiological reactions in systems, it may persist in aquatic systems for a long period. By analogy with aquatic iron (EPA, 1984), the residence time of manganese in aquatic ecosystems may be a few hundred years.

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Both chemical and microbiological interactions may cause speciation of manganese in soils. Soil pH and Eh will influence manganese speciation. It has been reported that in acid water saturated soils, manganese passes freely into solution and may leach into groundwater. Also, manganese can be leached readily from waste burial sites and other natural soils into groundwater (EPA, 1982).

The tendency of soluble manganese compounds to adsorb to soils and sediments depends mainly on the cation exchange capacity and the organic composition of the soil (Curtin et al. 1980; Kabata-Pendias and Pendias, 1984). The oxidation state of manganese in soils and sediments may be altered by microbial activity. Geering et al. (1969) observed that Mn(II) in suspensions of silt or clay loams from several areas of the United States was oxidized by microorganisms, leading to the precipitation of manganese minerals.

A.9.1.3 Biological Fate

The BCF for manganese in a species of edible fish (striped bass) has been reported to be less than 10 (EPA, 1982). Also, significant bioaccumulation of manganese may not occur with organisms in higher trophic levels (EPA, 1982).

In a study with channel catfish (*Ictalurus punctatus*) and green sunfish (*Lepomis cyanellus*), natural concentrations of manganese appeared to be greatest in the heart, followed by the gills, kidney, liver, and finally muscle (Mathis et al., 1977). Although no measured BCFs for fish were identified, Chapman et al. (1968) estimated a BCF of 25 for the edible portions of freshwater fish. A BCF of 1,300 is reported for the freshwater snail *(Physa sp.)* in a shallow freshwater pond (Mathis et al., 1977).

A.9.2 <u>Toxicity</u>

A.9.2.1 Aquatic Organisms

Data concerning toxicity of manganese to freshwater organisms are extremely limited. Manganese is a micronutrient for both animals and plants and it serves as an enzyme activator

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in plants. Manganese is not highly toxic to freshwater organisms. It is reported to stimulate growth of certain microorganisms in reservoirs and municipal water supplies when present in concentrations above 100 $\mu g/\ell$ (Mathis et al., 1977).

A concentration of 12,000 $\mu g/\ell$ of manganous chloride was determined to be fatal to *Fundulus* sp. following 6 days of exposure. However, data derived from other investigations indicate that this value may be exceptionally low. The 24-hour LC_{50} for *Orizas* sp. was 7,850,000 $\mu g/\ell$. While toxic effects were observed at 330,000 $\mu g/\ell$, mortality did not occur until 800,000 $\mu g/\ell$ for a variety of freshwater fish species (Mathis et al., 1977).

Manganese difluoride has a reported LC_{50} of 500,000 $\mu g/\ell$ for the freshwater trench (*Tinca tinca*), and manganese nitrate has a minimum lethal concentration of 40,000 $\mu g/\ell$ for sticklebacks (*genus species*) (McKee and Wolf, 1963).

For daphnia the threshold concentration for toxicity for manganous chloride is reported to be 50,000 $\mu g/\ell$ (McKee and Wolf, 1963). No other data regarding toxic effects of manganese to invertebrates were available in the literature reviewed.

No data were available concerning toxicity of manganese to aquatic plants.

Summary of Available Criteria

EPA freshwater criteria for the protection of aquatic life are unavailable for manganese. McKee and Wolf (1963) suggest that 1,000 $\mu g/\ell$ is not deleterious to fish and aquatic life. The Colorado State criteria are 1,000 $\mu g/\ell$ (CDH, 1991).

A.9.2.2 Terrestrial and Riparian Organisms

Adequate data for the characterization of toxicity of manganese to wildlife or riparian organisms were not available in the literature reviewed.

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Manganese is the least toxic of the essential metals. Mice, rats, and rabbits can tolerate approximately 1,000 ppm Mn(II) in their drinking water. At concentrations greater than 1,000 ppm, manganese is toxic to varying degrees, depending upon the manganese species and its oxidation state. Permanganate appears to be a highly toxic form of manganese because of its high solubility and powerful oxidizing action on tissues (Venugopal and Luckey, 1978).

Reported LD_{50} oral toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) for manganese in laboratory mammals include:

- 200 for guinea pigs exposed to manganese fluoride (mild toxicity).
- 1,090 for mice exposed to potassium permanganate.

Manganese in water may be significantly bioconcentrated at lower trophic levels. Folsom et al. (1963) estimated the BCF of manganese was 2,500 to 6,300 for phytoplankton, 300 to 5,500 for marine algae, and 35 to 930 for coastal fish. In general, these data indicate that lower organisms such as algae have larger BCF values than higher organisms. Thus, biomagnification of manganese in the food chain may not be significant (EPA, 1984).

Concentrations of manganese less than 15 to 25 ppm in soil are considered deficient for plant survival, whereas concentrations of 300 to 2,000 ppm in soil are considered toxic to plants (Bodek et al., 1988).

Manganese toxicity can vary depending upon the route of exposure. When ingested, manganese is considered to be among the least toxic of the trace elements (Integrated Risk Information System, 1992). The maximum chronic tolerated level of manganese in grazing animal diet is 400 to 2,000 mg/kg (Bodek et al., 1988). This is approximately 16 to 80 mg/kg body weight assurming a dietary rate of 0.04 kg/kg body weight Several toxicity studies on manganese have been performed on laboratory animals. Most of these have been inhalation exposures demonstrating an effect on both the brain and lungs. Several oral studies have been performed in rodents that demonstrated biochemical changes in the brain following exposure to

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1 mg $MnCl_2$-$4H_2O$/ml in drinking water (approximately 38.9 mg Mn/kg body weight/day) (Lai et al., 1981 and 1982; Leung et al., 1981). Levels of significant oral exposure to manganese for terrestrial organisms are presented in Table A.9-1.

In animals, most studies indicate that manganese compounds have low acute oral toxicity. With exposure through feed, daily oral doses of 930 mg/kg body weight/day (as manganous sulfate) did not cause significant mortality in rats until after 16 months of exposure (Hejtmancik et al. 1987a), and chronic exposure of mice to 810 mg manganese/kg/day did not cause increased mortality within 24 months (Hejtmancik et al., 1987b).

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Table A.9-1

Levels of Significant Oral Exposure to Manganese for Terrestrial Organisms

Species	Route	Exposure Frequency	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Effect	
Mouse	Feed	6 months	2,300	NA	NA	Gianutsos and Murray, 1982
Rat	Feed	224 days	180	NA	NA	Carter et al., 1980
Rat	Water	10 weeks	12	NA	NA	Wassermann and Wassermann, 1977
Rat	Water	2 months	NA	600	Increased GABA levels	Bonilla, 1978
Rat	Water	30 days	NA	140	Altered behavior and neurotransmitter levels	Chandra, 1983
Mouse	Feed	90 days	NA	140	Decrease in activity	Gray and Laskey, 1980
Rat	Feed	103 weeks	290	930	Death	Hejtmancik et al., 1987a

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A.10 MERCURY

A.10.1 Environmental Chemistry

Mercury (Hg), although a relatively rare element, is ubiquitous in the earth's crust, occurring at levels from 10 to 500 ppb as a sulfide, chloride, or oxide. However, mercury can form organic compounds that bioaccumulates in the food chain and become a significant toxicological concern. Naturally occurring levels of mercury and its compounds in surface and groundwaters are less than 0.5 $\mu g/\ell$ (EPA, 1987).

A.10.1.1 Physical Properties

Mercury is a metal that exists as a liquid at room temperature. In the environment, mercury exists in three oxidation states: 0 (elemental), I (mercurous compounds), and II (mercuric compounds). Metallic mercury has a vapor pressure of 1.2×10^{-3} millimeters (mm) mercury at 20°C, and a water solubility of 81.3 $\mu g/\ell$ at 30°C (Callahan et al., 1979). In the I state, the mercurous salts are not very soluble in water. In the II state, mercuric salts are more water soluble (Weast, 1980). Besides a variety of inorganic compounds, mercury forms a number of compounds with organic ligands. These compounds are toxicologically and environmentally significant since they are able to accumulate throughout the food chain.

A.10.1.2 Chemical Properties

Mercury is strongly bound to soil and is attached predominantly to soil organic matter. Therefore, the mobility of mercury and compounds in soil is minimal even in soils contaminated by mercury fungicides. The probability of groundwater contamination resulting from mercury migration through soils is unlikely due to its immobility (EPA, 1980).

Three aspects of mercury chemistry influence its activity in soil and distinguish it from other metals. These include the volatility of elemental mercury, an accessible redox chemistry whereby free mercury can be produced in soils, and the biomethylation of mercury producing very toxic and often volatile compounds (i.e., $(CH_3)_2Hg$). Organic mercury compounds are

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likely to occur in soils. Hydroxy and chloro-mercurial compounds occur depending on the soil pH and chloride concentration. There is a strong interaction between mercury and sulfur that can form mercury-sulphur organic compounds in humic acid (Adriano, 1986).

Approximately 4 to 50% of mercury is associated with surface active organic materials. Mercury ions and the metal are rapidly sorbed by sediments. Factors that influence the sorption of mercury include surface area, organic matter, and grain size. In addition, mercury is often bonded to sites containing sulphur (Moore and Ramamoorthy, 1984).

The aquatic fate of mercury and compounds has been studied extensively. Photolysis, chemical speciation, volatilization, sorption, and biotransformation are all important processes in the aquatic media. Adsorption onto the surface of particulate matter and subsequent sedimentation probably constitutes the most important mercury removal mechanism in the aquatic system (Callahan et al., 1979). Part of the precipitated mercury may be transformed into organic mercurial compounds through biotransformation and may re-enter the aquatic phase (EPA, 1980).

The dominant inorganic species of mercury in freshwater is $Hg(OH)_2$, which forms at a pH of approximately 6. Mercury-organic ligand complexes can be formed. Mercury-humic species are the main form of organic mercury in freshwater, and the major species likely to occur is CH_3HgOH (Fergusson, 1990).

Forms of mercury include elemental mercury, its vapor, inorganic mercurial salts ($HgCl_2$, Hg_2Cl_2 and $Hg(NO_3)22$), organic mercurial salts (monoalkyl halides, dialklyl mercury compounds), and organic mercurials (diphenyl mercury and mercurochrome). Organometallic compounds such as $CH_3Hg(I)$ behave like ionic compounds and are formed from inorganic mercurial salts by microbial action; thus, toxicity of simple dialkyl mercurial compounds and $CH_3Hg(I)$ salts are included. Alkyl mercury salts are more toxic than elemental (mercury liquid or vapor) and inorganic Hg(II) salts (Venugopal and Luckey, 1978).

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Mercury is found in most soil types existing as a large variety of minerals. The most common are the sulfides cinnabar and meta-cinnabar (Adriano, 1986). Mercury is found in one of three stable oxidation states: 0, I, and II. Elemental mercury is rare; however, the other forms may be transformed to elemental mercury. Mercury also exists in nature in the form of inorganic or organic complexes or as stable mercuric sulfides (Adriano, 1986).

Mercury in soil is slightly mobile as organocomplexes. Uptake is promoted by methylation and possibly limited by the presence of lime, sulfur, and phosphates (Bodek et al., 1988).

Background concentrations of mercury in soils range from 0.01 to 0.8 ppm (Bodek et al.,1988).

A.10.1.3 Biological Fate

The BCF for mercury is high because uptake is fast and depuration is very slow. Due to this slow depuration rate, methyl mercury is the major form of mercury residues in aquatic organisms. Elimination of methyl mercury is very slow due to the sulfhydryl groups in proteins which securely bond to mercury and require demethylation before they can be eliminated. Thus, the biological half-life of mercury in fish is estimated to be 2 to 3 years (EPA, 1980).

Methyl mercury is more water soluble and 10 to 20 times more toxic to both mammals and aquatic life than other forms of mercury. Even though most mercury in the water column is divalent inorganic mercury, methyl mercury comprises most of the mercury residue in tissues of aquatic organisms (EPA, 1980).

BCF values for mercury in aquatic organisms are available for organic and inorganic mercury. BCF vary from 250 for mussel to 63,000 for fathead minnows (EPA, 1980).

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A.10.2 Toxicity

Mercury has long been recognized as one of the most toxic of the heavy metals, but was only recently identified as a serious pollutant in the aquatic environment. Due to the methylation process, mercury is one of the few pollutants that adversely affects aquatic life through direct toxicity and adversely affects the use of aquatic life from bioaccumulation into edible tissue (Heinz, 1976). Data on the acute toxicity of mercury are available for over 50 species of freshwater and saltwater organisms. Table A.10-1 summarizes literature values that were determined important for this ecological evaluation.

A.10.2.1 Aquatic Organisms

Acute toxicity data for divalent inorganic mercury reveal a broad spectrum of acute toxicity data (Table A.10-1). Few data are available for various organic mercury compounds and mercurous nitrate, but they all appear to be 4 to 31 times more acutely toxic than inorganic mercury (EPA, 1980). MacLeod and Pessah (1973) studied the effect of temperature on the acute toxicity of mercuric chloride to rainbow trout. At 5, 10, and 15°C the LC_{50} values were 400, 280, and 220 $\mu g/\ell$, respectively.

Chronic toxicity data indicates that methyl mercury is the most chronically toxic of the mercury compounds tested. Chronic concentrations that resulted in adverse effects in brook trout were 0.52 $\mu g/\ell$ (EPA, 1980; Eisler, 1987).

Invertebrates are sensitive to the adverse effects of mercury. LC_{50} data range from 2.2 $\mu g/\ell$ for a daphnid to 2,000 $\mu g/\ell$ for a stonefly and mayfly. Chronic values for *Daphnia* were less than 0.04 $\mu g/\ell$ (EPA, 1980; Eisler, 1987).

Freshwater plants appear to be less sensitive than the most sensitive freshwater animals to both divalent inorganic mercury and methyl mercury. The toxicity of mercury to algal community structure was assessed by Singh and Rai (1991). Maximum inhibition of algal number was

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Table A.10-1

Toxic Effects of Mercury to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Plants					
Algae	*Chlorella vulgaris*	Mercuric chloride	1030	EC_{50} cell division inhibition	Rosko and Rachlin, 1977
Algae	*Scenedesmus quadricauda*	Mercuric chloride	70	Growth inhibition	Bringmann, 1975
Invertebrates					
Crayfish	*Orconectes limosus*	Mercuric chloride	50	LC_{50} (sp. mean acute value)	Boutet and Chalsemartin, 1973
Crayfish	*Faxonella clypeatus*	Mercuric chloride	1000	LC_{50}	Helt and Fingerman, 1977
Mayfly	*Ephemeralla subvaria*	Mercuric chloride	2000	LC_{50} (sp. mean acute value)	Warnick and Bell, 1969
Damselfly	Unknown	Mercuric nitrate	1200	LC_{50} (sp. mean acute value)	Rehwoldt et al., 1973
Stonefly	*Acroneuria lycorias*	Mercuric chloride	2000	LC_{50} (sp mean acute value)	Warnick and Bell, 1969
Caddisfly	*Hydropsyche betteni*	Mercuric chloride	2000	LC_{50} (sp mean acute value)	Warnick and Bell, 1969
Midge	*Chironomus sp.*	Mercuric nitrate	20	LC_{50} (sp. mean acute value)	Rehwoldt et al., 1973
Cladoceran	*Daphnia magna*	Mercuric chloride	0.72 - 1.28	Chronic limit	Biesinter et al., 1982
Cladoceran	*Daphnia magna*	Mercuric chloride	0.91 - 1.82	Chronic limit	Biesinger et al., 1982
Cladoceran	*Daphnia magna*	Mercuric chloride	6.7	EC_{50}	Biesinger and Christenson, 1976
Cladoceran	*Daphnia magna*	Mercuric chloride	13	LC_{50}	Bringmann and Kuhn, 1977
Cladoceran	*Daphnia magna*	Methylmercuric chloride	0.52 - 0.87	Chronic limit	Biesinger et al., 1982

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Table A.10-1 (Continued)

Toxic Effects of Mercury to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Amphibians					
Leopard frog	*Rana pipiens*	Mercuric chloride	7.3	EC_{50} death, deformity	Birge et al., 1979
Narrow-mouthed toad	*Gastrophyrnye carolinensis*	Mercuric chloride	1 - 1.3	EC_{50} death, deformity	Birge et al., 1979
Green toad	*Bufo fowleri*	Mercuric chloride	65.9	EC_{50} death, deformity	Birge et al. 1979
Vertebrates					
Rainbow trout	*Onchorhynchus mykiss*	Mercuric chloride	275	LC_{50} (sp. mean acute value	Lock et al., 1981
Rainbow trout	*Onchorhynchus mykiss*	Methylmercuric chloride	24	LC_{50} (sp. mean acute value) - larvae	Wobeser, 1973
Rainbow trout	*Onchorhynchus mykiss*	Methylmercuric chloride	42	LC_{50} (sp. mean acute value) - juvenile	Wobeser, 1973
Brook trout	*Salvelinus fontinalis*	Methylmercuric chloride	0.29 - 0.93	Chronic limit	McKim et al., 1976
Brook trout	*Salvelinus fontinalis*	Methylmercuric chloride	84	LC_{50} - juvenile	McKim et al., 1976

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observed at 0.8 mg/ℓ. Unicellular forms were sensitive, and the decline in algal number was concentration dependent.

Rainbow trout and brook trout were tested for methyl mercury toxicity. Rainbow trout (larvae and juveniles) were the most sensitive species to acute toxicity (24 μg/ℓ and 25 μg/ℓ, respectively). Juvenile and yearling brook trout were slightly more resistant having acute values of 84 and 64 μg/ℓ, respectively (EPA, 1980).

Summary of Available Criteria

Numerous factors such as temperature, dissolved oxygen, hardness, pH, alkalinity, and organic complexation capacity probably affect the acute and chronic toxicity and bioaccumulation of the various forms of mercury. Too few data are available, however, to show that quantitative relationships exist between these variables to mercury toxicity; therefore, existing criteria do not incorporate these factors.

EPA AWQC recommend that the 4-day average concentration of dissolved mercury should not exceed 0.012 μg/ℓ more than once every 3 years on the average, and the 1-hour average concentration should not exceed 2.4 μg/ℓ more than once every 3 years on the average.

The State of Colorado acute criterion is 2.4 μg/ℓ, and the chronic criterion is 0.1 μg/ℓ (CDH, 1991).

A.10.2.2 Terrestrial and Riparian Organisms

Both organic and inorganic forms of mercury are reported to be teratogenic and embryotoxic in experimental animals. Toxicity may involve liver, heart, gonads, pancreas, and gastrointestinal tract. Inorganic mercury is generally less acutely toxic than organic mercury compounds but it does affect the central nervous system adversely.

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Visible signs of mercury exposure for plants include stunting of seedlings and roots, chlorosis, browning of leaf points, and reduction in growth (Fergusson, 1990). Biochemical processes that are affected by mercury exposure include cell membrane permeability, protein synthesis, bonding of thiol and sulfhydral groups, photosynthesis, stomatal opening, and transpiration (Peterson and Girling, 1981). Concentrations in leaves ranging from 1 to 3 $\mu g/g$ have been correlated to toxic effects (Kabata-Pendias and Pendias, 1984). Mercury acts antagonistically towards phosphorous and potassium in plants (Fergusson, 1990).

Mercury transfer to plants occurs almost totally as elemental vapor. Concentrations of 1 to 2 ppm mercury in soil are considered toxic to plants (Bodek et al., 1988).

Studies involving amphibian species have revealed that the narrow-mouthed toad was the most sensitive species tested; the 7-day EC_{50} value was 1.3 $\mu g/\ell$. The southern gray tree frog and the squirrel tree frog were slightly more resistant (EC_{50} of 2.4 $\mu g/\ell$), and the Fowlers toad and pig frog were the most resistant (65.9 $\mu g/\ell$ and 67.2 $\mu g/\ell$, respectively) (EPA, 1980).

Studies of two species of worms (tiger worms and earthworms) exposed to methyl mercury revealed that earthworms were more sensitive. Earthworms exhibited 50% mortality at soil concentrations of 0.79 mg/kg. A 100% mortality was observed at 5.0 mg/kg in 60-day exposure tests (Abassi and Soni, 1983). Tiger worms exhibited 21% mortality at 5.0 mg/kg and 100% mortality at 25 mg/kg (Abassi and Soni, 1983). Methyl mercury hydroxide administered in the diet to *Drosophila melanogaster* at 5 mg/ℓ failed to cause chromosomal disfunction. Methyl and phenyl mercury produced small increases in the rate of point mutations (Ramel, 1972).

Inorganic mercury and its chemical compounds are relatively biologically inert. However, the addition of an organic group to elemental mercury to form methyl mercury (CH3Hg or MeHg) markedly increases lipid solubility and rates of transfer of mercury across biological membranes. The organic mercury compounds pose the greatest threat to animal health (Wren, 1986).

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Albanus et al. (1972) fed cats a homogenate of pike from a mercury-contaminated lake in Sweden. The methyl mercury level in the fish was about 6.0 μg/g and exposure to cats was approximately 0.45 mg mercury/kg body weight/day. Changes in behavior and movement in the cats were observed 4 to 11 days before onset of convulsions, which occurred between 60 and 73 days after the experiment.

Eaton et al. (1980) fed cats liver of ringed seals that contained an average of 26.2 μg/g of mercury (range 5.2 to 87.5 μg/g) of which only 3% of the mercury was in the methyl mercury form. Three groups of cats were fed 100 g, 50 g, and 25 g of seal liver daily. At the end of 90 days, no neurological or pathological disorders were noted in any of the cats fed seal liver.

To assess the secondary effects of mercury poisoning in wild carnivores, Hanko et al., (1970) fed methyl mercury contaminated wheat (average 8 μg/g) to chickens. The chickens remained healthy after 40 days of exposure. Chickens were then fed to mountain lions; no adverse effects were observed in the lions.

O'Conner and Nielson (1980) examined the effect of methyl mercury on adult male river otters. Methyl mercuric hydroxide was added to food to give levels of 2, 4, and 8 μg/g methyl mercury in the diet of the experimental animals which provided an exposure of 9.3, 17, and 37 mg mercury/kg body weight/day. The mean survival time in the experimental groups was 184, 117, and 54 days, respectively.

A comparison of the effect of similar dietary levels of methyl mercury on different mammals in the literature are presented in Table A.10-2.

Mitsumori et al. (1981) fed groups of 60 male and 60 female SPRICR mice 0, 15 or 30 ppm methyl mercury chloride in the diet for up to 78 weeks. The majority of the 30 ppm groups died from neurotoxicity by week 26. Histopathology of all animals after 53 weeks revealed renal tumors in the 15 ppm exposure group. Using a dietary intake rate of 0.120 kg/kg body weight (Sax, 1984), a value of 1.8 mg/kg body weight is estimated from the 15 ppm dietary exposure.

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Table A.10-2

Toxic Effects of Methyl Mercury on Terrestrial Organisms

Organism (Group)	Exposure Rate (days)	Concentration ($\mu g/g$)	Effect	Reference
Otters	117	4.00	Mortality	O'Conner and Nileson, 1983
Ferrets	58	5.00	Mortality	Hanko et al., 1976
Mink	100	5.70	No effect	Jernelov et al., 1976
Mink	26-29	4.80	Mortality on some test animals	Wobeser et al., 1976
Mink	30	5.00	Mortality	Aulerich, 1974
Cats	60-75	6.00	Convulsions	Albanus et al., 1972
Cats	78	0.25	Mortality	Eaton et al., 1980

Source: Eisler, R. 1987. *Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review.* U.S. Fish Wildlife Service Biol. Rep. 85(1.10). 90 pp.

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Mercuric salts cause reproductive dysfunction in Japanese quail when they were fed mercuric chloride at 125 ppm in drinking water (Cogburn et al., 1973). Thinning of egg shells and mortality of young was observed in hens fed 1 to 8 ppm mercuric chloride in the diet (Cogburn et al., 1973). The dietary intake rate of 0.175 kg/kg body weight for chickens (Sax, 1984) was used to predict a dose of 0.175 mg/kg body weight for reproduction effects.

Juvenile starlings inadvertently poisoned with mercury-contaminated commercial feed (1.1 mg Hg/kg of feed) developed kidney lesions within 8 weeks after consuming the contaminated diet (Nicholson and Osborn, 1984). Applying a dietary intake rate for chickens of 0.175 kg/kg body weight (Sax, 1984) results in a toxic dose of 0.19 mg/kg body weight.

Other reported reproductive effects of chronic dietary inorganic mercury exposure in birds include:

- Delayed testicular development in young male quail at a level of 2 ppm in the diet (Hill and Soares, 1984).
- Gonadal atresia and a reduced number of mating attempts in quail exposed to 125 ppm mercury in the drinking water for 8 weeks (Thaxton and Parkhurst, 1973).
- Decreased fertility of eggs in young hens fed 8 ppm or greater dietary mercury (Hill and Shafner, 1976).

Concentrations of mercury greater than 125 ppm in drinking water were required to cause depressed growth rates and increased mortality in young chickens (Parkhust and Thaxton, 1973).

The bioaccumulation and bioconcentration potential of mercury in terrestrial animals, molluscs and crustacea is high to very high, while the bioaccumulation potential is low to limited for higher plants, mosses, lichens, and algae (Bodek et al., 1988). Mercury biomagnifies in terrestrial carnivores, aquatic plants, herbivores, and carnivores (Bodek et al., 1988).

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Reported oral LD_{50} toxicity values (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory animals include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Mercuric Oxide	Mice	22
Mercuric Oxide	Rats	18
Mercuric Chloride	Rats	37
Mercuric Iodide	Mice	80
Mercuric Sulfate	Mice	40
Mercuric Acetate	Mice	62
Mercurous Iodide	Mice	110
Mercurous Nitrate	Rats	297
Mercurous Nitrate	Mice	388
Ethyl Mercuric Chloride	Rats	30
Mercuric Acetate	Mice	6
Phenyl Mercuric Acetate	Mice	50

Other established oral toxicity values (Venugopal and Luckey, 1978) from the literature include LD_{100} of 15 mg/kg for dogs exposed to mercuric chloride.

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A.11 NICKEL

A.11.1 Environmental Chemistry

A.11.1.1 Physical Properties

Nickel is a hard, silvery white metal has properties that make it very desirable for combining with other metals to form mixtures called alloys. Some of the metals that nickel is alloyed with are iron, copper, chromium, and zinc. Most nickel is used to make stainless steel. Nickel also combines with other substances such as chlorine, sulfur, and oxygen to form nickel compounds. Many of these compounds dissolve fairly easily in water and have a characteristic green color (Cotton and Wilkinson, 1980).

Nickel occurs naturally in the earth's crust and is also emitted from volcanos. Nickel is not readily degraded in the environment. Most nickel will strongly attach to particles containing iron or manganese. Under acidic conditions, nickel is more mobile in soil and may seep into groundwater. Nickel does not appear to concentrate in fish or in terrestrial plants (Cotton and Wilkinson, 1980).

A.11.1.2 Chemical Properties

Nickel is a transition metal in group VIII of the periodic table. While nickel can exist in oxidation states of -I, 0, II, III, and IV, its only important oxidation state is nickel (II) under environmental conditions (Cotton and Wilkinson, 1980). Nickel has a molecular weight of 58.69 amu, is insoluble in water, and has a density of 8.9 g/cm^3 (Cotton and Wilkinson, 1980).

Much of the nickel released into waterways as runoff is associated with particulate matter; it is transported and settles out in areas of active sedimentation such as the mouth of a river (Bowman et al., 1981). During a 4-month study of Lake Onondaga in Syracuse, New York, 36% of the nickel in the lake was lost to sediment (Young et al., 1982). Seventy-five % of the nickel loaded into the polluted lake was soluble and remained so in the lake. Nickel is strongly adsorbed at mineral surfaces such as oxides and hydrous oxides of iron, manganese, and

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aluminum (Evans, 1989; Rai and Zachara, 1984). Such adsorption plays an important role in controlling the concentration of nickel in natural waters.

Nickel is strongly adsorbed by soil, although to a lesser degree than lead, copper, and zinc (Rai and Zachara, 1984). There are many adsorbing species in soil, and many factors affect the extent to which nickel is adsorbed, so the adsorption of nickel by soil is site specific. Soil properties such as texture, bulk density, pH, organic matter content, and the type and amount of clay minerals and certain hydroxides influence the retention and release of metals by soil (Richter and Theis, 1980).

Amorphous oxides of iron and manganese and to a lesser extent clay minerals are the most important adsorbents in soil. In alkaline soils, adsorption may be irreversible (Rai and Zachara, 1984) which limits nickel's availability and mobility in these soils. Cations such as Ca(II) and Mg(II) have been reported to reduce adsorption due to competition for binding sites, whereas anions like sulfate reduce adsorption as a result of complexation. Nickel adsorption depends strongly on pH.

A.11.1.3 Biochemical Properties

Nickel is not accumulated in significant amounts by aquatic organisms (Birge and Beach, 1980); Callahan et al., 1978). The range of BCFs appears to be 40 to 100 in fish and 100 to 259 in invertebrates (Callahan et al., 1978). The concentration of nickel in a major carnivorous fish in New York state, the lake trout was the lowest and the concentration did not increase appreciably with the age of the fish (Birge and Beach, 1980). The mean BCF for the carnivorous fish was 36.

There are few studies available concerning nickel levels in birds and animals that might indicate whether nickel biomagnifies in the food chain, and these are generally out-dated (Jenkins, 1980). Two more recent studies concerning levels in voles and rabbits living on sludge-amended land did not indicate any accumulation of nickel in these herbivores or in the plants they fed upon (Alberici et al., 1989; Dressler et al., 1986). The lack of significant bioaccumulation of nickel

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in aquatic organisms, voles and rabbits would indicate that nickel is not biomagnified in the food chain.

A.11.2 Toxicity

A.11.2.1 Aquatic Organisms

The acute toxicity of nickel to nine invertebrate species has been reported by EPA (1986). Under comparable water hardness conditions (40 to 45 mg $CaCO_3/\ell$) the reported median lethal or median effective concentrations (LC_{50} or EC_{50}) of nickel in static test conditions range from 51 $\mu g/\ell$ in the cladoceran, *Daphnia magna* (Biesinger and Christensen, 1972) to 33,500 $\mu g/\ell$ in the stonefly, *Acroneuria lycorias* (Warnick and Bell, 1969). The acute toxicity of nickel to invertebrates generally decreases with increased water hardness. In *Daphnia magna,* for example, acute toxicity values range from 510 $\mu g/\ell$ in soft water, as reported above to a value of 4,970 $\mu g/\ell$ in hard water (EPA, 1986).

Similarly, the chronic toxicity of nickel is influenced by water hardness. Chronic toxicity values derived from life-cycle tests with *Daphnia magna* ranged from 14.8 $\mu g/\ell$ in soft water to 357 $\mu g/\ell$ in hard water (EPA, 1986). The acute-chronic ratios of nickel calculated by EPA (1986) based on the above studies are between 122 (soft water) and 14 (hard water).

Nickel has been observed to reduce growth in several species of freshwater algae at concentrations between 100 and 700 $\mu g/\ell$ (Hutchinson, 1973; Hutchinson and Stokes, 1975). Wang and Wood (1984) tested six strains of algae and one Euglena species for their sensitivity to nickel toxicity and for their ability to bioaccumulate nickel. The cyanobacteria tested were found to be more sensitive to nickel toxicity than the green algae or the Euglena species. Wang and Wood (1984) also reported concentration factors for the species studied that ranged from 0 to 3,000 and varied significantly with pH. Most strains tested accumulate nickel optimally at approximately a pH of 8.0. EPA (1986) reported a bioconcentration factor of 9.8 for green algae exposed for 6 days to nickel at a concentration of 1,000 $\mu g/\ell$. Bioconcentration in water hyacinths (*Eichhornia crassipes*) exposed for 12 days to nickel concentrations of 1,000 to 8,000 $\mu g/\ell$ ranged from 256 to 438 in the roots and 174 to 576 in the tops of the plants (EPA, 1986).

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Nickel toxicity to aquatic vertebrates also decreases with increasing water hardness. Acute toxicity in fathead minnows (*Pimephales promelas*), range from 4,580 $\mu g/\ell$ in soft water to 44,500 $\mu g/\ell$ in hard water (Pickering and Henderson, 1966). In acute toxicity tests conducted under similar water hardness conditions (20 to 26 mg Ca CO3/ℓ), reported LC_{50} values range from 2,480 $\mu g/\ell$ in rock bass (*Ambloplites rupestris*) (Lind et al., manuscript) to 9,820 $\mu g/\ell$ in goldfish (*Carassius auratus*) (Pickering and Henderson, 1966).

Chronic toxicity information concerning nickel to aquatic vertebrates is limited. Pickering (1974) reported a chronic toxicity value of 527 $\mu g/\ell$ (at a water hardness of 210 mg CaCO3/ℓ) in fathead minnows (*Pimephales promelas*) exposed to nickel for a lifetime. Based upon this chronic toxicity value and a mean acute toxicity value of 26,500 $\mu g/\ell$ (at a water hardness of 210 mg CaCO3/ℓ), EPA (1986) calculated and acute–chronic ratio of 53. Using the results derived by Lind et al., (manuscript) from early life-stage tests of fathead minnows and nickel in soft water, EPA (1986) also calculated a chronic value of 217 $\mu g/\ell$, and an acute–chronic ratio of 24.

Nebeker et al. (1985) examined early life stage sensitivity in rainbow trout and identified an NOAEL for early life stage exposure. Newly fertilized eggs were the most sensitive to nickel exposure with a NOAEL of 35 $\mu g/\ell$, followed in sensitivity by eyed eggs and larval fish, each with a NOAEL of 134 $\mu g/\ell$.

A maximum bioconcentration factor of 106 was identified for fathead minnows exposed for 30 days to nickel at a concentration of 21 $\mu g/\ell$ (EPA, 1986).

A.11.2.2 Terrestrial and Riparian Organisms

Oral LD_{50} values have been reported for rats and mice. Rats died after gavage exposure to 8.6 mg nickel/kg body weight/day as nickel chloride hexahydrate for 91 days (American Biogenics Corporation, 1986). Clinical signs included lethargy, ataxia, irregular breathing, cool body temperature, and salivation. No increase in mortality was observed in mice exposed to 150 mg nickel/kg body weight/day as nickel sulfate in the drinking water for 6 months (Dieter et al.,

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1988). Rats chronically exposed to nickel in the diet exhibited no hepatic effects (NOAEL) at a concentration of 5 mg nickel/kg body weight/day. A LOAEL value of 50 mg nickel/kg body weight/day was derived from this investigation (Ambrose et al., 1976). In a subchronic exposure with mice, a NOAEL value of 44 mg/nickel/kg body weight/day, and a LOAEL of 108 mg nickel/kg body weight/day was derived based upon renal effects (Dieter et al., 1988). Lower NOAEL and LOAEL values for subchronic gavage studies were derived based upon hematopoietic and dermal/ocular effects. The reported NOAEL and LOAEL values were 1.2 and 8.6 mg nickel/kg body weight/day respectively (American Biogenics Corp. 1986).

Levels of nickel in soil considered toxic to grazing animals are between 50 to 300 mg/kg dry diet (NAS, 1980).

Phytotoxicity studies have demonstrated that nickel toxicity to certain species of plants is pH dependent. At a concentration of 50 ppm in soil, a greater than 30% reduction in mustard yield was observed at a pH of 5.7, while no effect on mustard yield was observed at a pH of 6.4 (Patterson, 1971). A NOAEL of 5.5 ppm was established by Patterson (1971). Other phytotoxicity studies have shown that 1,100 ppm nickel in soil causes reduction in oat and mustard crop yield (effects are soil pH dependent, and concentrations of 20 to 60 ppm are slightly to severely toxic to Heine wheat (Patterson, 1971). Cottenie (1977) determined that 100 ppm nickel in soils will cause toxicity to plants. The USDA maximum recommended cumulative sludge metal application level to prevent phytotoxicity for nickel is 50 to 200 kg/hectacre (ha) (Chaney, 1985).

An LC_{50} value of 757 ppm nickel in soil was derived from a 2 week artificial soil test for earthworms (*E. feotida*) (Neuhauser, 1985). At a concentration of 500 ppm, a significant reduction and growth and reproduction was observed in earthworms (*E. feotida*) exposed over a 6 week duration (Neuhauser, 1984).

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A.12 SELENIUM

A.12.1 Environmental Chemistry

The abundance of selenium (Se) in the earth's crust is reportedly about 0.05 to 0.09 ppm. Selenium is present in detectable but highly variable amounts in soils usually ranging in concentration from 0.1 to 2 ppm total selenium (Adriano, 1986). Selenium in natural freshwater exists at concentrations of 1 $\mu g/\ell$ (Weast, 1981).

A.12.1.1 Physical Properties

Selenium (atomic no. 34) has an atomic weight of 78.96, a boiling point of 684.9°C, a melting point of 217°C, and a specific gravity of 4.26 to 4.81 g (cm^3) (Adriano, 1986). Metallic selenium is insoluble in water, whereas selenium crystals are slightly soluble in carbon disulfide and ether. Amorphous forms are soluble in carbon disulfide, methylene iodide, benzene, and quinoline (Weast, 1981).

A.12.1.2 Chemical Properties

The availability of an element in soil is related to soil properties such as pH and Eh. In general, as the soil pH increases, the amount of available selenium will decrease. In contrast, the amount of available selenium will increase as soil Eh decreases (Fergusson, 1990).

Selenium is stable in four valence states: II, 0, IV, and VI. Elemental selenium can be considered inert in the aquatic environment, and deposition of this form appears to be a major sink for selenium in natural waters (Weast, 1981).

Both Se(IV) and Se(VI) occur in freshwater depending on the pH. Both of these species are very soluble, and it is probably the most abundant form of selenium released into the aquatic environment (Weast, 1981). Under reducing conditions and at low pH, elemental selenium or metal selenides can be formed. Assimilation into organic forms is also an important chemical process for selenium in freshwater (Fergusson, 1990).

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Chemical speciation patterns affect the transport of selenium in soil. In poorly aerated, acidic soils, insoluble forms predominate. In well-aerated, alkaline soils, soluble forms of selenium are subject to leaching and selenium compounds are readily taken up by plants (EPA, 1980; EPA, 1984). The availability of selenium in soil is inhibited by acid and neutral conditions, while alkaline and well-oxidized soils promote selenium activity (Bodek et al., 1988).

Selenium is strongly adsorbed to hydrous metal oxides, while clays and organic materials have a lesser affinity (Adriano, 1986). Sorption by bed sediments or suspended solids, and precipitation with hydrous iron oxides are probably the major control on mobility of selenium in aerobic waters. However, most selenium in aquatic systems is probably transported as the dissolved species (Weast, 1981).

Experimental studies indicate that selenium is quite mobile in clays, especially under alkaline conditions (EPA, 1984).

A.12.1.3 Biological Fate

Inorganic selenium can be biotransformed to organic selenium in the food chain (Ingersoll, et al., 1990). Selenium can be methylated by a variety of organisms, including benthic microflora. In a reducing environment, hydrogen selenide (H_2Se) may be formed. Both the methylated forms and hydrogen selenide are volatile and can be released to the atmosphere. Consequently, remobilization of selenium from aquatic and terrestrial systems, through biotransformation to volatile forms and subsequent atmospheric transport, can result in significant recycling (EPA, 1980).

Selenium is bioaccumulated by aquatic and terrestrial organisms. Although dietary intake is thought to be the most important source of selenium in many organisms, little biomagnification appears to take place. Conversion of selenium to inert and insoluble forms may occur in terrestrial and aquatic organisms (EPA, 1980).

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The BCF for *Daphnia* ranged from 322 at 5 $\mu g/\ell$ to 71 at 711 $\mu g/\ell$ (Ingersoll, et al., 1990). Daphnids accumulated selenium from water to concentrations that may result in food chain affects. BCFs differed significantly by exposure concentration, there was a decrease in the BCF value as the exposure concentration increased, but stabilization occurred at a concentration of 156 $\mu g/\ell$ (Ingersoll, et al., 1990).

The liver is the main storage organ in fish (Hermanutz, et al., 1992). Tissue residues suggested as indicators of potential adverse effects are 40 mg/kg in viscera, and 17.5 mg/kg in skeletal muscle; these concentrations would potentially adversely affect survival (Lemly, 1985). Lower levels of 15.8 in liver, and 2.8 mg/kg in skeletal muscle were correlated with adverse health effects in another study (Hermanutz, et al., 1992). The concentration of selenium in ovaries is critical because of parental transfer to offspring. Progeny were adversely affected at maternal ovary concentrations of 4.4 to 4.5 mg/kg (Hermanutz, et al., 1992).

A.12.2 Toxicity

As an essential element, selenium is required for the glutathione peroxidase enzyme that protects tissue from oxidative damage arising from organic peroxides and hydroxy radicals. Two examples of selenium toxicity in livestock are alkali disease and the blind staggers. Blind staggers is characterized by anorexia, emaciation, and collapse and can be produced by eating plants with 100 to 100,000 $\mu g/g$ selenium content. A chronic effect for exposure to lower concentrations, alkali disease is the result of consuming plants with a selenium content of 25 to 50 $\mu g/g$ and is characterized by weight loss, fibrosis of the heart, liver, and kidneys, and impaired reproductive capacity (Lucky and Venugopal, 1978).

Selenium is antagonistic with phosphorous, sulfur, manganese, copper, zinc, and cadmium in plants and is synergistic towards vanadium (Fergusson, 1990).

The bioaccumulation and bioconcentration potential for selenium is high for higher plants, moderate for terrestrial animals, and low for aquatic invertebrates, mosses, lichens, and algae.

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Biominification of selenium has been observed in aquatic and terrestrial plants, and terrestrial herbivores (Bodek et al., 1988).

A.12.2.1 Aquatic Organisms

The selenite form is generally more acutely toxic to early life stages of aquatic organisms, and effects are more pronounced at elevated temperatures. Selenium salts may be converted to methylated forms by microorganisms, and these are readily accumulated by aquatic vertebrates. Chronic toxicity has no trend between the selenite and selenate forms (Ingersoll, et al., 1990). Toxicity for selenium is summarized in Table A.12-1.

For freshwater algae species, it has been demonstrated that selenite, selenate, selenomethionine, and selenopurine are all toxic, but that sulfate has a significant protective role against selenium toxicity (Eisler, 1985). At water concentrations of 47 to 53 $\mu g/\ell$, selenium was associated with anemia and reduced hatching of rainbow trout (Hodson et al., 1980). At 250 $\mu g/\ell$ selenium, growth was reduced in rainbow trout fry after exposure for 21 days (Adams, 1976).

Hermanutz, et al. (1992) evaluated the effects of selenium on bluegill in outdoor experimental streams. Adults exposed to 10 $\mu g/\ell$ had significantly ($p \leq 0.05$) reduced survival, and at 30 $\mu g/\ell$ complete mortality. Reproductive effects were observed when adults were exposed for 40 weeks prior to spawning. These effects included decreased embryo and larval survival, and produced larva with a high occurrence of edema, lordosis, and internal hemorrhaging. Results of this and other field studies indicate that laboratory tests have underestimated selenium toxicity.

Chronic exposure to 837 $\mu g/\ell$ selenium delayed the emergence time of the adult midge at a 6 to 1 ratio of selenate to selenite (Ingersoll et al., 1990). Adverse effects in *Daphnia* were observed at concentrations greater than or equal to 156 $\mu g/\ell$ (Ingersoll, et al., 1990).

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Table A.12-1

Toxic Effects of Selenium to Aquatic Organisms

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Plants					
Algae	*Anabaena variabilis*	Selenium salt	15,000 - 17,000	LC_{50} (96 hour)	Kumar and Prakash, 1971
Algae	*Oedogonium cardiacum*	Selenium salt	< 100	LC_{50}	Nassos et al., 1980
Invertebrates					
Cladoceran	*Daphnia magna*	Selenium salt	710	LC_{50} (sp. mean acute value)	Halter et al, 1980
Cladoceran	*Daphnia magna*	Selenium salt	240	LC_{50}	EPA, 1980
Amphibian					
Frog	*Xenopus laevis* (embryo)	Selenium salt	20,000	LC_{50} (27 hour)	Browne and Dumont, 1979
Frog	*Xenopus Laevis* (tadpole)	Selenium salt	1,500	LC_{50} (7 day)	Browne and Dumont, 1979
Fish					
Rainbow trout	*Onchorhynchus mykiss*	Selenium salt	4,200 - 4,500	LC_{50} (96 hour)	Adams, 1976
Rainbow trout	*Onchorhynchus mykiss*	Selenium salt	12,500	LC_{50} (96 hour)	EPA, 1980
Brook trout	*Salvelinus fontinalis*	Selenium salt	10,200	LC_{50} (96 hour)	Cardwell et al., 1976

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Sublethal effects were observed at 348 $\mu g/\ell$. The MATC for *Daphnia* ranged from 343 to 359 $\mu g/\ell$ for selenite, and 36 to 1,225 $\mu g/\ell$ for selenate (Ingersoll et al., 1990).

Amphibians are adversely affected by selenium. Frogs (*Xenopus laevis*) had cranial and vertebral deformities, and lowered survival during development in water with selenium concentrations of 2,000 $\mu g/\ell$ or higher (Browne and Dumont, 1979).

Summary of Available Criteria

The available data for inorganic selenate indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 760 $\mu g/\ell$, and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of inorganic selenate to sensitive freshwater aquatic life.

EPA (1980; 1990) recommended that the 4-day average concentration of dissolved selenium should not exceed 5 $\mu g/\ell$ more than once every 3 years on the average, and the 1-hour average concentration should not exceed 20 $\mu g/\ell$ more than once every 3 years on the average.

The State of Colorado standards for the protection of aquatic life for acute exposure to selenium are 135 $\mu g/\ell$, and the chronic criterion is 17 $\mu g/\ell$ (CDH, 1991).

A.12.2.2 Terrestrial and Riparian Organisms

There is an optimum concentration for a nutrient element below which the element is deficient and above which the element is toxic. Selenium is considered deficient for plants at concentrations less than 0.02 mg/ℓ in nutrient solution, optimum at concentrations less than 1 mg/ℓ, and toxic at levels from 1 to 2 mg/ℓ (Kabata-Pendias and Pendias, 1984). Levels of 5 to 100 ppm selenium in soil are considered toxic to plants. Selenium is taken up by plant roots and distributed to growing tips and concentrated in the roots (Bodek et al., 1988).

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Visible signs of selenium toxicity to plants includes: interveined chlorosis, black spots at high selenium concentrations, bleaching and yellowing of young leaves, and pink spots on roots (Fergusson, 1990). Biochemical processes that are affected by selenium exposure include competition for sites with essential metabolites, and replacement of essential atoms (Peterson and Girling, 1981), although 0.01 to 2 ppm is considered background for plants (Bodek et al., 1988). Some food and forage crops growing on certain seleniferous soils can accumulate selenium to concentrations as high as 1,000 ppm.

Chronic selenium toxicity can occur in grazing animals that consume plants containing 3 to 25 ppm selenium over a long period of time. This is approximately 0.12 to 1 mg/kg body weight obtained by using a dietary intake rate of 0.04 mg/kg body weight. Symptoms of chronic poisoning ("alkali" disease) include lack of vitality, loss of hair, sterility, hoof deformity, lameness, anemia, and fatty necrosis of the liver. Acute toxic effects including impairment of vision, weakness of limbs, and respiratory failure may occur in livestock consuming 100 to 1,000 ppm of selenium. There are reports that consumption of plants containing 400 to 800 ppm have been lethal to sheep, hogs, and calves. There are no reports of increased cancer rates among livestock in selenium-contaminated areas (EPA, 1984).

The maximum chronic tolerated level of selenium in a grazing animal diet is 2 mg/kg (Bodek et al., 1988).

Selenium compounds are the most toxic of the Group VI elements. Selenium toxicity, or selenosis is caused by both organic and inorganic forms of selenium (Venugopal and Luckey, 1978). The effects of selenosis include increased pancreatic weight, metabolic rate, and hemolysis. Chronic inorganic selenosis causes follicular skin rash, inflammation of the perivascular lymph channels, hemolytic anemia, serum bilirubin, and damage to the spleen, pancreas, and liver. The specific symptoms in cattle include gastroenteritis and plyencephalomalacia; in sheep, myocardial degeneration and fibrosis, pulmonary congestion, and edema; in rats, hepatitis, nephritis, and myocarditis (Venugopal and Luckey, 1978).

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Reported oral LD_{50} toxicity values (mg/kg body weight) in the literature (Venugopal and Luckey, 1978) include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Selenium Disulfide	Rats	370
Sodium Selenite	Mice	7
Sodium Selenite	Guinea Pigs	5.1
Sodium Selenite	Rats	7
Sodium Selenite	Rabbits	2.25
Selenium Metal	Rats	13
Sodium Selenate	Dogs	4.0

Other reported toxicity values include (Venuugopal and Luckey, 1978):

Compound	Test Animal	LD_{50} (mg/kg bw)
Selenium Disulfide	Mice	38
Selenous	Rats	25
Sodium Selenite	Dogs	4.0
Sodium Selenate	Rabbits	7.0

Elemental selenium given orally to rats has an LD_{50} of 6,700 mg/kg body weight (Cummins and Kumura, 1971); however, some selenium compounds appear to be more toxic. For instance, the oral LD_{50} for selenium sulfide in the rat is 138 mg/kg body weight (Cummins and Kimura, 1971).

Although selenium is an essential trace element, excessive amounts cause reproductive failure and death in animals. In laboratory studies with birds, concentrations of 25 ppm selenium in the diet caused mortality (Heinz et al., 1987), and concentrations of 5 ppm caused reproductive effects in chickens (Ort and Latshaw, 1978). Applying a dietary intake rate of 0.175 kg/kg

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body weight (Sax, 1984) results in a dose of 0.88 mg/kg body weight for chickens fed 55 ppm in diet.

Investigations involving the exposure of adult mallards fed selenomethionine demonstrated that liver and breast tissue will quickly accumulate high levels of selenium. Liver was calculated to take 7 to 8 days to reach 95% of its peak concentration, while breast muscle took 81 days to reach 95% of its peak concentration, when ducks were exposed to 10 ppm selenium in feed. The ducks quickly eliminated selenium once untreated feed was supplied (Heinz et al., 1990).

In a study by Heinz and Sanderson (1990) adult male mallards were given a choice between an untreated diet (control) and a diet containing 5, 10, or 20 ppm selenium as selenomethionine dissolved in water and mixed in the diet. Concentrations of 10 and 20 ppm were avoided. Avoidance appeared to be caused by a conditioned response probably due to selenium-related illness, and not due to taste aversion.

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A.13 SILVER

A.13.1 Environmental Chemistry

Silver (As) is the 66th most abundant element in the earth's crust and is present at an average concentration of 0.07 ppm (Adriano, 1986). The general range of silver in normal soils has been reported as less than 0.1 ppm to 1 ppm, and some values exceed 5 ppm. For the conterminous United States, Shacklette and Boerngen (1984) found that surficial mineral soils averaged 0.7 ppm silver, while soils high in organic matter had a range of 2 to 5 ppm silver.

A.13.1.1 Physical Properties

Silver (atomic no. 47) has an atomic weight of 107.868, a boiling point of 2212°C, a melting point of 961.93°C, and a specific gravity of 10.5 g/cm^3 at 20°C. Elemental silver is relatively insoluble in water (somewhat soluble in alkali cyanide solutions); however, some of its compounds are water soluble (Adriano, 1986).

A.13.1.2 Chemical Properties

Silver can exist in several chemical forms in aqueous systems. Metallic silver, which has very low solubility, is stable over much of the Eh to pH range for water. Concentrations of hydrated silver cations, usually present as the univalent species, may be controlled by reaction with chloride, bromide, and iodide ions to give insoluble silver halides (Weast, 1981). Precipitation of silver chloride may exert a major control on solubility of silver in situations where chloride concentrations are relatively high. Under reducing conditions often found in sediments, formation of insoluble silver sulfides and metallic silver may control levels of soluble silver species.

Silver is strongly sorbed by manganese dioxide, ferric hydroxide, and clay minerals. Sorption is probably the dominant process leading to removal of dissolved silver from the water column. In general, concentrations of silver are higher in sediments than in overlying waters. For

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instance, these concentrations were reported to differ by a factor of 1,000 in an alpine lake (EPA, 1980).

Silver has a low solubility in water of 0.1 to 10 mg/ℓ, depending on pH and the chloride concentration (Hem, 1970). Trace amounts of silver are found in natural and finished waters originating from natural sources and from industrial waste. Data from 1,577 samples of well and surface waters from 130 points in the United States showed detectable silver concentrations (0.1 μg/ℓ or greater) in only 104 samples. The highest concentrations were noted in the St. Lawrence and Colorado Rivers (Durum and Hafty, 1961).

Soil concentrations of silver vary greatly dependent on the underlying geologic formation. Granite igneous rocks in Nevada contain up to 50 mg/kg silver, and coal fly ash may contain up to 15 mg/kg silver (Nordberg and Gerhardsson, 1988).

Adsorption appears to be the dominant process leading to partitioning of silver into sediments. Silver concentrations in lake sediments and nearby soils were found to correlate with organic content. Silver concentrations in sediments were 100 times the concentration of overlying waters (Freeman, 1977). The affinity of silver for three clays decreased in the following order: montmorillonite > Illite > kaolinite. However, silver has a stronger affinity to MnO_2 and $Fe(OH)_3$ (Kharkar et al., 1968).

Colorado streams showed a high correlation between silver and manganese content (Kharkar et al., 1968).

Photolysis, volatilization, atmospheric transport, and biotransformation do not appear to be important fate or transport processes for silver (EPA, 1980).

A.13.1.3 Biological Fate

Bioaccumulation of silver by aquatic plants, invertebrates, and vertebrates occurs readily and appears to depend primarily on sorption/desorption from sediments. However, the amount of

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silver partitioned to the biota appears to be minor in comparison with the amount partitioned to the sediments. Little biomagnification occurs (EPA, 1980).

BCFs for three invertebrate species have also been determined. The maximum BCF for each species was 15 in the stonefly *Claasenia sabulosu*, 170 in the stonefly *Pteronarcys californica*, and 240 in the mayfly *(Ephemerella grandis)* (EPA, 1980). EPA (1980) reported a BCF of less than 1 for bluegill sunfish, and a BCF of 3,080 for edible portions of freshwater fish.

A.13.2 Toxicity

A.13.2.1 Aquatic Organisms

Toxicity data indicate that the acute toxicity of silver decreases with increasing water hardness. However, no definitive relationship between water hardness and chronic toxicity has yet been established (EPA, 1980).

Silver is one of the most toxic metals to aquatic organisms, and was particularly toxic to developmental stages of rainbow trout. Median lethal concentrations for freshwater fish ranged from 3.9 $\mu g/\ell$ for fathead minnows in soft water, to 280 $\mu g/\ell$ for rainbow trout in hard water. Chronic exposure investigations have demonstrated significant growth and survival reduction in rainbow trout due to silver exposure. The EPA (1980) reported values for the acute toxicity of silver in four freshwater invertebrate species. LC_{50} or EC_{50} were derived under similar test conditions and were reported to range from 0.25 $\mu g/\ell$ for *Daphnia* to 4,500 $\mu g/\ell$ for the scud (*Gammarus pseudolimnaeus*). Based upon data obtained from life-cycle studies with *Daphnia,* chronic toxicity values ranging between 2.6 and 29 $\mu g/\ell$ have been established.

Silver has been observed to cause adverse effects in freshwater plants at concentrations ranging from 30 to 7,500 $\mu g/\ell$ (EPA, 1980). Toxic effects often involve lethality of inhibition of growth or biochemical function. Stokes et al. (1973) found that 30 $\mu g/\ell$ inhibited the growth of *Chlorella vulgaris,* with complete inhibition at 100 $\mu g/\ell$. Brown and Rattigan (1979) observed inhibition of oxygen evolution in *Elodea canadensis* at silver concentrations of 100 $\mu g/\ell$.

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Studies have shown that silver bioaccumulates in aquatic organisms at relatively low concentrations because most silver compounds are sparingly soluble in water. Bioconcentration factors ranging from 10 to 100 were reported for largemouth bass and bluegill (Coleman and Cearley, 1974; Cearley, 1971).

Soluble silver compounds that ionize readily are quite toxic to fish. The LC_{50} ranges from 0.2 mg Ag/ℓ for young eels to 0.003 mg Ag/ℓ for salmon fry (Terhaar et al., 1972). Silver complex, as in thiosulfate which occurs in photographic processing effluents, are at most only slightly toxic to fish. The 96-hour LC_{50} is greater than 250 mg for thiosulfate complex (Bard et al., 1976).

Davies et al. (1978) observed a mean 96-hour LC_{50} of 6.5 μg/ℓ and 13.0 μg/ℓ in softwater (approximately 26 mg/ℓ hardness as $CaCO_3$) and hardwater (350 mg/ℓ hardness as $CaCO_3$) respectively, to rainbow trout. The long-term no effect concentration for silver added to water as silver nitrate was between 0.09 and 0.17 μg/ℓ after 18 months exposure in soft water.

Summary of Available Criteria

EPA has proposed a recommended criteria that the 4-day average concentration of dissolved silver should not exceed 0.12 μg/ℓ more than once every 3 years on the average, and the 1-hour average concentration should not exceed 0.92 μg/ℓ more than once every 3 years on the average.

Since the toxicity of silver is hardness dependent, the EPA (1980) recommends that the concentration (μg/ℓ) of total recoverable silver should not exceed the numerical value given by $e^{(1.72[\ln(\text{hardness})]-6.52)}$ at any time. The available data indicate that chronic toxicity to freshwater aquatic life may occur at concentrations as low as 0.12 μg/ℓ.

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The State of Colorado chronic standard is given by $e^{(1.72[\ln(hardness)]-9.06)}$; the standard specifically for trout is $e^{(1.72[\ln(hardness)]-10.51)}$. The state aute standard is $0.5e^{(1.72[\ln(hardness)]-6.52)}$ (CDH, 1991).

A.13.2.2 Terrestrial and Riparian Organisms

Silver ions are considered one of the most toxic heavy metal ions to microorganisms, fish, and invertebrates. However, since silver readily forms insoluble compounds, it is considered harmless in the natural terrestrial environment (Adriano, 1986).

Excess silver can induce selenium, vitamin E, and copper deficiency symptoms in animals fed nutritionally adequate diets, and can aggravate deficiency symptoms in animals whose diets lack one or more of these nutrients. These effects are reported in dogs, sheep, pigs, chicks, turkey poults, and ducklings (EPA, 1980)

A single oral dose of 420 mg/kg of silver colloid did not result in any deaths in rats. Mortality was noted only after repeated daily oral ingestion of 1,680 mg/kg for 4 days (Dequidt et al., 1974). Grasso et al. (1969) observed a rapidly fatal liver necrosis beginning on day 14 after addition of silver acetate to the diet of rats (130 to 1,000 ppm silver acetate, or 4 to 33 mg/kg/day) or to the drinking water (1,500 ppm silver acetate or 97.5 mg/kg/day assuming a 200 g rat consumes 20 mℓ/day water).

Van Vleet (1976) reported that four weanling swine fed a diet containing adequate selenium and vitamin E, and 0.5% silver acetate (3,250 ppm silver, or 130 mg/kg/day) for 4 weeks developed anorexia, diarrhea, and growth depression; three of the four pigs died. The LOAEL and NOAEL were 130 and 52 mg/kg/day silver, respectively.

In a study conducted by Walker (1971), rats were exposed to silver in food for 4, 6, 8, 10, 12, 16, 25, or 60 weeks. No effects were observed at 65 mg/kg/day, and the exposure was stopped after 12 weeks.

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Olcott (1948) reported a ratio of 1 to 1,000 solution of silver nitrate in drinking water (63.5 mg/kg/day (assuming a 200 g rat consumes 20 mℓ water/day) given to rats for 218 days induced intense pigmentation of many tissues. No shortening of the lifespan or reduction of body weight occurred. A NOAEL of 63.5 mg/kg/day was identified.

Reported oral LD_{50} values (mg/kg body weight) (Venugopal and Luckey, 1978) for laboratory animals exposed to silver in the diet include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Silver Metal Colloid	Rats	100
Silver Cyanide	Rats	123
Silver Oxide	Rats	2,820
Silver Fluoride	Guinea Pigs	300

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A.14 THALLIUM

A.14.1 Environmental Chemistry

A.14.1.1 Physical Properties

Thallium (Tl) (atomic no. 81) is a group II metal and has an atomic weight of 204.37, a boiling point of 1,457°C, a melting point of 303.5°C, and a specific gravity of 11.85 g/cm^3. Thallium and its compounds are relatively insoluble in water (Adriano, 1986).

Although thallium does not occur in a free state in nature, several minerals contain thallium as a major constituent. Thallium is produced mainly as a by-product of zinc and lead smelting and refining. Due to the volatile nature of most thallium compounds, the element becomes enriched on the flue dusts during the processing of ores (Adriano, 1986)

Thallium is a naturally occurring element with a mean concentration in the earth's crust of 1 ppm (Adriano, 1986). Thallium can exist in two oxidation states of I and III. Generally inorganic Tl(I) compounds are more stable than the III forms. However, Tl(III) can form stable complexes with many ligands (Manzo and Sabbioni, 1988).

Thallium is strongly adsorbed by montmorillontric clays, especially in alkaline solution, and has been found in relatively high concentrations in sediments in freshwater lakes (Bodek et al., 1988).

In a study of thallium movement in water, sand, vegetation, and fish of an aquatic ecosystem, concentrations of thallium decreased slowly in the water and increased tenfold in the vegetation and fish. Definite transport of thallium occurred among water, fish, and vegetation, but no significant transport was seen between the sand and the other ecosystem components. Thallium concentration in sand remained essentially constant throughout the experiments (Wallwork-Barber et al., 1985).

Thallium occurs in small amounts in all living organisms; natural levels in plants are reported to be between 0.01 and 3800 ppm ash weight, 0.5 ppm being typical for most species

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(Sabbioni and Manzo, 1980). Bioconcentration factors reported by various investigations range from 10 to 150,000 for marine plants and animals (Bodek et al., 1988).

A.14.1.2 Chemical Properties

Thallium as Tl(I) is the stable form in soils and is highly associated with potassium (K(I)). Thallium may replace potassium in soils such as clays where potassium is sorbed (Fergusson, 1990).

In reducing environments, thallium may be precipitated as the metal or as thallium sulfide. However much of the thallium present in aquatic systems is likely to remain in solution and be transported to the oceans (Weast, 1981). Active removal of some dissolved thallium by sorption to clay minerals and hydrous metal oxidizes present in bed sediments is probably an important environmental fate process.

There is no evidence to suggest that photolysis or volatilization are important environmental processes. There is speculation that thallium may be methylated under aerobic conditions by electrophilic attack, although biotransformation does not appear to be an important process in aquatic systems (EPA, 1980).

A.14.1.3 Biological Fate

Thallium is readily taken up by aquatic organisms, and bioaccumulation may be an important fate process. Results of limited studies with algae suggest that thallium may be biomagnified. A bioconcentration factor of 1×10^5 was reported for freshwater fish (EPA, 1980).

A.14.2 Toxicity

A.14.2.1 Aquatic Organisms

Little is known about the behavior of thallium in freshwater aquatic systems. Thallium is known to kill fish slowly at concentrations of 1 to 60 mg/ℓ (Zitco et al., 1975). It is lethal to

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aquatic insects and invertebrates at 2 to 4 mg/ℓ, and algae may be affect at concentrations as low as 0.1 mg/ℓ (Delling and Healey, 1926; EPA, 1980).

Hardness of water, which normally affects the toxicity of metals to fish, may not have an effect on thallium toxicity because of the low complexing ability of thallium. For the same reason, humic acid has little effect on thallium toxicity (Zitco, 1975).

Acute and chronic toxicity of thallium for freshwater aquatic life occurs at concentrations as low as 1,400 and 40 μg/ℓ, respectively. Toxic effects would be expected to occur at lower concentrations among species more sensitive than those tested (EPA, 1980).

The available data for thallium indicates that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 1,400 and 40 μg/ℓ, respectively (EPA, 1980). There is sufficient data to indicate that thallium is chronically toxic to fish and invertebrate species at concentrations as low as 20 μg/ℓ (EPA, 1980).

Daphnia magna and the fathead minnow have similar acute sensitivity to thallium. LC_{50} values for both species were in the range of 910 to 2,180 μg/ℓ (Table A.14-1). The chronic values for these two species were also similar, with levels of 130 and 57 μg/ℓ, respectively (EPA, 1980).

Algae were also sensitive; effects occurred at concentrations as low as 100 μg/ℓ. There were 50% reductions in chlorophyll-a and cell numbers of an alga at concentrations of 110 and 100 μg/ℓ, respectively (EPA, 1980).

Summary of Available Criteria

Due to a lack of comprehensive data, there is no established federal AWQC for the protection of aquatic life (EPA, 1980). The Colorado State chronic standard is 15 μg/ℓ (CDH, 1991); no acute standard is available.

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Table A.14-1

Toxic Effects of Thallium for Aquatic Life

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Invertebrates					
Cladoceran	*Daphnia magna*	Thallium salt	2180	LC_{50}	EPA, 1978
Cladoceran	*Daphnia magna*	Thallium salt	910	LC_{50}	Kimball, Manuscript
Cladoceran	*Daphnia magna*	Thallium salt	100 - 181	Chronic value	Kimball, Manuscript
Vertebrates					
Fathead minnow	*Pimephales promelas*	Thallium salt	1800	LC_{50}	Kimball, Manuscript
Fathead minnow	*Pimephales promelas*	Thallium salt	<40	Chronic value	EPA, 1978
Fathead minnow	*Pimephales promelas*	Thallium salt	40 - 81	Chronic value	Kimball, Manuscript

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A.14.2.2 Terrestrial and Riparian Organisms

Thallium is neither essential not stimulatory in either man or animals. It has been considered the most highly toxic cumulative cation. Incidences of poisoning from thallium-laden rodenticides have been documented (Venugopal and Luckey, 1978). Mammals were found to be more sensitive to thallium toxicity than to mercury, cadmium, lead, copper, or zinc. The toxicity of thallium is exceeded only by methyl mercury (Wallwork-Barber et al., 1985).

In laboratory studies, guinea pigs and rabbits appear to be the most susceptible to thallium toxicity. In addition, male rats are more affected by thallium exposure than female rats (Venugopal and Luckey, 1978). Some of the reported LD_{50} oral toxicity values (mg/kg body weight) presented in this literature include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Thallium Oxide	Rats	40.6
Thallium Chloride	Mice	24
Thallium Nitrate	Mice	32.5
Thallous Sulfate	Rats	15.8
Thallium Acetate	Mice	53.2
Thallium Acetate	Rats	41.3
Thallium Oxide	Rabbits	31.2
Thallium Oxide	Dogs	31.2
Thallic Oxide	Guinea Pigs	5.2
Thallium Bromide	Mice	29

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Other LD_{100} oral toxicity values derived from thallium exposure to laboratory animals include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Thallium Nitrate	Dogs	45
Thallous Sulfate	Rats	25
Thallous Selenite	Rats	50
Thallium Acetate	Dogs	25.8
Thallium Acetate	Rats	24.5

In a study conducted by Formigli et al. (1986), male rats were exposed to 10 ppm $TlSO_4$ in drinking water (0.7 mg Tl/kg/day) for 30 to 60 days. Numerous reproductive abnomalities were observed.

A NOAEL of 0.25 mg/kg/day for thallium sulfate and a NOAEL of 0.26 mg/kg/day for thallium nitrate was established by Integrated Risk Information System (1992) based on a rat study in which organisms were exposed to 0, 0.1, 0.05, and 0.25 mg/kg/day by gavage (approximately 0, 0.008, 0.04, and 0.2 mg Tl/kg/day).

Results demonstrated no mortality, but dose related increases in the incidence of alopecia, lacrimation, and exophthalmos were observed. No differences between control groups and treated were observed in body weights, body weight gain, good consumption, or organ weights. Moderate dose related changes were observed in some blood chemistry parameters: increased SGOT, LDH and sodium levels, and decreased blood sugar levels (EPA, 1986).

Based on the results of this study the highest dose of 0.25 mg/kg/day thallium sulfate (0.20 mg/kg/day thallium) was considered a NOAEL. Using the molecular weight of thallium nitrate to thallium (266/204.4) for this conversion, this NOAEL was converted to 0.26 mg/kg/day $TLNO_3$ (EPA, 1986).

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The acute oral LD_{50} values for various thallium compounds in mice and rats range from 10 mg Tl/kg for thallium sulfate in rats to 46 mg Tl/kg for thallium sulfate in mice (Danilewica et al., 1979; Downs et al., 1960; National Institute for Occupational Safety and Health, 1985; Truhaut, 1959).

Thallous acetate administered in the diet of 100 g young Wistar rats for 15 weeks at levels of 5, 15, or 50 mg/kg of diet (i.e., 0.4, 1.2 or 3.9 mg Tl/kg/day, respectively) produced high mortality: 20, 40, and 100%, respectively, for the three dosage groups. Due to the high mortality rate among all groups, a true no-effect level could not be established (Downs et al., 1960).

In a comparison study, thallic oxide was administered in the diet to groups of five 100 g young Wistar rats for 15 weeks at levels of 20, 35, 50, 100, and 500 mg/kg diet (i.e., 1.8, 3.1, 4.5, 9.0, and 44.8 mg Tl/kg body weight, respectively). High mortality resulted at all dose levels (Downs et al., 1960).

Thallium sulfate administered in the drinking water to 80 female Sprague-Dawley rats at 10 mg Tl/kg (about 1.4 mg Tl/kg/day) for 36 weeks produced 21 percent mortality and 20 percent incidence of alopecia (Manzo et al., 1983). Because the study employed one dose level, and because the one dose used caused a high number of deaths, a no-effect level could not be established.

In mammalian species, the acute LD_{50} for thallium compounds for all routes of administration ranges between about 5 and 70 mg/kg body weight (Friberg et al., 1986).

Due to their similar chemistry, thallium will displace the potassium ion in plant cells (Fergusson, 1990). Visible signs of thallium toxicity in plants includes a reduction in growth (Fergusson, 1990). Biochemical processes which are affected by thallium exposure include bonding of thiol and SH groups, replacement of essential atoms, inhibition of certain enzymes, photosynthesis, stomatal opening, and transpiration (Peterson and Girling, 1981).

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Thallium causes neurological effects in mammals where symptoms ranging from mild peripheral neuropathy to irreversible coma and death. Thallium possibly alters catecholamine metals and can activate ATP-ase with ten times the affinity of potassium (Bank et al., 1972). Histological examination reveals damage in motor cortex, substantial nitra and globas pallidus; axons in the peripheral nervous system degenerate, swell and fragment with demyelination.

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A.15 ZINC

A.15.1 Environmental Chemistry

Zinc (Zn) is one of the most abundant essential trace elements in tissues. It is a constituent of all cells, and several enzymes depend upon it as a cofactor. Zinc interacts with cadmium in the geosphere and biosphere; both are group IIB metals that occur in the same ore types. Zinc is an essential element, whereas cadmium is not. In soils, the ratio of cadmium to zinc varies from approximately 1 to 300, to 1 to 2,900 by mole (Schroeder et al., 1967).

Zinc is the 24th most abundant element in the earth's crust, and the average soil concentration world-wide is estimated to be 70 ppm (Adriano, 1986). The average range for total zinc in normal soils is between 10 to 300 ppm, although wider ranges of 1 to 900 ppm with an average of 90 ppm have been reported (Adriano, 1986).

Background concentrations of zinc in soils and plants have been reported to range from 10 to 300 ppm, and 8 to 150 ppm, respectively (Bodek et al., 1988).

A.15.1.1 Physical Properties

Zinc (atomic no. 30) has an atomic weight of 65.38, a boiling point of 970°C, a melting point of 419.58°C, and a specific gravity of 7.133 g/cm^3 at 25°C (Weast, 1981). Zinc is a bluish-white metal that is relatively soft with a density of 7.133 g/cm^3. Elemental zinc is water insoluble, however zinc salts are soluble. Elemental zinc is soluble in acid and alkali solutions (Weast, 1981).

A.15.1.2 Chemical Properties

Zinc can occur in both suspended and dissolved forms. Dissolved zinc may occur as the free (hydrated) zinc ion or as dissolved complexes and compounds with varying degrees of stability and toxicity. Suspended (undissolved) zinc may be dissolved following minor changes in water chemistry or may be sorbed to suspended matter (EPA, 1980). The predominant fate of zinc

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in aerobic aquatic systems is sorption of the divalent cation by hydrous iron and manganese oxides, clay minerals, and organic material. The efficiency of these materials in removing zinc from solution varies according to their compositions and concentrations, the pH, the concentrations of complexing ligands, and the concentration of zinc (EPA, 1980).

Concentrations of zinc in suspended and bed sediments always exceed concentrations in ambient water. In reducing environments, precipitation of zinc sulfide limits the mobility of zinc. However, under aerobic conditions, precipitation of zinc compounds is probably important only where zinc is present in high concentrations (EPA, 1980). Zinc tends to be more readily sorbed at higher pH than at lower pH and tends to be desorbed from sediments as salinity increases. Compounds of zinc with the common ligands in surface waters are soluble in most neutral and acidic solutions, so that zinc is readily transported in most unpolluted, relatively organic-free waters (Weast, 1981).

The relative mobility of zinc in soil is determined by the same factors that affect transport in aquatic systems. Atmospheric transport of zinc is also possible. However, except near sources such as smelters, zinc concentrations in air are relatively low (EPA, 1980).

Zinc availability in soils is inhibited by high soil pH, high clay content, high cation exchange capacity, and high phosphate content. Zinc availability is promoted by low soil pH, whereby it can be present in the form of Zn(II) (Bodek et al., 1988).

A.15.1.3 Biological Fate

Since zinc is an essential nutrient, it is strongly bioaccumulated even in the absence of abnormally high ambient concentrations; however, zinc does not appear to be biomagnified. Although zinc is actively bioaccumulated in aquatic systems, the biota appear to represent a relatively minor sink compared to the sediments. Since it is actively bioaccumulated, the environmental concentrations of zinc probably exhibit seasonal fluctuations (EPA, 1980; EPA, 1984).

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Zinc is bioaccumulated by freshwater fish through food and water. Zinc was found to bioaccumulate in freshwater animal tissues from 51 to 1,130 times the concentration present in the water. Internal organs and bones accumulate much higher zinc levels than edible muscle tissue (Phillips and Russo, 1978). Bioconcentration factors in aquatic organisms range from 51 for Atlantic salmon to 1,130 for the mayfly (*Ephemerella grandis*) (EPA, 1986).

A.15.2 Toxicity

A.15.2.1 Aquatic Organisms

Acute toxicity values are available for over 43 species of freshwater animals. Data indicate that the toxicity of zinc decreases as hardness increases, and increases as temperature increases (EPA, 1980). Toxicity data are summarized in Table A.15-1.

Zinc is an essential micronutrient important for cell growth and differentiation. Exposure to sublethal concentrations of zinc causes extensive edema and necrosis of liver tissue. Death often results from gill necrosis and hypoxia. Zinc toxicity is dependent on water hardness. The pH may affect zinc toxicity, in which available zinc will increase in water as the pH decreases (Campbell and Stokes, 1985).

Several toxicity studies indicate that zinc toxicity increases with decreased pH, whereas other studies have indicated less toxicity and/or less uptake with a decreased pH (Campbell and Stokes, 1985; Sprague, 1985). Modeling predicts little variation in zinc speciation over a pH range from 4 to 7, and experimental data also supports this conclusion. Thus, zinc toxicity for different pH is most likely to be function of organism response to changing pH.

The most sensitive vertebrate species in acute assays is the rainbow trout. The 96-hour LC_{50} for rainbow trout is 90 $\mu g/\ell$. Rabe and Sapplington (1970) demonstrated trout to be among the most sensitive of the freshwater fish species. The most tolerant species is the fathead minnow. The 96-hour LC_{50} for fathead minnows is 35,500 $\mu g/\ell$ at a water hardness of 200 mg/ℓ (EPA, 1980).

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Table A.15-1

Toxic Effects of Zinc for Aquatic Life

Organism (Group)	Species	Chemical	Concentration (mg/ℓ)	Effect	Reference
Invertebrates					
Cladoceran	*Daphnia magna*	Zinc sulfate	280 (hardness 45 mg/ℓ $CaCO_3$)	LC_{50}	Cairns, et al., 1978
Cladoceran	*Daphnia magna*	Zinc chloride	100 (hardness 45 mgℓ $CaCO_3$)	LC_{50}	Biesinger and Christensen, 1972
Midge	*Chironomus sp.*	Zinc sulfate	18,200 (hardness 50 mg/ℓ $CaCO_3$)	LC_{50}	Rehwoldt, et al., 1973
Vertebrates					
Rainbow trout	*Onchorhynchus mykiss*	Zinc chloride	815 (hardness 22 mg/ℓ $CaCO_3$)	LC_{50}	Chapman and Stevens, 1978
Rainbow trout	*Onchorhynchus mykiss*	Zinc chloride	1,760 (hardness 83 mg/ℓ $CaCO_3$)	LC_{50}	Chapman and Stevens, 1978
Rainbow trout	*Onchorhynchus mykiss*	Zinc sulfate	2,510 (hardness 178 mg/ℓ $CaCO_3$)	LC_{50}	Holcomb and Andrew, 1978
Rainbow trout	*Onchorhynchus mykiss*	Zinc sulfate	140 - 547 (hardness 26 mg/ℓ $CaCO_3$)	LC_{50}	Sinley, et al., 1974

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In acute metal toxicity studies with salmonid species, gill damage and increased mucus secretion are often observed; mortality is often related to respiratory disturbance resulting in hypoxia, or body ion depletion due to ionoregulatory disturbance (Handy and Eddy, 1990). Handy and Eddy (1990) identified gill and mucus (but not skin) as important initial uptake sites in rainbow trout for zinc in water. Increased plasma concentrations were also observed indicating absorption across gills. Dietary uptake is also important, as starved fish exhibited zinc loss (Handy and Eddy, 1990).

For chronic exposures, MATCs based on life cycle, partial life cycle, or early life stage tests for fish range from 47 $\mu g/\ell$ for the flagfish (*Jordanella floridae*) at a water hardness of 44 mg/ℓ, to 852 $\mu g/\ell$ for the brook trout *Salvelinus fontinalis* at a water hardness of 45 mg/ℓ. In an early life stage test with rainbow trout, an MATC of 277 $\mu g/\ell$ was reported at a water hardness of 26 mg/ℓ $CaCO_3$. Chronic values for a variety of fish species range from 36.41 $\mu g/\ell$ for the flagfish to 854.7 $\mu g/\ell$ for brook trout (EPA, 1980).

Studies of acute effects of zinc to invertebrates report 48-hour EC_{50} ranging from 100 $\mu g/\ell$ for *Daphnia* at a water hardness of 45 mg/ℓ, to 58,100 $\mu g/\ell$ for a caddisfly at a water hardness of 50 mg/ℓ (Rehwoldt et al., 1973).

In life cycle studies with daphnia the MATCs were 47, 47, 85, and 135 $\mu g/\ell$ at water hardnesses of 104, 211, 45, and 52 mg/ℓ $CaCO_3$, respectively. Chronic toxicity values for invertebrates range from 46.73 $\mu g/\ell$ for *Daphnia* to greater than 5,243 $\mu g/\ell$ for the caddisfly (*Clistomia magnificia*) (EPA, 1980).

The sensitivity range for freshwater plants to zinc is greater than that for fish and invertebrates. Growth of the algae (*Selenastrum capricornutum*) was inhibited at concentrations of 30 $\mu g/\ell$ (Brown and Rattigan, 1979). Reported toxicity values for plants range from a 7-day EC_{50} of 30 $\mu g/\ell$ in algae to a 28-day LC_{50} of 67,700 $\mu g/\ell$ for the duckweed (*Lemna minor*) (Brown and Rattigan, 1979). Some species of algae are particularly sensitive to zinc exposure for instance. Growth of *Selenastrum capricornutum* was inhibited by

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30 $\mu g/\ell$ in 7 days, whereas the 4-day EC_{50} for several other species of green algae (*Chlorella salina* and *Scenedesmos guadricauda*) exceeded 200,000 $\mu g/\ell$ (EPA, 1986).

The relationship between cadmium and zinc toxicity is not always clear. For example, some authors report synergistic effects (Hutchinson and Czyrska, 1975), whereas others report additive or antagonistic reactions (Huebert and Shay, 1992) depending on the plant species. Bennett and Brooks (1989) reported zinc antagonism of cadmium toxicity at non-toxic zinc levels. Huebert and Shay (1992) indicate that zinc reduced the inhibitory effect of cadmium on multiplication rate and final yield in the aquatic macrophyte (*Lemna trisulca*) in a dose dependent manner. At the highest zinc levels however, zinc promoted cadmium uptake from the media. This led the authors to speculate that at non-toxic zinc concentrations, cadmium is excluded from binding sites on cell walls, but as zinc concentrations exceed optimum for the species, this mechanism breaks down. Cadmium toxicity, however, was mitigated by zinc even when uptake was not possibly by exclusion of cadmium from internal binding sites.

Summary of Available Criteria

Freshwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration of zinc (in $\mu g/\ell$) does not exceed the numerical value given by $e^{(0.8473[\ln(hardness)]+0.7614)}$ more than once every 3 years on the average, and if the 1-hour average concentration (in $\mu g/\ell$) does not exceed the numerical value given by $e^{(0.8473[\ln(hardness)]+0.8604)}$ more than once every 3 years on the average (EPA, 1980). The federal AWQC for the protection of freshwater aquatic life are 120 $\mu g/\ell$ (acute) and 110 $\mu g/\ell$ (chronic) at 100 mg/ℓ hardness (EPA, 1980).

The State of Colorado standards are derived from the equations given above for the EPA criteria. The 4-day average corresponds to the chronic state standard, and the 1-hour average corresponds to the acute state standard (CDH, 1991).

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A.15.2.2 Terrestrial and Riparian Organisms

Zinc is one of the essential nutrients for plant growth. Toxic zinc concentrations for corn and cowpeas in acidic southeastern United States soils ranged from 450 to 1,400 ppm and 180 to 700 ppm extractable zinc, respectively. In pot experiments, with different soils types and zinc concentrations of up to 250 ppm zinc in soil, zinc concentrations of 792 ppm in corn, 523 in lettuce, and 702 ppm in alfalfa were associated with yield depressions (Adriano, 1986). In a solution culture, the 5-day acute toxicity threshold for lettuce was 4.3 mg Zn/ℓ, above which root growth ceased (Adriano, 1986).

Concentrations of zinc less than 10 to 20 ppm in soil are considered deficient for plant survival. Concentrations greater than 300 to 1,500 ppm zinc in soil are considered toxic to plants (Bodek et al., 1988).

Zinc toxicity by oral administration is low, and water insoluble zinc salts are practically nontoxic. In rats, acute zinc intoxication causes anemia of the hypochromic microcytic type, decreased erythrocyte production, formation of immature erythrocytes, and increased leukocyte production (Sutton and Nelson, 1937).

Most animals have a high tolerance for zinc, but ruminants are more susceptible to zinc intoxication than monogastric animals. Beef cattle and lambs tolerate 500 μg of zinc per gram of feed (Ott et al., 1966a, 1966b); this is approximately 20 mg/kg body weight. Pigs and rats tolerate up to 0.10 percent zinc in diet (approximately 75 mg/kg body weight assuming a dietary intake rate of 0.75 kg/kg body weight) without harmful effects, but when fed 0.5 percent zinc in the diet, the rats become anemic, grow poorly and have high mortality (Sutton and Nelson, 1937; Lewis et al., 1957). Zinc poisoning has occurred in cattle. In one incident, food contaminated with zinc at a concentration of 20 g/kg caused severe enteritis and some deaths at an estimated intake of 140 g of zinc per cow per day for approximately 2 days (Adriano, 1986). For a 500 kg cow, this is equivalent to 280 mg/kg body weight. Decreased food intake and weight gain were observed in pigs given dietary zinc at concentrations greater than 1,000 mg/kg. In addition, laboratory studies of foals living near lead-zinc smelters

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suggest that excessive exposure to zinc may produce bone changes, joint afflictions, and lameness.

The maximum chronic tolerated level of zinc in grazing animal diet is 300 to 1,000 mg/kg (Bodek et al., 1988). Assuming a dietary intake rate of 0.04 kg/kg body weight (Sax, 1984), this is 12 to 40 mg/kg body weight.

Reported LD_{50} values (mg/kg body weight) for laboratory animals exposed to zinc salts orally (Venugopal and Luckey, 1978) and LD_{100} toxicity values (mg/kg body weight) derived from oral zinc sulfate exposure to laboratory animals include:

Compound	Test Animal	LD_{50} (mg/kg bw)
Zinc Chloride	Mice	350
Zinc Chloride	Rats	350
Zinc Chloride	Guinea Pigs	250
Zinc Phosphide	Rats	45.7
Zinc Acetate	Rats	2,460
Compound	Test Animal	LD_{50} (mg/kg bw)
Zinc Sulfate	Rabbits	2,100
Zinc Sulfate	Rats	2,200

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APPENDIX B

ANALYTICAL DATA UTILIZED IN THE ECOLOGICAL RISK ASSESSMENT

Appendix B-1

Dissolved Metals Concentrations - 1991 Data

SAMPLE	DATE	SILVER	ALUMINUM	ARSENIC	BARIUM	CADMIUM	CHROMIU	COPPER	IRON	MANGANESE	NICKEL	LEAD	ANTIMONY	SELENIUM	ZINC
AG01	05/02/91	0.5 U	9.0 B	8.4 B	10.0 U	5.2	10.0 U	38.4	121.0	-999.0	20.0 U	55.8	40.0 U	1.0 UN	-999.0
AR01	04/30/91	0.5 U	50.0 U	1.0 U	62.1 U	0.3 B	10.0 U	2.0 B	-999.0	93.9	20.0 U	1.0 U	40.0 U	1.0 U	18.6 BN
AR01	06/13/91	0.5 U	50.0 U	1.0 U	16.6 B	0.3 B	10.0 U	4.3 B	72.4 B	26.2	20.0 U	1.0 U	40.0 U	1.0 U	45.8
AR01	07/25/91	0.5 U	50.0 U	1.0 U	41.2 B	0.2 B	10.0 U	4.5 B	124.0	45.3	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
AR01	09/18/91	0.5 U	50.0 U	1.0 U	63.4 BE	0.4 B	10.0 U	1.0 B*	20.0 U	56.7	20.0 U	1.0 UN	-999.0 *	-999.0	17.4 B
AR01	03/25/92	0.1 U	68.3 B	1.0 U	68.6 B	0.1 U	1.0 U	1.0 U	75.6 B	22.9	2.9 B	1.0 U	2.2 B	1.0 U	10.0 U
AR01	03/25/92	0.1 U	50.0 U	1.0 U	72.1 B	0.1 U	1.0 U	1.3 B	52.7 B	23.6	2.4 B	1.0 U	1.4 B	1.0 U	10.0 U
AR02	04/30/91	0.5 U	50.0 U	1.0 U	65.8	0.7 B	10.0 U	3.0 B	-999.0	92.6	20.0 U	1.0 U	40.0 U	1.0 U	30.0 N
AR02	06/12/91	0.5 U	50.0 U	1.0 U	19.4 B	3.9 S	10.0 U	8.2 W	85.2 BE	23.8	20.0 U	1.0 U	40.0 U	1.0 U	43.7
AR02	07/24/91	0.5 U	50.0 U	1.0 U	45.5 B	0.2 B	10.0 U	2.0 B	67.2 B	43.7	20.0 U	1.0 U	40.0 U	1.1 B	157.0
AR02	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/12/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR02	09/17/91	0.5 U	50.0 U	1.2 B	63.3 B	0.1 B	10.0 U	2.8 B	92.3 B	50.0	20.0 U	-999.0 N*	3.1 B	1.0 UN	10.0 U
AR02	03/24/92	0.1 U	50.0 U	1.0 U	68.5 B	0.1 U	1.0 U	3.6 B	24.2 B	14.9 B	2.2 B	1.0 U	5.3 B	1.0 U	10.0 U
AR03	04/30/91	0.5 U	50.0 U	1.0 U	60.1 B	1.2	10.0 U	5.1 B	-999.0	-999.0	20.0 U	1.0 U	40.0 U	1.0 U	140.0 N
AR03	06/12/91	0.5 U	50.0 U	1.0 U	23.6 B	1.6	10.0 U	5.5 B	89.1 BE	202.0	20.0 U	1.0 U	40.0 U	1.0 U	376.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0

-1 = NO DATA AVAILABLE; -999 = REJECTED DATA

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Appendix B-1 (continued)

Dissolved Metals Concentrations - 1991 Data

SAMPLE	DATE	SILVER	ALUMINUM	ARSENIC	BARIUM	CADMIUM	CHROMIU	COPPER	IRON	MANGANESE	NICKEL	LEAD	ANTIMONY	SELENIUM	ZINC
AR03	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/12/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR03	09/12/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	04/29/91	0.5 U	50.0 U	1.0 U	24.9 B	1.5	10.0 U	13.1	-999.0	-999.0	20.0 U	1.2 B	40.0 U	1.0 U	-999.0 N
AR04	06/11/91	0.5 UE	50.0 U	1.0 U	26.9 B	1.5	10.0 U	8.7 B	91.5 B	115.0	20.0 U	1.0 U	40.0 U	1.0 U	202.0
AR04	07/23/91	0.5 U	50.0 U	1.0 U	32.7 B	1.0 B	10.0 U	3.8 B	139.0	160.0	20.0 U	1.0 U	40.0 U	1.0 U	46.1
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR04	09/16/91	0.1 U	50.0 U	1.0 U	36.1 B	-999.0 N	10.0 U	2.2 B	92.9 B	110.0	20.0 U	1.0 U	9.8 B	-999.0 N	10.0 B
AR04	03/23/92	0.1 U	50.0 U	1.0 U	37.3 B	0.2 B	1.0 U	4.7 B	54.5 B	143.0	1.6 B	1.0 U	5.9 B	1.0 U	10.0 U
AR05	04/29/91	3.2	50.0 U	23.0	24.6 B	2.7	10.0 U	33.4	-999.0	-999.0	20.0 U	18.6 S	40.0 U	11.6 W	-999.0 N
AR05	06/11/91	0.5 U	50.0 U	1.0 U	28.1 B	1.8 B	10.0 U	10.2 W	89.6 B	97.5	20.0 U	1.0 U	40.0 U	1.0 U	189.0
AR05	07/23/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR05	09/16/91	10.0 U	50.0 U	1.0 U	43.1 B	-999.0 N	10.0 U	1.0 U	57.1 B	60.5	20.0 U	20.0 U	40.0 U	-999.0 N	10.0 U
AR05	03/23/92	0.1 U	50.0 U	1.0 U	39.4 B	0.2 B	1.0 U	1.7 B	58.1 B	62.3	2.2 B	1.0 U	2.8 B	1.0 U	10.0 U
AR05	03/23/92	0.1 U	50.0 U	1.0 U	41.8 B	0.3 B	1.0 U	1.9 B	45.3 B	63.7	2.1 B	1.0 U	1.6 B	1.0 U	10.0 U
AR3A	07/24/91	0.5 U	50.0 U	1.0 U	52.2 B	1.0	10.0	2.4 B	45.1 B	165.0	20.0 U	1.0 U	40.0 U	1.0 U	122.0
AR3A	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3A	09/16/91	0.1 U	50.0 U	1.0 U	59.8 B	-999.0 N	10.0 U	1.0 U	41.8 B	112.0	20.0 U	1.0 U	9.8 B	-999.0 N	10.6 B
AR3A	03/23/92	0.1 U	50.0 U	1.0 U	65.3 B	0.9 B	1.0 U	5.8 B	20.0 U	387.0	3.3 B	1.0 U	5.6 B	1.0 U	32.3
AR3E	07/24/91	0.5 U	50.0 U	1.0 U	46.4 B	1.2	10.0 U	5.2 B	44.1 B	324.0	20.0 U	1.0 U	40.0 U	1.0 U	167.0
AR3E	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0

-1 = NO DATA AVAILABLE; -999 = REJECTED DATA

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Appendix B-1 (continued)

Dissolved Metals Concentrations - 1991 Data

SAMPLE	DATE	SILVER	ALUMINUM	ARSENIC	BARIUM	CADMIUM	CHROMIU	COPPER	IRON	MANGANESE	NICKEL	LEAD	ANTIMONY	SELENIUM	ZINC
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
AR3E	09/17/91	0.5 U	50.0 U	1.1 B	60.1 B	0.5 B	10.0 U	1.4 B	78.7 B	195.0	20.0 U	-999.0 N*	7.2 B	1.0 UN	14.9 B
AR3E	03/23/92	0.1 U	50.0 U	1.0 U	63.6 B	1.4 B	1.0 U	2.1 B	20.0 U	699.0	3.4 B	1.0 U	2.4 B	1.0 U	399.0
AR3W	07/24/91	0.5 U	50.0 U	1.0 U	43.9 B	0.3 B	10.0 U	3.8 B	63.0 B	48.7	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
AR3W	09/17/91	0.5 U	50.0 U	1.0 U	63.0 B	0.1 B	10.0 U	5.0 U	72.9 B	87.6	20.0 U	-999.0 N*	5.4 B	1.0 UN	11.2 B
AR3W	03/23/92	0.1 U	50.0 U	1.0 U	59.0 B	0.8 B	1.0 U	13.7 B	20.0 U	426.0	3.1 B	1.0 U	8.5 B	1.1 B	116.0
CG01	05/03/91	0.5	701.0	1.0 U	31.0 B	122.0	10.0 U	31.1	200.0	2130.0	20.0 U	687.0	40.0 U	10.0	12400.0
CG01	06/12/91	0.5 U	22500.0	1.0 U	13.7 B	225.0	10.0 U	713.0	2340.0 E	6570.0	29.5 B	997.0	40.0 U	1.0 U	30500.0
CG02	05/03/91	0.5 U	202.0	1.0 U	28.2 B	163.0	31.9 U	10.6	8700.0	-999.0	20.0 U	35.1	40.0 U	10.0 UN	-999.0
CG02	06/12/91	0.5 U	7800.0	1.0 U	17.7 B	257.0	10.0 U	355.0	1200.0 E	7850.0	26.8 B	492.0	40.0 U	1.0 U	34100.0
CG02	03/24/92	0.1 U	50.0 U	1.0 U	23.6 B	4.3 B	1.0 U	7.7 B	30.1 B	123.0	8.0 B	1.0 U	5.9 B	2.0 B	353.0
CG03	05/03/91	0.5 U	50.0 U	1.0 U	44.2 B	66.1	10.0 U	3.8	229.0	6660.0	20.0 U	1.0	40.0 U	1.0 U	21100.0
CG03	06/12/91	0.5 U	300.0	1.0 U	37.2 B	327.0	10.0 U	874.0	2320.0 E	17300.0	-999.0	37.8 W	40.0 U	1.0 U	71400.0
CG03	07/24/91	0.5 U	50.0 U	1.2 B	36.1 B	0.1 U	10.0 U	2.0 B	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
CG03	09/17/91	0.5 U	50.0 U	1.0 U	33.3 B	98.2	10.0 U	2.4 B	223.0	12800.0	36.6 B	-999.0 N	8.1 B	5.0 UE	40500.0
CG03	03/24/92	0.1 U	50.0 U	1.0 U	46.7 B	10.4	1.0 U	5.9 B	77.6 B	693.0	5.2 B	1.7 B	6.0 B	2.3 B	2780.0
CG04	05/02/91	0.5 U	8920.0	1.3 B	65.7 B	146.0	10.0 U	888.0	90600.0	-999.0	50.1	2240.0	40.0 U	1.0 UN	-999.0
CG04	06/12/91	0.5 U	2880.0	1.0 B	35.5 B	313.0	10.0 U	863.0	6400.0 E	27800.0	47.9	171.0	40.0 U	1.5 B	71900.0
CG04	07/24/91	0.5 U	50.0 U	1.0 U	45.1 B	78.4	10.0 U	4.2 B	65.1 B	16400.0	25.8 B	1.0 U	40.0 U	7.0 B	9920.0
CG04	09/17/91	0.5 U	716.0	1.0 U	24.2 B	83.0	10.0 U	88.3	15700.0	59800.0	96.1	-999.0	12.8	5.0 UN	55200.0
CG04	03/24/92	0.1 U	50.0 U	1.0 U	39.4 B	26.8	1.0 U	8.1 B	2470.0	12500.0	24.7 B	1.0 U	9.4 B	3.7 B	13400.0
CG04	03/24/92	0.1 U	50.0 U	1.1 B	38.4 B	28.3	1.0 U	5.3 B	2460.0	12400.0	18.0 B	1.0 U	4.8 B	4.4 B	13300.0
CG05	05/02/91	0.5 U	6110.0	1.0 U	34.0 B	88.4 S	10.0 U	333.0	22900.0	-999.0	38.6 B	433.0	40.0 U	1.0 UE	-999.0
CG05	06/12/91	0.5 U	323.0	1.0 U	32.8 B	254.0	10.0 U	580.0	4800.0 E	23200.0	43.0	107.0 B+	40.0 U	1.0 U	59400.0
CG05	07/24/91	0.5 U	50.0 U	1.0 U	61.1 B	88.6	10.0 U	9.8 B	20.0 U	11700.0	20.0 U	3.6 W	40.0 U	5.0 U	7770.0
CG05	09/17/91	0.5 U	50.0 U	1.0 U	25.8 B	46.9	10.0 U	13.1	140.0	22600.0	38.4 B	-999.0	10.0 U	5.0 UN	26500.0
CG05	03/24/92	0.1 U	50.0 U	1.3 B	40.6 B	31.3	1.0 U	4.7 B	413.0	11800.0	17.5 B	1.0 U	5.4 B	5.1	14200.0
CG06	05/01/91	0.5 U	50.0 U	1.0 U	20.0 B	47.8	10.0 U	8.7 B	20.0 U	9680.0	20.0 U	1.0 U	40.0 U	1.0 U	11800.0
CG06	06/12/91	0.5 U	50.0 U	1.0 U	29.9 B	137.0	10.0 U	20.7	20.0 UE	15100.0	28.6 B	5.0 U	40.0 U	1.2 B	33400.0
CG06	07/24/91	0.5 U	50.0 U	1.0 U	37.1 B	9.1	10.0 U	6.8 B	20.0 U	4310.0	20.0 U	1.0 U	40.0 U	1.0 U	114.0
CG06	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/24/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0

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Appendix B-1 (continued)

Dissolved Metals Concentrations - 1991 Data

SAMPLE	DATE	SILVER	ALUMINUM	ARSENIC	BARIUM	CADMIUM	CHROMIU	COPPER	IRON	MANGANESE	NICKEL	LEAD	ANTIMONY	SELENIUM	ZINC
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	08/30/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/11/91	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
CG06	09/17/91	0.5 U	50.0 U	1.0 U	20.1 B	2.9 S	10.0 U	10.8	20.0 U	3590.0	20.0 U	-999.0	8.6 B	1.0 UN	35.0
CG06	03/24/92	0.1 U	50.0 U	1.1 B	33.6 B	8.0	1.0 U	5.2 B	20.0 U	3980.0	8.7 B	1.0 U	8.4 B	1.0 U	3090.0
EF01	05/01/91	0.5 U	50.0 U	1.0 U	104.0 B	0.3 B	10.0 U	1.4 B	44.9 B	18.7	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EF01	06/13/91	0.5 U	50.0 U	1.0 U	33.6 B	0.1 U	10.0 U	2.6 B	43.1 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EF01	09/18/91	0.5 U	50.0 U	1.0 U	75.5 BE	0.1 U	10.0 U	1.0 U*	20.0 U	10.6 B	20.0 U	1.0 UN	3.2 B	-999.0	10.0 U
EF02	05/01/91	0.5 U	50.0 U	1.0 U	99.5 U	0.9 B	10.0 U	2.5 B	20.0 U	425.0	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EF02	06/13/91	0.5 U	50.0 U	1.0 U	32.8 B	0.3 B	10.0 U	3.8 B	40.1 B	36.5	20.0 U	1.0 U	40.0 U	1.0 U	19.9 B
EF02	09/18/91	0.5 U	50.0 U	1.0 U	77.0 BE	0.2 B	10.0 U	1.0 U*	20.0 U	78.1	20.0 U	1.0 UN	3.9 B	-999.0	10.0 U
EF02	03/25/92	0.1 U	50.0 U	1.0 U	95.6 B	0.1 U	1.0 U	23.4 B	20.0 U	47.3	8.1 B	1.0 U	8.5 B	1.6 B	10.0 U
EG01	06/13/91	0.5 U	50.0 U	1.0 U	49.6 B	0.5 B	10.0 U	3.6 B	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EG01	09/18/91	0.5 U	50.0 U	1.0 U	69.3 BE	0.3 B	10.0 U	1.0 U*	20.0 U	10.0 U	20.0 U	1.0 UN	4.1 BS	-999.0	10.0 U
EG02	06/13/91	0.5 U	50.0 U	1.0 U	50.8 B	0.5 B	10.0 U	4.0 B	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM01	06/11/91	0.5 U	50.0 U	1.0 U	66.3 B	0.1 U	10.0 U	1.4 B	45.0 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM01	07/23/91	0.5 U	50.0 U	1.0 U	83.0 B	0.1 U	10.0 U	1.0 U	37.8 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM01	09/16/91	0.1 U	50.0 U	1.0 U	65.6 B	-999.0 N	10.0 U	3.2 B	29.1 B	10.0 U	20.0 U	1.0 U	11.5 M	-999.0 N	10.0 U
EM02	05/03/91	0.5 U	50.0 U	1.0 U	96.0 B	0.1 U	10.0 U	2.9 B	98.1 B	33.5	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM02	06/11/91	0.5 U	50.0 U	1.0 U	77.0 B	0.1 U	10.0 U	1.4 B	28.4 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM02	07/23/91	0.5 U	95.6 B	1.0 U	86.3 B	0.1 B	10.0 U	1.0 U	-999.0	-999.0	20.0 U	2.2 B	40.0 U	1.0 U	10.0 U
EM02	09/16/91	0.1 U	50.0 U	1.0 U	85.6 B	-999.0 N	10.0 U	1.0 U	20.0 U	10.2 B	20.0 U	1.0 U	7.2 B	-999.0 N	10.0 U
EM02	03/23/92	0.1 U	50.0 U	1.0 U	82.5 B	0.1 U	1.0 U	3.7 B	20.0 U	10.0 U	1.8 B	1.0 U	4.8 B	1.0 U	10.0 U
EM03	04/30/91	0.5 U	50.0 U	1.0 U	-999.0	0.1 B	10.0 U	2.1 B	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 UN
EM03	06/11/91	0.5 U	50.0 U	1.0 U	86.6 B	0.1 U	10.0 U	3.0 B	99.4 B	10.3 B	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM03	07/23/91	0.5 U	50.0 U	1.2 B	76.0 B	0.1 U	10.0 U	1.0 B	129.0	28.6	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM03	09/16/91	0.1 U	50.0 U	1.0 U	87.3 B	-999.0 N	10.0 U	1.0 U	89.6 B	29.3	20.0 U	1.0 U	6.1 B	-999.0 N	10.0 U
EM04	04/29/91	0.5 U	50.0 U	1.4 B	95.3 B	0.1 B	10.0 U	25.0	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 UN
EM04	06/11/91	0.5 U	50.0 U	1.0 U	108.0 B	0.1 U	10.0 U	1.2 B	58.2 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM04	07/23/91	0.5 U	50.0 U	1.0 U	87.2 B	0.1 U	10.0 U	1.3 B	75.6 B	21.8	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
EM04	09/16/91	0.1 U	50.0 U	1.0 U	107.0 B	-999.0 N	10.0 U	1.0 U	59.5 B	30.4	20.0 U	1.0 U	9.4 B	-999.0 N	10.0 U
EM04	03/23/92	0.1 U	50.0 U	1.0 U	133.0 B	0.1 U	1.0 U	5.2 B	54.1 B	31.0	2.6 B	1.0 U	6.4 B	1.0 U	10.0 U
GG01	05/02/91	0.5 U	85.8 B	11.2	10.0 U	1.9	10.0 U	9.7 B	61.0 B	-999.0	20.0 U	15.5	40.0 U	1.0 UN	-999.0
HC01	04/30/91	0.5 U	50.0 U	1.0 U	21.7 B	0.1 B	10.0 U	23.8	63.2 B	10.3 B	20.0 U	1.0 U	40.0 U	1.0 U	10.0 UN
HC01	06/11/91	0.5 U	50.0 U	1.0 U	13.6 B	0.1 U	10.0 U	4.1 B	20.0 U	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
HC01	07/23/91	0.5 U	50.0 U	1.0 U	13.7 B	0.1 U	10.0 U	1.4 B	32.5 B	10.0 U	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
HC01	09/16/91	0.1 U	50.0 U	1.0 U	16.7 B	-999.0 N	10.0 U	1.0 U	20.0 U	10.0 U	20.0 U	1.0 U	6.0 B	-999.0 N	10.0 U

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Appendix B-1 (continued)

Dissolved Metals Concentrations - 1991 Data

SAMPLE	DATE	SILVER	ALUMINUM	ARSENIC	BARIUM	CADMIUM	CHROMIU	COPPER	IRON	MANGANESE	NICKEL	LEAD	ANTIMONY	SELENIUM	ZINC
IG01	04/29/91	0.5 U	50.0 U	1.0 U	59.3 B	0.1 B	10.0 U	1.2 B	63.2 B	10.0 U	20.0 U	3.5 W	40.0 U	1.0 U	-999.0 N
IG01	06/11/91	0.5 U	50.0 U	1.0 U	37.0 B	0.2 B	10.0 U	7.5 B	20.0 U	11.1 B	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
IG01	07/23/91	0.5 U	50.0 U	1.0 U	40.3 B	0.2 B	10.0 U	2.0 B	20.0 U	28.6	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
IG01	09/16/91	0.1 U	50.0 U	1.0 U	44.1 B	-999.0 N	10.0 U	1.0 U	20.0 U	11.6 B	20.0 U	1.0 U	5.2 B	-999.0 N	10.0 U
LF01	05/01/91	0.5 U	50.0 U	1.0 U	200.0 U	0.1 B	10.0 U	3.4 B	292.0	70.0	40.0 U	1.0 U	40.0 U	1.0 U	20.0 U
LF01	06/11/91	0.5 U	50.0 U	1.0 U	10.4 B	1.5	10.0 U	4.5 B	202.0	54.7	20.0 U	1.0 U	40.0 U	1.0 U	10.0 U
LF01	07/23/91	0.5 U	50.0 U	1.0 U	10.0 U	0.1 U	10.0 U	2.0 U	292.0	58.8	20.0 U	1.0 U	40.0 U	1.0 U	13.4 U
LF01	09/16/91	0.1 U	50.0 U	1.0 U	10.9 B	-999.0 N	10.0 U	1.0 U	206.0	15.5	20.0 U	1.0 U	-999.0 W	-999.0 N	10.0 U
LF01	03/23/92	0.1 U	50.0 U	1.0 U	18.1 B	0.1 U	1.0 U	5.2 B	152.0	77.3	1.0 U	1.0 U	5.9 B	1.0 U	10.0 U
MG01	05/01/91	0.5 U	77.3 B	1.2 B	143.0 B	63.2	10.0 U	43.4	48.8 B	448.0	20.0 U	226.0	40.0 U	1.0 U	3280.0
OG01	05/02/91	0.5 U	33500.0	15.8 B	45.8 B	110.0	10.0 U	893.0	728000.0	257000.0	192.0	-999.0	40.0 U	10.0 UN	114000.0
OG01	06/12/91	0.7 W	********	-999.0 +	250.0 U	555.0	250.0 U	8420.0	********* E	*********	916.0 B	-999.0	1000.0 U	10.0 U	644000.0
OG01	07/24/91	-999.0 E	96200.0	268.0 S	500.0 U	664.0	500.0 U	2270.0	*********	*********	1000.0 U	1.0 U	2000.0 U	20.0 UE	-999.0
PG01	05/02/91	0.5 U	88.1 B	3.8 B	35.4 B	2.9	10.0 U	13.2	105.0	-999.0	20.0 U	12.9	40.0 U	1.0 U	-999.0
SD01	05/02/91	0.5 U	50.0 U	1.0 U	33.0 B	136.0	10.0 U	16.8	71.5 B	-999.0	20.0 U	8.6	40.0 U	1.0 UN	-999.0
SD01	06/12/91	0.5 U	11400.0	1.4 B	46.5 B	410.0	10.0 U	910.0	10500.0 E	15600.0	20.4 B	899.0	40.0 U	1.0 U	52400.0
SD01	07/24/91	0.5 U	50.0 U	1.0 U	48.3 B	4.7 B	10.0 U	2.8 B	236.0	348.0	20.0 U	1.0 U	40.0 U	1.0 U	177.0
SD01	03/24/92	0.1 U	71.1 B	3.2 B	25.6 B	1.7 B	1.0 U	13.7 B	26.2 B	138.0	3.5 B	4.6	11.5 B	1.0 U	56.8
SG01	05/02/91	0.5 U	50.0 U	1.0 U	42.6 B	165.0	10.0 U	38.9	49.7 B	-999.0	20.0 U	313.0	40.0 U	1.0 UN	-999.0
SG01	06/12/91	0.5 U	12300.0	1.0 U	34.7 B	393.0	10.0 U	801.0	17600.0 E	14800.0	27.1 B	208.0	40.0 U	1.0 U	50700.0
SG01	07/24/91	5.4	52200.0	1110.0	100.0 U	934.0	100.0 U	7880.0	*********	50600.0	200.0 U	1420.0	400.0 U	5.0 UE	94800.0
TC01-1	05/01/91	0.5 U	50.0 U	1.0 U	25.8 U	0.6 B	10.0 U	6.1 B	720.0	52.1	20.0 U	1.0 U	40.0 U	1.0 U	52.0
TC01-1	06/13/91	0.5 U	66.2 B	1.0 U	10.0 U	0.4 B	10.0 U	8.6 B	119.0	33.3	20.0 U	1.0 U	40.0 U	1.0 U	64.9
TC01-2	09/18/91	0.5 U	50.0 U	1.0 U	11.5 BE	0.1 U	10.0 U	7.7 B*	169.0	20.6	20.0 U	1.6 BN	3.6 B*	-999.0 UN	10.6 B
TC01-3	03/25/92	0.1 U	50.0 U	1.0 U	108.0 B	0.4 B	523.0	19.6 B	106.0	16.4	37.5 B	2.4 B	7.2 B	1.0 U	10.0 U

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Appendix B-2

Dissolved Metal Concentration for July 1992 through August 1993 Data

Station	Date	Aluminum	Antimony	Arsenic	Barium	Cadmium	Chromium	Copper	Iron	Lead	Manganese	Mercury	Nickel	Selenium	Silver	Zinc
		ug/l														
AR02	07/21/92															
AR03E	07/21/92	25 U	8.8 J	0.5 U	50.1 J	0.7 J	0.5 U	6.4 J	10 U	0.5 U	262	0.1 U	2 J	0.5 U	0.05 U	5 U
AR03W	07/21/92	25 U	2.5 J	0.5 U	53.9 J	0.11 J	0.5 U	1.7 J	10 U	0.5 U	21	0.1 U	1.7 J	0.5 U	0.05 U	5 U
CG03	07/21/92	25 U	21 J	0.5 U	42.4 J	9	0.5 U	1.6 J	10 U	0.5 U	633 J	0.1 U	12.6 J	0.5 J	0.05 U	2850
CG04	07/21/92	25 U	11.4 J	0.5 U	39.9 J	14.2	0.5 U	13.3 J	927	0.5 U	6910	0.1 U	21.3 J	1.1 J	0.05 U	7520
CG05	07/21/92	25 U	2.3 J	0.5 U	43.1 J	13.5	0.5 U	0.5 U	10 U	0.5 U	6310	0.1 U	12.7 J	2.6 J	0.05 U	6640
CG06	07/21/92	25 U	2.7 J	0.5 U	30.1 J	6.4	0.5 U	5 J	10 U	0.5 U	3890	0.1 U	11.9 J	0.5 U	0.05 U	2060
AR02	08/18/92	25 U	12.5 J	0.5 U	40.3 J	0.05 U	0.5 U	7.5 J	86.3 J	0.5 U	21.3	0.1 U	1.2 J	0.5 U	0.1 J	5 U
AR03E	08/18/92	25 U	18.6 J	0.5 U	47.4 J	0.8 J	0.5 U	5.2 J	61.8 J	0.5 U	391	0.1 U	1.9 J	0.5 U	0.1 J	16.8 J
AR03W	08/18/92	25 U	12.4 J	0.5 U	42.8 J	0.05 U	0.5 U	12.8 J	93.1 J	0.5 U	22.7	0.1 U	1.2 J	0.5 U	0.1 J	5 U
CG03	08/18/92	25 U	13 J	0.5 U	45.7 J	3.2 J	0.5 U	12.6 J	10 U	0.5 U	235	0.1 U	5.8 J	0.5 U	0.1 J	733
CG04	08/18/92	25 U	11.4 J	0.5 U	41 J	12.9	0.5 U	13 J	119	0.5 U	5550	0.1 U	13.6 J	0.5 U	0.1 J	6040
CG05	08/18/92	25 U	16.4 J	0.5 U	36.4 J	12.3	0.5 U	5.1 J	10 U	0.5 U	6420 J	0.1 U	11.3 J	0.5 U	0.1 J	5610
CG06	08/18/92	25 U	11.1 J	0.5 U	18.3 J	4.5 J	0.5 U	14.2 J	10 U	0.5 U	3380	0.1 U	6.3 J	0.5 U	0.1 J	957
AR02	09/15/92	25 U	12.8 J	0.5 U	47.9 J	0.15 J	0.5 U	7.3 J	26 J	0.5 U	20.9	0.1 U	0.5 U	0.5 U	0.05 UJ	5 U
AR03E	09/15/92	25 U	23.4 J	0.5 U	52.8 J	1.5 J	0.5 U	3.5 J	10 U	0.5 U	483	0.1 U	1.5 J	0.5 U	0.05 UJ	24.7
AR03W	09/15/92	25 U	12.7 J	0.5 U	46.2 J	0.3 J	0.5 U	5.7 J	20 J	0.5 U	43.5	0.1 U	0.5 U	0.5 U	0.05 UJ	5 U
CG03	09/15/92	25 U	12.2 J	0.5 U	41.5 J	3.5 J	0.5 U	7.3 J	10 U	0.5 U	372	0.1 U	4.1 J	0.5 U	0.05 UJ	1150
CG04	09/15/92	25 U	13 J	0.5 U	35.8 J	0	0.5 U	9.9 J	920	0.5 U	7880	0.1 U	12.5 J	0.5 U	0.05 UJ	6460
CG05	09/15/92	25 U	11.8 J	0.5 U	33.4 J	11.8	0.5 U	9.2 J	57.5 J	0.5 U	5850	0.1 U	10.7 J	0.5 U	0.05 UJ	5540
CG06	09/15/92	25 U	15 J	0.5 U	27.5 J	6.7	0.5 U	5.2 J	10 U	0.5 U	4090	0.1 U	7.3 J	0.5 UJ	0.05 UJ	2460
AR02	10/13/92	25 U	12.3 U	0.5 U	52.2 J	0.11 J	0.5 U	5.9 J	20 J	0.5 U	26	0.2	0.5 U	0.5 U	0.05 U	14.3 J
AR03E	10/13/92	25 U	2.4 J	0.5 U	65.3 J	0.67 J	0.5 U	2.3 J	10 U	0.5 U	277	0.1 U	1.2 J	0.5 U	0.05 U	5 U
AR03W	10/13/92	25 U	13.2 J	0.5 U	51.9 J	0.35 J	0.5 U	8.5 J	10 U	0.5 U	35.9	0.1 U	0.5 U	0.5 U	0.05 U	5 U
CG03	10/13/92	25 U	10.8 J	0.5 U	39.3 J	5.3	0.5 U	4.9 J	77.2 J	0.5 U	571	0.2	4.6 J	0.5 U	0.05 U	2080 J
CG04	10/13/92	25 U	10.8 J	0.5 U	37.6 J	15.7	0.5 U	10.8 J	2070	0.5 U	11000	0.1 U	15.1 J	0.5 U	0.05 U	9480
AR02	11/20/92	62.4 U	0.5 U	0.5 U	67.2 J	0.1 U	0.5 U	1.2 J	71.1 J	0.5 U	22.1	0.1 U	0.5 U	1 U	0.05 U	5 U
AR3A	11/20/92	25 U	8.8 J	0.5 U	48.8 J	1.3 J	0.5 U	10 J	63.5 J	0.5 U	203	0.1 U	1 J	1 U	0.05 U	232
CG03	11/20/92	25 U	9.5 J	0.5 U	40.1 J	4.8 J	0.5 U	7.8 J	52.5 J	0.5 U	411	0.1 U	2.4 J	1 U	0.05 U	1510
CG04	11/20/92	25 U	7.3 J	0.5 U	34.3 J	20.4	0.5 U	14 J	2820	0.5 U	12400 J	0.1 U	16.3 J	1 U	0.05 U	11400 J
CG05	11/20/92	25 U	6.8 J	0.5 U	28.2 J	25.2	0.5 U	9.8 J	1710	0.5 U	14600	0.1 U	18.7 J	1 U	0.05 U	14300
CG06	11/20/92	25 U	8.1 J	1.1 J	13.4 J	7.2	0.5 U	9.4 J	10 U	0.5 U	4350 J	0.1 U	6.7 J	1 U	0.05 J	3270
AR02	12/15/92	25 U	8.6 J	0.5 U	52.2 J	0.1 U	0.5 U	6.3 J	52 J	0.5 U	14.5 J	0.1 U	1.1 U	1 U	0.05 U	5 U
AR3A	12/15/92	58.8 J	7.1 J	0.5 U	54 J	1.1 J	0.5 U	7.4 J	68.6 J	0.5 U	254	0.1 U	1.5 J	1 U	0.12 J	315
CG05	12/15/92	25 U	7.5 J	0.5 U	32.6 J	23.8	0.5 U	23.4 J	1700	0.5 U	12100	0.1 U	17.4 J	1 U	0.05 U	13400
CG06	12/15/92	25 U	8.2 U	1.2 J	14 J	4.7 J	0.5 U	14.3 J	10 U	0.5 U	3130	0.1 U	6.4 J	1 U	0.05 U	1880
AR3A	01/18/93	25 U	4.55 U	0.5 U	23.6 U	0.265 U	0.5 U	5.9 U	23.9 U	1.5 U	189	0.1 U	0.85 U	1 U	0.055 U	20
CG06	01/18/93	25 U	3.85 U	0.65 U	9.65 U	1.65 U	0.5 U	6.65 U	10 UJ	0.5 U	2180	0.1 U	2.25 U	1 U	0.05 U	1020
AR3A	02/17/93	25 U	4.55 U	0.5 U	24.25 U	0.465 U	0.5 U	8.7 U	10.45 U	0.5 U	501	0.1 U	1.3 U	1 U	0.05 UJ	179
CG06	02/17/93	25 U	3.35 U	0.5 U	15.85 U	2.45 U	0.5 U	6.25 U	10 U	2.15	3430	0.1 U	2.9 U	1 U	0.05 UJ	2050

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Appendix B-2 (Continued)

Dissolved Metal Concentration for July 1992 through August 1993 Data

Station	Date	Aluminum	Antimony	Arsenic	Barium	Cadmium	Chromium	Copper	Iron	Lead	Manganese	Mercury	Nickel	Selenium	Silver	Zinc
		ug/l														
AR02	03/08/93	25 U	13.65 U	0.5 U	28.6 U	0.1 U	0.5 U	2.35 U	10 U	0.5 U	16.1	0.1 U	0.85 U	1 U	0.05 UJ	5 U
AR3A	03/08/93	25 U	11.55 U	0.5 U	26.75 U	0.38 U	0.5 U	15.35	10 U	0.5 U	336	0.1 U	1.35 U	1 U	0.05 UJ	73.8
CG06	03/08/93	25 U	12.7 U	0.55 U	14.05 U	5.3	0.5 U	11.4 U	10 U	0.5 U	3580	0.1 U	3.65 U	1 U	0.05 UJ	1880
AR02	04/19/93	25 U	12.45 U	0.5 U	26.95 U	0.1 U	0.5 U	2.2 U	26.6 U	0.5 U	20.5	0.1 U	1.95 U	1 U	0.05 U	5 U
AR3A	04/19/93	25 U	11 U	0.5 U	25.5 U	1 U	0.5 U	1.9 U	26.35 U	0.5 U	1480	0.1 U	1.8 U	1 U	0.05 U	831
CG06	04/19/93	25 U	9.6 U	0.5 U	14.75 U	11.8	0.5 U	6.35 U	78 U	0.5 U	9990	0.1 U	13.75 U	1 U	0.05 U	6870
AR02	05/03/93	25 U	12.85 U	0.5 U	20.85 U	0.1 U	0.5 U	3.45 U	132 J	0.5 U	20.4	0.1 U	0.5 U	1 U	0.05 U	5.3 U
AR3A	05/03/93	33.95 U	10.7 U	0.5 U	21.95 U	5.1	0.5 U	12.35 U	630	4.1	1780 J	0.1 U	1.3 U	1 U	0.05 U	1430 J
CG06	05/03/93	6800.	12.35 U	0.5 U	21.35 U	109	0.5 U	416	31200	275	36300	0.1 U	40.9 U	1 U	0.095 U	35900
AR02	06/08/93	25 U	7.8 U	0.5 U	11.35 U	0.235 U	0.5 U	7.6 U	28.25 U	0.5 U	30.6	0.1 UJ	1 U	1 U	0.05 U	40.3
AR3A	06/08/93	25 U	1.2 U	0.5 U	18.05 U	0.9 U	0.5 U	3.25 U	35.1 U	0.9 U	118	0.1 UJ	1.05 U	1 U	0.05 UJ	231 J
CG06	06/08/93	1370	0.65 U	0.5 U	23.05 U	81.4	0.5 U	238	831	113	5580	0.1 UJ	8.55 UJ	1 U	0.05 UJ	14900 J
AR02	07/14/93	25 U	12.95 U	0.5 U	11.95 U	0.165 U	0.5 U	5.1 U	28.4 U	0.5 U	23.8	0.1 U	0.5 U	1 U	0.05 U	5 U
AR3A	07/14/93	25 U	12.45 U	0.5 U	13.6 U	0.36 U	0.5 U	2.65 U	25.5 U	0.5 U	92.6	0.1 U	0.5 U	1 U	0.05 U	55.9
CG06	07/14/93	25 U	12.45 U	0.55 U	14.95 U	12.5 U	0.5 U	3 U	20.3 U	0.5 U	6290	0.1 U	4.85 U	1 U	0.05 U	4640
AR02	08/25/93	25 U	5.05 U	0.5 U	27.7 U	0.1 U	0.5 U	1.65 U	10 U	0.5 U	49.4 J	0.1 U	0.5 U	1 U	0.05 U	5 U
AR3A	08/25/93	25 U	8.1 U	0.5 U	27.9 U	0.75 U	0.5 U	3.15 U	12.7 U	0.5 U	265	0.1 U	0.7 U	1 U	0.05 U	269
CG06	08/25/93															

Note: "U" qualified data have been divided by 2 to obtain 1/2 the detection limit.
All "R" qualified data have been deleted.

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Appendix B-3

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Cyanide		Silver		Aluminum		Arsenic		Barium		Calcium		Cadmium		Chromium		Copper	
AG01	F	05/02/91	1.00	UN J	43.40		23800.00	J	574.00		812.00				52.80	J	17.10		1050.00	J
AR01	F	04/30/91	1.00	U	0.28	U	53.00	B J	0.55	U	60.90	B J			0.60	B J	5.00	U	2.80	B J
AR01	F	06/13/91	1.00	U UJ	0.28	U J	408.00		1.40	B J	35.20	B J			0.59	B J	5.00	U J	8.00	B J
AR01	F	07/25/91	1.00	U J	0.28	U	92.80	B J	0.55	U	43.60	B J			0.40	BW J	5.50	U	5.20	B J
AR01	F	09/18/91				R	27.50	U	0.55	U	68.50	B* J			0.40	B* J	5.50	U	0.55	U
AR02	F	04/30/91	1.00	U	0.28	U	25.00	U	0.55	U	63.90	B J			0.62	B J	5.00	U	4.70	B J
AR02	F	06/12/91		UN R	0.28	U	1010.00		0.55	UN J	46.00	B J			0.94	B J	5.00	U J	8.00	B J
AR02	F	07/24/91			0.28	U	78.40	B J	0.55	U	51.80	B J			0.55	B J	5.50	U J	2.40	BW J
AR02	F	08/24/91			0.28	UN J			1.10	B J			23200.00		0.96	BN J			1.10	B J
AR02	F	08/24/91			0.28	UN J			0.55	U			23100.00		0.37	BN J			0.55	U
AR02	F	08/24/91			0.28	UN J			1.10				23100.00		0.26	BN J			0.55	U
AR02	F	08/24/91																		
AR02	F	08/30/91			0.28	U			1.90	B J			21800.00		0.26	B J				BN R
AR02	F	08/30/91			0.28	U			0.55	U			21900.00		0.18	B J				BN R
AR02	F	08/30/91			0.28	U			0.55	U			22300.00		0.14	B J				BNW
AR02	F	08/30/91			0.28	U			0.55	U			22200.00		0.12	B				BN R
AR02	F	08/30/91			0.28	U			0.55	U			22500.00		0.12	B J				BN J
AR02	F	08/30/91			0.28	U			0.55	U			22700.00		0.16	B J				BN R
AR02	F	08/30/91			0.28	U			0.55	U			22700.00		0.12	B J				BN R
AR02	F	08/30/91			0.28	U			0.55	U			22600.00		0.06	U				BN R
AR02	F	08/30/91			0.28	U			0.55	U			22700.00		0.06	U				BN R
AR02	F	08/30/91			0.28	UW			0.55	U			22800.00		0.11	B J				BNW
AR02	F	08/30/91			0.28	U			0.55	U			22700.00		0.25	B J				BN R
AR02	F	08/30/91			0.28	U			0.55	U			23500.00		0.18	B J				BN R
AR02	F	09/11/91			0.28	U J			0.55	U J			22400.00	J	0.48	BN J			3.70	BW J
AR02	F	09/11/91			0.28	U J			0.55	U J			22800.00	J	0.29	BN J			7.30	BW J
AR02	F	09/11/91			13.50				0.55	U J			22300.00	J	0.27	BN J			6.50	BW J
AR02	F	09/11/91			0.28	U J			0.55	U J			23700.00	J	0.32	BN J			6.80	BW J
AR02	F	09/11/91			5.70				0.55	U J			22300.00	J	0.28	BN J			7.20	BW J
AR02	F	09/11/91			0.28	U J			0.55	U J			22800.00	J	0.40	BN J			6.60	BW J
AR02	F	09/11/91			0.28	U J			0.55	U J			22200.00	J	0.24	BN J			7.50	BW J
AR02	F	09/11/91			0.28	U J			0.55	U J			22300.00	J	0.32	BN J			8.40	BW
AR02	F	09/11/91			0.28	U J			0.55	U J			22600.00	J	0.33	BN J			6.50	B J
AR02	F	09/11/91			0.28	U J			0.55	U J			22900.00	J	0.19	BN J			1.80	B J
AR02	F	09/11/91			0.28	U J			0.55	U J			23000.00	J	0.23	BN J			7.30	B J
AR02	F	09/12/91			0.28	U J			1.10	J			22900.00	J	0.30	BN J			1.20	B J
AR02	F	09/17/91			0.28	U	27.50	U	0.55	U	61.70	B J			0.40	B J	5.50	U	1.20	BW J
AR03	F	04/30/91	1.00	U	0.55		25.00	U	0.55	U	61.60	B J			0.88	B J	5.00	U	5.30	B J
AR03	F	06/12/91		UN R	0.28	U	934.00		0.55	UN J	47.30	B J			3.00	J	5.00	U J		+S R
AR03	F	09/11/91			0.28	UN J			1.20	B J			24100.00	J	6.40				44.50	BNS J
AR03	F	09/11/91			0.28	UN J			0.55	U			23200.00	J	4.40	S			46.40	BNS J
AR03	F	09/11/91			0.28	UN J			0.55	U			23700.00	J	6.40				27.70	NS
AR03	F	09/11/91			0.28	UN J			0.55	U			23400.00	J	3.80	S			25.70	NS
AR03	F	09/11/91			0.28	UN J			0.55	U			23400.00	J	2.70	S			21.10	BNS J
AR03	F	09/11/91			0.28	UN J			0.55	U			23700.00	J	5.70	S			46.80	BNS J
AR03	F	09/11/91			0.28	UN J			0.55	UW			23400.00	J	5.60	S			38.00	BN J
AR03	F	09/11/91			0.28	UN			0.55	UW			24400.00	J	5.40				35.50	BNW
AR03	F	09/11/91			0.28	UN J			0.55	U			25100.00	J	6.80				26.70	NS
AR03	F	09/11/91			5.40	BN			150.00	S			34400.00		251.00	S J			687.00	N
AR03	F	09/12/91			1.90	UN J			59.00	S			33200.00	J	85.20				333.00	N

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Cyanide		Silver		Aluminum		Arsenic		Barium		Calcium		Cadmium		Chromium		Copper	
AR03	F	09/12/91			1.20	BN J			5.80	BW J			29300.00	J	25.10				92.00	N
AR04	F	04/29/91	1.00	U	0.28	U	319.00		1.60	B J	47.20	B J			3.60	S J	5.00	U	36.10	
AR04	F	06/11/91	11.00	* J	0.28	U J	306.00		0.55	U J	32.40	B J			1.70	J	5.00	U J	6.80	BW J
AR04	F	07/23/91	1.00	U J	0.28	U	147.00		0.55	U	38.50	B J			1.50		5.50	U J	8.90	B J
AR04	F	09/11/91			0.28	UN			0.55	UN J			24100.00	J	2.60	S				BNW
AR04	F	09/11/91																		
AR04	F	09/11/91			0.28	UN U			0.55	UN			23000.00	J	2.60	S			3.20	BNW
AR04	F	09/11/91			0.28	UN U			0.55	UN			23700.00	J	7.10	S			6.70	BNW
AR04	F	09/11/91			0.28	UN U			0.55	UN			23600.00	J	3.20	S			2.90	BNW
AR04	F	09/11/91			0.28	UN U			0.55	UN			23000.00	J	2.30	S			3.00	BNW
AR04	F	09/11/91			0.28	UN			0.55	UN			24000.00	J	2.80	S			5.90	BN J
AR04	F	09/11/91			0.28	UN			0.55	UN			23300.00	J	3.60	S			8.80	BN J
AR04	F	09/11/91			0.28	UN			0.55	UN			23600.00	J	3.40	S			6.90	BN J
AR04	F	09/11/91			0.28	UN			0.55	UN J			24000.00		4.30	S			8.60	BN J
AR04	F	09/11/91			3.90	N			44.40	N J			30000.00		96.40	J			254.00	
AR04	F	09/11/91			2.30	NS			27.70	N J			29200.00	J	58.70				213.00	
AR04	F	09/16/91	5.00	BN J		N R	27.50	U	0.55	U J	41.80	B J			0.80	B J	5.50	U	5.60	BW J
AR05	F	04/29/91	1.00	U	0.28	U	161.00		2.10	B J	46.60	B J			1.90	J	5.00	U	9.50	B J
AR05	F	06/11/91	11.00	* J	0.28	U J	451.00		1.10	J	18.40	U J			2.70	S J	5.00	U J	11.40	UW J
AR05	F	07/23/91	1.00	U J	0.28	U	160.00		0.55	U	43.00	B J			1.50		11.00	J	9.70	B J
AR05	F	09/16/91	1.00	UN J		N R	27.50	U	0.55	U J	47.30	B J			0.99	B J	5.50	U	5.10	BN J
AR3A	F	07/24/91			0.28	U	114.00		0.55	U	53.50	B J			2.30	S	5.50	U J	6.40	B J
AR3A	F	08/24/91			0.28	UN J			0.55	U			25100.00		3.10	NS J			8.90	B J
AR3A	F	08/24/91			0.28	UN J			0.55	U			24500.00		1.60	NS J			3.20	B J
AR3A	F	08/24/91			0.28	UN J			0.55	U			24600.00		1.60	NS J			0.55	U
AR3A	F	08/24/91			0.86	BN J			0.55	U			24300.00			N+ R			0.55	U
AR3A	F	08/30/91			0.28	U			1.30	B J			24500.00		5.50	NS J			3.50	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			24700.00		4.40	NS J			3.50	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			24800.00		4.70	NS J			3.80	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			24900.00		5.20	NS J			10.00	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			25000.00		5.60	NS J			6.90	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			25100.00		5.10	NS J			5.10	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			25500.00		5.50	NS J			5.00	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			25400.00		6.60	NS J			5.10	B J
AR3A	F	08/30/91			0.28	U			0.55	U J			26400.00		8.80	NS J			25.40	S
AR3A	F	08/30/91			0.28	U			0.55	U J			26000.00		7.50	NS J			5.70	B J
AR3A	F	08/30/91			0.28	U			0.55	U			27300.00		7.80	NS J			6.40	B J
AR3A	F	08/30/91			0.28	UN			0.55	U			25900.00		6.90	NS J			8.90	B J
AR3A	F	09/16/91	53.00	N J		N R	27.50	U	0.55	U J	62.80	B J			1.10	J	5.50	U	5.10	BW J
AR3E	F	07/24/91			0.28	U	174.00		0.55	U	54.50	B J			3.40	S	5.50	U J	11.20	
AR3E	F	08/24/91			0.28	UN J			0.55	U			25000.00		2.00	NS J			0.55	U
AR3E	F	08/24/91			0.28	UN J			0.55	U			25000.00		1.10	N J			0.55	U
AR3E	F	08/24/91			0.28	UN J			0.55	U			24800.00		2.00	NS J			0.55	U
AR3E	F	08/24/91			0.28	UN J			0.55	U			24700.00		0.97	BN J			0.55	U
AR3E	F	08/30/91			0.28	UN J			0.55	U			25800.00		7.00	N			9.50	B J
AR3E	F	08/30/91			0.28	UN J			0.55	U			25600.00		7.40	N			13.00	
AR3E	F	08/30/91			0.28	UN J			0.55	U			26100.00		9.00	N			28.20	
AR3E	F	08/30/91			0.28	UN J			2.90	B J			26400.00		7.80	N			13.30	
AR3E	F	08/30/91			0.28	UN J			0.55	U			26200.00		7.80	N			5.80	B J
AR3E	F	08/30/91			0.28	UN J			0.55	U			26500.00		8.60	N			7.60	B J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Cyanide		Silver		Aluminum		Arsenic		Barium		Calcium		Cadmium		Chromium		Copper	
AR3E	F	08/30/91			7.00	N J			2.50	B			27100.00		10.10	N			11.90	
AR3E	F	08/30/91			0.28	UN J			0.55	U			28100.00		15.60	N			70.40	
AR3E	F	08/30/91			0.28	UN J			0.55	U			27900.00		11.10	N			14.60	
AR3E	F	08/30/91			0.28	UN J			0.55	U			27600.00		10.00	N			12.00	
AR3E	F	08/30/91			0.28	UN J			1.20	B J			27000.00		10.00	N			16.30	
AR3E	F	08/30/91			0.28	UN J			1.90	B			27200.00		9.80	N			31.20	S
AR3E	F	09/17/91			0.28	UW	66.40	B J	0.55	U	61.30	B J					5.50	U	5.10	BW J
AR3W	F	07/24/91			0.28	U	112.00		0.55	U	54.50	B J			0.84	B J	5.50	U	3.50	BW J
AR3W	F	09/17/91			0.28	UW	66.50	B J	0.55	UW	61.50	B J			0.59	B J	5.50	U	2.10	BW J
CG01	F	05/03/91	1.00	UN J	7.10		2010.00	J	24.50		71.70	B J			97.50	J	5.00	U	94.00	J
CG01	F	06/12/91		UN R	1.20	S	20100.00	J	0.55	UN J	14.00	B J			198.00	J	5.00	U J	639.00	J
CG02	F	05/03/91	1.00	UN J	11.70		5350.00	J	263.00		127.00	B J			154.00	J	5.00	U	208.00	J
CG02	F	19910612		UN R	1.50	S	7440.00	J	13.80	N J	25.50	B J			236.00		5.00	U J	330.00	
CG03	F	05/03/91	1.00	UN J	0.28	UW	490.00	J	3.30	B J	48.20	B J			67.50	J	5.00	U	51.90	J
CG03	F	06/12/91		UN R	0.28	U	1320.00		0.55	UN J	33.40	B J			290.00	J	5.00	U J	822.00	
CG03	F	07/24/91			0.28	UW	922.00		1.20	B J	47.30	B J			19.20	S	5.50	U J	117.00	
CG03	F	09/17/91			0.28	U	116.00		1.40	BW J	31.70	B J			87.20		5.50	U	13.60	
CG04	F	05/02/91	38.00	N J	74.00		30500.00	J	253.00		488.00				151.00	J	5.00	U	2420.00	J
CG04	F	06/12/91		UN R	0.28	U	3140.00		0.55	UN J	33.90	B J			276.00	J	5.00	U J	772.00	J
CG04	F	07/24/91			9.00		3770.00		28.90	S	96.60	B J			148.00	S J	5.50	U J	346.00	
CG04	F	09/17/91			0.28	UW	2100.00		0.55	U	24.00	B J			77.10		5.50	U	102.00	
CG05	F	05/02/91	1.00	UN J	3.30		7290.00	J	19.60		65.40	B J			87.70	J	5.00	U	404.00	J
CG05	F	06/12/91		UN R	0.28	U	2500.00		0.55	UN J	31.60	B J			217.00	J	5.00	U J	557.00	
CG05	F	07/24/91			318.00		46100.00		627.00		1600.00				208.00		58.00	J	4210.00	
CG05	F	09/17/91			0.28	UW	1830.00		3.80	BW J	25.30	B J			49.80		5.50	U	66.00	
CG06	F	05/01/91	1.00	U J	20.50		6600.00		122.00		228.00				57.30		5.00	U J	494.00	J
CG06	F	06/12/91		UN R	0.28	U	1430.00		0.55	U J	34.80	BN J			138.00		5.00	U J	300.00	
CG06	F	07/24/91			2.20	S	1840.00		11.40		66.80	B J			186.00	S	5.50	U J	16.90	S
CG06	F	08/24/91			0.28	UN J			1.10				70300.00		29.50	NS J			10.55	U
CG06	F	08/24/91			0.28	UN J			1.10	BW J			67400.00		31.50	NS J			30.80	
CG06	F	08/24/91			0.28	UN J			1.50	BW J			60900.00		31.10	NS J			30.80	
CG06	F	08/24/91			0.28	UN J			1.40	BW J			55000.00		23.60	NS J			30.50	
CG06	F	08/30/91				UN R			0.55	UNW			72000.00		71.80					R
CG06	F	08/30/91				UN R			0.55	UN J			71900.00		74.40					R
CG06	F	08/30/91				UN R			2.10	BNW			77600.00		90.10				747.00	
CG06	F	08/30/91				UN R			0.55	UN J			76800.00		86.00					R
CG06	F	08/30/91				UN R			0.55	UN J			77600.00		85.90					R
CG06	F	08/30/91				UN R			0.55	UNW			78400.00		88.80					R
CG06	F	08/30/91				UN R			0.55	UNW			78000.00		96.80					R
CG06	F	08/30/91				NS R			6.80	BNW			88700.00		127.00				620.00	
CG06	F	08/30/91				BNW			4.10	BNW			89100.00		112.00					R
CG06	F	08/30/91				BNW			4.30	BNW			81700.00		99.90					R
CG06	F	08/30/91				BNW			5.10	BNW			81700.00		96.60					R
CG06	F	08/30/91				NS R			9.70	BNW			81700.00		95.60				400.00	
CG06	F	09/11/91			0.28	U*			2.40	BW J			58300.00		44.30				96.80	BN J
CG06	F	09/11/91			0.28	U*			2.70	B J			55000.00	J	44.50				107.00	BN J
CG06	F	09/11/91			0.28	U*			2.60	B J			52600.00	J	41.50				104.00	BN J
CG06	F	09/11/91			0.28	U*			2.90	BW J			50500.00	J	40.70				128.00	N J
CG06	F	09/11/91			0.56	B* J			4.00	B			48900.00	J	40.20				139.00	N J
CG06	F	09/11/91			0.64	BW*			6.10	BW J			48900.00	J	51.90				306.00	N J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Cyanide		Silver		Aluminum		Arsenic		Barium		Calcium		Cadmium		Chromium		Copper	
CG06	F	09/11/91			1.80	NS			8.30	BW J			50200.00	J	55.70				270.00	N J
CG06	F	09/11/91			1.40	S*			7.20	B J			66700.00	J	63.10				227.00	N J
CG06	F	09/11/91			2.30	S*			17.30	BS J			61300.00	J	78.50				360.00	N J
CG06	F	09/11/91			7.40	S*			878.00	BS J			155000.00	J	1530.00				6870.00	
CG06	F	09/11/91			22.90	NS			296.00	BS+			131000.00	J	698.00				2910.00	
CG06	F	09/11/91			14.20	S*			108.00	BS J			116000.00	J	227.00				764.00	
CG06	F	09/17/91			0.28	U	93.30	B J	1.50	B J	36.00	B J			5.60		5.50	U	11.40	
EF01	F	05/01/91	1.00	U J	0.28	U J	25.00	U	0.55	U	105.00	B J			0.28	B J	5.00	U	5.40	B J
EF01	F	06/13/91	1.00	U UJ	0.28	U J	374.00		1.80	B J	44.70	B J			0.66	B J	5.00	U J	5.40	B J
EF01	F	09/18/91				R	27.50	U	0.55	U	79.10	B* J			0.06	U*	5.50	U	0.55	U
EF02	F	05/01/91	1.00	U J	0.28	U J	25.00	U	0.55	U	100.00	B J			1.60	S J	5.00	U	2.80	BN J
EF02	F	06/13/91	1.00	U UJ	0.28	U J	357.00		1.40	B J	48.60	B J			0.56	B J	5.00	U J	5.20	B J
EF02	F	09/18/91				R	27.50	U	0.55	U	81.80	B* J			0.56	B* J	5.50	U	0.55	U
EG01	F	06/13/91	1.00	U UJ	0.28	UW J	58.60	B J	2.70	BW J	52.10	B J			0.87	B J	5.00	U J	8.10	B J
EG01	F	09/18/91				R	27.50	U	0.55	U	75.50	B* J			0.48	B* J	5.50	U	0.55	U
EG02	F	06/13/91	1.00	U UJ	0.28	U J	80.70	B J	1.30	B J	50.60	B J			0.80	B J	5.00	U J	7.50	B J
EM01	F	06/11/91	1.00	U* UJ	0.28	U J	159.00		0.55	U J	73.50	B J			0.20	B J	5.00	U J	3.60	BW J
EM01	F	07/23/91	1.00	U J	0.28	U	97.80	B	0.55	U	94.80	B			0.06	U	5.50	U J	4.60	B
EM01	F	09/16/91	1.00	UN J		N R	62.40	B J	0.55	U J	84.70	B J			0.06	U J	5.50	U	5.60	BW J
EM02	F	05/03/91	1.00	UN J	0.28	UW	360.00		1.20	B J	105.00	B J			1.10	J	5.00	U	3.30	B J
EM02	F	06/11/91	1.00	U* UJ	0.28	U J	96.20	B	0.55	U J	81.70	B J			0.19	B J	5.00	U J	1.20	B J
EM02	F	07/23/91	1.00	U J	0.28	U	27.50	U	0.55	U	80.50	B J			0.06	U	11.60	J	2.60	B J
EM02	F	09/16/91	1.00	UN J		NW	27.50	U	0.55	U J	88.80	B J			0.13	B J	5.50	U	4.10	BW J
EM03	F	04/30/91	1.00	U	0.28	U	86.40	B J	0.55	U		R			0.28	B J	5.00	U	17.00	
EM03	F	06/11/91	1.00	U* UJ	0.28	U J	268.00		0.55	U J	97.30	B J			0.16	B J	5.00	U J	2.90	BW J
EM03	F	07/23/91	1.00	U J	2.30		438.00		1.80	B J	107.00	B J			0.27	B J	5.50	U J	5.00	B J
EM03	F	09/16/91	1.00	UN J		N R	73.10	B J	1.20	B J	91.50	B J			0.06	UW J	5.50	U J	6.30	BW J
EM04	F	04/29/91	1.00	U	0.28	U	1620.00		5.70	B J		R			2.60	J	5.00	U	10.90	B J
EM04	F	06/11/91	1.00	U* UJ	0.28	U J	387.00		1.30	BW	115.00	B J			0.34	B J	5.00	U J	5.70	BW J
EM04	F	07/23/91	1.00	U J	0.28	U	767.00		1.60	B J	126.00	B J			0.34	B J	5.50	U J	2.90	B J
EM04	F	09/16/91	15.00	N J		NW	55.30	B J	0.55	U J	119.00	B J			0.06	U J	5.50	U	4.80	BW J
GG01	F	05/02/91	1.00	UN J	2.30		2790.00	J	40.60		109.00	B J			4.60	J	5.00	U	80.80	J
HC01	F	04/30/91	1.00	U	0.28	U	25.00	U	0.55	U	22.00	B J			0.27	B J	5.00	U	2.10	B J
HC01	F	06/11/91	6.00	B* J	0.28	U J	72.90	B J	0.55	U J	14.00	B J			0.22	B J	5.00	U J	4.40	BW J
HC01	F	07/23/91	1.00	U J	0.28	U	62.00	B J	0.55	U	17.60	B J			0.06	UW	5.50	U J	1.50	B J
HC01	F	09/16/91	1.00	UN J		N R	27.50	U	0.55	U J	24.10	B J			0.06	U J	5.50	U	5.00	B J
IG01	F	04/29/91	1.00	U	0.28	U	25.00	U	0.55	U	57.80	B J			0.32	B J	5.00	U	0.55	UW
IG01	F	06/11/91	6.00	B* J	0.28	U J	191.00		1.20	B J	45.60	B J			0.64	BW J	5.00	U J	5.10	BW J
IG01	F	07/23/91	1.00	U J	0.28	U	214.00		0.55	U	52.70	B J			0.62	B J	5.50	U J	5.80	B J
IG01	F	09/16/91	4.00	BN J		NW	27.50	U	0.55	UW J	44.20	B J			0.33	B J	5.50	U	3.00	BW J
LF01	F	05/01/91	1.00	U J	0.28	U J	83.10	B J	0.55	U	18.70	B J			0.25	B J	5.00	U	3.20	BN J
LF01	F	06/11/91	7.00	B* J	0.28	U J	180.00		0.55	U J	12.80	B J			0.32	B J	5.00	U J	4.90	BW J
LF01	F	07/23/91	1.00	U J	0.28	U	108.00	B J	0.55	U	15.10	B J			0.18	B J	5.50	U J	4.30	B J
LF01	F	09/16/91	1.00	UN J		NW	64.60	B J	0.55	U J	16.80	B J			0.14	B J	5.50	U	2.40	B J
MG01	F	05/01/91	1.00	U J	85.80		55300.00		28.80		1310.00				170.00		55.80		1100.00	J
OG01	F	05/02/91	3.00	BN J	12.90	S	381000.00	J	245.00		2610.00				89.50	J	50.00	U	10200.00	J
OG01	F	06/12/91		BN R	1.90	S	312000.00		33.60	BNW	125.00	U			557.00	S J	125.00	U J	7800.00	
OG01	F	07/24/91			17.00	S	103000.00		601.00	S	275.00	U			549.00	S	275.00	U J	2340.00	
PG01	F	05/02/91	1.00	UN J	14.80	W	8470.00	J	54.00		369.00				33.40	J	5.00	U	180.00	J
SD01	F	05/02/91	1.00	UN J	381.00	J	52800.00	J	581.00		2300.00				772.00	J	50.00		12900.00	J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																					
Station	Type	Date	Cyanide			Silver			Aluminum			Arsenic			Barium			Calcium	Cadmium			Chromium			Copper		
SD01	F	06/12/91		UN	R	5.20	BW	J	11800.00			20.70	NS	J	76.30	B	J		357.00		J	5.00	U	J	838.00		
SD01	F	07/24/91				0.74	BW		322.00			2.80	B	J	55.70	B	J		7.00		S	5.50	U	J	17.00		
SG01	F	05/02/91	1.00	UN	J	8.80			1660.00		J	18.20			98.20	B	J		163.00		J	5.00	U		145.00		J
SG01	F	06/12/91		UN	R	0.81	B	J	10800.00		J	2.10	BNW		31.80	B	J		344.00		J	5.00	U	J	709.00		J
SG01	F	07/24/91				144.00			65800.00			1130.00			350.00	B	J		959.00		S	110.00	U	J	7970.00		
TC01	F	05/01/94	1.00	U	J	0.28	U	J	25.00	U		0.55	U		24.40	B	J		0.44	B	J	5.00	U	J	3.70	BN	J
TC01	F	06/13/91	1.00	U	UJ	0.28	U	J	179.00			1.30	B	J	12.50	B	J		0.67	B	J	5.00	U	J	6.30	B	J
TC01	F	09/18/91						R	58.20	B	J	0.55	U		20.90	B*	J		0.23	B*	J	5.50	U		0.55	U	
TC10	F	09/18/91																									

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Iron		Mercury		Magnesium		Manganese		Nickel		Lead		Antimony		Selenium		Zinc	
AG01	F	05/02/91	25600.00	J	1.90				973.00		21.80	B J	11100.00	J	20.00	U	2.75	UN	1190.00	J
AR01	F	04/30/91	563.00	E J	0.10	U			97.40		10.00	U	0.55	U J	20.00	U	0.55	U		R
AR01	F	06/13/91	624.00		0.10	UN UJ			83.70		10.00	U	6.50	J	20.00	U J	0.55	UW	115.00	J
AR01	F	07/25/91	245.00		0.10	UN J			49.10		11.00	U	1.80	BNW J	22.00	U	2.80	BW J	85.40	
AR01	F	09/18/91	174.00		0.10	U			72.30	J	11.00	U J		R	0.55	UNW		R	145.00	* J
AR02	F	04/30/91	493.00	E J	0.10	U			89.60		10.00	U	1.10	B UJ	20.00	U	0.55	U		R
AR02	F	06/12/91	1230.00		0.10	UN J			150.00		10.00	U	12.20	J	20.00	U		UNW	166.00	E J
AR02	F	07/24/91	171.00		0.10	U			45.20		11.00	U	1.90	B J	22.00	U	1.10	U	258.00	E J
AR02	F	08/24/91	107.00	B J			8950.00		45.90				0.55	U					82.30	J
AR02	F	08/24/91	106.00	B J			9020.00		44.10	J			0.55	U					83.10	J
AR02	F	08/24/91	121.00				8970.00		44.10	J			0.55	U					82.50	J
AR02	F	08/24/91																		
AR02	F	08/30/91	88.40	B J			8540.00		32.00				2.00	BW J					59.50	
AR02	F	08/30/91	87.60	B J			8500.00		33.10				0.55	U					61.20	
AR02	F	08/30/91	87.40	B J			8530.00		33.90				0.55	U					58.50	
AR02	F	08/30/91	44.75	U J			8450.00		33.50				0.55	U					56.60	
AR02	F	08/30/91	96.40	B J			8550.00		33.90				0.55	U					54.90	
AR02	F	08/30/91	112.00				8640.00		33.90				0.55	UW					58.90	
AR02	F	08/30/91	97.60	B J			8600.00		34.20				0.55	UW					57.00	
AR02	F	08/30/91	88.40	B J			8470.00		31.30				0.55	UW					55.50	
AR02	F	08/30/91	94.50	B J			550.00	U	34.20				0.55	UW					58.00	
AR02	F	08/30/91	93.60	B J			8610.00		32.00				0.55	UW					56.80	
AR02	F	08/30/91	89.30	B J			8510.00		32.80				0.55	UW					57.00	
AR02	F	08/30/91	93.80	B J			8830.00		34.20				0.55	UW					61.00	
AR02	F	09/11/91	117.00	J			8600.00		48.50	J			0.55	U J					77.50	J
AR02	F	09/11/91	108.00	B J			8790.00		48.00	J			0.55	U J					84.50	J
AR02	F	09/11/91	119.00	J			8570.00		48.00	J			0.55	U J					85.00	J
AR02	F	09/11/91	124.00	J			9060.00		49.40	J			0.55	UW J					86.40	J
AR02	F	09/11/91	122.00	J			8460.00		48.00	J			0.55	U J					81.70	J
AR02	F	09/11/91	116.00	J			8680.00		48.00	J			0.55	U J					86.40	J
AR02	F	09/11/91	103.00	B J			8510.00		42.50	J			0.55	U J					82.50	J
AR02	F	09/11/91	119.00	J			8520.00		46.20	J			0.55	U J					85.20	J
AR02	F	09/11/91	114.00	J			8740.00		46.20	J			0.55	U J					81.20	J
AR02	F	09/11/91	122.00	J			8910.00	J	48.00	J			0.55	U J					81.70	J
AR02	F	09/11/91	119.00	J			8890.00		47.50	J			0.55	U J					89.20	J
AR02	F	09/12/91	127.00	J			8840.00		48.90	J			0.55	U J					89.40	J
AR02	F	09/17/91	144.00		0.10	U			50.00		11.00	U	0.55	U	0.55	U		N R	97.30	E
AR03	F	04/30/91	561.00	E J	0.10	U				R	10.00	U	4.50	J	20.00	U	1.90	B J		R
AR03	F	06/12/91	1520.00		0.10	UN J			322.00		10.00	U	16.60	J	20.00	U		UNW	570.00	E J
AR03	F	09/11/91	420.00	* J			9780.00		466.00				6.40	NW					1050.00	E J
AR03	F	09/11/91	314.00	* J			9420.00		379.00				7.80	NW					734.00	E J
AR03	F	09/11/91	285.00	* J			9640.00		409.00				5.20	N					924.00	E J
AR03	F	09/11/91	287.00	* J			9440.00		341.00				6.90	NW					711.00	E J
AR03	F	09/11/91	194.00	* J			9440.00		300.00				3.60	N					585.00	E J
AR03	F	09/11/91	460.00	*			9630.00		409.00				16.00	NS					1040.00	E J
AR03	F	09/11/91	695.00	* J			9530.00		344.00				23.40	NS					1160.00	E J
AR03	F	09/11/91	577.00	* J			10100.00		553.00				32.70	NS					1160.00	E J
AR03	F	09/11/91	372.00	* J			10400.00		608.00				27.70	NS					1330.00	E J
AR03	F	09/11/91	58600.00	* J			16400.00		5290.00				1200.00	N					37900.00	E J
AR03	F	09/12/91	23000.00	* J			14400.00		3170.00				1090.00	N					15300.00	E J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																			
Station	Type	Date	Iron		Mercury		Magnesium		Manganese		Nickel		Lead		Antimony		Selenium		Zinc	
AR03	F	09/12/91	4380.00	* J			12400.00		1640.00				409.00	NS					5170.00	E J
AR04	F	04/29/91	1680.00	E J	0.10	U UJ			540.00		10.00	U	27.50	S J	20.00	U	0.55	U		R
AR04	F	06/11/91	567.00	J	0.10	UN			162.00	J	10.00	U	12.00	J	20.00	U	0.55	UW J	344.00	J
AR04	F	07/23/91	486.00		0.10	UN J			182.00		11.00	U	8.20		22.00	U	3.60	BW J	298.00	
AR04	F	09/11/91	204.00	N J			9550.00	J	296.00	N J			3.60						543.00	EN J
AR04	F	09/11/91																		
AR04	F	09/11/91	612.00	N J			9070.00	J	255.00	N J			3.60						549.00	EN J
AR04	F	09/11/91	300.00	N J			9400.00	J	301.00	N J			15.60						862.00	EEN J
AR04	F	09/11/91	232.00	N J			9400.00	J	243.00	N J			4.60						564.00	EN J
AR04	F	09/11/91	219.00	N J			9190.00	J	208.00	N J			3.90						481.00	EN J
AR04	F	09/11/91	835.00	N J			9740.00	J	266.00	N			8.70						690.00	EN J
AR04	F	09/11/91	431.00				9340.00	J	236.00	N			13.00						762.00	EN J
AR04	F	09/11/91	370.00	N			9590.00	J	326.00	N			14.70						740.00	EN J
AR04	F	09/11/91	469.00	N J			9680.00	J	334.00	N J			26.90	S					791.00	EN J
AR04	F	09/11/91	30000.00	N J			13000.00	J	3390.00	N J			813.00	J					23400.00	EN J
AR04	F	09/11/91	19300.00	N J			12400.00		2370.00	N J			452.00	J					11700.00	EN J
AR04	F	09/16/91	335.00		0.10	U J			123.00		11.00	U	4.50	NW J	1.10	BW J	0.55	UN U	200.00	
AR05	F	04/29/91	815.00	E J	0.10	U UJ			228.00		10.00	U	19.20	J	20.00	U	4.10	BW J		R
AR05	F	06/11/91	879.00	J	0.10	UN			201.00	J	10.00	U	25.80	S J	20.00	U	0.55	UW J	385.00	J
AR05	F	07/23/91	515.00		0.10	UN J			159.00		11.00	U	14.20		22.00	U	0.55	U	329.00	
AR05	F	09/16/91	209.00		0.10	U J			68.90		11.00	U	2.40	BNW J	0.55	UW J	0.55	UN U	186.00	
AR3A	F	07/24/91	305.00		0.10	U			228.00		11.00	U	6.30		22.00	U	0.55	U J	604.00	E J
AR3A	F	08/24/91	122.00				9960.00		297.00				0.55	U					326.00	
AR3A	F	08/24/91	108.00	B J			9720.00		298.00				0.55	U					327.00	
AR3A	F	08/24/91	122.00				9780.00		308.00				1.30	B J					330.00	
AR3A	F	08/24/91	132.00				9600.00		282.00				1.30	B J					330.00	
AR3A	F	08/30/91	103.00	B J			9640.00		432.00				0.55	U J					737.00	E J
AR3A	F	08/30/91	113.00				9710.00		465.00				1.30	BW J					799.00	E J
AR3A	F	08/30/91	122.00				9800.00		453.00				1.40	BW J					827.00	E J
AR3A	F	08/30/91	153.00				9880.00		543.00				2.20	BW J					869.00	E J
AR3A	F	08/30/91	139.00				9870.00		538.00				2.40	BW J					883.00	E J
AR3A	F	08/30/91	127.00				9910.00		528.00				2.10	BW J					934.00	E J
AR3A	F	08/30/91	119.00				10100.00		569.00				1.40	BW J					1030.00	E J
AR3A	F	08/30/91	132.00				10100.00		610.00				2.10	BW J					1170.00	E J
AR3A	F	08/30/91	443.00				10500.00		736.00				4.60	S					1800.00	E J
AR3A	F	08/30/91	99.50	B J			10400.00		694.00				1.80	BW J					1350.00	E J
AR3A	F	08/30/91	162.00				10900.00		693.00				2.60	BW J					1360.00	E J
AR3A	F	08/30/91	153.00				10200.00		616.00				4.70	W					1190.00	E J
AR3A	F	09/16/91	199.00		0.10	U J			120.00		11.00	U	2.40	BN J	1.20	BW J	0.55	UN U	198.00	
AR3E	F	07/24/91	479.00		0.10	U			460.00		11.00	U	11.20		22.00	U	0.55	UW J	957.00	E J
AR3E	F	08/24/91	124.00				10100.00		511.00				1.10	B J					422.00	
AR3E	F	08/24/91	122.00				10000.00		510.00				1.10	U					414.00	
AR3E	F	08/24/91	166.00				9950.00		441.00				1.20	B J					413.00	
AR3E	F	08/24/91	120.00				9900.00		340.00				1.10	U					369.00	
AR3E	F	08/30/91	138.00				10600.00		828.00					BN R					1430.00	E J
AR3E	F	08/30/91	210.00				10500.00		792.00					NW R					1470.00	E J
AR3E	F	08/30/91	135.00				11000.00		1000.00					BNW R					1940.00	E J
AR3E	F	08/30/91	5300.00				11000.00		991.00					NW R					1460.00	E J
AR3E	F	08/30/91	589.00				10900.00		927.00					BNW R					1520.00	E J
AR3E	F	08/30/91	158.00				11000.00		1010.00					BNW R					1730.00	E J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																	
Station	Type	Date	Iron		Mercury		Magnesium		Manganese		Nickel		Lead		Antimony		Selenium		Zinc	
AR3E	F	08/30/91	213.00				11400.00		1130.00					NW R					2110.00	E J
AR3E	F	08/30/91	1160.00				11800.00		1310.00					N R					3600.00	E J
AR3E	F	08/30/91	306.00				11600.00		1230.00					N R					2380.00	E J
AR3E	F	08/30/91	256.00				11500.00		1130.00					NW R					2110.00	E J
AR3E	F	08/30/91	326.00				11300.00		1070.00					N R					1980.00	E J
AR3E	F	08/30/91	364.00				11300.00		1090.00					N R					1980.00	E J
AR3E	F	09/17/91	171.00		0.10	U			187.00		11.00	U	1.80	B J	0.55	U		N R	229.00	E
AR3W	F	07/24/91	206.00		0.10	U			75.30		11.00	U	2.20	B J	22.00	U	0.55	U J	311.00	E J
AR3W	F	09/17/91	152.00		0.10	U			88.10		11.00	U	1.40	B J	0.55	U		N R	139.00	E
CG01	F	05/03/91	8710.00	J	0.10	U			1810.00	J	10.00	U	2960.00	J	20.00	U	0.55	UNW	10300.00	J
CG01	F	06/12/91	3100.00		0.10	UN J			6050.00		10.00	U	942.00		20.00	U		UNW	26100.00	E J
CG02	F	05/03/91	40900.00	J	0.30				13300.00		24.40	B J	3190.00	J	20.00	U	2.75	UNW	17900.00	J
CG02	F	19910612	4160.00		0.10	UN J			7450.00		10.00	U	618.00		20.00	U		UNW	31000.00	E J
CG03	F	05/03/91	2920.00	J	0.10	U			6460.00	J	22.20	B J	96.70	J	20.00	U	0.55	UNW	20200.00	J
CG03	F	06/12/91	2550.00		0.10	UN J			15600.00	J		B R	62.20		20.00	U		UNW	61200.00	E J
CG03	F	07/24/91	707.00		0.10	U			1440.00		11.00	U	22.40	S	22.00	U	0.55	UW J	4730.00	E J
CG03	F	09/17/91	1970.00		0.10	U			11300.00		36.20	B J	41.80		0.55	U		N R	37300.00	E
CG04	F	05/02/91	211000.00	J	2.30				61900.00		65.20		10500.00	J	20.00	U	2.75	UNW	51100.00	J
CG04	F	06/12/91	8560.00		0.10	U J			24900.00	J	39.40	B J	167.00		20.00	U		UNW	61200.00	E J
CG04	F	07/24/91	26300.00		0.20				15800.00		11.00	U J	1840.00	J	22.00	U	2.75	UW J	17900.00	E J
CG04	F	09/17/91	21800.00		0.10	U			57300.00		88.00		315.00	N J	0.55	U		NW	51100.00	E
CG05	F	05/02/91	39500.00	J	0.50				34800.00		47.90		1180.00	J	20.00	U	8.50	NW	32700.00	J
CG05	F	06/12/91	8250.00		0.10	UN J			20700.00	J	28.40	B J	212.00		20.00	U		UNW	50700.00	E J
CG05	F	07/24/91	270000.00		0.70				23200.00		55.00	U	38800.00	J	110.00	U	2.75	UW J	43600.00	E J
CG05	F	09/17/91	15200.00		0.10	U			19500.00		40.80	B J	276.00	N J	0.55	U		NW	27700.00	E
CG06	F	05/01/91	42000.00		1.30				9770.00		10.00	U	3550.00		20.00	U	11.80	B J	15700.00	
CG06	F	06/12/91	4940.00		0.10	UN J			13600.00	J	10.00	U	123.00		20.00	U		UNW	32500.00	E J
CG06	F	07/24/91	8580.00		0.10	U			6160.00		11.00	U	451.00		22.00	U	2.50	BS J	9690.00	E J
CG06	F	08/24/91	1440.00				33500.00		9850.00				42.20	S					6840.00	
CG06	F	08/24/91	2050.00				32500.00		9880.00				51.00	S					7400.00	
CG06	F	08/24/91	2270.00				29000.00		7940.00				86.80	S J					7200.00	
CG06	F	08/24/91	2110.00				25700.00		6100.00				1.60	BS J					5980.00	
CG06	F	08/30/91	2280.00				35100.00		10200.00				45.30						18400.00	
CG06	F	08/30/91	3290.00				35900.00		10400.00				93.80						19300.00	
CG06	F	08/30/91	7880.00				41400.00		12700.00				152.00	S					25900.00	
CG06	F	08/30/91	4000.00				38800.00		11800.00				103.00						21800.00	
CG06	F	08/30/91	2820.00				38700.00		11700.00				60.90	S					22100.00	
CG06	F	08/30/91	2980.00				39500.00		12400.00				49.90						23100.00	
CG06	F	08/30/91	4730.00				39000.00		12100.00				119.00						26900.00	
CG06	F	08/30/91	11200.00				44100.00		14000.00				478.00						36300.00	
CG06	F	08/30/91	6000.00				44200.00		14200.00				305.00						29800.00	
CG06	F	08/30/91	6940.00				40100.00		12600.00				298.00						26000.00	
CG06	F	08/30/91	7220.00				40600.00		12700.00				287.00						25700.00	
CG06	F	08/30/91	10900.00				40700.00		13000.00				418.00						25800.00	
CG06	F	09/11/91	2990.00				27600.00	J	7120.00				66.60						11900.00	
CG06	F	09/11/91	3540.00				26200.00	J	6490.00				111.00						11500.00	
CG06	F	09/11/91	3170.00				24800.00	J	5970.00				102.00						11300.00	
CG06	F	09/11/91	3720.00				24000.00	J	5750.00				130.00						11000.00	
CG06	F	09/11/91	4810.00				23100.00	J	5290.00				169.00						11100.00	
CG06	F	09/11/91	8010.00				23300.00	J	5270.00				363.00	*					14900.00	

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

			ug/l																			
Station	Type	Date	Iron		Mercury		Magnesium		Manganese		Nickel		Lead		Antimony		Selenium		Zinc	
CG06	F	09/11/91	8970.00				23300.00	J	5270.00				397.00	*					16500.00	
CG06	F	09/11/91	8170.00				32000.00	J	8200.00				483.00	*					17300.00	
CG06	F	09/11/91	14000.00				30200.00		7300.00				1020.00	*					21200.00	
CG06	F	09/11/91	634000.00				89500.00	J	52700.00				25300.00	*					342000.00	
CG06	F	09/11/91	195000.00				63300.00	J	30100.00				10200.00	*					143000.00	
CG06	F	09/11/91	51300.00				56500.00	J	21300.00				3470.00	*					59100.00	
CG06	F	09/17/91	697.00		0.10	U			3530.00		11.00	U	30.20	B J	0.55	UW		N R	2840.00	E
EF01	F	05/01/91	387.00		0.10	U			46.00		10.00	U	1.70	B J	20.00	U	2.10	B J	10.00	U
EF01	F	06/13/91	525.00		0.10	UN UJ			49.20		10.00	U	4.60	J	20.00	U J	0.55	UW	29.20	J
EF01	F	09/18/91	89.40	B J	0.10	U			17.40	J	11.00	U J		R	2.20	BNW	0.55	UN J	5.50	U* J
EF02	F	05/01/91	270.00		0.10	U			410.00		10.00	U	1.50	B J	20.00	U	4.50	B J	689.00	
EF02	F	06/13/91	574.00		0.10	UN UJ			88.00		10.00	U	6.00	J	20.00	U J	0.55	U	101.00	J
EF02	F	09/18/91	102.00	B J	0.10	U			88.90	J	11.00	U J		R	0.55	UNW	1.20	BN J	176.00	J
EG01	F	06/13/91	86.10	B J	0.10	UN UJ			13.80	B J	10.00	U	8.20	W J	20.00	U J	0.55	UW	101.00	J
EG01	F	09/18/91	22.00	U	0.10	U			5.50	U J	11.00	U J		R	0.55	UNW	0.55	UN J	60.90	*
EG02	F	06/13/91	94.20	B J	0.10	UN UJ			5.00	U	10.00	U	8.60	J	20.00	U J	0.55	U	113.00	J
EM01	F	06/11/91	292.00	J	0.10	UN			16.40	J	10.00	U	2.70	B J	20.00	U	0.55	U J	14.80	B J
EM01	F	07/23/91	171.00		0.10	UN J			14.00	B	11.00	U	1.40	B	22.00	U	0.55	UW	5.50	U
EM01	F	09/16/91	214.00		0.10	U J			16.30	B J	11.00	U	0.55	UN J	0.55	UW J	0.55	UN U	19.70	B J
EM02	F	05/03/91	1090.00	J	0.10	U			104.00	J	10.00	U	15.80	J	20.00	U	0.55	UNW	37.50	J
EM02	F	06/11/91	237.00	J	0.10	UN			15.50	J	10.00	U	2.00	B J	20.00	U	0.55	U J	5.00	U J
EM02	F	07/23/91		B R	0.10	UN J				B R	11.00	U	0.55	U	22.00	U	0.55	UW	5.50	U
EM02	F	09/16/91	204.00		0.10	U J			22.50		11.00	U	0.55	UN J	0.55	UW J	0.55	UW	16.00	B J
EM03	F	04/30/91	220.00	E J	0.10	U			14.80	B J	10.00	U	4.50	J	20.00	U	0.55	U	17.20	B J
EM03	F	06/11/91	548.00	J	0.10	UN			88.80	J	10.00	U	4.70	J	20.00	U	0.55	UW J	5.00	U J
EM03	F	07/23/91	873.00		0.10	UN J			161.00		11.00	U	12.30		22.00	U	0.55	UW	30.30	
EM03	F	09/16/91	489.00		0.10	U J			68.90		11.00	U	18.70	N J	1.30	B J	0.55	UN U	14.90	B J
EM04	F	04/29/91	2520.00	E J	0.10	U UJ			179.00		10.00	U	103.00	J	20.00	U	0.55	UW	81.40	J
EM04	F	06/11/91	769.00	J	0.10	UN			80.20	J	10.00	U	12.30	J	20.00	U	0.55	UW J	19.80	B J
EM04	F	07/23/91	1270.00		0.10	UN J			138.00		11.00	U	15.00		22.00	U	2.75	U	30.60	
EM04	F	09/16/91	377.00		0.10	U J			39.20		11.00	U	3.20	BN J	0.55	U J	0.55	UN U	28.70	
GG01	F	05/02/91	4000.00	J	0.10	U			325.00		10.00	U	932.00	J	20.00	U	1.10	UN	1180.00	J
HC01	F	04/30/91	56.50	BE J	0.10	U			5.00	U	10.00	U	0.55	U J	20.00	U	0.55	U	11.70	B J
HC01	F	06/11/91	65.60	B J	0.10	UN			5.00	U J	10.00	U	1.30	BW J	20.00	U	0.55	U J	10.20	B J
HC01	F	07/23/91	68.60	B J	0.10	UN J			5.50	U	11.00	U	0.55	U	22.00	U	0.55	UW	5.50	U
HC01	F	09/16/91	81.80	B J	0.10	U J			5.50	U	11.00	U	0.55	UN J	0.55	UW J	0.55	UNW	5.50	U
IG01	F	04/29/91	79.30	BE J	0.10	U UJ			5.00	U	10.00	U	5.60	W J	20.00	U	0.55	U	14.40	B J
IG01	F	06/11/91	391.00	J	0.10	UN			122.00	J	10.00	U	27.90	S J	20.00	U	0.55	UW J	117.00	J
IG01	F	07/23/91	411.00		0.10	UN J			126.00		11.00	U	21.30		22.00	U	0.55	U	113.00	
IG01	F	09/16/91	52.30	B J	0.10	U J			19.10		11.00	U	2.90	BNW J	0.55	U J	2.75	UN U	243.00	
LF01	F	05/01/91	690.00		0.10	U			88.80		10.00	U	0.55	U	20.00	U	0.55	U	25.00	J
LF01	F	06/11/91	494.00	J	0.10	UN			66.40	J	10.00	U	1.20	B J	20.00	U	0.55	UW J	25.00	J
LF01	F	07/23/91	638.00		0.10	UN J			85.20		11.00	U	0.55	UW	22.00	U	0.55	UW	25.40	
LF01	F	09/16/91	410.00		0.10	U J			30.20		11.00	U	1.50	BN J	0.55	U J	0.55	UN U	20.70	B J
MG01	F	05/01/91	99900.00		2.90				3720.00		45.90		16500.00		20.00	U	5.50	U J	14100.00	
OG01	F	05/02/91	8130000.00	J	0.30				2510000.00		1600.00		25500.00	J	200.00	U	2.75	UEN	1110000.00	J
OG01	F	06/12/91	2760000.00		0.10	UN J			1490000.00		726.00	B J		S R	500.00	U		BEN	634000.00	E J
OG01	F	07/24/91	3710000.00		0.30				1260000.00		550.00	U	3600.00		1100.00	U			559000.00	E J
PG01	F	05/02/91	17700.00	J	0.60				2060.00		10.00	U	2750.00	J	20.00	U	0.55	UN	4360.00	J
SD01	F	05/02/91	290000.00	J	9.00				33100.00		113.00		55900.00	J	100.00	U	2.75	UNW	106000.00	J

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Appendix B-3 (continued)

Total Metal Concentration for 1991 through 1993 Data

Station	Type	Date	Iron	Mercury	Magnesium	Manganese	Nickel	Lead	Antimony	Selenium	Zinc
			ug/l								
SD01	F	06/12/91	31200.00	0.10 UN J		14200.00	10.00 U	1170.00	20.00 U	UNW	44600.00 E J
SD01	F	07/24/91	1060.00	0.10 U		309.00 J	11.00 U	131.00	22.00 U	0.55 U J	512.00 E J
SG01	F	05/02/91	10800.00 J	0.10 U		4160.00	10.00 U	2490.00 J	20.00 U	0.55 UNW	15600.00 J
SG01	F	06/12/91	18200.00	0.10 UN J		13400.00	10.00 U	203.00	20.00 U	UNW	43500.00 E J
SG01	F	07/24/91	1170000.00	0.94		50000.00	220.00 U	8990.00	440.00 U	13.75 UW	92800.00 E J
TC01	F	05/01/94	909.00	0.10 U		52.20	10.00 U	1.10 U	20.00 U	0.55 U	94.00
TC01	F	06/13/91	270.00	0.10 UN UJ		51.70	10.00 U	2.00 B J	20.00 U J	0.55 UW	88.80 J
TC01	F	09/18/91	361.00	0.10 U.		29.10	11.00 U J	R	0.55 UNW	0.55 UNW	33.50 * J
TC10	F	09/18/91									

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