DOE/AL/62350-57
REV. 2

UMTRA GROUND WATER PROJECT

BASELINE RISK ASSESSMENT OF GROUNDWATER CONTAMINATION AT THE URANIUM MILL TAILINGS SITE NEAR GUNNISON, COLORADO

June 1996

Prepared by the
U.S. Department of Energy
Albuquerque, New Mexico

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June 1996

Prepared for
U.S. Department of Energy
Grand Junction Projects Office
Albuquerque, New Mexico

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Albuquerque, New Mexico

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CITIZENS' SUMMARY

INTRODUCTION

The Uranium Mill Tailings Remedial Action (UMTRA) Project consists of the Surface Project (phase I), and the Ground Water Project (phase II). For the UMTRA Project site located near Gunnison, Colorado (the Gunnison site), Surface Project cleanup occurred from 1992 to 1995. The mill tailings and radioactively contaminated soils and materials resulting from uranium processing were removed from their original locations and taken to a disposal cell about 6 miles (10 kilometers) east of the town of Gunnison and 0.4 mile (0.6 kilometer) south of the county solid waste landfill. Removing the radioactive source material reduced radon and other radiation emissions and minimized further contamination of ground water beneath the Gunnison site.

Phase II of the UMTRA Project evaluates the nature and extent of ground water contamination resulting from uranium processing and its effect on human health and the environment, and determines site-specific ground water compliance strategies in accordance with the U.S. Environmental Protection Agency (EPA) ground water standards established for the project. A risk assessment was conducted for the Gunnison site because human health risks could occur from drinking water pumped from a hypothetical well drilled in the contaminated ground water area. Environmental risks may result if plants or animals are exposed to contaminated ground water or if surface water is mixed with contaminated ground water.

Risk assessment is the process of describing a source of contamination, showing how that contamination may reach the public and the environment, calculating the amount of contamination exposure, and characterizing possible health or environmental effects that may result from this exposure.

This report is the second site-specific risk assessment document prepared for the Ground Water Project at the Gunnison site. A preliminary risk assessment was conducted in 1990 to determine whether long-term use of ground water from private wells near the Gunnison site had the potential for adverse health effects. Due to the results of that preliminary risk assessment, the residents were provided bottled water on an interim basis. In July 1994, the residents and the nearby Valco cement/concrete plant were given the option to connect to a new alternate water supply system, eliminating the bottled water option.

This document evaluates current and potential future impacts to the public and the environment from exposure to contaminated ground water. The results of this evaluation and further site characterization will be used to determine whether more action is needed to protect human health and the environment and to comply with the EPA standards.

RISK SUMMARY

Because no one is drinking ground water contaminated by the site, no human health risks are currently associated with the affected ground water. Existing data indicate human and livestock health problems would not be expected from the use of ground water from private wells near the site. This favorable risk situation will continue if land and water use at the site do not change. Changes may or may not create future risks; therefore, when specific land uses are determined for the site area, they should be evaluated to identify the potential health and environmental risks that might result from site-affected ground water.

The site impact to the surface water bodies near the site has not been determined, although site-affected ground water discharges into these water bodies. Fishing in the Gunnison River, Tomichi Creek, or a campground pond near the site would not result in human health risks because of the great dilution capability of these water bodies.

To evaluate future risks, this report assesses the worst-case results of placing a drinking water well in the areas where ground water is most contaminated by site activities. Water from this hypothetical well then is evaluated as the sole source of drinking water for people and livestock or as a source of irrigation water for crops. In addition, the possible impacts to plants and animals from this water is considered. This risk assessment shows that under those conditions, people, livestock, and plants could experience health problems. Based on these findings, the worst ground water from the alluvial aquifer beneath the Gunnison site should not be used, and the less contaminated water from other areas of the site should be evaluated prior to any use.

GROUND WATER QUALITY

Ground water is present in the alluvial aquifer. It ranges from 72 feet to greater than 130 feet deep (approximately 20 to 40 meters) in the Gunnison site vicinity. The alluvial aquifer is underlain by a discontinuous unit of unknown extent and thickness of low-permeability rock known as the Brushy Basin Member. This low-permeability rock forms a layer that prevents ground water movement from the alluvial aquifer downward.

Background ground water quality

Background ground water quality is the ground water quality that would be present in the area if uranium milling activities had not occurred. The alluvial aquifer is a source of potable water in the site area. The Brushy Basin Member is not a water resource because it contains very little water.

Site-related ground water quality

The major constituents in the alluvial aquifer that resulted from uranium processing at the Gunnison site are cadmium, cobalt, iron, manganese, sulfate, and uranium. Contamination apparently extends southwest some distance from the source. It discharges into the Gunnison River, Tomichi Creek, and possibly to a campground pond near the Gunnison site where it is quickly diluted. Thus, surface water and sediment from these streams and a campground pond do not show any signs of site-related contamination.

HUMAN HEALTH RISK ASSESSMENT

Methods

This risk assessment starts by identifying constituents in ground water contaminated by the uranium milling process. This is done by comparing water quality data from wells drilled on and downgradient from the Gunnison site to water quality data from background ground water wells. Next, it examines potential human health problems that could result from drinking water containing these constituents; eating meat and/or drinking milk from cattle that had drunk the ground water; eating domestic garden produce irrigated with the water; and/or eating fish from the Gunnison River, Tomichi Creek, or a campground pond near the Gunnison site.

Both current and potential future human health risks are evaluated here. To evaluate current potential risk, it must be determined if the ground water is now used for drinking water or to water livestock or domestic gardens. To evaluate potential future risk, the risk assessors assumed a drinking water well is drilled into the most contaminated area under the Gunnison site, and that people receive all their drinking water from this well. To evaluate the potential risk of consuming of meat and milk from cattle watered with this water or from eating vegetables from a domestic garden irrigated with the ground water, they also assumed the most contaminated portion of ground water from the alluvial aquifer is the sole source of livestock and irrigation water, and the domestic garden is the source of all produce in the diet.

Health risks other than cancer were specially evaluated for children, as this age group is more likely to experience health problems because of their ratio of contaminant to body weight. Thus, children 1 to 10 years old are the age group most likely to experience health problems after drinking contaminated water. Infants (0 to 1 year) generally drink less water than children, but are sensitive to constituents such as sulfate. To estimate cancer risks, a lifetime exposure was assumed and these risks were evaluated for adults.

The seriousness of potential health effects varies because constituent levels in ground water vary from one well sampling round to the next. People vary in body weight, water consumption, and the way their bodies react to chemical exposure. These differences are considered in this risk assessment whenever possible. Graphs show the exposure levels that are possible and the most current scientific information on the type of health effects that may result from a hypothetical exposure (see Section 6.0).

Results

Because no one uses the affected ground water from the Gunnison site area for drinking and/or bathing, it currently causes no health problems. Although some private wells downgradient of the Gunnison site potentially have been affected by tailings-related constituents, all current residences were connected to the new water supply system, which provides good water for domestic use. Existing data do not indicate human or livestock health problems should be expected from watering livestock or crops with ground water from private wells near the Gunnison site. This favorable risk situation will remain the same in the future if land and water uses near the Gunnison site do not change. It is unlikely that people will use the contaminated ground water from the aquifer for drinking in the future, because the alternate water supply system is readily available. Nevertheless, based on the concentrations of constituents found in the most contaminated wells at the former mill site, if people began drinking the contaminated ground water possible human health risks could result. Table CS.1 lists the possible health problems that could be expected. Only people whose sole source of drinking water comes from a hypothetical well placed in the most contaminated ground water would be expected to experience the health problems described in this table. Consequently, it gives the upper limit of possible risks; real future risk probably would be lower.

Table CS.1 Hypothetical future health effects from drinking site-affected ground water, Gunnison, Colorado, site

Constituent	Short-term	Long-term
Cadmium	None	None
Cobalt	None	None
Iron	Unknown	Skin pigmentation, liver problems, and diabetes.
Lead-210	None	On average, increased lifetime cancer risk is within the range recommended by EPA as acceptable; however, the maximum estimate could slightly exceed the upper end of this range.
Manganese	None	Neurological symptoms include memory loss, irritability, muscle rigidity, tremors.
Polonium-210	None	Excess lifetime cancer risk is within the range recommended by EPA as acceptable.
Sulfate	Laxative effects or diarrhea, particularly in infants.	None
Thorium-230	None	Excess lifetime cancer risk is well below the lower end of the range recommended by EPA as acceptable.
Uranium	None	Although possible, health problems are not expected from chemical toxicity; increased lifetime cancer risk slightly exceeds the maximum recommended by EPA as acceptable.

Note: These effects could vary from person to person depending on the amount of water each drinks, body weight, dietary habits, and individual sensitivities such as preexisting kidney, liver, or heart diseases, and other factors.

The most notable possible health hazards in this ground water are sulfate, iron, and manganese. The sulfate levels in the contaminated ground water could cause diarrhea, particularly when ingested by infants. However, these levels make the water taste and smell very unpleasant, discouraging its use. Ingesting iron in the amounts found in the ground water could cause adverse effects ranging from increased skin pigmentation to liver damage, and diabetes, following long-term consumption. The manganese levels could affect the nervous system, cause mental disturbances, memory loss, irritability, and Parkinson's-like symptoms, including muscle rigidity and tremors. Although the uranium levels are not expected to be toxic to the kidneys, uranium and lead-210 found in the water could increase the lifetime cancer risk to slightly higher than the maximum acceptable value recommended by the EPA.

Based on this risk assessment, no adverse health effects would be expected after 1) consuming milk and meat from livestock grazed and/or watered on pastureland downgradient of the Gunnison site, or 2) eating garden produce watered with contaminated ground water. In addition, the available data show no health problems attributed to the former mill tailings would be expected from eating fish caught in the Gunnison River, Tomichi Creek, or a campground pond near the Gunnison site.

ECOLOGICAL AND LIVESTOCK RISK ASSESSMENT

Methods

The ecological risk assessment presented in this document is a screening level assessment that relies on limited environmental sampling and current scientific literature. Ecological risk assessment has many uncertainties due to limited scientific knowledge, such as little data on how some chemicals affect plants and animals, and the impact of a mixture of chemicals on plants and animals, which is not fully understood. Furthermore, due to limited environmental sampling at the Gunnison site, no information is available about site-specific constituent levels in plants or animals.

To determine the possible effects of these site-related constituents on grazing livestock, wildlife, and crops, this assessment compares the constituent concentrations in ground water, surface water, and sediment to available guidelines from regulatory agencies and literature values. It also evaluates surface water and sediment in the Gunnison River and Tomichi Creek near the site and a campground pond, and plant uptake of constituents from ground water, assuming plant roots extend into the alluvial aquifer and take up the most contaminated ground water. This risk assessment examines the use and impact of this ground water as a sole source for watering livestock and irrigating crops, as well as its possible impact on wildlife.

Livestock and agricultural results

Based on available information, the only possible health problem, if any, to livestock could come from sulfate, if the most contaminated ground water were used as a sole drinking water source. This water should not be used for continuous irrigation because its cobalt, iron, and manganese levels could harm agricultural crops. However, the existing data suggest that surface water in the Gunnison River, Tomichi Creek, and the campground pond is safe for livestock watering and crop irrigation.

Ecological results

The roots of vegetation growing above the alluvial aquifer may extend into the shallow contaminated ground water. However, based on literature values for several constituents, no ecological threat exists to plants in contact with soil-saturated with the most contaminated ground water in the alluvial aquifer. Limited environmental sampling and available guidelines from regulatory agencies were not sufficient to fully evaluate the possible long-term impacts of the affected ground water on terrestrial plants and animals.

CONCLUSIONS

Site-related contamination of ground water below the Gunnison site is limited to the alluvial aquifer; the contamination extends about 7000 ft (2200 meters) downgradient and toward the confluence of the Gunnison River and Tomichi Creek. Although several private wells are located in the area of ground water contamination, water from these wells is not used for domestic purposes because good quality water is available from a new water supply system. However, the water from these wells is safe for both livestock and irrigated crops. Consequently, no current human health risks from contaminated ground water are associated with the Gunnison site. Also, based on limited sampling, no human health risks are associated with ground water discharging to the streams and a campground pond near the Gunnison site.

This risk assessment has determined that there could be certain health problems in people and livestock if in the future the most contaminated ground water beneath the Gunnison site were used as a sole drinking or irrigation source. Therefore, no one should drink this water or use it for irrigating crops or watering livestock. The Gunnison site evaluation is ongoing and will include further characterization of ground water quality, water levels, and ground water movement. This risk assessment and future investigations will be used to determine how to deal with the contaminated ground water. Because some residents on the west side of the Gunnison River have not elected to connect to the alternate water supply system, a plan to monitor private wells in that area was developed and implemented. The public will be kept informed and notified of adverse changes in water quality.

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LIST OF ACRONYMS AND ABBREVIATIONS

Acronym	Definition
ACL	alternate concentration limit
BCF	bioconcentration factor
CDPHE	Colorado Department of Public Health and Environment
DHHS	U.S. Department of Health and Human Services
DOE	U.S. Department of Energy
EA	environmental assessment
EPA	U.S. Environmental Protection Agency
ER-L	effects range-low
ER-M	effects range-median
FWS	Fish and Wildlife Service
HEAST	Health Effects Assessment Summary Table
IARC	International Agency for Research on Cancer
IRIS	Integrated Risk Information System
Kd	soil-to-water distribution coefficient
LOAEL	lowest-observed-adverse-effect level
MCL	maximum concentration limit
NCP	National Contingency Plan
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
RAP	remedial action plan
RDA	recommended daily allowance
RfD	reference dose
SF	slope factor
SQC	sediment quality criteria
T&E	threatened and endangered
TDS	total dissolved solids
UMTRA	Uranium Mill Tailings Remedial Action
UMTRCA	Uranium Mill Tailings Radiation Control Act

1.0 INTRODUCTION

The purpose of this baseline risk assessment is to determine whether ground water contamination at the Gunnison, Colorado, uranium mill tailings site could adversely affect human health and the environment. The Gunnison site is 1 of 24 designated uranium mill tailings sites undergoing remediation in accordance with the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA) under the oversight of the U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project (42 USC §7901 *et seq.*). The remediation of surface contamination at the Gunnison processing site was completed in 1995 with relocation and stabilization of the tailings in a disposal cell located about 6 miles (10 kilometers) east of the town of Gunnison and 0.4 mile (0.6 kilometer) south of the county solid waste landfill.

A preliminary baseline risk assessment was conducted in 1990 to determine whether long-term use of ground water from domestic wells near the site could adversely affect public health (DOE, 1990). Section 2.7 summarizes the preliminary risk assessment results. Because of its results, area residents and the nearby Valco cement/concrete plant were provided bottled water until they were connected to a new alternate water supply system and the bottled water option was eliminated. This second risk assessment is baseline in that it describes existing ground water conditions at the site. This document evaluates pathways associated with surface water, sediment, and fish tissue; these data were not available for the first (preliminary) assessment of ground water-related risks. It also identifies major exposure pathways and evaluates the present and future potential for human health and environmental risks related to ground water contamination that may need attention before the site is characterized further. This risk assessment is based on available data from wells at the Gunnison site (the former tailings pile, ore storage, and mill yard areas).

This risk assessment follows the basic framework outlined by the U.S. Environmental Protection Agency (EPA) for evaluating hazardous waste sites to assess potential health and environmental impacts (EPA, 1989a). The framework consists of the following steps:

- Data evaluation.
  - Combining existing data from various site investigations.
  - Comparing sample results with background and tailings source data.
  - Selecting appropriate chemical data for the risk assessment.
- Exposure assessment.
  - Characterizing exposure settings.
  - Identifying exposure pathways.
  - Quantifying exposure.

- Toxicity assessment.
  - Identifying toxicity values.
  - Evaluating noncarcinogenic effects.
  - Evaluating carcinogenic effects from radionuclides and chemical carcinogens.
- Public health risk characterization.
  - Comparing toxicity ranges to predicted exposure ranges.
  - Combining risks across exposure pathways and multiple constituents.
  - Characterizing uncertainties.
- Environmental risk assessment.
  - Characterizing potential biota exposure pathways.
  - Identifying potential ecological receptors.
  - Evaluating environmental risk qualitatively.

This framework is incorporated in the methodology developed to evaluate current human health risks at UMTRA Project sites and to estimate risks from potential future use of contaminated ground water or surface water near the former processing site (DOE, 1994).

The DOE is authorized to conduct ground water remediation under the 1988 UMTRCA Amendments Act (42 USC §7922 *et seq.*) and will determine site-specific ground water compliance strategies for each site. This risk assessment will support decisions made for the UMTRA Ground Water Project. It also provides information that will be used to determine a site-specific ground water compliance strategy for the Gunnison site.

2.0 SITE DESCRIPTION

2.1 SITE BACKGROUND

The processing site is adjacent to the city of Gunnison in Gunnison County, Colorado, between the Gunnison River and Tomichi Creek (Figure 2.1). The mill was constructed in the late 1950s to produce uranium to sell to the U.S. Atomic Energy Commission (predecessor to the DOE) and operated from February 1958 until April 1962. Ore was trucked to the mill from mines in the Cochetopa Pass area, about 25 mi (40 km) southeast of Gunnison. The ore was ground and then leached with sulfuric acid and sodium chlorate. After leaching, the uranium-rich solutions and waste solids were separated by a four-stage countercurrent classifier and thickener circuit. The uranium solutions then were treated by solvent extraction to concentrate and recover the uranium; the solids were dumped in what became the tailings pile. During its 4 years of operation, the mill processed about 540,000 dry tons (490,000 metric tons) of ore with an average grade of 0.15 percent uranium oxide (FBDU, 1981).

The designated site covered 61 acres (ac) (24 hectares [ha]); approximately 35 ac (14 ha) were occupied by the rectangularly shaped tailings pile and approximately 20 more ac (8 ha) were occupied by the former mill structures, the former ore storage area, and miscellaneous areas (Figure 2.2). Windblown contaminated areas within and adjacent to the designated site occupied an additional 17 ac (6.8 ha).

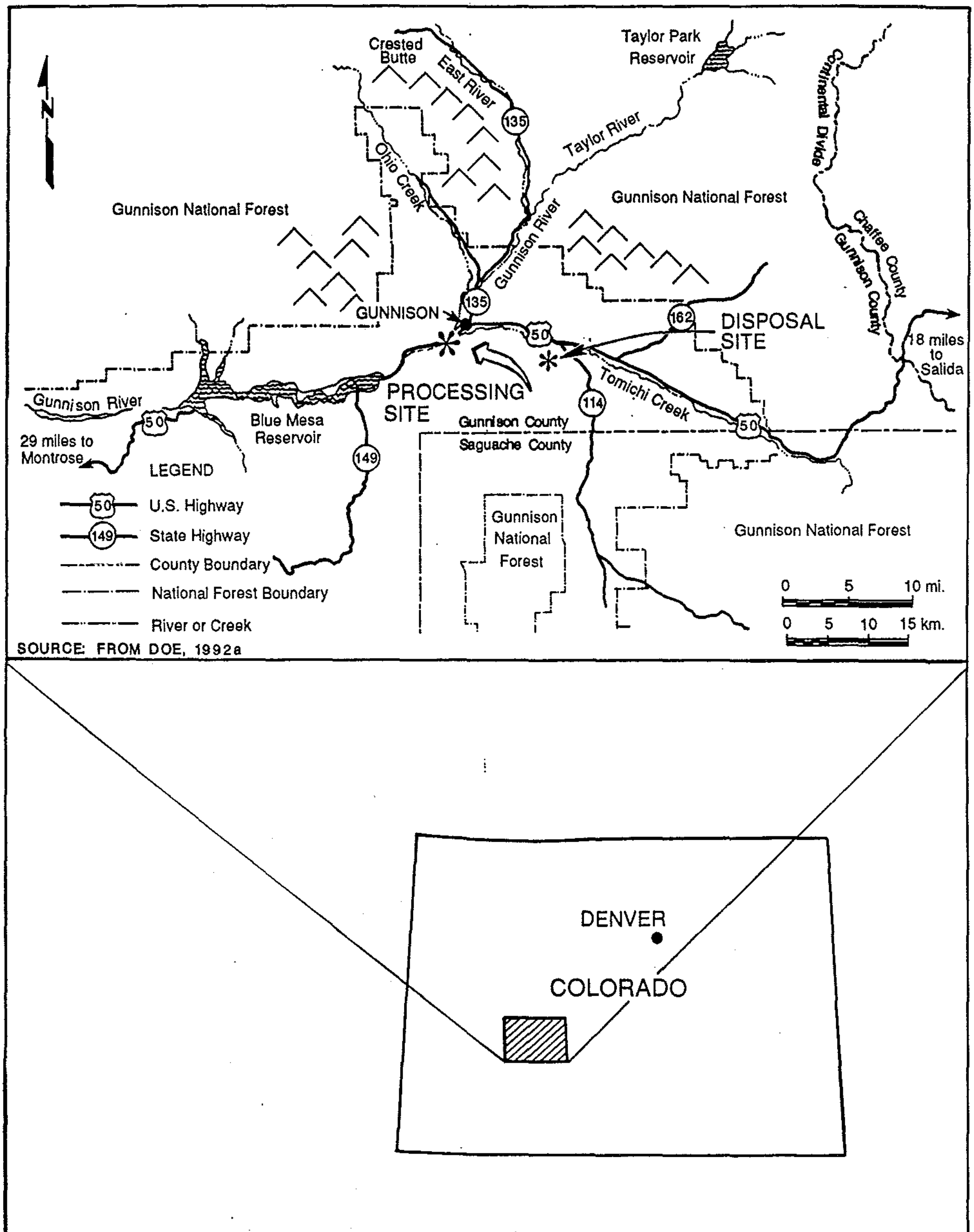
Uranium mill tailings, contaminated vicinity property materials, demolition debris (which was stored on the site), and windblown materials were relocated to a permanent disposal site during 1992 and 1995. This disposal site was selected based on public input, environmental considerations, and design opportunities acceptable to the regulatory agencies involved. The selected site is approximately 6 mi (10 km) east of the town of Gunnison and 0.4 mi (0.6 km) south of the county solid waste landfill.

2.2 CLIMATE

The Gunnison area is characterized by low humidity, frequent sunny days, and large daily and seasonal temperature ranges. The average annual temperature is 37 degrees Fahrenheit (°F) (2.8 degrees Celsius [°C]) and ranges from an average temperature of 10°F (-12°C) in January to 62°F (17°C) in July. The average annual precipitation is 11 inches (28 centimeters [cm]). Maximum rainfall occurs during July and August, while the least rainfall occurs from April through June. Thunderstorms are common during the summer. The average annual snowfall accumulation is 58 inches (150 cm), with the largest amount falling during January (NOAA, 1984). Prevailing winds are from the southwest and south.

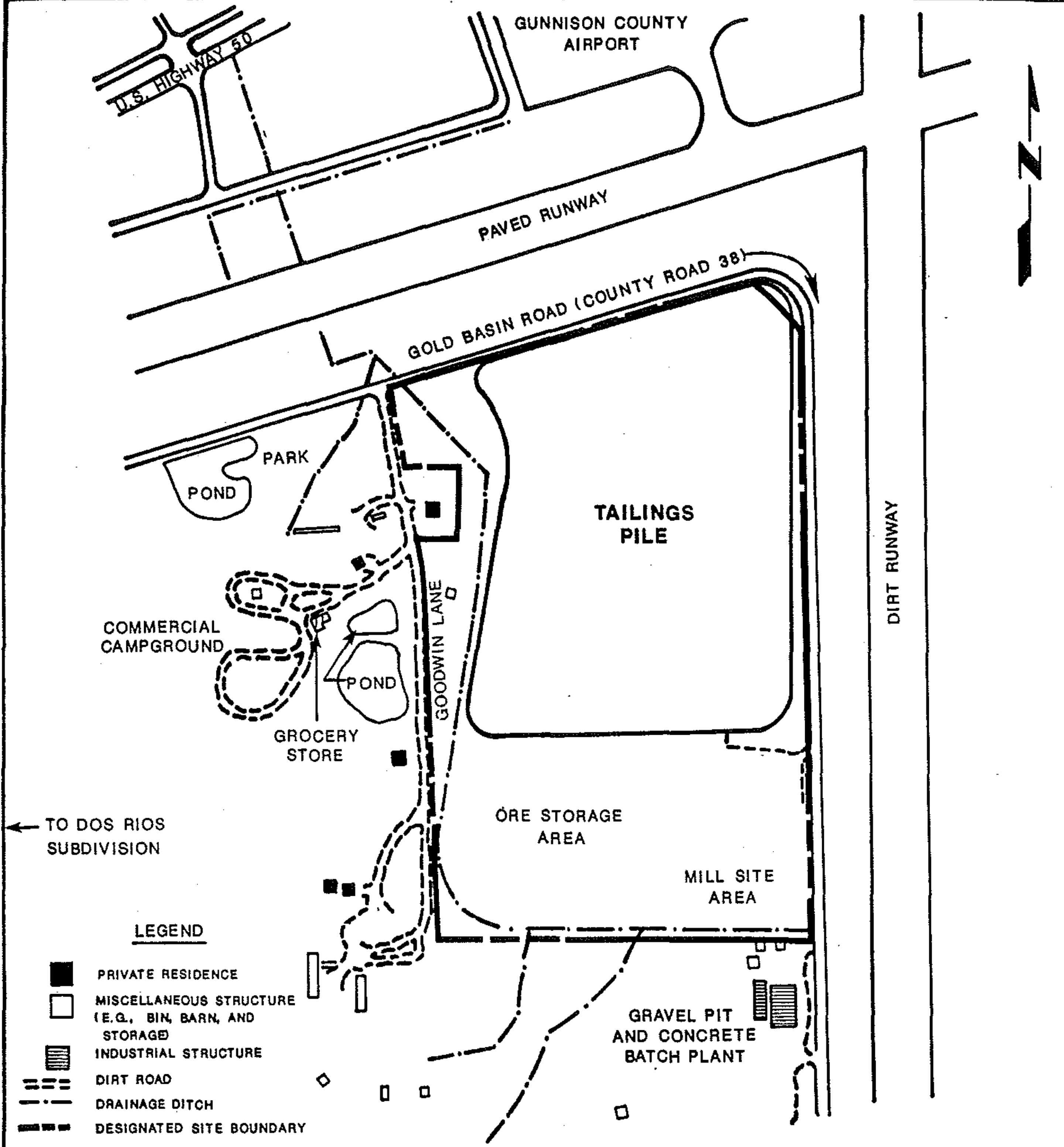
2.3 HYDROGEOLOGY

Widespread recent floodplain and terrace deposits (hereafter referred to as alluvium) associated with the Gunnison River and Tomichi Creek underlie the Gunnison processing site and surrounding area. The alluvium is composed of poorly sorted material ranging in size from clay-sized material to cobbles and occasional boulders. The alluvium generally tends to become more clayey with depth. Figure 2.3 provides a cross-sectional view of the



SOURCE: FROM DOE, 1992a

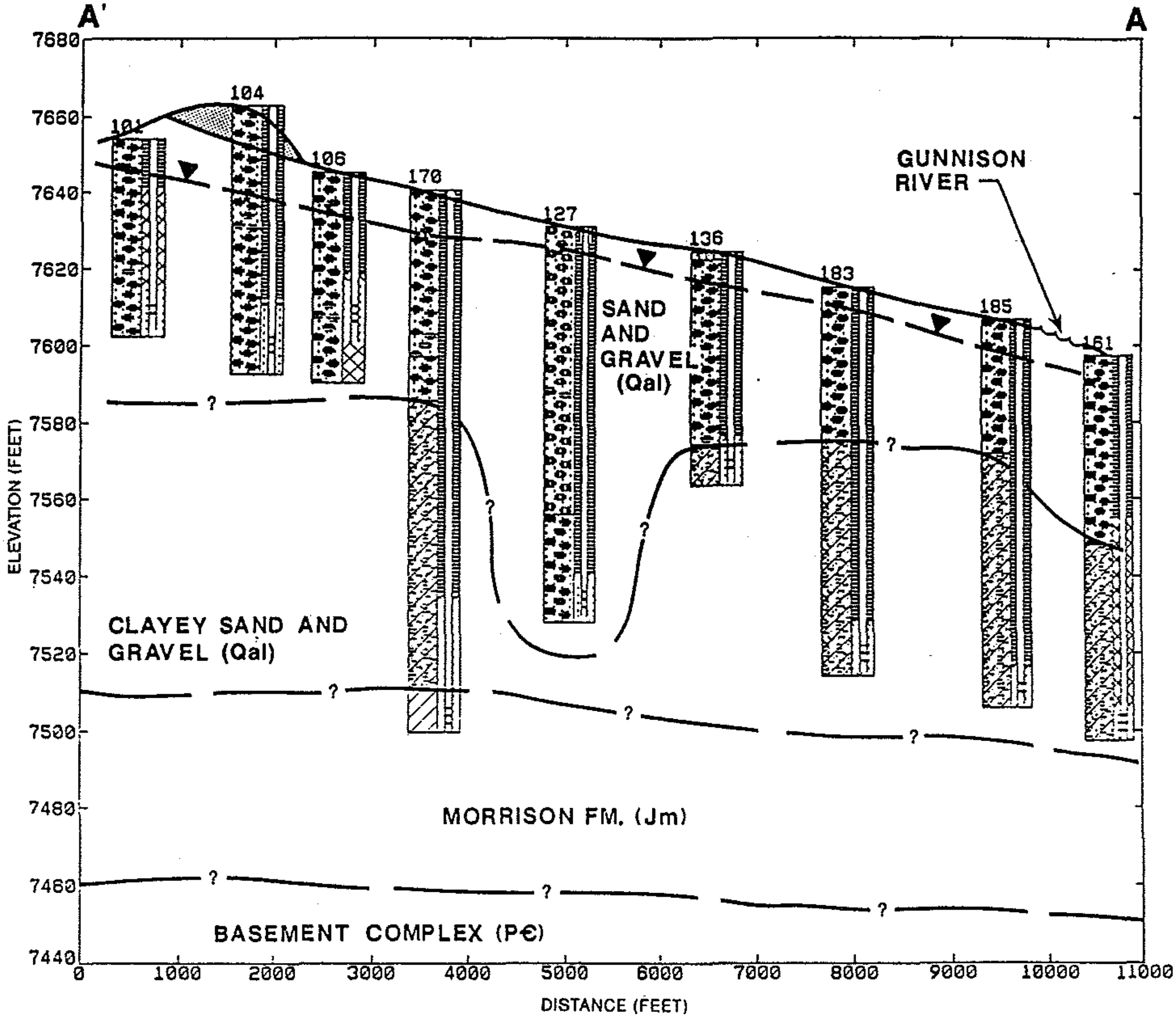
FIGURE 2.1
GUNNISON SITE LOCATION MAP, COLORADO



SOURCE: FROM DOE, 1992b

NOT TO SCALE

FIGURE 2.2
LAND USE IN THE VICINITY OF THE
GUNNISON PROCESSING SITE NEAR GUNNISON, COLORADO
(BEFORE REMEDIAL ACTION)



LEGEND

Qal	QUATERNARY ALLUVIUM
Jm	JURASSIC MORRISON FORMATION
P€	PRE-CAMBRIAN BASEMENT COMPLEX
—▼—	WATER TABLE, WATER LEVEL ELV. MEAS. IN FEBRUARY 1991
▨	TAILINGS PILE
—?—	PROJECTED FORMATION BOUNDARY

NOTES: 1. SEE FIGURE 2.5 FOR CROSS SECTION LOCATION.
2. VERTICAL SCALE IS EXAGGERATED.

(1:40

SOURCE: FROM DOE, 1992a

FIGURE 2.3
GEOLOGIC CROSS SECTION AT
THE GUNNISON PROCESSING SITE, COLORADO

subsurface materials. The thickness of the alluvium is extensive, ranging from 72 ft (22 m) to greater than 130 ft (40 m). The alluvium extends beyond the bottom of most of the boreholes and wells drilled by the DOE.

Underlying the alluvium is a discontinuous unit of unknown extent and thickness, known as the Brushy Basin Member of the Morrison Formation. Three DOE monitor wells (122, 170, and 194) encountered the Brushy Basin Member of the Morrison Formation. Lithologic logs of these wells describe the unit as a soft to moderately hard shale. The formation is composed of low-permeability shale that separates the overlying alluvium from the deeper units.

During previous site characterization activities, the DOE placed monitor well clusters upgradient and downgradient of the Gunnison processing site. These clusters generally consist of a shallow well (approximately 10 to 25 ft [3 to 8 m] deep below land surface), an intermediate well (approximately 40 to 60 ft [12 to 18 m] deep below land surface), and a deep well (approximately 90 to 100 ft [27 to 30 m] deep below land surface). The bottom 5 ft (1.5 m) of each well is screened. Figure 2.4 shows static water level elevations measured in the shallow monitor wells. This water table contour map shows that shallow ground water flows southwest as a subdued topographic replica of the land surface. The direction of horizontal ground water movement is comparable in the intermediate and deep monitor wells. The alluvial aquifer is generally unconfined. However, noncontinuous layers of low hydraulic conductivity silt and clay may create semiconfined conditions with increasing depth in the alluvium. Static water levels in each well of the monitor well clusters generally indicate a slight downward vertical gradient. Possible causes for these downward vertical gradients include 1) flood irrigation techniques used by landowners in the area, 2) that the area around Gunnison functions as a regional recharge area for the aquifer, and 3) dewatering activities from a nearby gravel pit operations. However, since ground water elevations in the alluvium fluctuate seasonally at the site, these downward vertical gradients may reverse locally in accordance with changes in recharge and discharge. The seasonal high water table usually occurs in early summer, and the seasonal low water table usually occurs in late winter.

The alluvial aquifer receives recharge from upgradient underflow, precipitation, and snowmelt. In addition, seasonal recharge occurs from the Gunnison River, Tomichi Creek, and irrigation ditches around the site. Ground water discharge is expected to occur as a result of localized pumping, shallow baseflow to the Gunnison River and/or Tomichi Creek, and deep ground water underflow that follows the southwest course of the Gunnison River downgradient of the Dos Rios subdivision.

Estimated hydraulic properties at the Gunnison processing site were presented in the remedial action plan (RAP) (DOE, 1992b). A 48-hour constant-rate, discharge aquifer performance test was performed in October 1983 in an 8-inch (20-cm)-diameter well completed in the alluvial aquifer to a total depth of 50 ft (15 m). The results were analyzed using the Theis-type curve-fitting method and the Jacob-Cooper Approximation and are presented in Appendix C to Attachment 3 of the Gunnison RAP (DOE, 1992b). The calculated range of transmissivity for the alluvial aquifer varies from 420 to 9600 square feet per day (ft^2/day) (39 to 890 square meters per day [m^2/day]) Hydraulic conductivities were calculated by dividing the transmissivity for each observation well by the saturated thickness of the

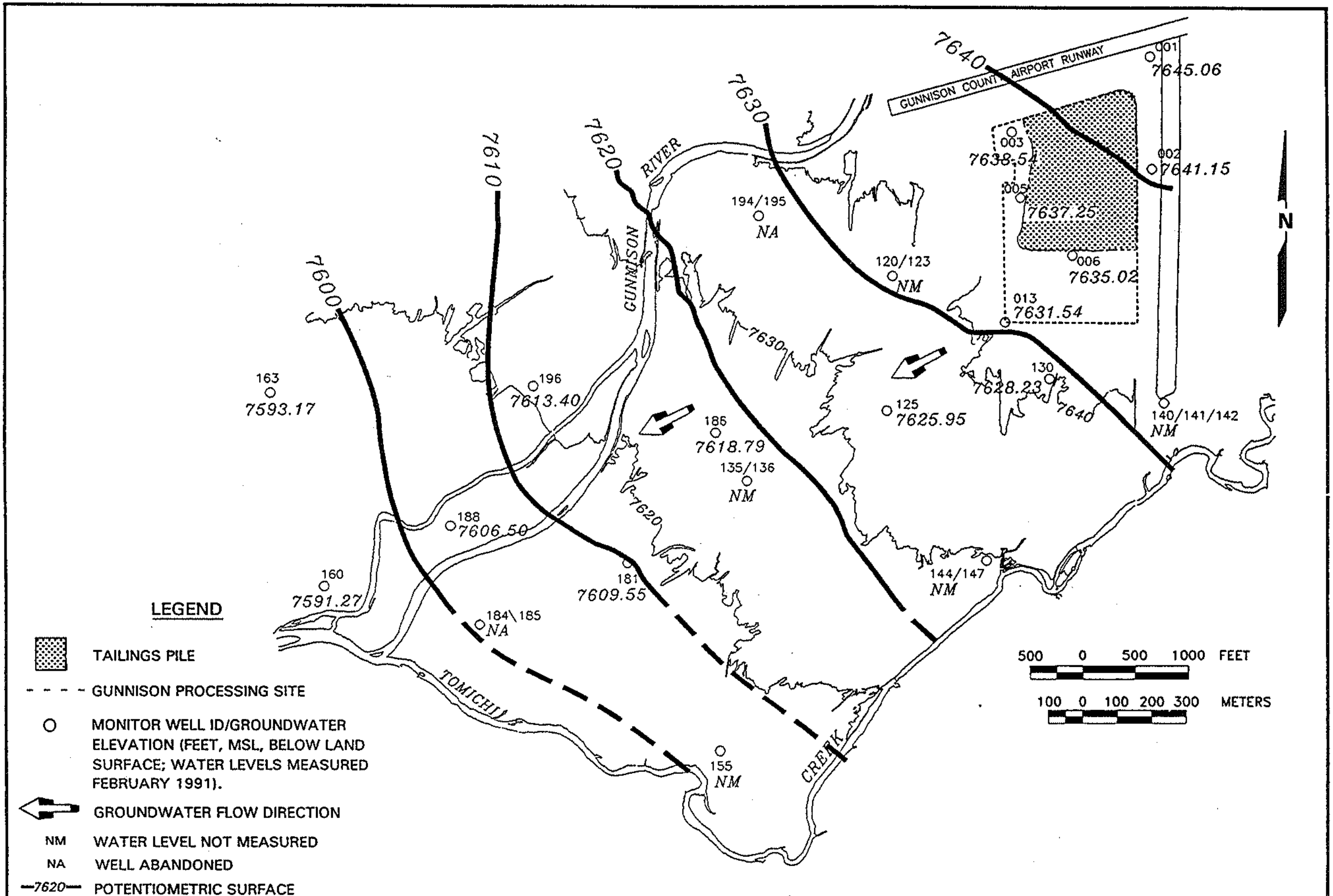


FIGURE 2.4
WATER TABLE ELEVATIONS IN THE ALLUVIAL AQUIFER
GUNNISON PROCESSING SITE, GUNNISON, COLORADO

SOURCE: FROM DOE 1992a

I:\FL3.01\J BRANNA 4-11-94 ACAD BX11GUN2.DWG GUN BRA CADGUN01.DRW pumping well (40.5 ft [12.3 m]). Hydraulic conductivities range from 10 to 240 ft/day (0.0035 to 0.084 centimeters per second [cm/s]). The wide range of hydraulic conductivity values results from the variation in size of the poorly sorted subsurface material, from clay to gravels and cobbles with occasional boulders, which is related to depositional facies.

Storage coefficient values range from 7×10^{-4} to 1×10^{-3}, based on calculations from the 1983 aquifer performance test. The calculated horizontal hydraulic gradient ranges from 3.8×10^{-3} to 4.8×10^{-3} (DOE, 1992b). Linear ground water velocities were calculated by applying Darcy's Law using the calculated range of hydraulic conductivities (10 to 240 ft/day [0.0035 to 0.085 cm/s]) and water table gradients (0.0038 to 0.0048) and the assumed porosity value (0.25 for gravelly alluvium) appropriate for the site (Freeze and Cherry, 1979; Davis and DeWeist, 1966). From this calculation, ground water velocities in the alluvial aquifer are estimated to range from 56 to 1700 ft/yr (1.1×10^{-4} to 33×10^{-4} cm/s), and average approximately 270 ft/yr (5.2×10^{-4} cm/s) (DOE, 1992b). Although ground water velocity is subject to localized variations from heterogeneities in the subsurface materials, the calculated average value appears to be representative of the alluvial ground water system, based on the distance of migration of the contaminant plume relative to the duration of time that the tailings have been placed on-site. Assuming that ground water moves at an approximate rate of 270 ft/yr, and that the tailings have been in place at the processing site for approximately 30 years, the leading edge of the contaminant plume should be expected to have moved roughly 8100 ft (2500 m) downgradient from the tailings pile. Based on actual water quality data for conservative (mobile) species (uranium and sulfate), the estimated ground water velocity value appears to be representative. The magnitude and extent of site-related contamination are discussed in detail in Section 3.2.

2.4 SURFACE WATER

The Gunnison processing site lies in the Gunnison River basin, 0.4 mi (0.6 km) east of the Gunnison River, 0.4 mi (0.6 km) northwest of Tomichi Creek, and 1.5 mi (2.4 km) above the confluence of the two. Drainage across the site is south and east toward Tomichi Creek. The site was bounded on the west by small storm drainage ditches and on the south and west by an irrigation ditch. Some sections of the irrigation ditch were relocated during site restoration.

The Gunnison River has a drainage basin of 1012 square miles (mi^2) (2621 square kilometers [km^2]) above its confluence with Tomichi Creek. The USGS reports an average flow of about 700 cubic feet per second (ft^3/s) (20 cubic meters per second [m^3/s]); the maximum recorded flow of the Gunnison River for the 55 years of record was 11,450 ft^3/s (324 m^3/s) in 1918 (USGS, 1984). Tomichi Creek has a drainage basin of 1061 mi^2 (2748 km^2) above its confluence with the Gunnison River and has an average annual flow of about 160 ft^3/s (5 m^3/s). A maximum flow of 1890 ft^3/s (54 m^3/s) was recorded in 1957 (USGS, 1984).

Snow generally melts from May through June in the Gunnison area. Based on recorded flow data, maximum flows occur in the Gunnison River Basin during the spring runoff. Runoff from snowmelt is occasionally augmented by rainstorms; however, precipitation in the spring is generally the lowest of the year (USGS, 1984).

2.5 LAND USE

The former uranium processing site is adjacent to the Gunnison city limits and the Gunnison County Airport. The main paved airport runway is within 200 feet (ft) [60 meters (m)] of the northern processing site boundary, and an emergency dirt runway is within 150 ft (46 m) of the eastern site boundary. On the north and east the site was bounded by former County Road 38 (Gold Basin Road), which was rerouted as part of the tailings removal process. The land between this county road and the runways is owned by the county for expansion of the airport. The processing site has been acquired by the state of Colorado. An operating commercial gravel pit and concrete batch plant are on private land immediately south of the site. On private land west of the site are a park, a commercial campground with a grocery store and shower house, a small pond that is used for fishing, and five private residences (DOE, 1990). There are more than 100 private wells within a mile (1.6 km) and downgradient of the site, between the Gunnison River and Tomichi Creek.

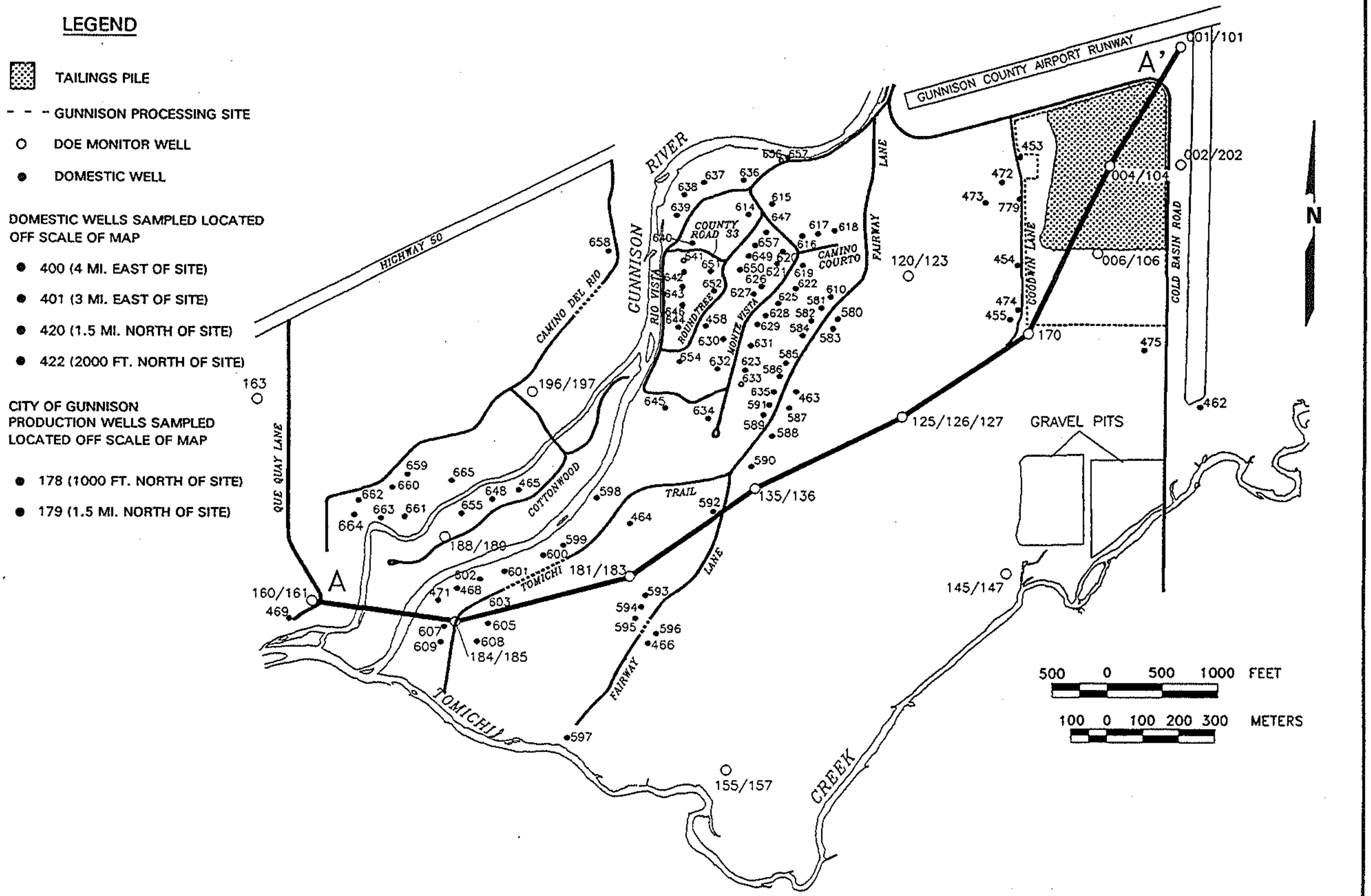
2.6 WATER USE

Figure 2.5 shows the well locations of residences along the western boundary of the Gunnison site and the locations of DOE monitor wells downgradient of the site. Figure 2.6 shows the locations of monitor wells at and upgradient of the processing site. Five private residences and a campground are located along Goodwin Lane. Previously, these houses and the campground obtained water from shallow wells completed in the alluvial aquifer. As an interim measure, these residents were provided the option of receiving bottled water. They were connected to the alternate water supply when it was completed in July 1994 (Figure 2.7).

Approximately 2000 ft (600 m) downgradient to the west and southwest of the site are Units 1 and 3 of the Dos Rios subdivision. The Dos Rios subdivision contains approximately 100 houses and a condominium complex east of the Gunnison River, most of which were built in the early 1970s. Previously, these houses obtained water from private wells completed in the alluvial aquifer. The closest domestic well in the subdivision lies approximately 1600 ft (490 m) from the processing site. The results of private water well sampling during April, July, and October 1990 showed that 21 of 100 private water wells sampled downgradient of the processing site had uranium concentrations that were elevated above background levels (DOE, 1992b). Uranium concentrations exceeded the ground water protection standard (0.044 milligrams per liter [mg/L]) in four of these private wells in 1990. As an interim measure, residents in the Dos Rios subdivision and along Goodwin Lane were provided the option of receiving bottled water. All residences potentially affected by ground water contamination from the processing site were connected to the alternate water supply system in July 1994 (Figure 2.7).

Adjoining the Gunnison site to the south is a gravel plant. This plant used an alluvial well for its office in addition to a batch plant well used to make concrete. The plant also was connected to the alternate water supply system in July 1994.

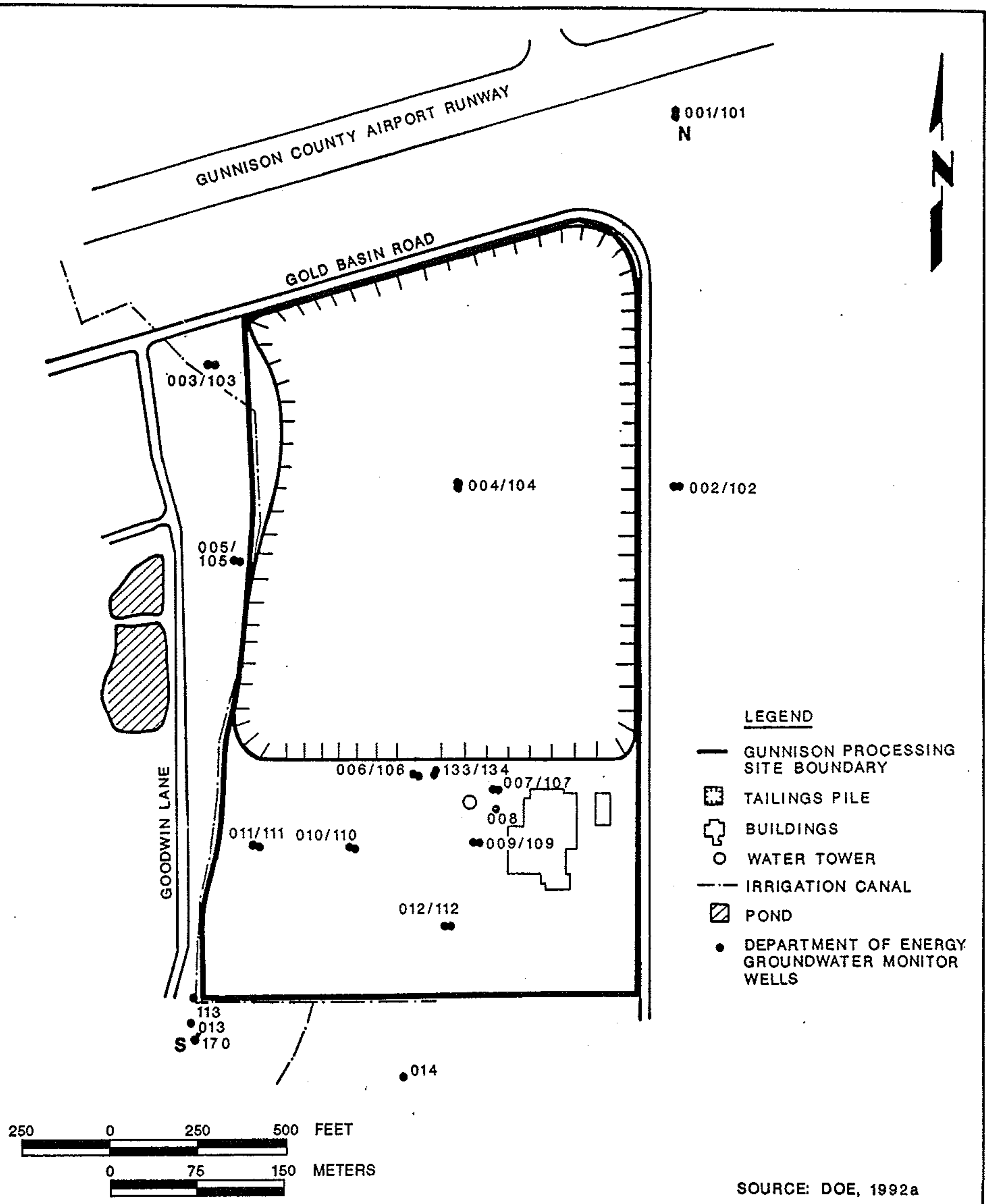
The airport runways border the site to the north and east. The airport building is approximately 0.5 mi (0.8 km) from the site, upgradient and on the distant side of the runways. Therefore, the airport is not considered a potential receptor.



SOURCE: MODIFIED FROM DOE, 1992a

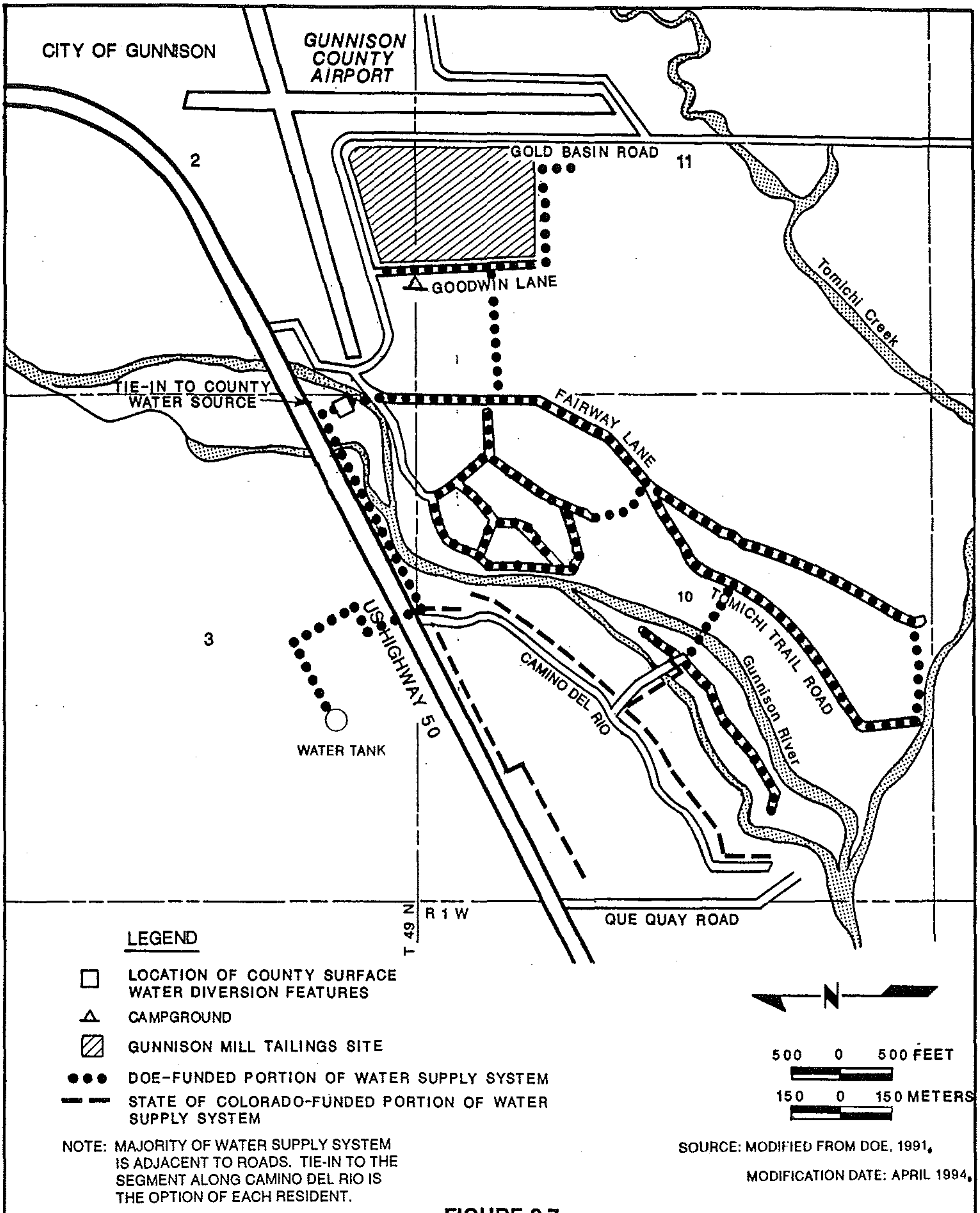
FIGURE 2.5
LOCATION OF SELECTED DOE MONITOR WELLS AND DOMESTIC WELLS IN THE VICINITY OF GUNNISON PROCESSING SITE, GUNNISON, COLORADO

I:\FL3.01\J BRANNA 4-11-94 ACAD 8X11GUN2.DWG GUN BRA CADGUN07.DRW



SOURCE: DOE, 1992a

FIGURE 2.6
MONITOR WELL LOCATION MAP OF THE GUNNISON PROCESSING SITE, COLORADO



NOTE: MAJORITY OF WATER SUPPLY SYSTEM IS ADJACENT TO ROADS. TIE-IN TO THE SEGMENT ALONG CAMINO DEL RIO IS THE OPTION OF EACH RESIDENT.

SOURCE: MODIFIED FROM DOE, 1991.

MODIFICATION DATE: APRIL 1994.

FIGURE 2.7
WATER SUPPLY SYSTEM COMPONENTS
NEAR GUNNISON, COLORADO

The city of Gunnison derives its entire municipal water supply from wells completed in the alluvial aquifer. All the city's wells are north and upgradient of the site and are not affected by constituents leaching from the former tailings pile.

2.7 PREVIOUS RISK ASSESSMENT SUMMARY

A preliminary baseline risk assessment was conducted during 1990 to determine whether long-term use of ground water from domestic wells near the Gunnison processing site had the potential for adverse health effects (DOE, 1990). The results of private well sampling for this baseline risk assessment during April, July, and October 1990 showed the uranium levels in 21 of 100 private water wells downgradient of the processing site were higher than background levels, currently estimated to be up to 0.0085 mg/L, which is the maximum observed natural concentration in the local area. Four of these wells also exceeded the MCL in Table A of 40 CFR Part 192 (0.044 mg/L). These water guidelines, which are in units of mass, can be converted to activity levels of 20 and 30 pCi/L for the health advisory and the proposed MCL, respectively, by multiplying by a conversion factor of 686. Other metals, including manganese, cadmium, and the uranium decay product lead-210, also were detected at levels above background.

In September 1990, based on the results of the Gunnison preliminary baseline risk assessment for ground water contamination (DOE, 1990), the DOE began making bottled water available to all downgradient users, including the entire Dos Rios subdivision, as a public health measure. The bottled water was intended as an interim action for residents with contaminated water wells, to allow time to determine a permanent solution; distribution was stopped in March 1996. The DOE evaluated ways of providing a permanent potable water supply system in an environmental assessment (EA) (DOE, 1991), which was approved in 1992. The DOE and the state funded construction of a water supply system, which began in 1992; it was completed in July 1994.

All residents on the east side of the Gunnison River (Dos Rios Units 1, 3, and the Island unit and Goodwin Lane) were connected to this alternate system. Residents on the west side of the Gunnison River (Dos Rios Unit 2 and along Que Quay Road) were given the option of connecting to the alternate water supply system; many residents declined this option. To ensure that private wells on the west side of the Gunnison River are not being adversely affected by the contaminated ground water potentially moving from the former uranium mill tailings site, the DOE has developed and implemented a plan to monitor private wells in this area; the public will be informed if adverse changes in water quality occur (DOE, 1995).

3.0 EXTENT OF CONTAMINATION

A map of the Gunnison processing site showing the locations of the tailings pile and the preremediation DOE monitor well network is presented in Figure 2.6. All available DOE monitor well water quality data between 1985 and 1993 were used to characterize the plume geometry and the geochemical processes active at this site (DOE, 1996). Upgradient DOE monitor wells 001, 101; private wells 400, 401, 402, 422; and city wells 178 and 179 were used to define the background water quality of the alluvial aquifer at the Gunnison processing site (DOE, 1996). The locations of these background wells are given in Figure 2.5. Water quality data for the years 1989 through 1993 from the background DOE monitor wells were used in the statistical characterization of background ground water. All available ground water quality data from the six non-DOE background wells (primarily collected between 1980 and 1990) were also used in this analysis. Ground water samples were collected in accordance with applicable standard operating procedures described in the Albuquerque Operations Manual (JEG, n.d.). The background ground water quality, magnitude of contamination, plume geometry, and constituents of potential concern for the Gunnison processing site are discussed below.

3.1 BACKGROUND WATER QUALITY DATA SUMMARY

The total dissolved solids (TDS) in the background ground water samples range from 196 to 428 mg/L, with a median value of 300 mg/L. Sulfate concentrations range from 11 to 38 mg/L, and alkalinity concentrations (expressed as $CaCO_3$) range from 130 to 300 mg/L, with median values of 18 and 236 mg/L, respectively.

Uranium was present at low levels in all the background ground water samples, with a maximum observed value of 0.0085 mg/L. The dominant cationic species is calcium, with concentrations that range up to 102 mg/L. The iron concentration in background ground water ranged from <0.03 to 2.7 mg/L, with a median of 0.28 mg/L. The high value for iron (2.7 mg/L) was considerably higher than other background iron values taken from these wells (median value = 0.28 mg/L). This single high iron value may represent a sampling or analysis anomaly. Manganese values ranged from <0.01 to 0.81 mg/L, with a median value of 0.06 mg/L. Although the upgradient background ground water samples were typically low in manganese and iron, there is some evidence that naturally elevated iron and manganese levels may have been present in some monitor wells downgradient of the tailings (see Section 3.2). Table 3.1 summarizes the minimum, median, and maximum values found for 43 chemical parameters in background ground water and plume-affected ground water for the Gunnison processing site.

3.2 MAGNITUDE AND EXTENT OF SITE-RELATED GROUND WATER CONTAMINATION

Ground water at the Gunnison processing site has been contaminated by acidic tailings leachate that contains high concentrations of uranium, iron, manganese, and sulfate, as well as many other constituents (Table 3.1), although background ground water at this site contains relatively low sulfate and uranium concentrations. Conversely, these constituents are present at relatively high levels in tailings effluent.

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
INORGANIC CONSTITUENTS						
Aluminum						
Background[b]	25	0.002-0.1	32	0.002	–	0.15
Plume[c]	14	0.05-0.1	7	<0.05	–	0.20
Ammonium						
Background	15	0.1	53	<0.1	0.1	0.4
Plume (133/134 only)	4	0.04	100	0.3	0.5	0.8
Antimony						
Background	13	0.003	31	<0.003	–	0.008
Plume	14	0.003	21	<0.003	–	0.010
Arsenic						
Background	25	0.001-0.1	4	<0.001	–	0.02
Plume	14	0.005-0.01	0	–	–	–
Barium						
Background	24	0.002-0.1	96	0.002	0.18	0.7
Plume	14	0.01-0.1	64	0.01	0.01	0.1
Beryllium						
Background	13	0.005-0.01	0	–	–	–
Plume	14	0.005-0.01	0	–	–	–
Boron						
Background	11	0.1	0	–	–	–
Plume	11	0.05-0.1	9	<0.05	–	1.0

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site (Continued)

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Bromide						
Background	11	0.1	0	–	–	–
Plume	11	0.1-1	27	<0.1	–	0.5
Cadmium						
Background[b]	24	0.0001-0.005	0	–	–	–
Plume[c]	14	0.0001-0.001	71	<0.001	0.001	0.004
Calcium						
Background	27	0.001-0.01	100	35	75	102
Plume	15	0.01-2	100	92	650	690
Chloride						
Background	27	0.2-1	100	2	14	31
Plume	15	0.5-1	100	10	15	20
Chromium						
Background	24	0.001-0.01	0	–	–	–
Plume	14	0.01-0.05	0	–	–	–
Cobalt						
Background	13	0.03-0.05	0	–	–	–
Plume (133/134 only)	4	0.03-0.05	100	0.19	0.32	0.37
Copper						
Background	22	0.001-0.02	9	<0.001	–	0.046
Plume	14	0.01-0.02	0	–	–	–
Cyanide						
Background	7	0.01	0	–	–	–
Plume	9	0.01	0	–	–	–

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site (Continued)

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Fluoride						
Background	21	0.1-1	48	<0.1	–	0.2
Plume	11	0.1	100	0.1	0.9	1.3
Iron						
Background[b]	27	0.001-0.03	89	<0.03	0.28	2.7
Plume[c] (133/134 only)	4	0.03-0.1	100	49	66	91
Lead						
Background	24	0.001-0.01	4	<0.001	–	0.012
Plume	14	0.003-0.03	7	<0.003	–	0.010
Magnesium						
Background	27	0.001-0.1	100	10	15	26
Plume	15	0.001-0.1	100	20	30	41
30-50 ft (9-15 m)[d]	13	0.001-0.5	100	30	50	94
Manganese						
Background	23	0.01-0.03	83	<0.01	0.06	0.81
Plume	15	0.01	100	0.05	3.5	7.0
30-50 ft (9-15 m)[d]	13	0.01-0.05	100	21	26	39
Mercury						
Background	12	0.0002-0.002	8	<0.000	–	0.0003
Plume	11	0.0002	0	–	–	–
Molybdenum						
Background	25	0.001-0.05	4	<0.001	–	0.02
Plume	14	0.01	14	<0.01	–	0.03

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site (Continued)

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum	Observed Median[a]	Observed Maximum
				(mg/L)		
Nickel						
Background	21	0.001-0.04	10	0.037	–	0.043
Plume (133/134 only)	4	0.04	100	0.10	0.16	0.18
Nitrate						
Background[b]	27	0.1-5	81	<0.1	3	49
Plume[c]	15	1-5	20	<1.0	–	6.2
Phosphate						
Background	15	0.1	67	<0.1	0.1	0.2
Plume	14	0.1	50	<0.1	–	0.1
Potassium						
Background	27	0.001-0.01	100	1.4	2.4	14
Plume	15	0.01-0.1	100	1.9	5.4	8.1
Selenium						
Background	25	0.002-0.03	0	–	–	–
Plume	14	0.001-0.05	7	<0.001	–	0.007
Silica						
Background	21	0.1-2	100	3	14	18
Plume (006)	10	0.1-2	100	15	17	21
30-50 ft (9-15 m)[d]	12	0.1-2	100	28	38	43
Silver						
Background	12	0.01	0	–	–	–
Plume	11	0.01	18	<0.01	–	0.01

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site (Continued)

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Sodium						
Background	27	0.001-0.002	100	3.2	7.5	19
Plume	15	0.002-5	100	7.4	15	21
30-50 ft (9-15 m)[d]	13	0.002-1	100	20	33	59
Strontium						
Background[b]	13	0.01-0.1	100	0.24	0.28	0.31
Plume[c]	14	0.01-0.1	100	0.30	0.74	1.0
Sulfate						
Background	29	0.1	100	11	18	38
Plume (133/134 only)	4	0.1-10	100	1470	1540	1590
Sulfide						
Background	15	0.1-1	40	<0.1	–	0.8
Plume	14	0.1-1	21	<0.1	–	1.2
Thallium						
Background	15	0.005-0.01	7	<0.005	–	0.01
Plume	15	0.005-0.1	7	<0.005	–	0.01
Tin						
Background	13	0.005-0.025	8	<0.005	–	0.04
Plume	14	0.005-0.05	0	–	–	–
Uranium						
Background	29	0.0003-0.003	100	0.000	0.003	0.0085
Plume (133/134 only)	4	0.0003-0.001	100	1.2	1.4	1.6
Vanadium						
Background	24	0.004-0.05	4	<0.004	–	0.01
Plume	14	0.01	21	<0.01	–	0.01

Table 3.1 Comparison of background and downgradient ground water quality for the alluvial aquifer, Gunnison, Colorado, site (Concluded)

Constituent	Number of samples	Detection limit (mg/L)	Percent above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Zinc						
Background	23	0.001-0.005	48	<0.005	-	0.096
Plume (133/134 only)	4	0.005	100	0.37	0.55	0.86
RADIONUCLIDES						
Lead-210						
Background	21			0.0	0.8	44
Plume	14			0.4	2.4	78
Polonium-210						
Background	7			0.0	0.0	0.3
Plume	9			0.0	0.4	1.6
Radium-226						
Background	25			0.0	0.1	6.2
Plume	15			0.0	0.1	1.3
Radium-228						
Background	17			0.0	0.1	1.8
Plume	17			0.0	0.4	8.3
Thorium-230						
Background	18			0.0	0.1	8.6
Plume	8			0.1	0.5	2.0

[a] The median is the 50th percentile of the sample data. The median cannot be calculated if 50 percent or less of the data are above detection. A dash "-" in the median column indicates that the median cannot be calculated.

[b] Background concentrations are from DOE monitor wells 001 and 101 (1989-1993 data), domestic wells 400, 401, 402, and 422 (1982-1984 data); and city of Gunnison production wells 178 and 179 (1984-1990 data).

[c] Plume concentrations are from DOE monitor wells 006, 133, and 134 (1989-1993 data) unless otherwise noted. These wells generally have screen depths of 10 to 20 ft (3 to 6 m).

[d] Concentrations are from deeper downgradient DOE monitor wells 106, 109, 110, 111, and 112 (1989-1993 data). These wells generally have screen depths of 30 to 50 ft (9 to 15 m).

Note: All samples were filtered.

Once introduced to the alkaline ground water typical of the Gunnison processing site, uranium and sulfate behave chemically as relatively conservative (mobile) components. Uranium and sulfate are excellent tracers of the extent of contamination and plume geometry at this site. Although the highest sulfate and uranium concentrations are found in ground water adjacent to the tailings pile, concentrations of these constituents that are above background levels are present in ground water for over 7000 ft (2100 m) downgradient from the tailings pile (Figures 3.1 and 3.2) (data for all wells). A review of the sulfate and uranium data of paired monitor wells 160 and 161 and monitor wells 163 and 188 suggests the possibility that the leading edges of the uranium and sulfate plumes have migrated under the river in the deeper portions (depth more than 30 ft [9 m] of the alluvial aquifer) (Figures 3.3 and 3.4). The DOE installed several shallow monitor wells on the west side of the Gunnison River in late 1994 to further characterize the extent of contamination in this area. Shallow ground water across the river (0 to 30 ft) has been tested in these wells by DOE and is not contaminated. Most of the private wells in the Dos Rios subdivision are drilled in the shallow zone, with depths ranging from 20 to 30 ft (6 to 9 m). Although shallow private wells east of the Gunnison River have shown signs of contamination, no private wells west of the Gunnison River or in the Island Unit (the area between the two branches of the Gunnison River in the Dos Rios subdivision) have shown elevated site-related constituent levels.

Contaminated ground water in the shallow portion of the aquifer is discharging to the Gunnison River, while low levels of contamination (e.g., 0.014 mg/L uranium) in the deeper portion of the aquifer is moving under the river near its confluence with Tomichi Creek. Factors that may cause contaminants to migrate into the deeper flow regime include flood irrigation downgradient of the site, dewatering of the Valco gravel pit, and density differences between site-affected and ambient ground water.

Although uranium and sulfate are the characteristic constituents of the distal parts of the Gunnison plume, plots of iron and manganese concentrations versus time (Figures 3.5 and 3.6) for some downgradient monitor wells suggest that elevated levels of these constituents may have also migrated downgradient from the tailings pile. The manganese and iron concentrations in ground water at the site are not as evenly distributed throughout the plume as those of sulfate and uranium. The possibility that the elevated manganese and iron levels in the far downgradient monitor wells are due to natural processes and not due to tailings contamination is discussed in Section 3.4.

3.3 CONSTITUENTS OF POTENTIAL CONCERN

The data presented in Sections 3.1 and 3.2 are summarized in Table 3.1. These data were used to compile a list of constituents of potential concern for the assessment of human health and environmental risk at the Gunnison site. A constituent was placed on the list of constituents of potential concern (Table 3.2, column 1) if concentrations of the constituent in downgradient monitor wells were, on average, higher than those in the background wells. The statistical comparison was made at the 0.05 level of significance (DOE, 1995). Some additional constituents listed in Table 3.1 show higher median and/or maximum concentrations

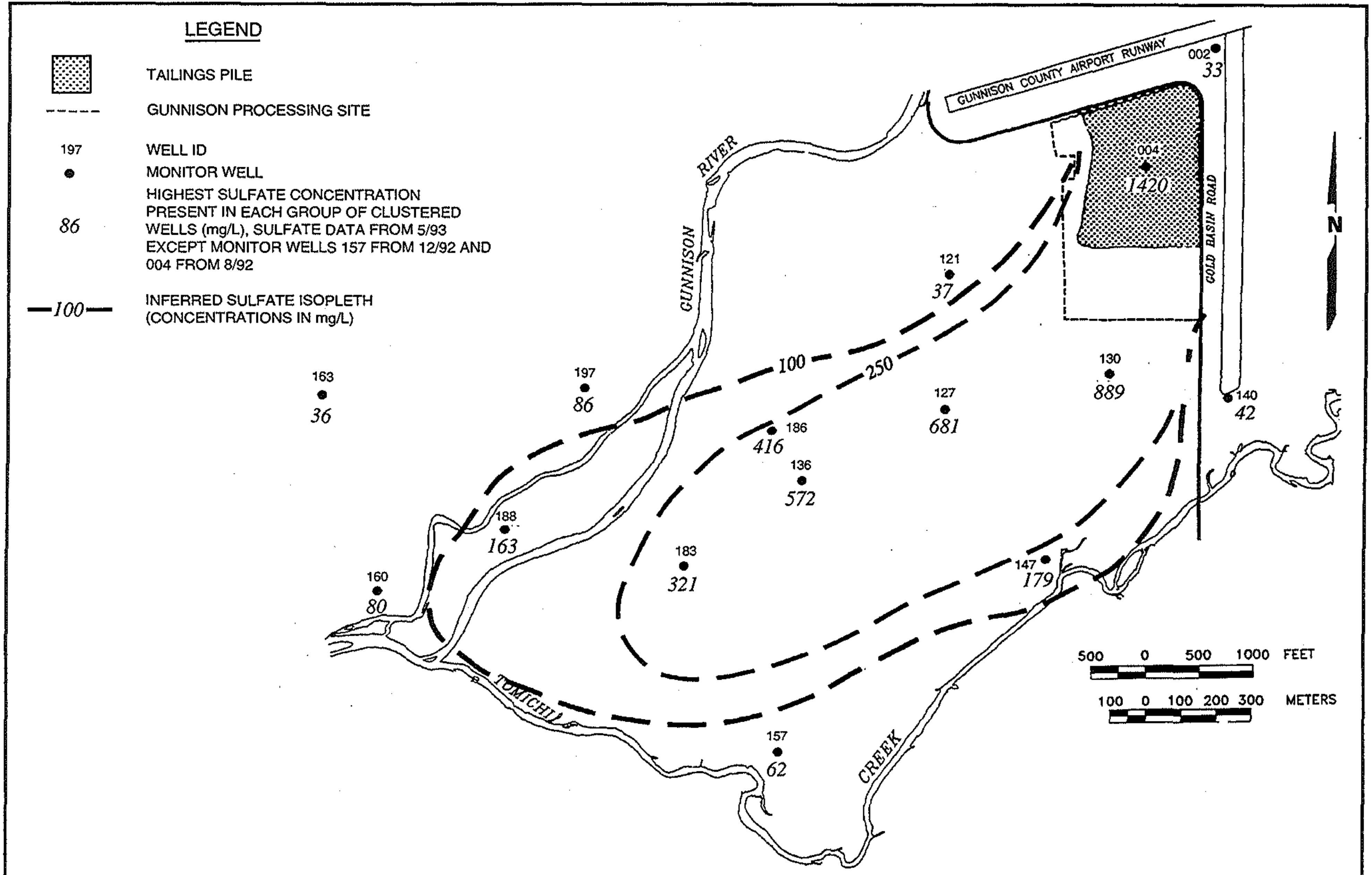


FIGURE 3.1
APPROXIMATE EXTENT OF SULFATE CONTAMINATION
GUNNISON PROCESSING SITE, GUNNISON, COLORADO

MAC: SITE/GUN/BRA/CADGUN09

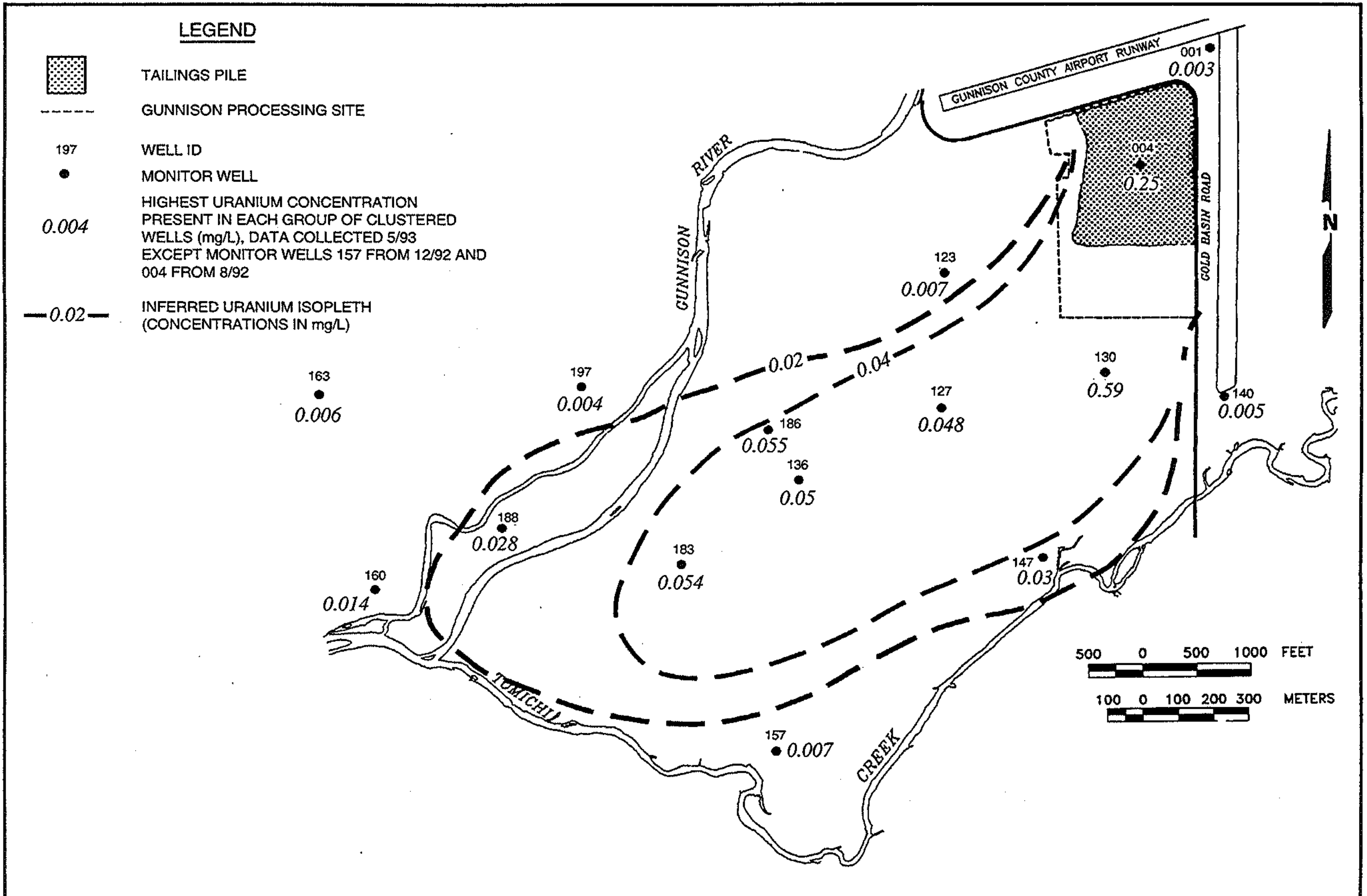
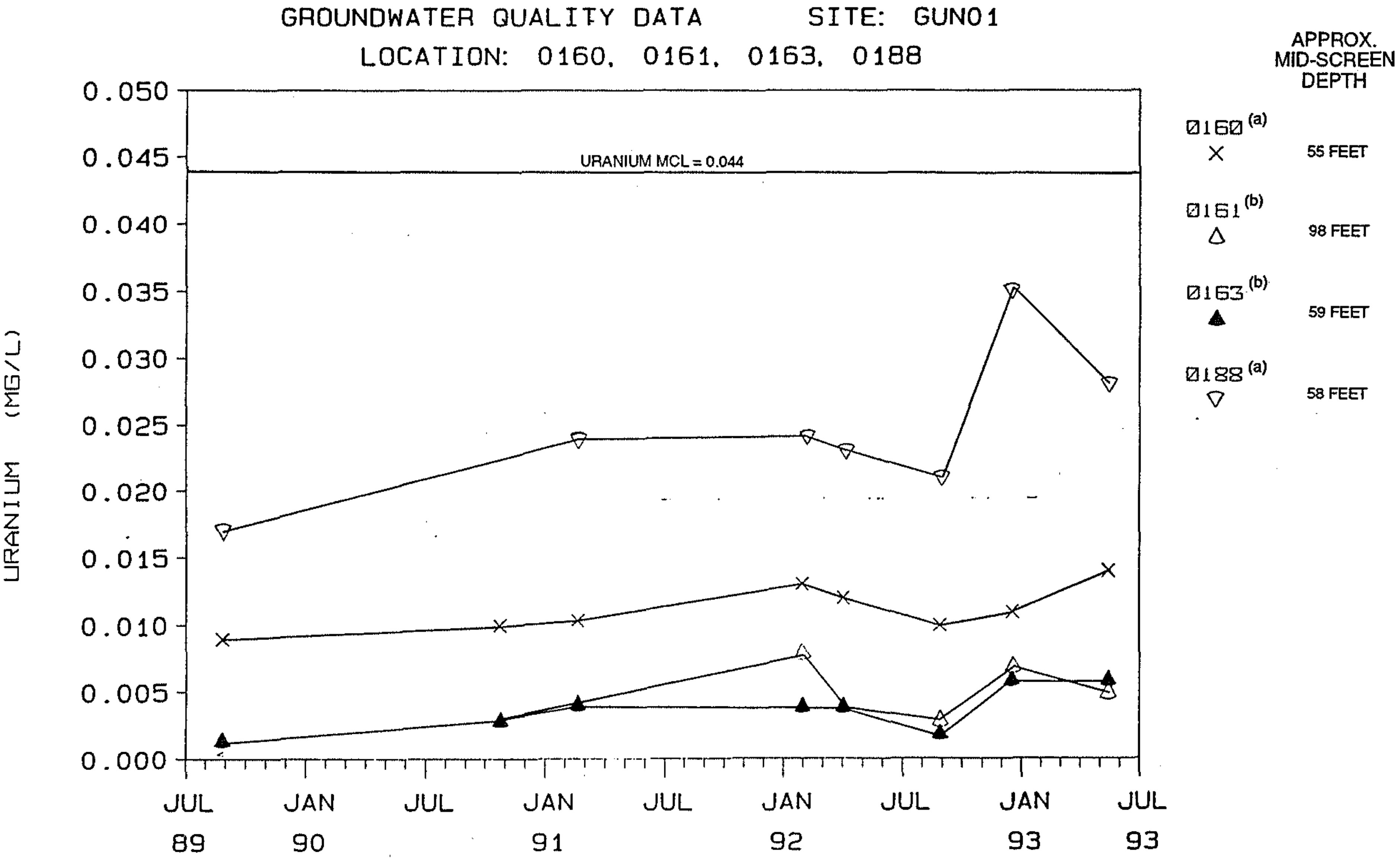


FIGURE 3.2
APPROXIMATE EXTENT OF URANIUM CONTAMINATION
GUNNISON PROCESSING SITE, GUNNISON, COLORADO

MAC: SITE/GUN/BRA/CADGUN10



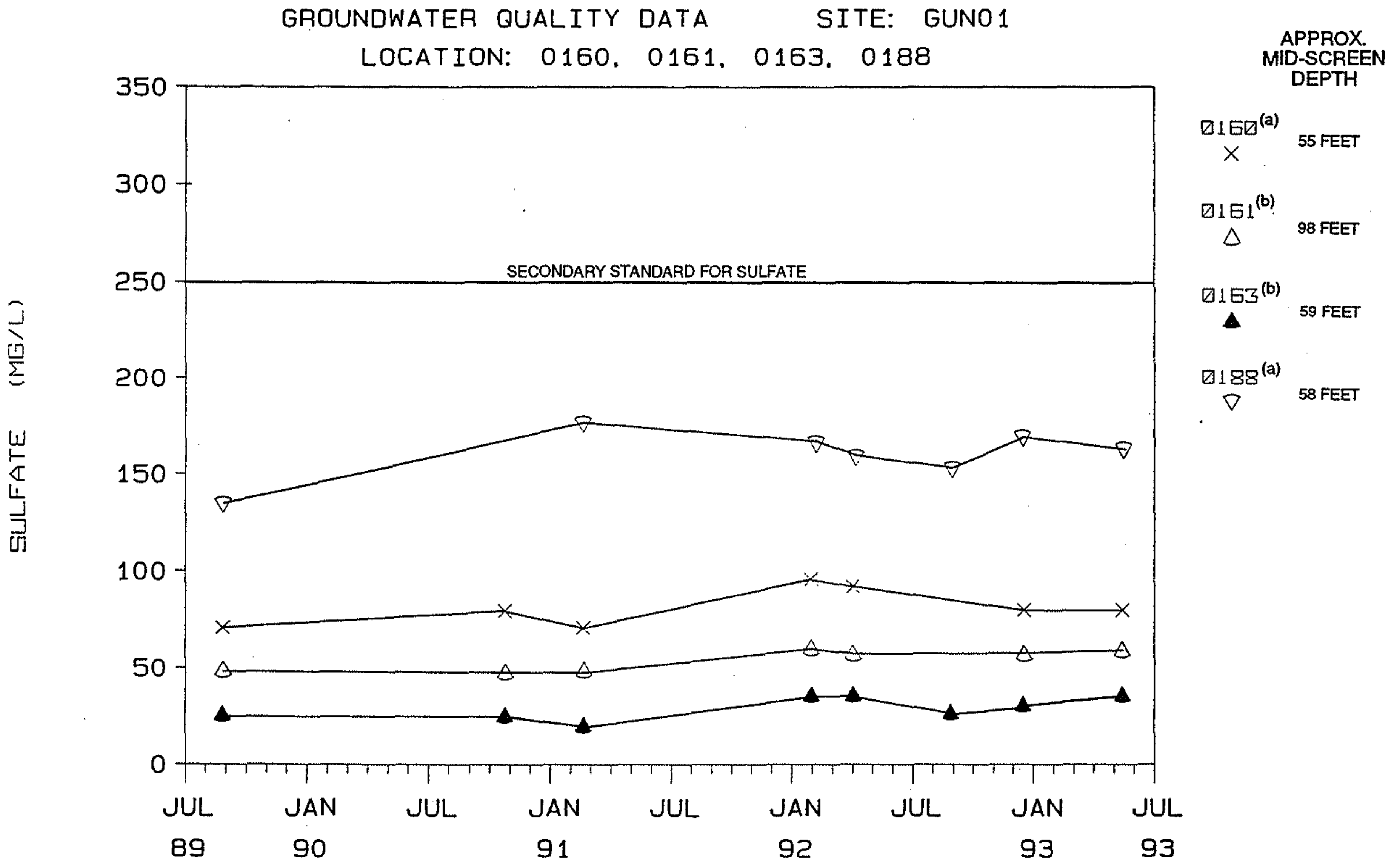
(a) POSSIBLE PLUME-AFFECTED MONITOR WELL

(b) PROBABLE UNAFFECTED MONITOR WELL

FIGURE 3.3
URANIUM CONCENTRATIONS VS. TIME IN MONITOR WELL PAIR (160, 161) AND MONITOR WELLS 163 AND 188 GUNNISON, COLORADO, SITE

PATH I:\FL3.01\GUNBRA

URA33A.DRW



NOTE: SUSPECTED ERRONEOUS SULFATE DATA COLLECTED AUGUST 1992
FOR MONITOR WELLS 160 AND 161 NOT USED IN PLOT.

(a) POSSIBLE PLUME-AFFECTED MONITOR WELL
(b) PROBABLE UNAFFECTED MONITOR WELL

FIGURE 3.4
SULFATE CONCENTRATIONS VS. TIME IN MONITOR WELL PAIR (160, 161)
AND MONITOR WELLS 163 AND 188
GUNNISON, COLORADO, SITE

PATH I:\FL3.01\GUNBRA

URA34A.DRW

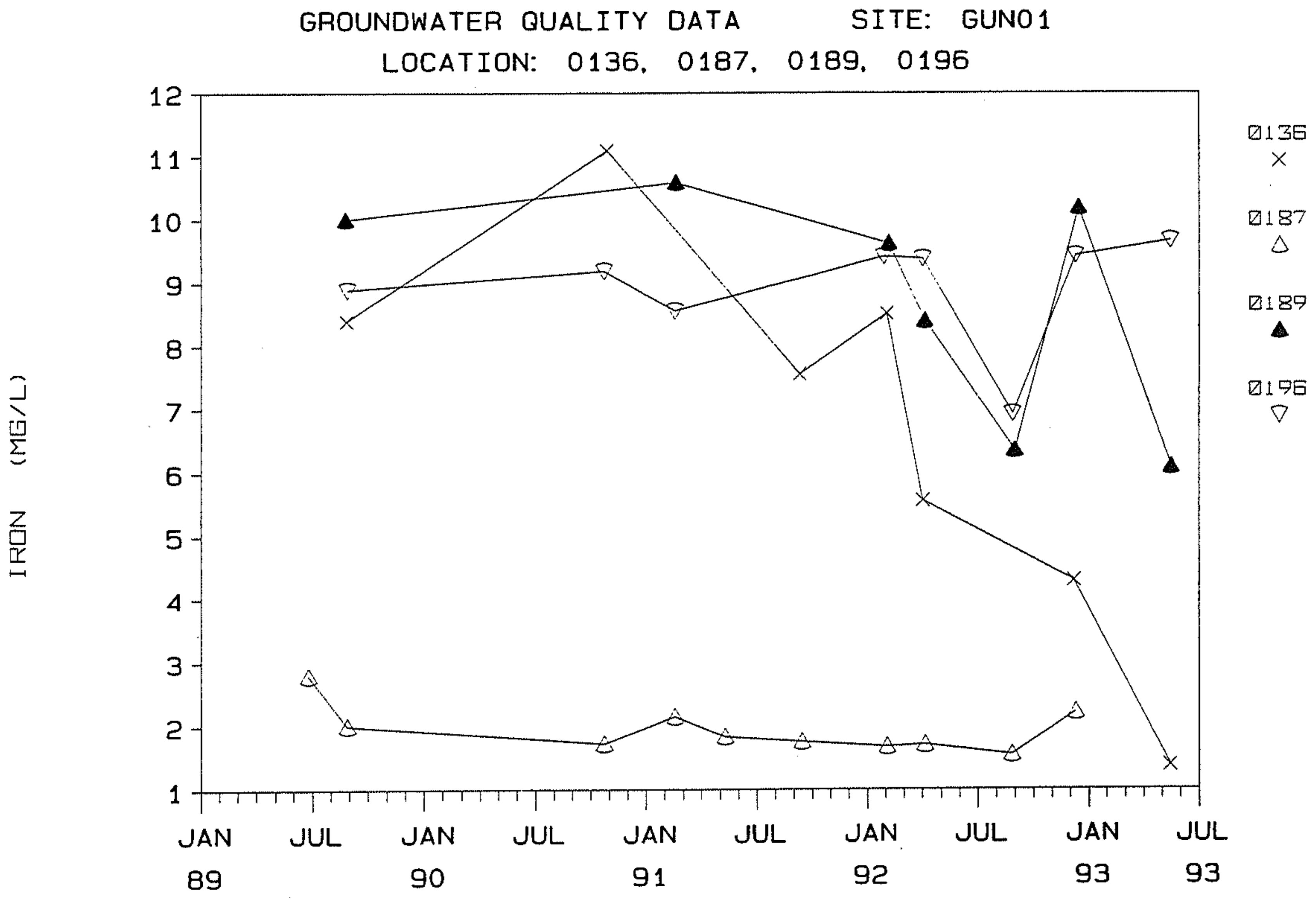


FIGURE 3.5
IRON CONCENTRATIONS VS. TIME IN SELECT DOWNGRADIENT MONITOR WELLS AT THE GUNNISON, COLORADO, SITE

PATH I:\FL3.01\GUNBRA

TIME35.DRW

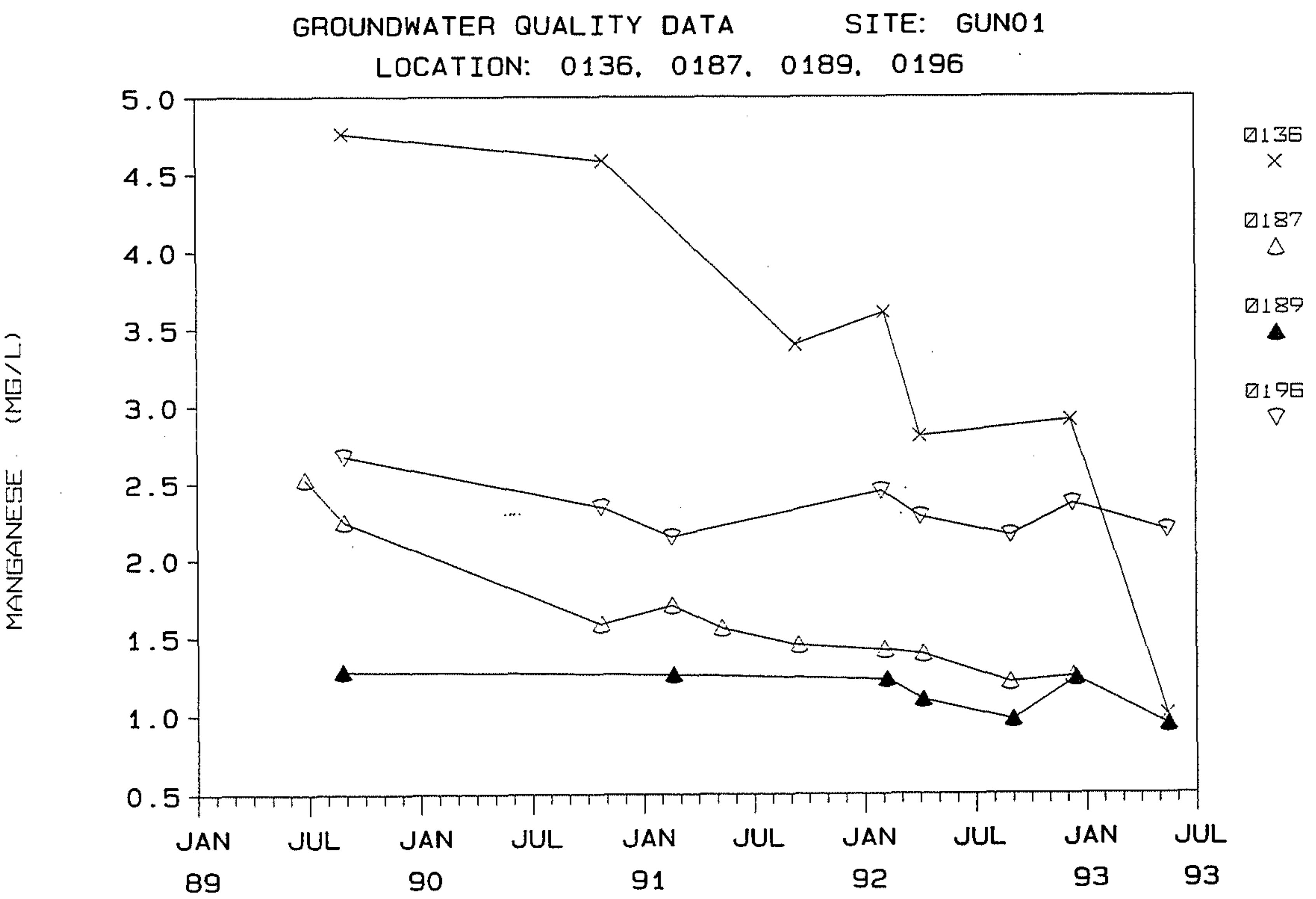


FIGURE 3.6
MANGANESE CONCENTRATIONS VS. TIME IN SELECT DOWNGRADIENT MONITOR WELLS AT THE GUNNISON, COLORADO, SITE

PATH I:\FL3.01\GUNBRA

TIME36.DRW

Table 3.2 Constituents of potential concern for human health evaluation

Constituent levels that exceed background	Constituent levels in nutritional range	Constituents of low toxicity and/or high dietary range	Constituents of potential concern
Ammonium		Ammonium	
Bromide		Bromide	
Calcium	Calcium		
Cadmium			Cadmium
Cobalt			Cobalt
Iron			Iron
Lead-210			Lead-210
Magnesium	Magnesium		
Manganese			Manganese
Nickel		Nickel	
Polonium-210			Polonium-210
Potassium	Potassium		
Silica		Silica	
Sodium	Sodium		
Strontium		Strontium	
Sulfate			Sulfate
Thorium-230			Thorium-230
Uranium			Uranium
Zinc	Zinc		

Note: Constituents of potential concern (column 4) are determined by eliminating the constituents listed in columns 2 and 3 from the list in column 1.

in downgradient monitor wells than were recorded in background wells (e.g., antimony, chloride, and radium-228). However, for these constituents the differences between background and downgradient data were not large enough or consistent enough to reach statistical significance.

A solvent extraction process, which used organic chemicals to recover uranium from the pregnant solution, was used at the site. Therefore, ground water was screened for the organic constituents listed in 40 CFR Part 264, Appendix IX, in 1988. No organic contamination, represented by the Appendix IX analyte list, was found at the site.

The constituents identified in column 1 of Table 3.2 were screened for their impact on human health using the criteria discussed below to develop a final list of constituents of potential concern for human health (DOE, 1996). Because environmental effects differ from effects on human health, the column 1 list of constituents (except bromide) is evaluated in the ecological assessment presented in Section 7.0. Several constituents that were detected above background were deleted from the final list of constituents of potential concern for human health because they are essential nutrients and the levels at which they are detected are within nutritional ranges. These chemicals include calcium, magnesium, potassium, sodium, and zinc.

Final screening of the remaining constituents was based on the very low toxicity and relatively high normal dietary intake compared to the values detected. These criteria were used to screen out ammonium, bromide, nickel, silica, and strontium.

After screening based on the criteria described above, the constituents remaining constituents of potential concern were cadmium, cobalt, iron, lead-210, manganese, polonium-210, sulfate, thorium-230, and uranium. These constituents form the basis of the human health portion of the risk assessment for Gunnison.

3.4 CONTAMINANT FATE AND TRANSPORT

There is evidence that the ground water in the alluvial aquifer at the Gunnison site is chemically stratified as a function of depth. Plots of constituents such as pH and alkalinity show distinct concentration differences as a function of depth in some monitor well clusters or pairs (e.g., background monitor wells 001/101 and far downgradient monitor wells 188/189) (Figures 3.7, 3.8, 3.9, and 3.10). These variations in water chemistry with depth are unlikely to be caused by plume effects alone and may be due, in part, to differences in ground water residence time and/or chemical or lithologic variations (e.g., more organic material and/or clay) in the aquifer matrix at various depths. Lithologic logs of drill holes at the Gunnison site confirm that the alluvial aquifer becomes more clayey with depth.

As plume constituents (e.g., uranium, sulfate, manganese) migrate through the aquifer, they interact with the natural, depth-related chemical variations present in the ground water at this site. These interactions may produce variations in constituent concentrations with depth that are not simply the result of physical dispersion. These and other chemical controls of specific constituents of potential concern for the Gunnison site are discussed below.

Sulfate

Sulfate concentrations in the highly contaminated parts of the plume immediately adjacent to the tailings are controlled primarily by gypsum solubility. Farther downgradient, sulfate concentrations are controlled largely by physical processes such as dispersion and dilution.

Uranium

Uranium is relatively mobile in oxidizing, alkaline ground water over a range of elevated pH values (e.g., between 6.5 and 8.5). Under these conditions, uranium can form stable carbonate complexes that facilitate transport in ground water. Higher ground water pHs tend to decrease the sorption of these uranium species by aquifer materials even further. These processes result in a uranium plume at the Gunnison site.

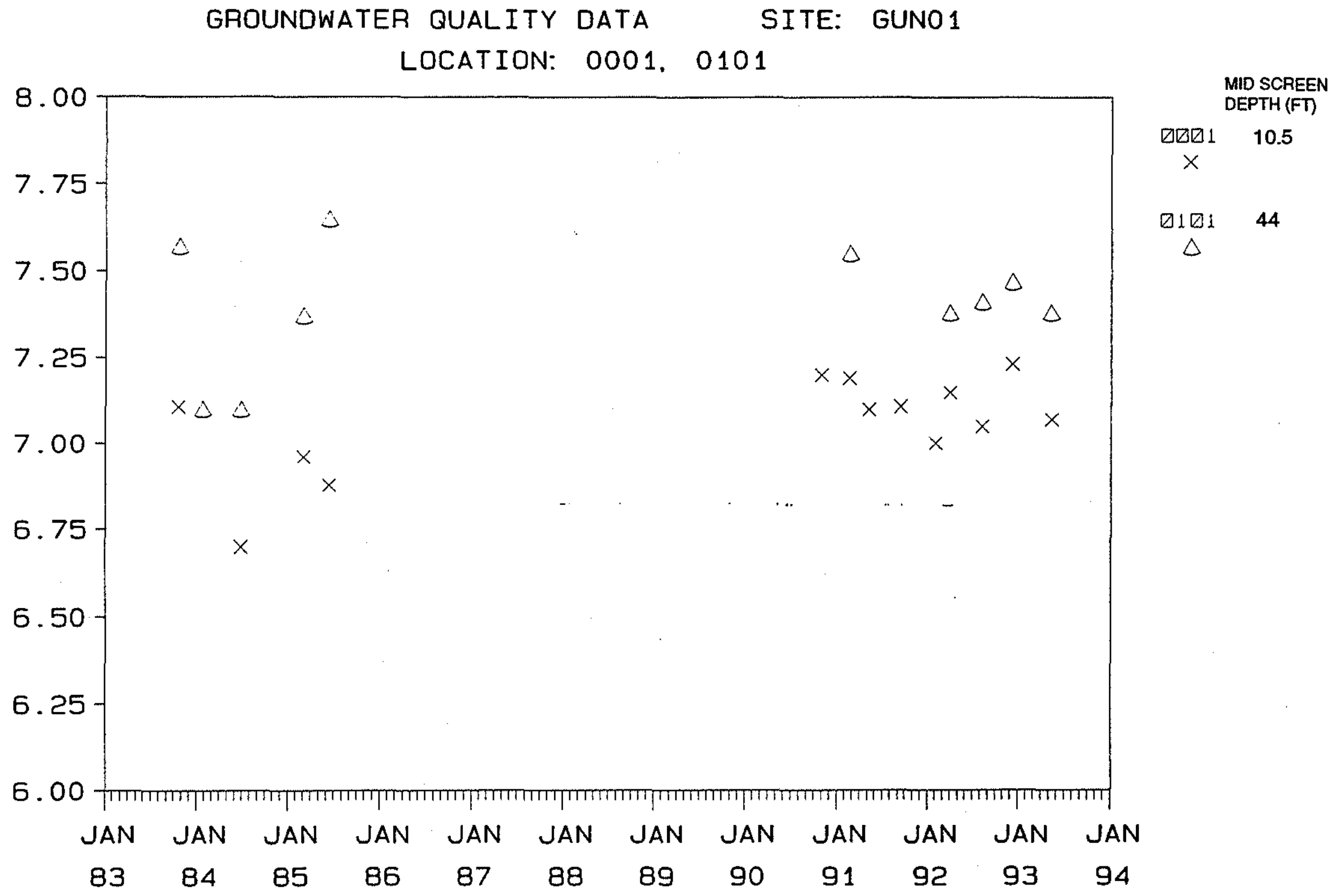


FIGURE 3.7
pH VS. TIME IN PAIRED BACKGROUND MONITOR WELLS 001 AND 101
GUNNISON, COLORADO, SITE

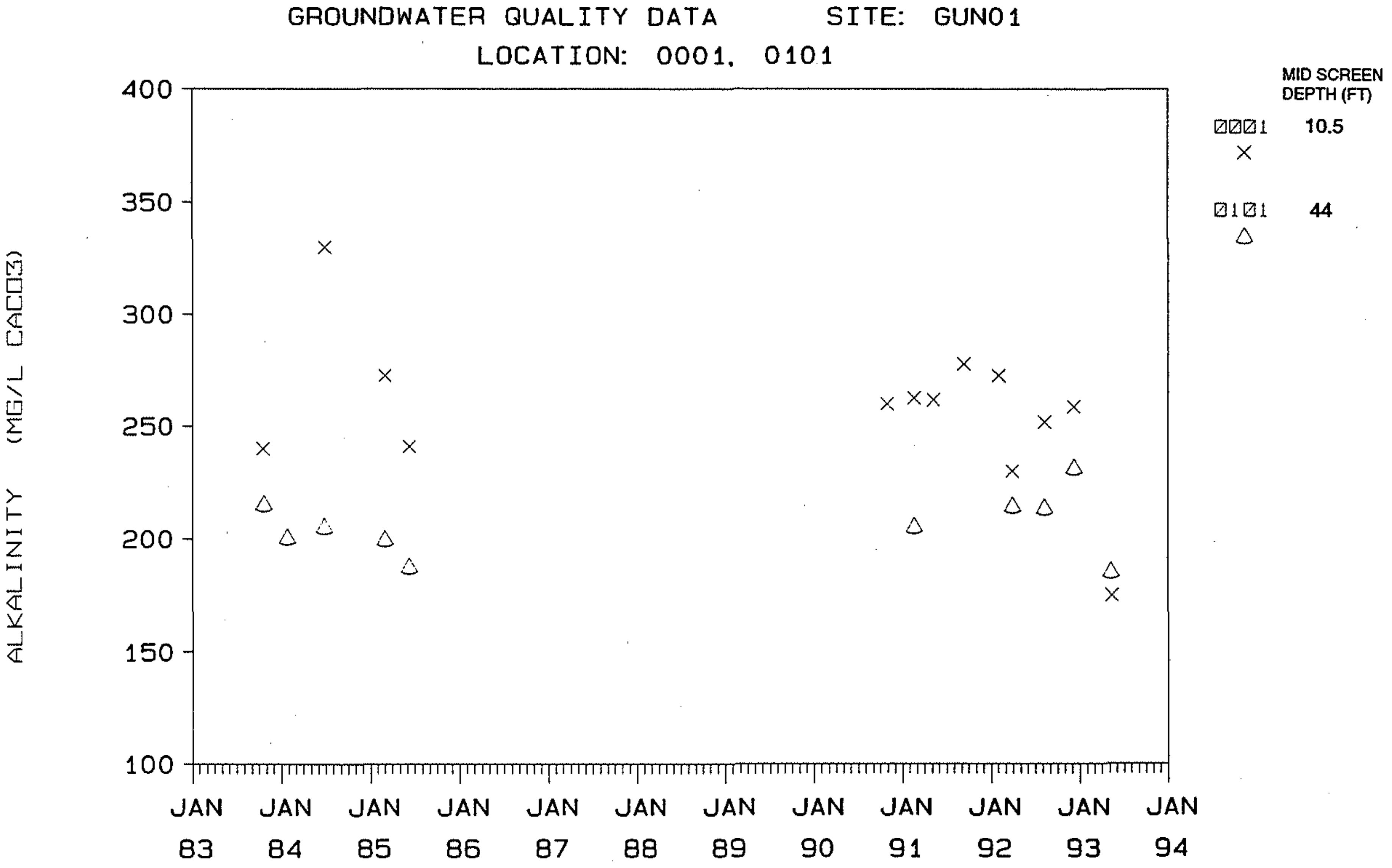


FIGURE 3.8
ALKALINITY VS. TIME IN PAIRED BACKGROUND MONITOR WELLS 001 AND 101
GUNNISON, COLORADO, SITE

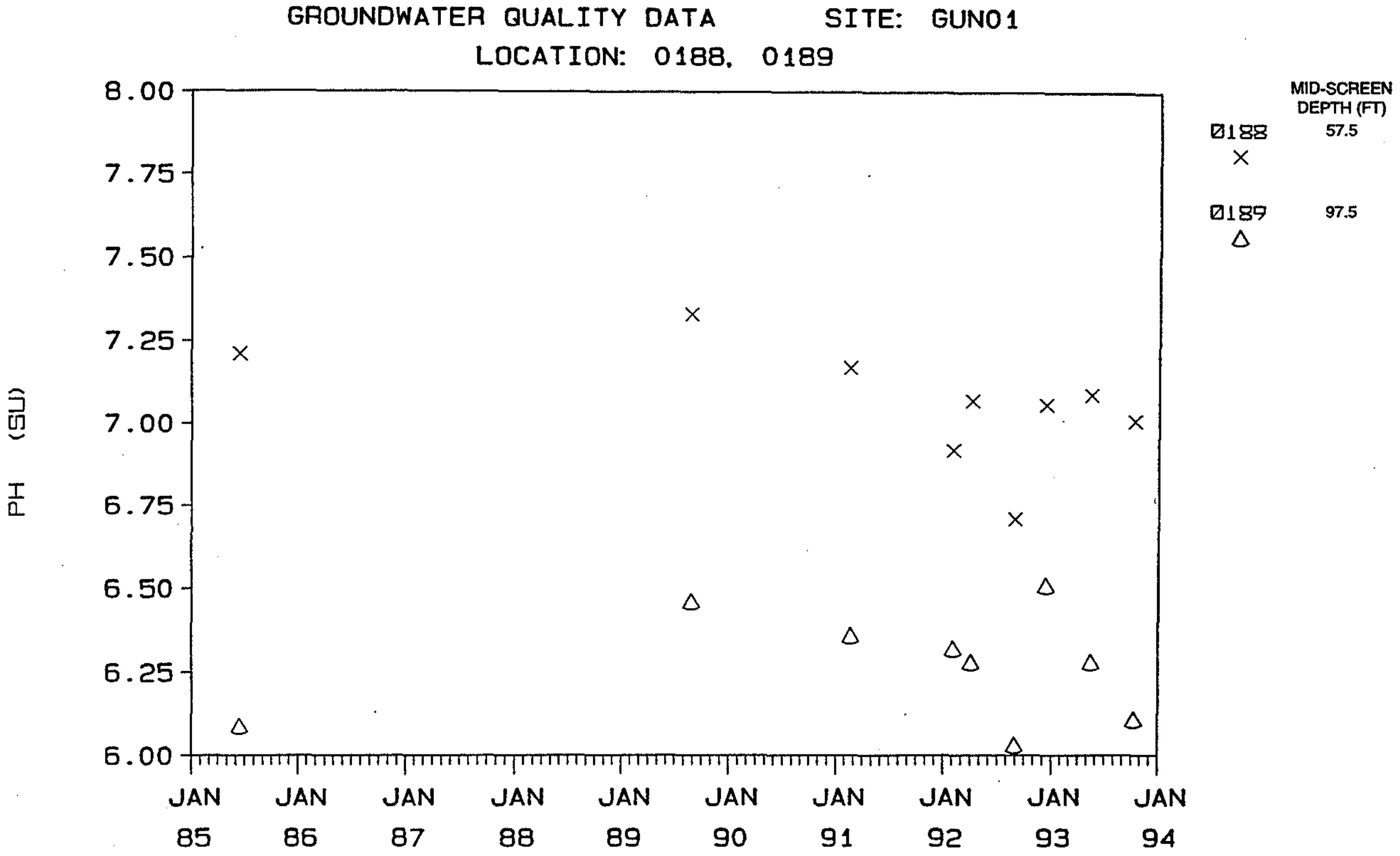


FIGURE 3.9
pH VS. TIME IN PAIRED FAR DOWNGRADIENT MONITOR WELLS 188 AND 189
GUNNISON, COLORADO, SITE

MAC: SITE/GUN/BRA/PHVSTIME(PASTEUP)

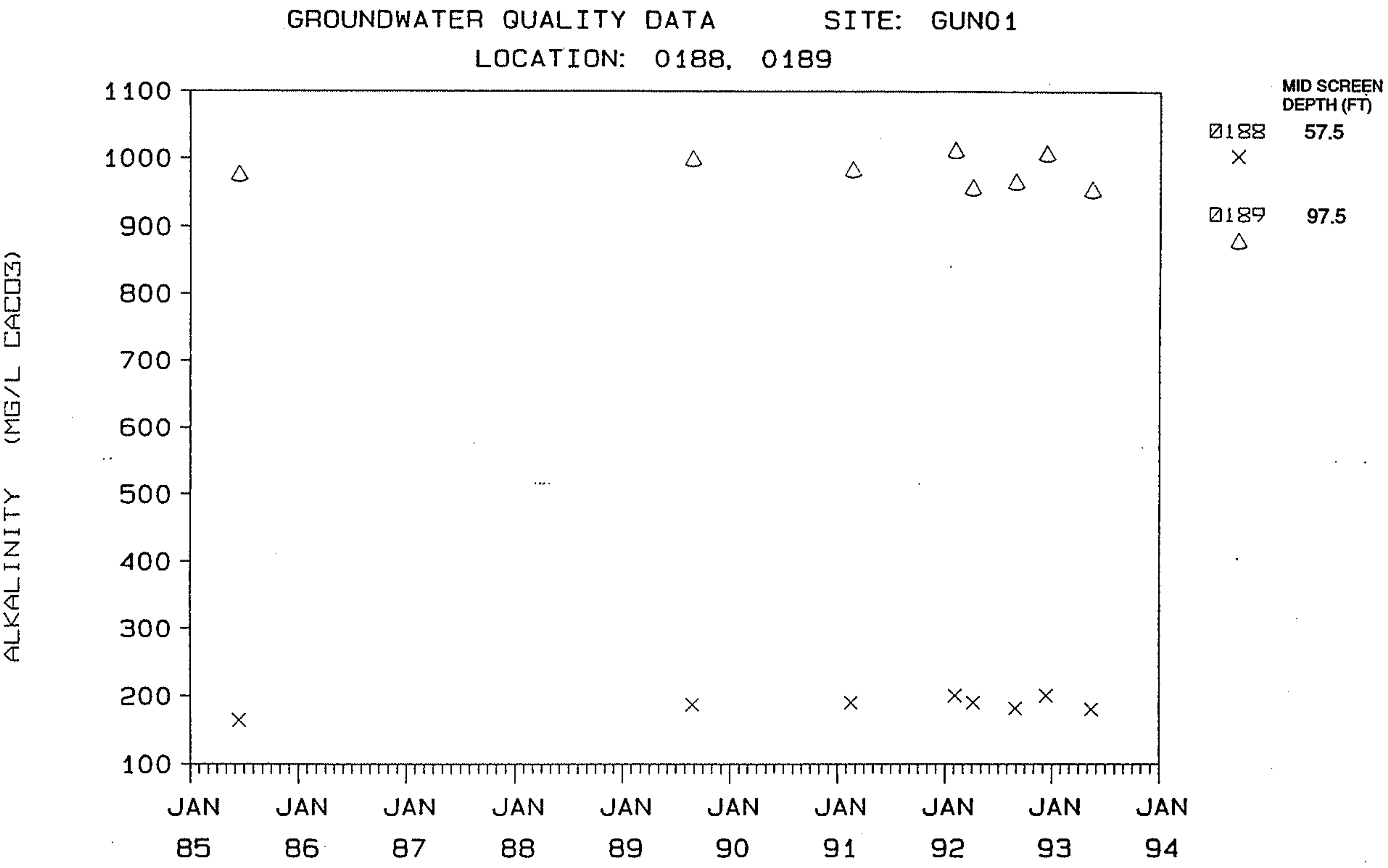


FIGURE 3.10
ALKALINITY VS. TIME IN PAIRED FAR DOWNGRADIENT MONITOR WELLS 188 AND 189
GUNNISON, COLORADO, SITE

Iron and manganese

Relatively high concentrations of iron (up to about 100 mg/L) are found in ground water sampled from monitor wells completed beneath or immediately adjacent to the former tailings (e.g., monitor wells 006, 007, and 009) (Figure 3.11). Within this area, iron concentrations generally are greatest nearest the surface (between 10 and 20 ft [3 and 6 m]), and decrease with increasing depth (between 30 and 60 ft [9 and 20 m]). High iron levels adjacent to the former tailings and ore storage area suggest the acidic tailings solutions are the source for the iron. The decrease in iron with increasing depth is not reflected by similar decreases in sulfate and manganese concentrations and cannot, therefore, be attributed to the dilution of tailings-contaminated ground water at these different depths. Instead, the decease likely reflects precipitation of iron oxyhydroxides in response to increasing pH. This precipitation is due to the dissolved iron, not stable in ground water, that is oxidizing and that has a pH above 6. Thus, in general, site-related contaminated ground water having concentrations of dissolved iron above background levels (ranging up to 5.6 mg/L) is limited to an area within the former processing site boundary. These levels will decrease with time due to continued precipitation of oxyhydroxides.

Very high manganese concentrations have also been found in ground water sampled along the southern margin of the tailings pile (about 80 mg/L in monitor wells 106 and 111). Aqueous species of Mn^{2+} are more stable than Fe^{2+} species under relatively oxidizing and alkaline conditions, and manganese should be more mobile than iron in the Gunnison alluvial aquifer. Elevated levels of manganese are found considerably deeper in the aquifer under and adjacent to the tailings pile than is iron. Nevertheless, the adsorption of manganese species onto the aquifer matrix and the precipitation of the mineral rhodochrosite ($MnCO_3$) should be important factors in eventually reducing manganese levels to near background levels in the Gunnison alluvial aquifer. The rate at which a combination of adsorption and precipitation processes will attenuate manganese at this site has not been quantified.

Water samples from some far downgradient monitor wells (e.g., 136, 187, 189, and 196) have shown iron concentrations up to 12 mg/L and manganese levels that recently were as high as 5 mg/L (Figures 3.5 and 3.6). Unlike sulfate and uranium, these iron and manganese occurrences are scattered throughout the area and do not define a contamination plume. Some of the highest iron and manganese concentrations occur in wells located far from the site while some of the lowest (less than 0.03 mg/L) occur in wells adjacent to the site. In addition, some of these high iron and manganese concentrations are not associated with elevated uranium or sulfate levels. Thus, these occurrences do not appear to be site related.

The occurrences of elevated iron and manganese concentrations in wells located beyond the site boundaries are all related to relatively high concentrations of dissolved organic carbon (DOC) and elevated alkalinity. For example, in well 196 the alkalinity is 1132 mg/L and the DOC is 112 mg/L. These values are considerably greater than any values measured directly beneath the former tailings pile where the maximum DOC value was 26 mg/L and the maximum alkalinity was 306 mg/L. This suggests the high concentrations of DOC and alkalinity are not related to the former tailings pile.

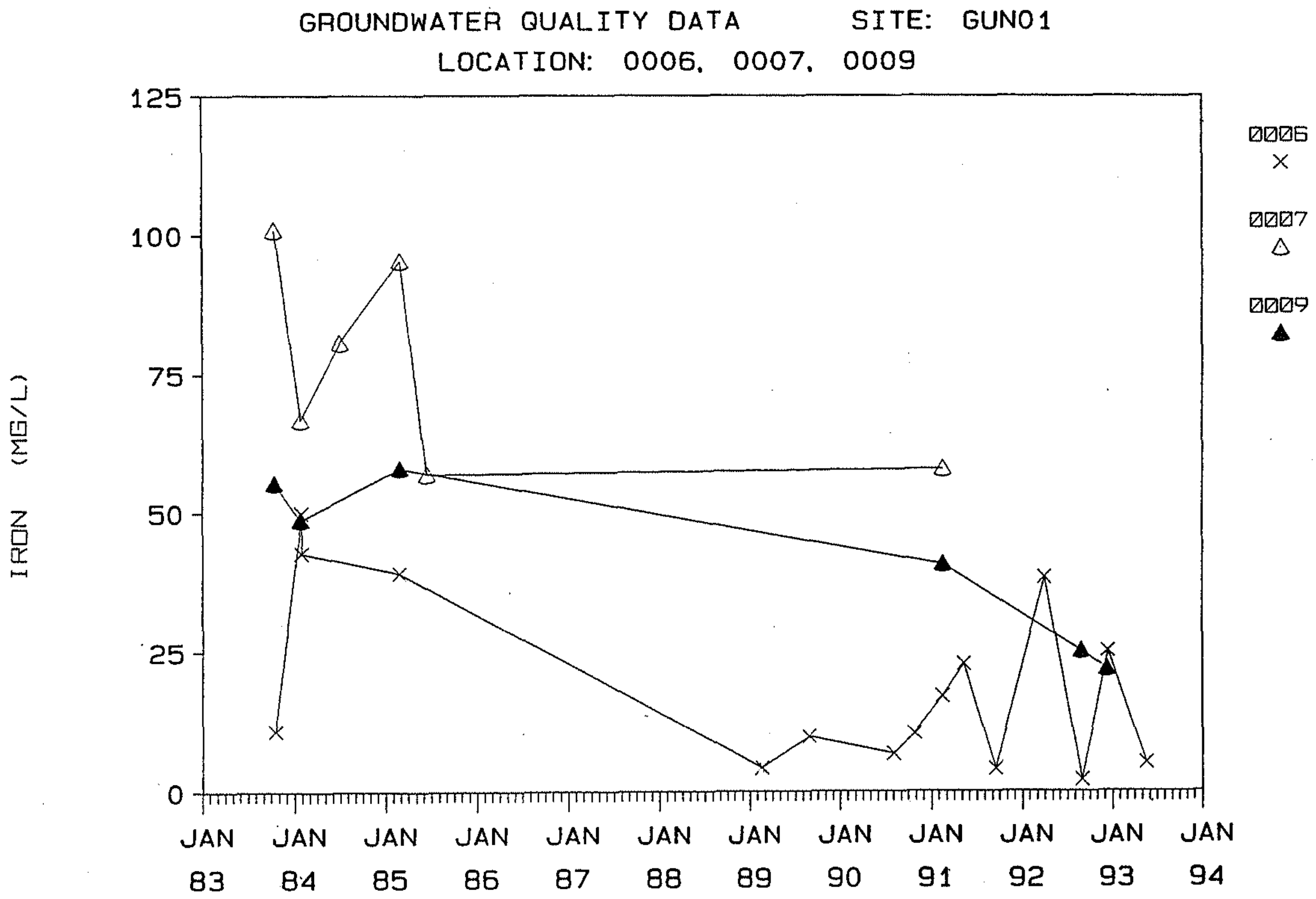


FIGURE 3.11
IRON CONCENTRATIONS VS. TIME FOR MONITOR WELLS 006, 007, AND 009
GUNNISON, COLORADO, SITE

The association of high levels of iron and manganese with elevated DOC and alkalinity is consistent with naturally reducing conditions caused by bacterial degradation of organic matter in the alluvial sediments. The elevated DOC indicates organic matter is present. The high alkalinity indicates bacterial metabolism of the organic matter. The presence of dissolved iron and manganese in association with the high DOC and alkalinity indicates the bacterial processes are producing locally reducing conditions. By contrast, elevated iron and manganese concentrations directly beneath the site are not associated with the high DOC and alkalinity. This is consistent with the acidic tailings solution as the source of these constituents in the immediate site vicinity rather than naturally occurring bacterial reduction.

Cobalt

Elevated cobalt levels have been introduced to the alluvial aquifer by acidic tailings leachate. However, because of its chemical similarity to iron and manganese, this constituent is typically strongly sequestered by the precipitation of manganese and iron hydroxides. At the typical pH range (6.0 to 8.0) observed for the ground water in the alluvial aquifer at Gunnison, cobalt should be present at levels at or below detection limits except in the area immediately adjacent to the tailings (where the aquifer is continually receiving tailings effluent).

Cadmium

Cadmium will be rapidly removed by the precipitation of the mineral octavite ($CdCO_3$) and by hydrolysis and adsorption reactions as the low-pH tailings leachate is neutralized by alkaline ground water and the calcite in the aquifer matrix. Dilution with background water should produce cadmium concentrations in downgradient ground water that are typically below detection limits, while detectable levels of cadmium should be restricted to the areas underneath or immediately adjacent to the tailings pile.

Lead-210

The fate and transport processes that affect lead-210 are those that affect lead concentrations in general. The following discussion of lead geochemistry was derived primarily from Rai and Zachara (1984). The dominant aqueous species of lead will be Pb^{2+} in acidic environments and Pb^{2+}-carbonate complexes in alkaline environments. The mineral cerrusite ($PbCO_3$) is a major control on lead concentrations in alkaline, carbonate-rich systems such as the alluvial aquifer at Gunnison. Iron and manganese hydroxides are strong adsorbents of lead and should also be major controls on lead solubility at Gunnison. Given these strong controls on lead solubility and transport in alkaline systems, lead-210, if introduced into the ground water system, should be rapidly removed from solution at Gunnison and stabilized in the aquifer matrix.

Polonium-210

Polonium-210 is produced by the beta decay of lead-210 (half-life equals 22 years) through the intermediate short-lived daughter bismuth-210 (half-life equals 5.02

days). Polonium-210 has a half-life of about 140 days, and it decays to stable lead-206. Because lead-210 has a much longer half-life than bismuth-210 or polonium-210, the distribution of polonium-210 in this aquifer will be controlled primarily by the distribution of lead-210.

Thorium-230

Thorium is moderately soluble and mobile in sulfate-rich, low-pH (up to a pH of about 4.5 to 5.0) aqueous solutions. However, once the pH of a thorium-bearing acidic solution rises to about 5.0, hydrolysis and precipitation of thorium occurs rapidly. Therefore, except in the immediate subpile region of the Gunnison tailings pile, thorium-230 levels should approach background levels.

3.5 SURFACE WATER MONITORING

The locations of five surface water sampling points are shown in Figure 3.12. Two sampling points are in the Gunnison River, one upstream (location 775) and one downstream (location 776) of the site. Tomichi Creek also has been sampled upstream (location 778) and downstream (location 777) of the site. The fifth sampling point is the pond (location 779), used for pay fishing at the campground. Filtered surface water samples were collected from each location in August 1989. In October 1990, unfiltered samples were collected from all the locations, except downstream of the site in Tomichi Creek. In May 1993, unfiltered samples were collected from each location (DOE, 1996).

Surface water data from the Gunnison River and Tomichi Creek show that most constituent concentrations detected at the downstream locations were not greater than their respective background (upstream) concentrations. Of the ground water constituents identified as exceeding background ground water quality values (Table 3.2), only silica was detected in Gunnison River water at a concentration above the background river water concentration. However, this is not considered significant because the downstream concentration was only 2 percent higher than the upstream concentration, and silica is not considered a site-related constituent. Three constituents from this list (calcium, magnesium, and sulfate) were detected in Tomichi Creek at concentrations slightly higher than at the background location. No statistical analysis can be conducted to determine whether these constituent concentrations are statistically above background concentrations because of the limited amount of surface water quality data. However, the downstream concentrations of calcium and magnesium are only slightly higher (approximately 8 percent) than the upstream concentrations, while the sulfate concentration is approximately 30 percent higher than the upstream concentration. Two constituents (iron and zinc) were detected in water from the campground pond at concentrations above background in the Gunnison River and Tomichi Creek. However, it is not known whether these concentrations represent a statistically significant elevation above background levels.

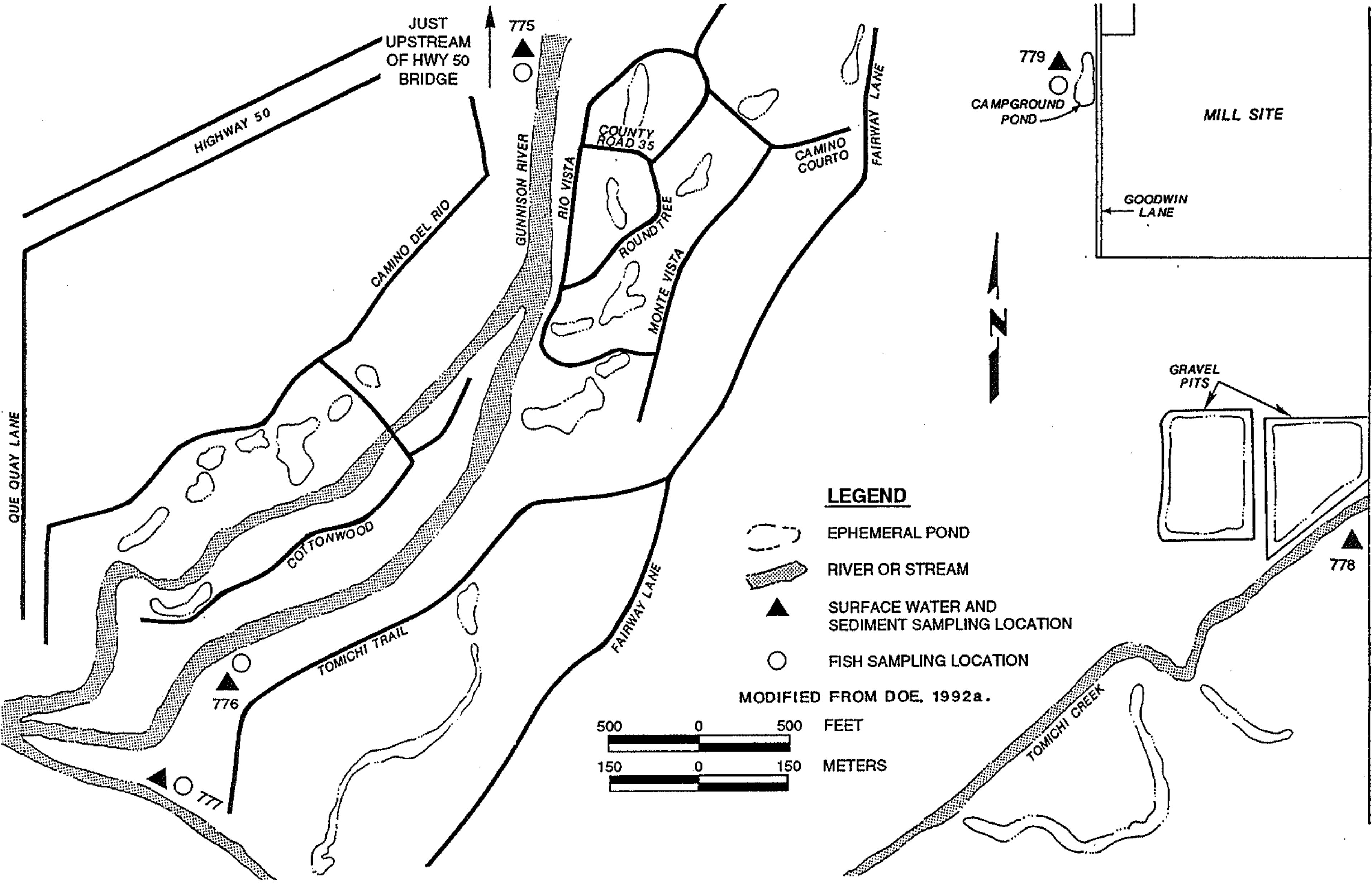


FIGURE 3.12
SURFACE WATER, SEDIMENT, AND FISH SAMPLING LOCATIONS
GUNNISON PROCESSING SITE VICINITY, GUNNISON, COLORADO

4.0 EXPOSURE ASSESSMENT

4.1 POTENTIALLY EXPOSED POPULATION

Exposure can occur only if there is both a source of contamination and a mechanism of transport to a receptor population or individual. Water supplied by a new water supply system is available to all residents downgradient of the site who have domestic wells potentially affected by the plume of contaminated ground water. Therefore, it is unlikely that water from these wells would be used for drinking, although these wells could be used to water livestock or irrigate garden plants. Using the livestock for food could create a potential exposure pathway to humans. Plants with roots in the alluvial aquifer or irrigated with water from this zone could take up and concentrate constituents, forming a potential pathway to humans through plant consumption. In the future, a domestic well could be constructed in the most contaminated portion of the alluvial aquifer, creating an exposure pathway through drinking and bathing water.

A future ground water use scenario at the former processing site is assumed for this risk assessment. This scenario evaluates domestic ground water use consistent with current use by the rural population in the region. The potentially exposed population includes individuals of the following age groups: infants (birth to 1 year old), children (1 to 10 years old), and adults (11 to 64 years old). These age groups were selected for the following reasons:

- Availability of survey data for population variables such as age, weight, and daily water intake.

- Similarity of toxicological variables, including responsiveness of sensitive subgroups (infants and children) to the constituents of potential concern, toxicant intake-to-body-weight ratios, and toxicokinetics (a study of the time course of absorption, distribution, metabolism, and excretion of a constituent in an individual's body).

Some individuals and/or subpopulations could be more vulnerable to exposure than the general population. These sensitive subpopulations could include infants, children, the elderly, or people with preexisting illness, such as diabetes or kidney insufficiency with the absence of diabetes.

4.2 EXPOSURE PATHWAYS

An exposure pathway describes the course a constituent takes from its source to the exposed individual or population. Therefore, the exposure pathway can be completed only if there is a source of contamination, a point of contact with a population or individual, and the way a constituent enters the body after contact (e.g., water ingestion). The tailings pile and contaminated soils and materials at the site were removed and relocated to a disposal cell about 6 mi (10 km) east of the town of Gunnison. Thus, soil or air exposure pathways (such as incidental soil ingestion, dermal contact with soil, or inhalation of particulates) are not considered here. This assessment evaluates both ground water and surface water/sediment pathways.

Ground water in the region is primarily used for household purposes such as drinking, cooking, and bathing. Other uses typical of the region that could indirectly lead to human exposure include irrigation and livestock watering. Figure 4.1 provides a conceptual model for potential ground water exposure pathways that could result from these uses and are believed to lead to the greatest potential exposures at the Gunnison site.

4.2.1 Drinking water ingestion

Although it is unlikely that ground water at the site will be used in the future for drinking purposes because of the availability of a new water supply system, this risk assessment will conservatively evaluate a hypothetical future use of the ground water from the most contaminated portion of the alluvial aquifer for drinking.

Drinking water ingestion is generally the dominant exposure pathway for ground water contaminated with metals and other nonvolatile compounds. For this evaluation, drinking water consumption includes water consumed for drinking, as well as water used for food preparation (e.g., reconstituted juices, soup, rice, and beans). For comparison of relative pathway significance, a screening level assessment of drinking water intake is shown in Table 4.1. These calculations are based on estimates of the maximum concentrations of constituents within the plume (i.e., the maximum detected concentration from the most contaminated plume wells). Therefore, the exposure estimates likely overestimate potential exposures.

4.2.2 Dermal absorption

Dermal absorption is the process by which chemicals coming into contact with the skin are absorbed into the blood vessels near the surface of the skin. Some compounds are absorbed easily in this manner, though metals do not possess the chemical properties that are conducive to skin absorption.

To evaluate this exposure route, a screening calculation was performed to determine if a dermal absorption pathway would be notable compared to the drinking water pathway for the constituents of potential concern (Table 4.1). Since chemical-specific absorption factors are not available for these constituents, they are assumed to absorb across the skin at the same rate as water. This assumption will probably overestimate any potential contribution from dermal absorption. Additionally, the concentration in water was assumed to be the maximum detected concentration from the most contaminated plume wells, which also will overestimate exposure.

Based on these results, dermal absorption was eliminated from more detailed quantitative evaluation because it contributed to less than 1 percent of the total intake from drinking water for all constituents.

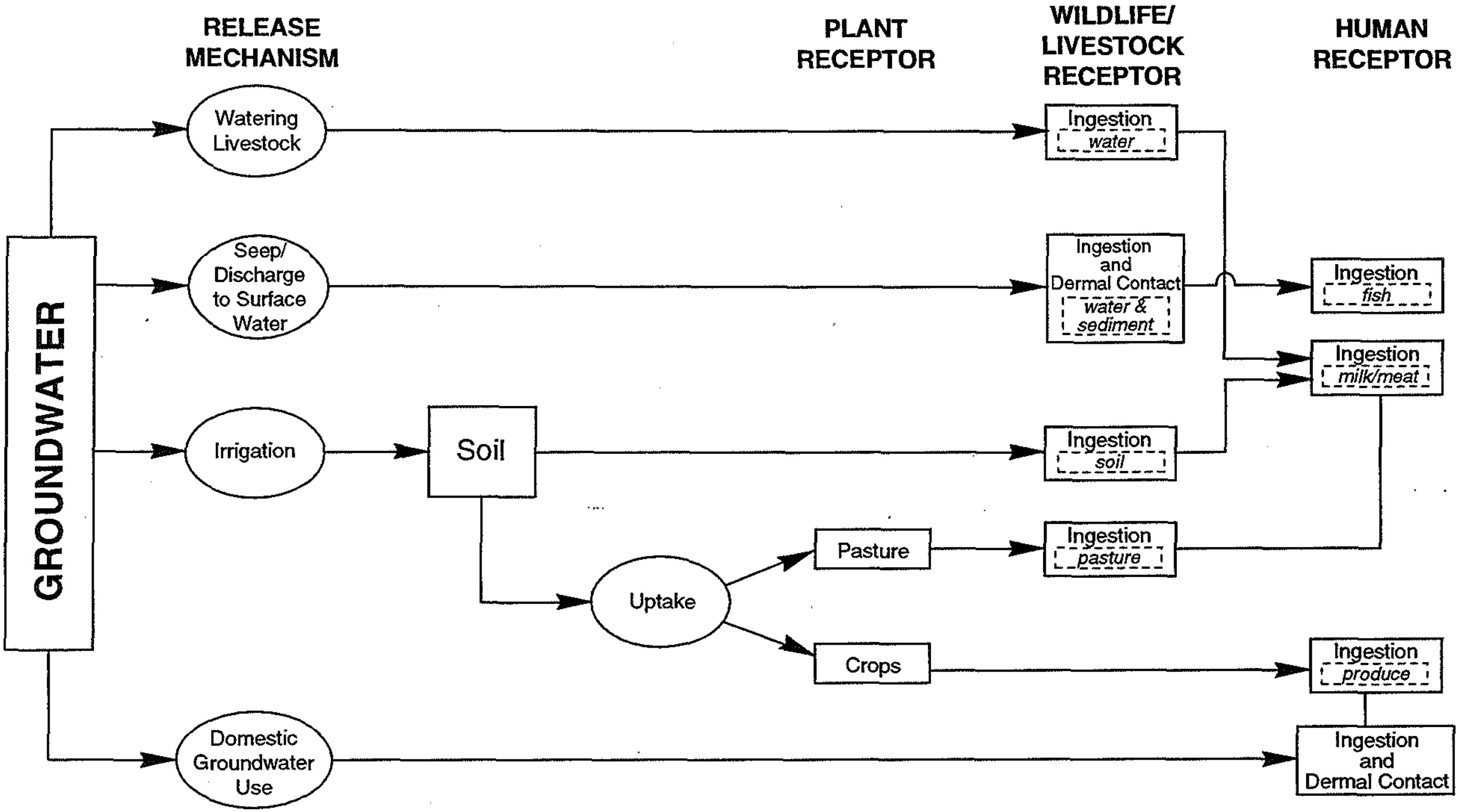


FIGURE 4.1
GUNNISON CONCEPTUAL MODEL

PREPARATION DATE: 9/13/93

MAC: SITE/GUN/BRA/CONCEPTMOD

Table 4.1 Exposure dose calculations for ground water ingestion and dermal contact, Gunnison, Colorado, site

Constituent of potential concern	Cw (mg/L)	Ground water exposure doses (mg/kg-day) Ingestion	Dermal contact	Ratio of dermal: ingestion[a]
Noncarcinogenic effects				
Cadmium	0.004	1E-04	2E-07	0.002
Cobalt	0.37	1E-02	2E-05	0.002
Iron	91	3E+00	5E-03	0.002
Manganese	39	1E+00	2E-03	0.002
Sulfate	1590	4E+01	9E-02	0.002
Uranium	1.6	4E-02	9E-05	0.002
Carcinogenic effects	(pCi/L)			
Lead-210	78	2E+06[b]	3E+03[b]	0.002
Polonium-210	1.6	3E+04[b]	7E+01[b]	0.002
Thorium-230	2.0	4E+04[b]	8E+01[b]	0.002
Uranium[c]	1100	2E+07[b]	5E+04[b]	0.002

Equation definitions for exposure dose calculations

Ingestion of ground water

Chemicals

Chronic daily intake (mg/kg-day) $= \dfrac{Cw \times IRw \times EF \times ED}{BW \times AT}$

Radionuclides

Lifetime intake (pCi/lifetime) $= Cw \times IRw \times EF \times ED$

Dermal contact with ground water

Chemicals

Chronic daily intake (mg/kg-day) $= \dfrac{(Cw \times SA \times Pc \times Cf) \times ET \times EF \times ED}{BW \times AT}$

Radionuclides

Lifetime intake (pCi/lifetime) $= (Cw \times SA \times Pc \times Cf) \times ET \times EF \times ED$

Table 4.1 Exposure dose calculations for ground water ingestion and dermal contact, Gunnison, Colorado, site (Concluded)

Where:
Cw = Constituent concentration in ground water (maximum concentration detected) (mg/L or pCi/L).
IRw = Ingestion rate for water (L/day) (2 L/day for an adult).
EF = Exposure frequency (350 days/year).
ED = Exposure duration (30 years for an adult).
BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days x ED for noncarcinogens).
SA = Skin surface area (19,400 square centimeters [cm^2]).
Pc = Dermal permeability constant (0.001 cm/hour).
Cf = Conversion factor (0.001 L/cm^3).
ET = Exposure time (0.2 hour/day).

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[a] This value is calculated by dividing the dermal contact exposure dose by the ground water ingestion exposure dose.
[b] Picocuries per lifetime.
[c] Uranium-234 and uranium-238 combined.

4.2.3 Ingestion of ground water-irrigated produce

This exposure route was also evaluated for its relative contribution to the drinking water ingestion route. The results of the screening calculation are shown in Table 4.2. The assumptions for this evaluation will probably overestimate the potential for exposure from this route, because it is assumed that this garden would be the source of all garden produce in the diet. The results of this screening show that for the constituents of potential concern at this site, ingesting garden vegetables and fruit irrigated with contaminated ground water would lead to potential exposures of 2 percent or less of that associated with drinking water ingestion. Thus, this pathway is eliminated from further quantitative evaluation, although the impact of this additional source will be discussed in Sections 6.1 and 6.2.

4.2.4 Ingestion of milk or meat from ground water-fed livestock

The relative contribution from ingesting milk from ground water-fed livestock is 4 percent or less for all constituents (Table 4.3). Constituent concentrations in milk and beef tissue are calculated using literature values for soil-to-plant bioconcentration factors (BCF) and feed-to-milk and feed-to-flesh transfer coefficients, because the site-specific values are not available for these variables. Ingesting meat from these animals would also contribute 4 percent or less of the exposure anticipated from drinking water for all constituents (Table 4.4). The contribution from these sources is not included in the exposure simulations, but the impact of these additional sources is discussed in Sections 6.1 and 6.2.

Table 4.2 Exposure dose calculations for ground water-irrigated garden produce ingestion, Gunnison, Colorado, site

Constituent of potential concern	Cw (mg/L)	Kd (L/kg)	Bv	Br	Garden produce ingestion exposure doses[a] (mg/kg-day)	Ratio of produce ingestion: water ingestion[b]
Noncarcinogenic effects						
Cadmium	0.004	15	0.55	0.15	2E-06	0.02
Cobalt	0.37	1.9	0.02	0.007	8E-07	0.00008
Iron	91	15	0.004	0.001	3E-04	0.0001
Manganese	39	17	0.25	0.05	2E-03	0.0002
Sulfate	1590	0	0.5	0.5	c	NA
Uranium	1.6	1	0.0085	0.004	8E-07	0.00002
Carcinogenic effects	**(pCi/L)**					
Lead-210	78	230	0.045	0.009	3E+04[d]	0.02
Polonium-210	1.6	5.9	0.025	0.0004	8E+00[d]	0.0003
Thorium-230	2.0	100	0.00085	0.000085	6E+00[d]	0.0002
Uranium	1100	1	0.0085	0.004	4E+02[d]	0.00002

Equation definitions for exposure dose calculations

Ingestion of garden produce irrigated with ground water

Chemicals

$$\text{Chronic daily intake (mg/kg-day)} = \frac{Cw \times Kd \times Bv \times \text{or } Br^{e} \times DF \times IRp \times FI \times EF \times ED}{BW \times AT}$$

Radionuclides

$$\text{Chronic daily intake (mg/kg-day)} = Cw \times Kd \times Bv \times \text{or } Br^{e} \times DF \times IRp \times FI \times EF \times ED$$

Where:

Cw = Constituent concentration in ground water (maximum concentration detected) (mg/L or pCi/L).
Kd = Soil-water partition coefficient (L/kg) (PNL, 1989), except for uranium. The Kd for uranium is a site-specific value.
Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless) (Baes et al., 1984).
Br = Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless) (Baes et al., 1984).
DF = Dry weight fraction of plant (0.066 unitless).
IRp = Ingestion rate for garden produce (0.05 kg/day for vegetative parts; 0.03 kg/day for reproductive parts).
FI = Fraction of garden produce ingested from contaminated source (1.0 unitless).
EF = Exposure frequency (350 days/year).

Table 4.2 Exposure dose calculations for ground water-irrigated garden produce ingestion, Gunnison, Colorado, site (Concluded)

ED = Exposure duration (30 years for an adult).
BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days x ED for noncarcinogens).

[a] Exposure doses shown are the sum of the vegetative parts plus the reproductive parts.

[b] This value is calculated by dividing the garden produce ingestion exposure dose by the ground water ingestion exposure dose.

[c] Value cannot be calculated because Kd is equal to zero.

[d] Picocuries per lifetime.

[e] Exposure doses due to vegetative parts and reproductive parts of garden produce are calculated separately, then summed for total intake.

NA - not applicable.

Table 4.3 Exposure dose calculations for ingestion of milk from ground water-fed livestock, Gunnison, Colorado, site

Constituent of potential concern	Cw (mg/L)	Kd (L/kg)	Bv	Fm	Milk ingestion exposure doses (mg/kg-day)	Ratio of milk ingestion: water ingestion[a]
Noncarcinogenic effects						
Cadmium	0.004	15	0.55	0.001	4E-06	0.04
Cobalt	0.37	1.9	0.02	0.002	2E-04	0.02
Iron	91	15	0.004	0.00025	6E-03	0.002
Manganese	39	17	0.25	0.00035	1E-03	0.001
Sulfate	1590	0	0.5	0.005	b	NA
Uranium	1.6	1	0.0085	0.0006	2E-04	0.005
Carcinogenic effects	**(pCi/L)**					
Lead-210	78	230	0.045	0.00025	2E+04[c]	0.01
Polonium-210	1.6	5.9	0.025	0.00035	1E+02[c]	0.003
Thorium-230	2.0	100	0.00085	0.000005	3E+00[c]	0.00008
Uranium	1100	1	0.0085	0.0006	1E+05[c]	0.005

Equation definitions for exposure dose calculations

Ingestion of milk from ground water-fed livestock

Chemicals

$$\text{Chronic daily intake (mg/kg-day)} = \frac{Cm \times IRm \times FI \times EF \times ED}{BW \times AT}$$

Radionuclides

$$\text{Chronic daily intake (mg/kg-day)} = Cm \times IRm \times FI \times EF \times ED$$

Where:

Cm = Constituent concentration in milk (mg/L), estimated using the following equation:
$Cm = Fm \times ([Qp \times Cp] + [Qs \times Cs] + [Qw \times Cw])$
where
Fm = Feed-to-milk transfer coefficient (days/kg) (Baes et al., 1984).
Qp = The quantity of pasture eaten by cattle per day (19 kg/day).
Qs = The quantity of soil eaten by cattle per day (0.38 kg/day).
Qw = The quantity of water consumed by cattle per day (56 L/day).
Cp = Constituent concentration in pasture (mg/kg). $Cp = Kd \times Cw \times Bv$.
Cs = Constituent concentration in soil (mg/kg). $Cs = Kd \times Cw$.
Cw = Constituent concentration in ground water (maximum concentration detected) (mg/L or pCi/L).
Kd = Soil-water partition coefficient (L/kg) (PNL, 1989), except for uranium. The Kd for uranium is a site-specific value.
Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless) (Baes et al., 1984).
EF = Exposure frequency (350 days/year).
ED = Exposure duration (30 years for an adult).

Table 4.3 Exposure dose calculations for ingestion of milk from ground water-fed livestock, Gunnison, Colorado, site (Concluded)

BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days x ED for noncarcinogens).
IRm = Ingestion rate of milk (0.3 kg/day).
FI = Fraction of milk ingested from contaminated source (1.0 unitless).

---

[a] This value is calculated by dividing the milk ingestion exposure dose by the ground water ingestion exposure dose.

[b] Value cannot be calculated because Kd is equal to zero.

[c] Picocuries per lifetime.

NA - not applicable.

Table 4.4 Exposure dose calculations for ingestion of meat from ground water-fed livestock, Gunnison, Colorado, site

Constituent of potential concern	Cw (mg/L)	Kd (L/kg)	Bv	Ff	Meat ingestion exposure doses (mg/kg-day)	Ratio of meat ingestion: water ingestion[a]
Noncarcinogenic effects						
Cadmium	0.004	15	0.55	0.00055	5E-07	0.005
Cobalt	0.37	1.9	0.02	0.02	4E-04	0.04
Iron	91	15	0.004	0.02	1E-01	0.03
Manganese	39	17	0.25	0.0004	4E-04	0.0004
Sulfate	1590	0	0.5	0.005	b	NA
Uranium	1.6	1	0.0085	0.0002	2E-05	0.0005
Carcinogenic effects	(pCi/L)					
Lead-210	78	230	0.045	0.0003	6E+03[c]	0.003
Polonium-210	1.6	5.9	0.025	0.0003	2E+01[c]	0.0007
Thorium-230	2.0	100	0.00085	0.000006	9E-01[c]	0.00002
Uranium	1100	1	0.0085	0.0002	1E+04[c]	0.0005

Equation definitions for exposure dose calculations

Ingestion of meat from ground water-fed livestock

Chemicals

$$\text{Chronic daily intake (mg/kg-day)} = \frac{Cm \times IRb \times FI \times EF \times ED}{BW \times AT}$$

Radionuclides

$$\text{Chronic daily intake (mg/kg-day)} = Cm \times IRb \times FI \times EF \times ED$$

Where:

Cb = Constituent concentration in beef (mg/kg), estimated using the following equation: $Cb = Ff \times ([Qp \times Cp] + [Qs \times Cs] + [Qw \times Cw])$ where

Ff = Feed-to-flesh transfer coefficient (days/kg) (Baes et al., 1984).

Qp = The quantity of pasture eaten by cattle per day (19 kg/day).

Qs = The quantity of soil eaten by cattle per day (0.38 kg/day).

Qw = The quantity of water consumed by cattle per day (56 L/day).

Cp = Constituent concentration in pasture (mg/kg). $Cp = Kd \times Cw \times Bv$.

Cs = Constituent concentration in soil (mg/kg). $Cs = Kd \times Cw$.

Cw = Constituent concentration in ground water (maximum concentration detected) (mg/L or pCi/L).

Kd = Soil-water partition coefficient (L/kg) (PNL, 1989), except for uranium. The Kd for uranium is a site-specific value.

Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless) (Baes et al., 1984).

EF = Exposure frequency (350 days/year).

ED = Exposure duration (30 years for an adult).

Table 4.4 Exposure dose calculations for ingestion of meat from ground water-fed livestock, Gunnison, Colorado, site (Concluded)

BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days x ED for noncarcinogens).
IRb = Ingestion rate of meat (0.075 kg/day).
FI = Fraction of meat ingested from contaminated source (1.0 unitless).

[a]This value is calculated by dividing the meat ingestion exposure dose by the ground water ingestion exposure dose.
[b]Value cannot be calculated because Kd is equal to zero.
[c]Picocuries per lifetime.

NA - not applicable.

4.2.5 Ingestion of fish

Sportfishing occurs in the reaches of the Gunnison River and Tomichi Creek in the site vicinity. Trout fishing also occurs in the stocked campground pond. Ingestion of fish that may have accumulated site-related constituents is a potential exposure pathway. Hook-and-line fishing was conducted at two locations in the Gunnison River (locations 775 and 776), and at one location in both Tomichi Creek (location 777) and the campground pond (location 779) in June 1993 (see Figure 3.12). In the Gunnison River, an approximately 0.75-pound (340-gram) brook trout was caught at location 775 and an approximately 1.25-pound (570-gram) rainbow trout was caught at location 776. One brook trout and one German brown trout, each approximately 0.33-pound (150-gram), were caught in Tomichi Creek at location 777. No fishing was conducted at location 778. One rainbow trout, approximately 1 pound (450-gram), was collected from the campground pond. The filleted muscle tissue was submitted to the laboratory and analyzed for manganese, molybdenum, uranium, and zinc (DOE, 1996). Compiled before this risk assessment was conducted, this list was based on a screening of maximum detected constituent concentrations in ground water. The results from the muscle tissue analyses are presented in Table 4.5.

Table 4.5 Constituents in muscle tissue of fish collected from surface water bodies in the site vicinity, Gunnison, Colorado, site

	Gunnison River location ID		Tomichi Creek location ID	Campground pond location ID
Constituent	775 (upstream)	776 (downstream)	777 (downstream)	779
Manganese	3.8	1.9	1.9	1.1
Molybdenum	0.26	<0.20	<0.20	<0.20
Uranium	<0.20	0.46	<0.20	<0.20
Zinc	28.1	24	28.4	26.6

Note: All concentrations reported in milligrams per kilogram dry weight (DW). Fish were collected 23 June 1993.

Manganese was detected at the highest concentration (3.8 mg/kg) in the fish collected from the upstream Gunnison River location. The concentrations in fish from the other locations ranged from 1.1 mg/kg (campground pond) to 1.9 mg/kg (downstream locations in both the river and creek). Molybdenum was detected only in the fish collected at the upstream Gunnison River location. Uranium was detected at a concentration of 0.46 mg/kg in fish muscle tissue from the downstream Gunnison River location. Uranium was not detected (<0.2 mg/kg) in fish from the other sampling locations. Zinc was detected at similar concentrations in fish tissue from all the sampling locations; the concentrations ranged from 24 to 28.4 mg/kg.

Whether the former uranium mill tailings site is contributing factor in this uranium level cannot be conclusively determined. Lack of fish bone tissue uranium concentrations (uranium is a bone-seeking element) and naturally occurring uranium ore deposits, coupled with tremendous uncertainty regarding habitat ranges of the fish over a lifespan, make a potential connection between fish tissue levels and site-related constituents difficult to determine. Nevertheless, the fish ingestion pathway is evaluated in this risk assessment.

To evaluate the fish ingestion exposure pathway, a screening calculation was performed to determine if the contribution of fish ingestion from each location tested would be notable compared to the drinking water pathway. For this screening pathway, if the constituent was not detected in a sample, one-half the detection limit was used to represent the exposure point concentration (Table 4.6). Based on the manganese and uranium results, the fish ingestion exposure pathway was eliminated from more detailed quantitative evaluation, since it contributed to less than 1 percent of the total dose from drinking water. The maximum ingestion doses calculated for zinc and molybdenum are within nutritional ranges. Therefore, no adverse health effects would be expected from the estimated levels.

Summary

In summary, the results from all of the screening pathways (water ingestion; dermal contact with water; garden produce, milk, meat, and fish ingestion) indicate drinking water ingestion is the dominant pathway; therefore, this pathway is further evaluated probabilistically in Section 4.4.

4.3 EXPOSURE CONCENTRATIONS

The exposure concentration of a constituent in ground water is defined as the concentration an individual is assumed to take in over a specific period. In this evaluation, the constituent concentrations are assumed to be in a steady state, although actual constituent concentrations (and therefore exposures) are expected to decrease with time after the tailings are removed. Nonetheless, these estimates are reasonable for chronic exposure soon after surface remediation. Chronic exposure for noncarcinogens is considered to be exposure for any period longer than 7 years.

Table 4.6 Exposure dose calculations for ingestion of fish from surface water bodies in the site vicinity, Gunnison, Colorado, site

	Gunnison River upstream		Gunnison River downstream		Tomichi Creek		Campground pond		
Constituent	Fish concentration (mg/kg)	Intake)mg/kg-day)	Fish concentration (mg/kg)	Intake (mg/kg-day)	Fish concentration (mg/kg)	Intake (mg/kg-day)	Fish concentration (mg/kg)	Intake (mg/kg-day)	Ratio of fish ingestion: water ingestion[a]
Manganese	3.8E+00	1.4E-03	1.9E+00	7.0E-04	1.9E+00	7.0E-04	1.1E+00	4.1E-04	0.007
Molybdenum	2.6E-01	9.6E-05	1.0E-01	3.7E-05	1.0E-01	3.7E-05	1.0E-01	3.7E-05	NA
Uranium	1.0E-01	3.7E-05	4.6E-01	1.7E-04	1.0E-01	3.7E-05	1.0E-01	3.7E-05	0.004
Zinc	2.8E+01	1.0E-02	2.4E+01	8.9E-03	2.8E+01	1.1E-02	2.7E+01	9.8E-03	NA

Equation definitions for exposure dose calculations

Ingestion of fish

Chemicals

$$\text{Chronic daily intake (mg/kg-day)} = \frac{Cf \times IR \times EF \times ED \times FI}{BW \times AT}$$

Radionuclides

$$\text{Liftime intake (pCi/lifetime)} = Cf \times IR \times EF \times ED \times FI$$

Where:

Cf = Constituent concentration in fish (mg/kg).
IR = Ingestion rate of fish (0.054 kg/day for an adult).
EF = Exposure frequency (350 days/year).
ED = Exposure duration (30 years for an adult).
FI = Fraction ingested from contaminated source (0.5 unitless).
BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days x ED for noncarcinogens).

---

[a] The maximum fish ingestion exposure dose was divided by the ground water ingestion exposure dose.

NA — not applicable; constituent was not identified as a constituent of concern in ground water.

For noncarcinogens, exposure concentrations are evaluated as a probability of occurrence based on ground water data collected from monitor wells 006, 133, and 134 for the constituents of potential concern (DOE, 1996). These wells have consistently shown the highest concentrations of most constituents since 1989. Concentrations of manganese are greatest in the deeper wells downgradient from the tailings pile; therefore, monitor wells 106, 109, 110, 111, and 112 were used to evaluate potential manganese exposures. All these wells are located on the processing site.

The probability distribution selected for each constituent reflected the same mean, median, standard deviation, and shape observed in historical water quality data (DOE, 1996). The upper tail of the distributions was truncated at the 99th percentile. For every constituent, this highest allowable concentration was higher than the maximum observed concentration in historical water quality data. The software package @RISK (Palisade Corporation, 1992) was used to generate the probability curves for the constituents of potential concern (Figures 4.2 through 4.7). The concentrations of radionuclides were represented by the median and maximum values observed in wells 006, 133, and 134.

4.4 ESTIMATION OF INTAKE

Within the population of future residents, individuals are expected to vary with respect to water consumption habits, stable body weight, and length of time they reside in the potential contamination zone. Consequently, health risks associated with ground water consumption will vary among members of this population. To adequately describe the range of potential risks to the future population, naturally occurring variability in daily water intake, body weight, and residence time were incorporated in this assessment through probability distributions; these distributions were generated from United States public health and census documents. All distributions were truncated at the upper and lower 0.01 percentile. Within the hypothetical population, values disallowed through this truncation may occur with a probability of less than 1 in 10,000.

The potential toxicity of noncarcinogenic constituents in drinking water depends primarily on long-term average daily consumption of the constituent per kilogram of body weight. For probability distributions of noncarcinogens, the chronic daily intake is calculated as follows:

$$\text{Intake (mg/kg-day)} = \frac{\underset{\text{(mg/L)}}{\text{Concentration}} \times \underset{\text{(L/day)}}{\text{ingestion rate}} \times \underset{\text{(days/year)}}{\text{exposure frequency}} \times \underset{\text{(years)}}{\text{exposure duration}}}{\text{Body weight (kg)} \times 365\text{ (days/year)} \times \text{exposure duration (years)}} \qquad \text{(Eq. 1)}$$

Potential carcinogenicity is thought to increase with total intake over time, instead of with average daily intake as for noncarcinogens. Also, body weight is relatively insignificant in determining risk from exposure. The only carcinogens elevated above background at the Gunnison site are the radionuclides lead-210, polonium-210, thorium-230, and uranium. Therefore, only calculations for radioactive carcinogens will be presented here. Intake of a carcinogenic substance

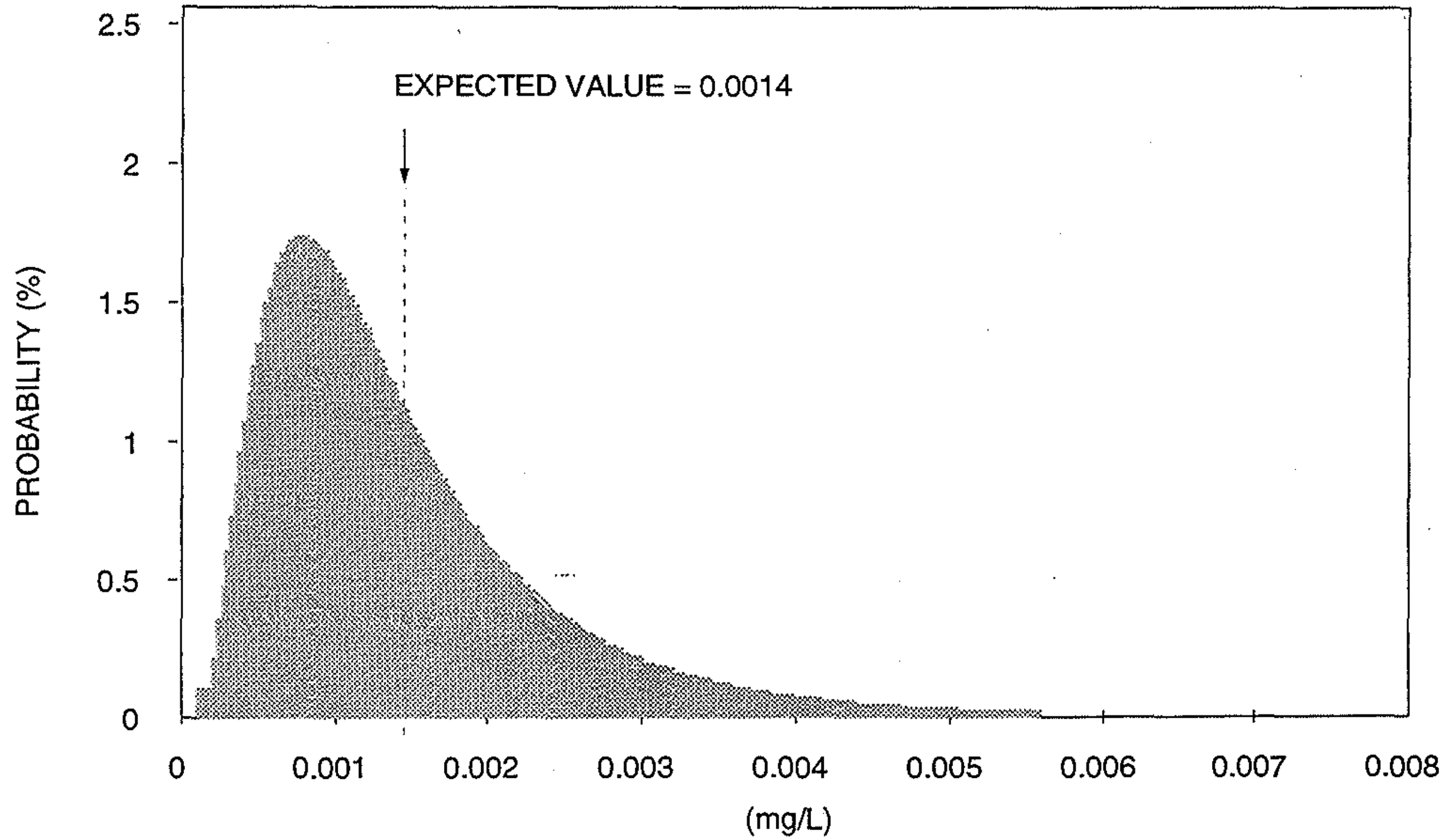

FIGURE 4.2
SIMULATED DISTRIBUTION OF CADMIUM CONCENTRATIONS
GUNNISON, COLORADO, SITE

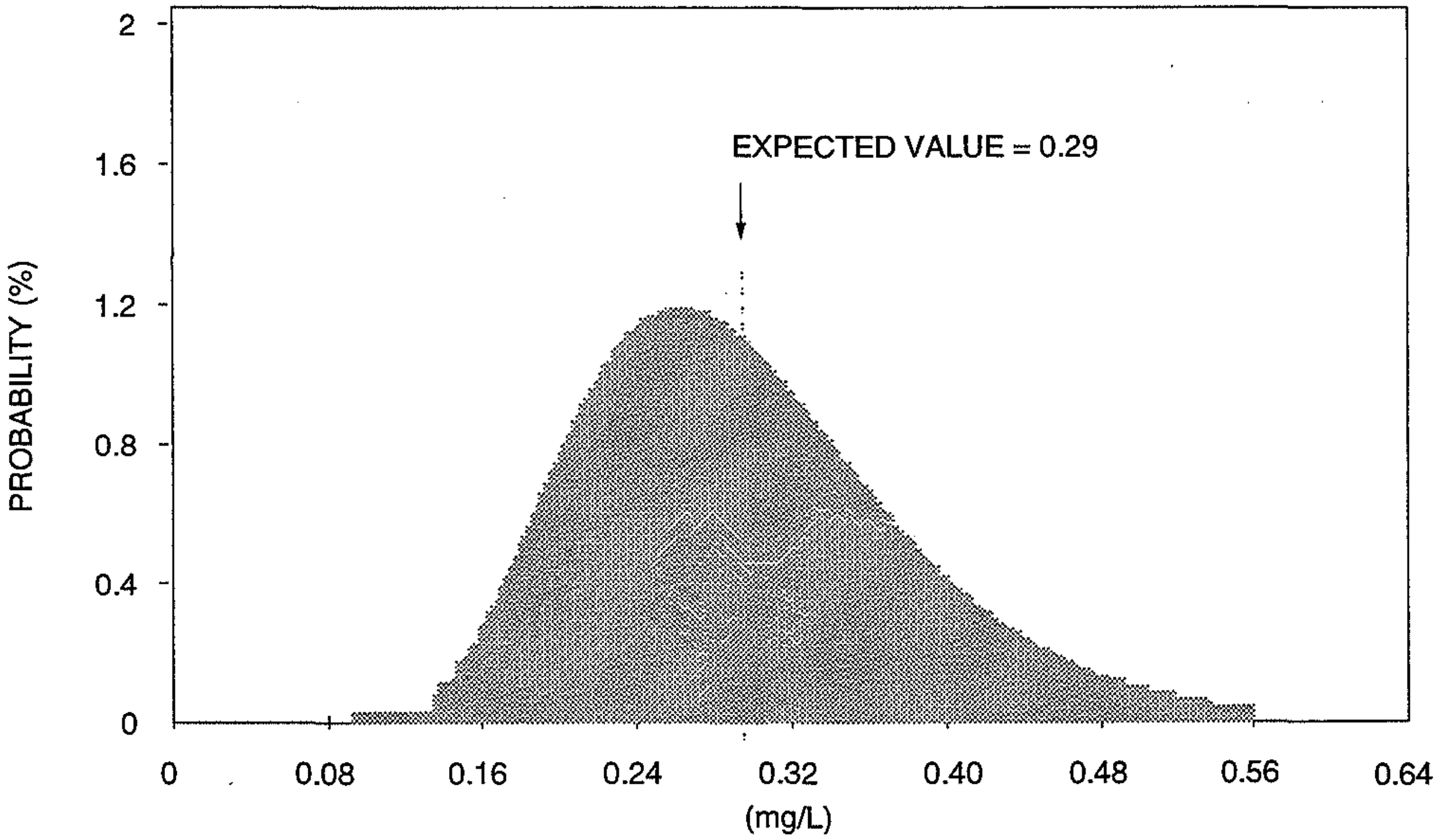

FIGURE 4.3
SIMULATED DISTRIBUTION OF COBALT CONCENTRATIONS
GUNNISON, COLORADO, SITE

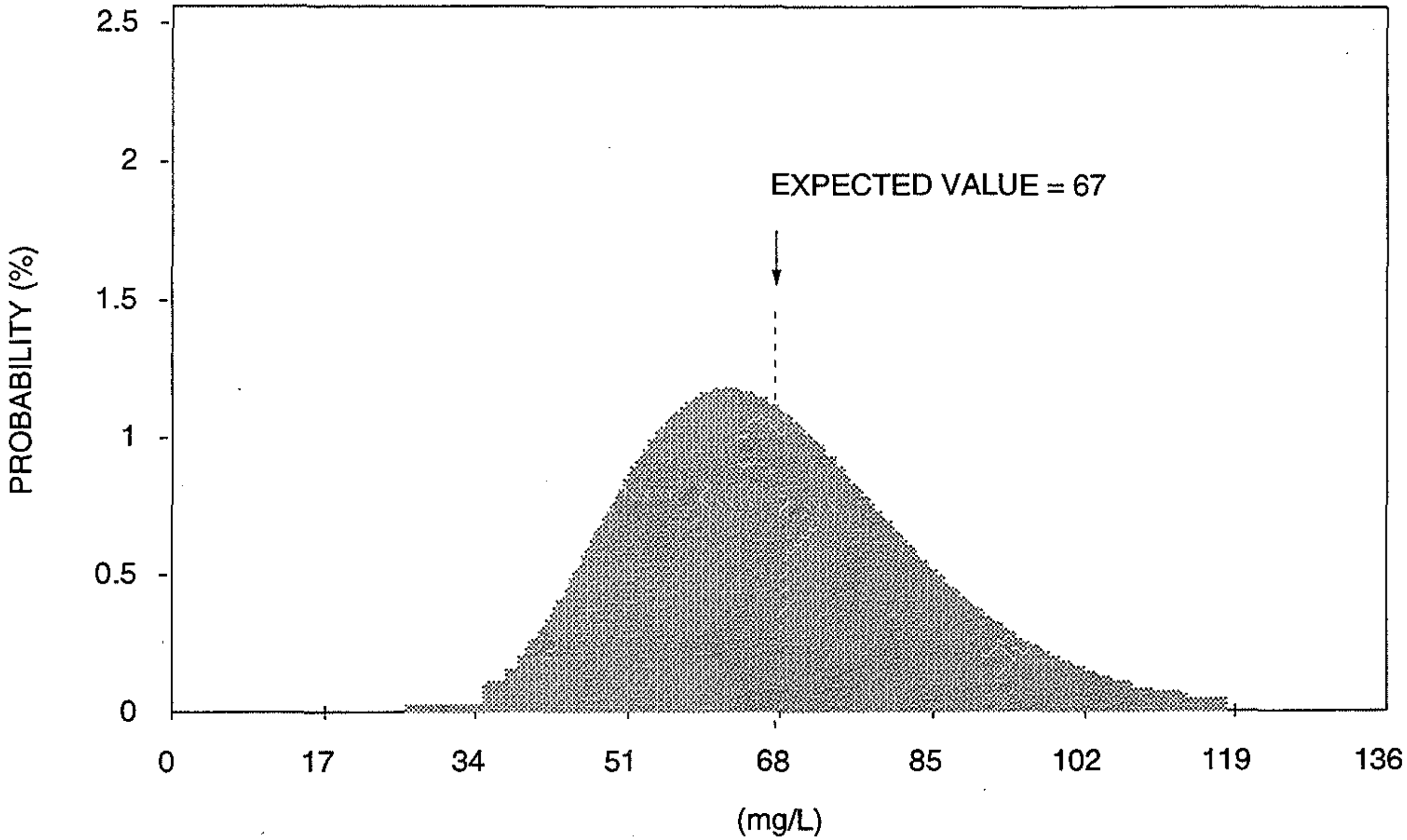

FIGURE 4.4
SIMULATED DISTRIBUTION OF IRON CONCENTRATIONS
GUNNISON, COLORADO, SITE

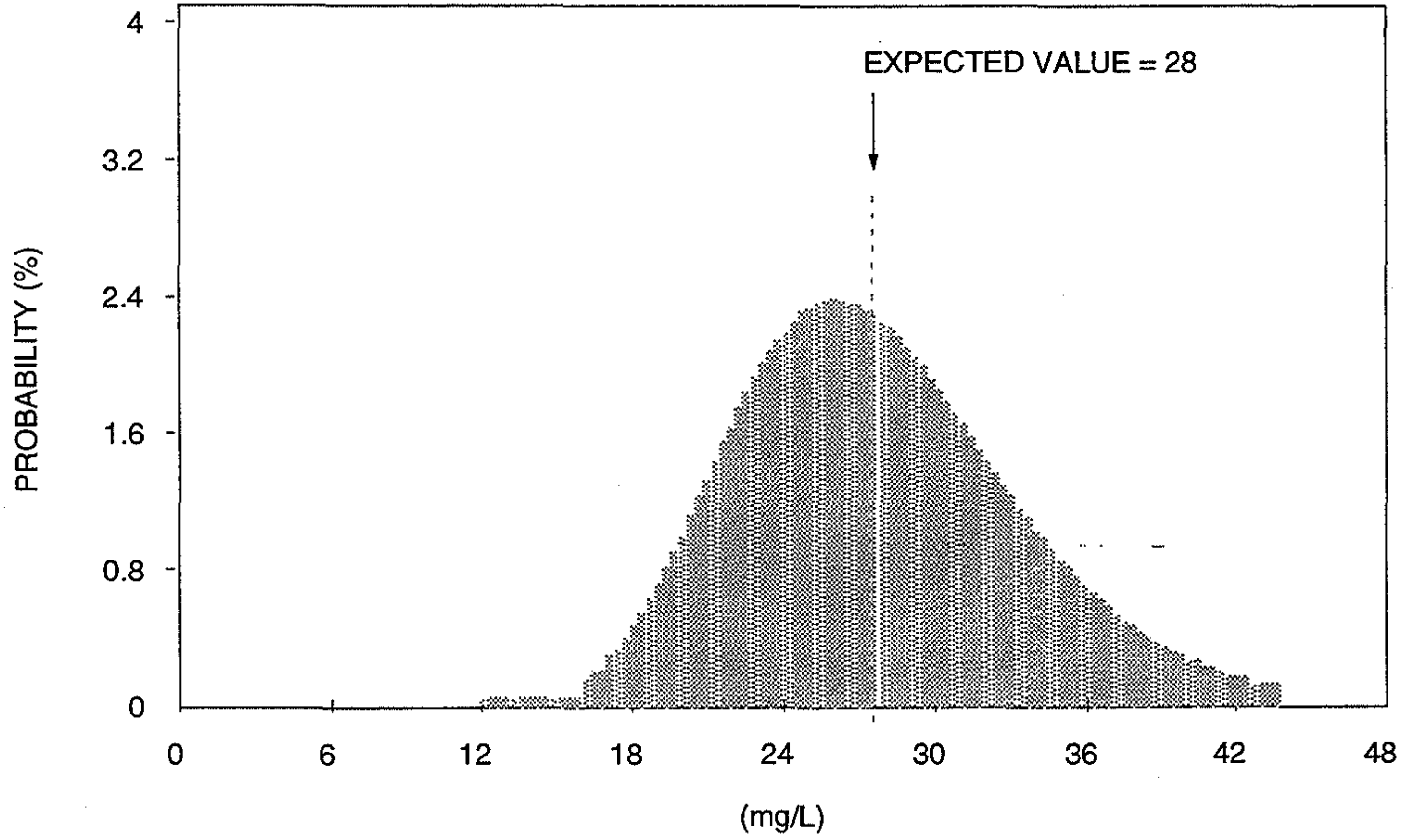

FIGURE 4.5
SIMULATED DISTRIBUTION OF MANGANESE CONCENTRATIONS
GUNNISON, COLORADO, SITE

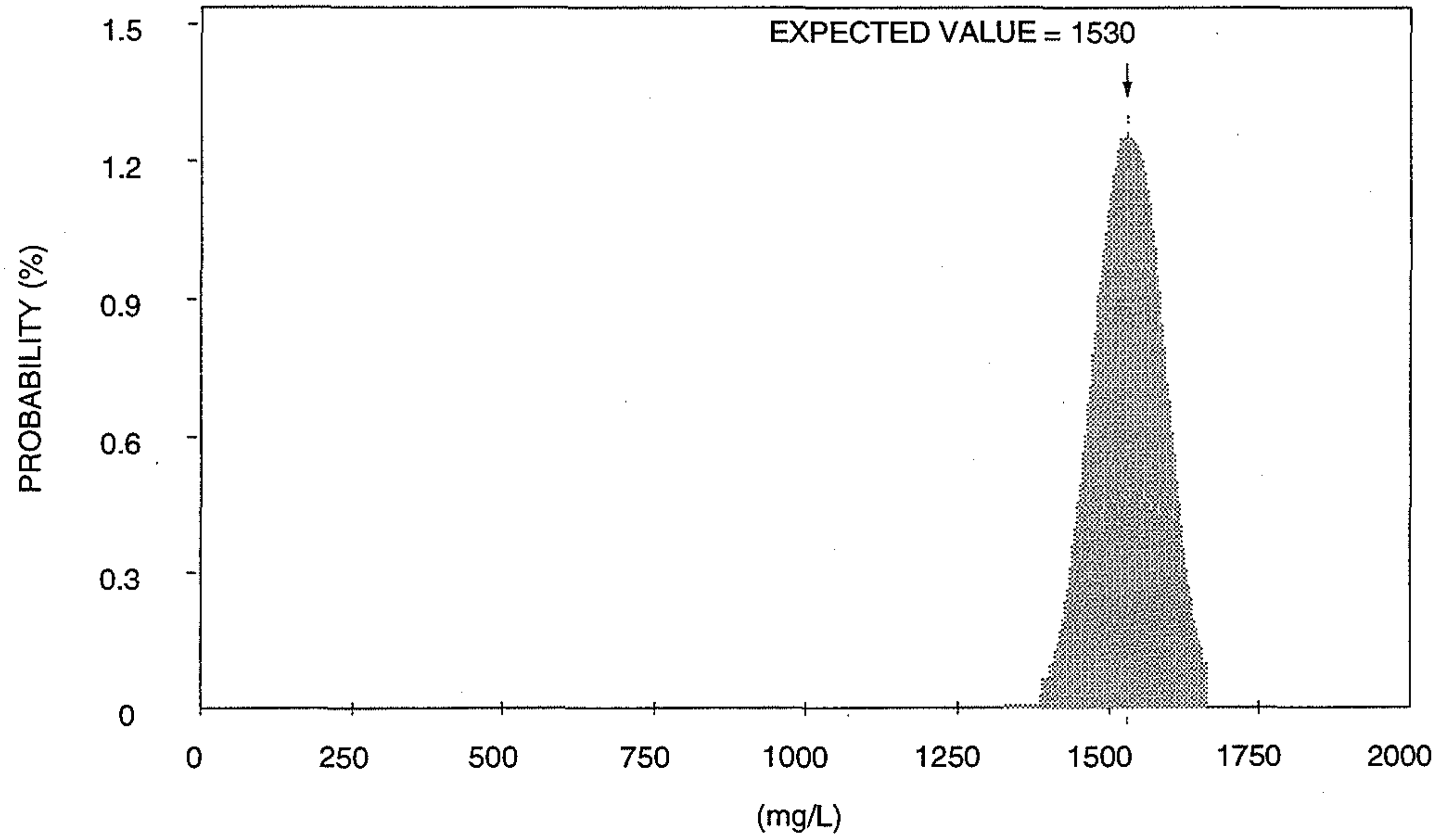

FIGURE 4.6
SIMULATED DISTRIBUTION OF SULFATE CONCENTRATIONS
GUNNISON, COLORADO, SITE

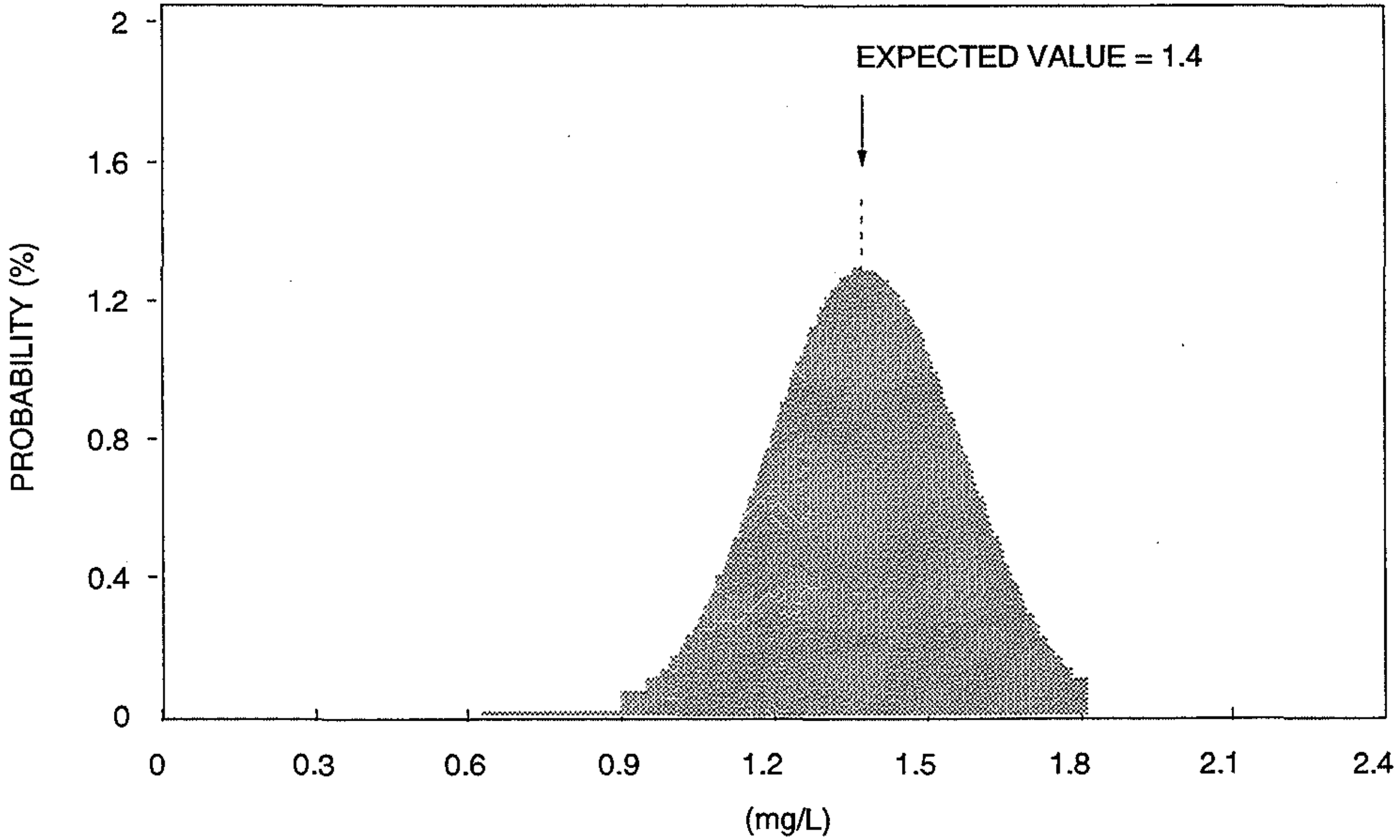

FIGURE 4.7
SIMULATED DISTRIBUTION OF URANIUM CONCENTRATIONS
GUNNISON, COLORADO, SITE

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is therefore quantified as total exposure to radioactivity throughout the residency period of an individual:

$$\text{Intake (pCi)} = \underset{\text{(pCi/L)}}{\text{Concentration}} \times \underset{\text{(L/day)}}{\text{ingestion rate}} \times \underset{\text{(days)}}{\text{exposure frequency}} \times \underset{\text{(years)}}{\text{exposure duration}} \qquad \text{(Eq. 2)}$$

Average daily intake (L/day)

Lognormal probability distributions were used to describe variation in average daily tap water intake among members of the population (Roseberry and Burmaster, 1992). These distributions were developed from data collected during the 1977-78 National Food Consumption Survey conducted by the U.S. Department of Agriculture (DOE, 1996). During the survey, total tap water consumption during a 3-day period was recorded for 26,081 survey participants nationwide (Figure 4.8). The exposure distribution for infants is based on tap water intake rates across a population that includes breast-fed and canned-formula-fed infants.

Body weight (kg)

Extensive national data on weights of males and females, by age, were collected by the National Health and Nutrition Survey between 1976 and 1980 (DOE, 1996). These data were used to develop lognormal probability distributions for body weight by age, separately by sex. The distributions for males and females were then combined using census data on the national ratio of males to females within each age group (Figure 4.9).

Exposure duration (years)

A survey of residents in the Dos Rios area indicated that residence time frequently exceeded 25 years (DOE, 1990). Therefore, a fixed lifetime exposure time of 30 years was used to model lifetime cancer risks. Because the concentrations of radionuclides in ground water at the Gunnison site are decreasing with time, evaluations of risk based on this exposure duration in combination with historic and current median and maximum concentrations of radionuclides should yield conservative estimates of carcinogenic risks for this site.

Derivation of intake distributions

Using exposure concentration distributions discussed in Section 4.3 and the intake parameter distributions described in this section, total intake distributions derived for the three age groups were generated for uranium. Figure 4.10 illustrates the effect of the different age group characterization on daily intake of noncarcinogens. This figure shows intake is greatest in the 1- to 10-year age group, although the intake for the 0- to 1-year age group is very similar. Therefore, the 1- to 10-year age group is used in risk evaluation unless one of the other age groups has demonstrated increased sensitivity to a particular constituent. No data are available to evaluate any sensitive subpopulations for any of the constituents of concern at

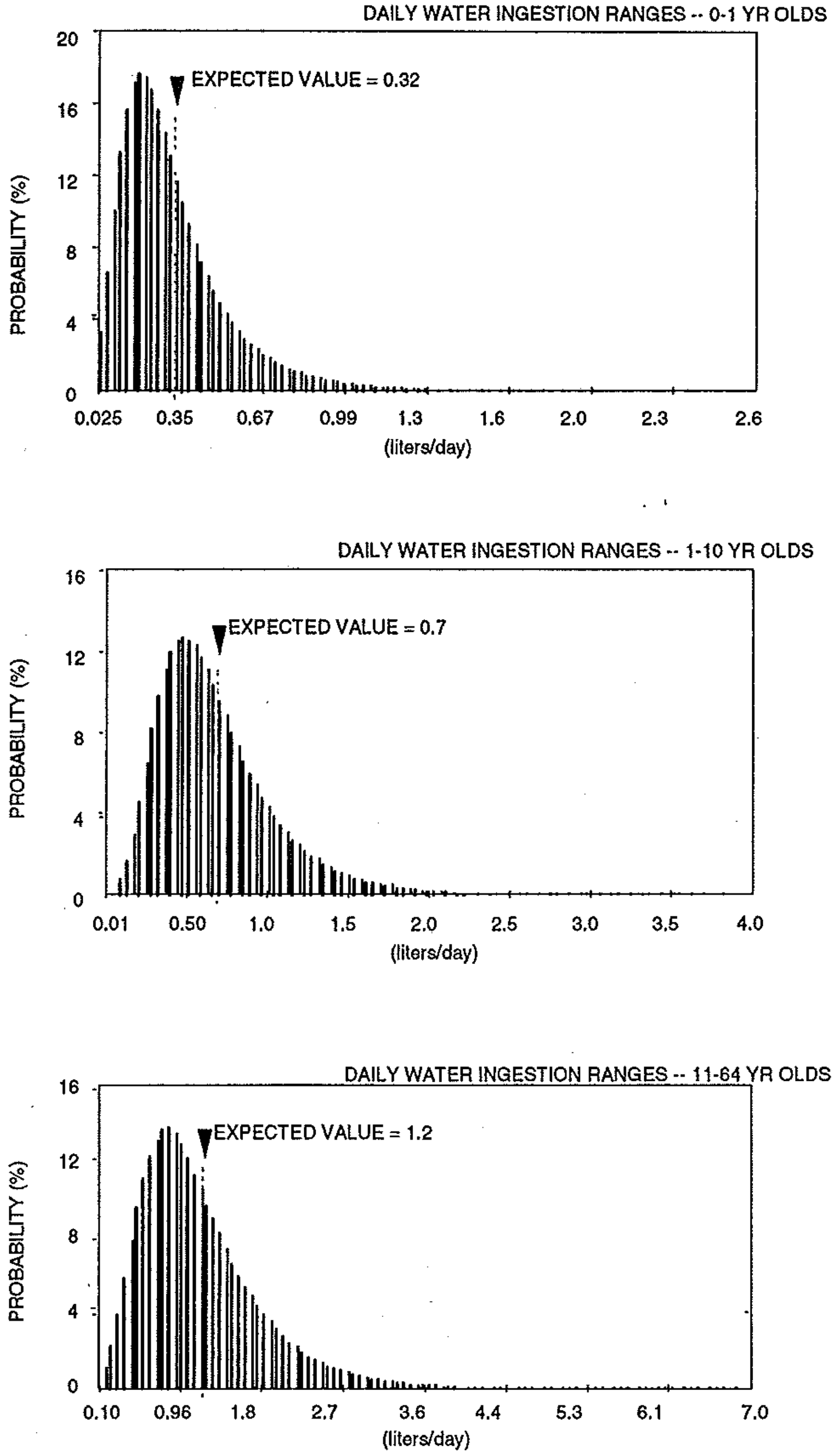


FIGURE 4.8
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES

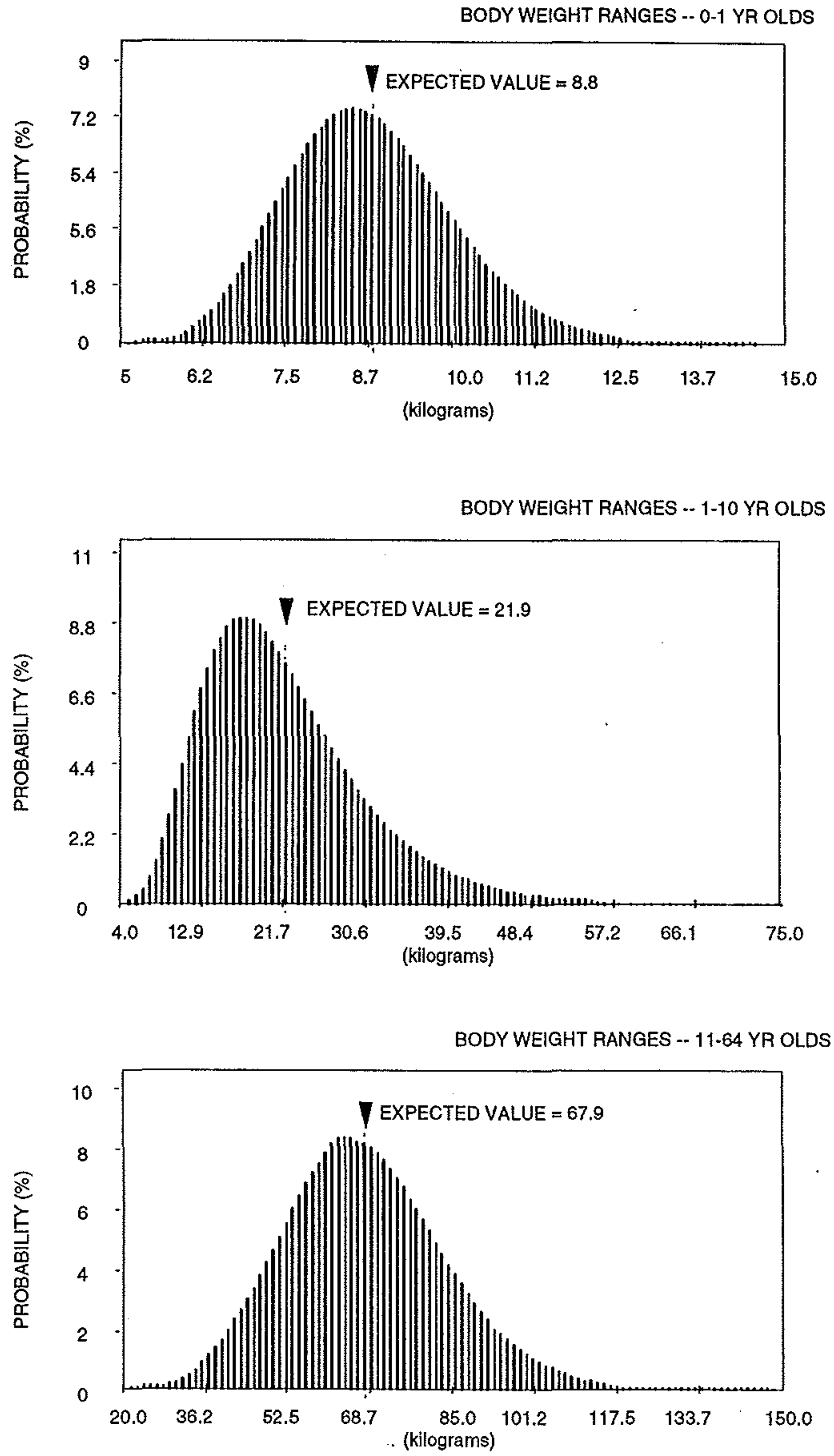
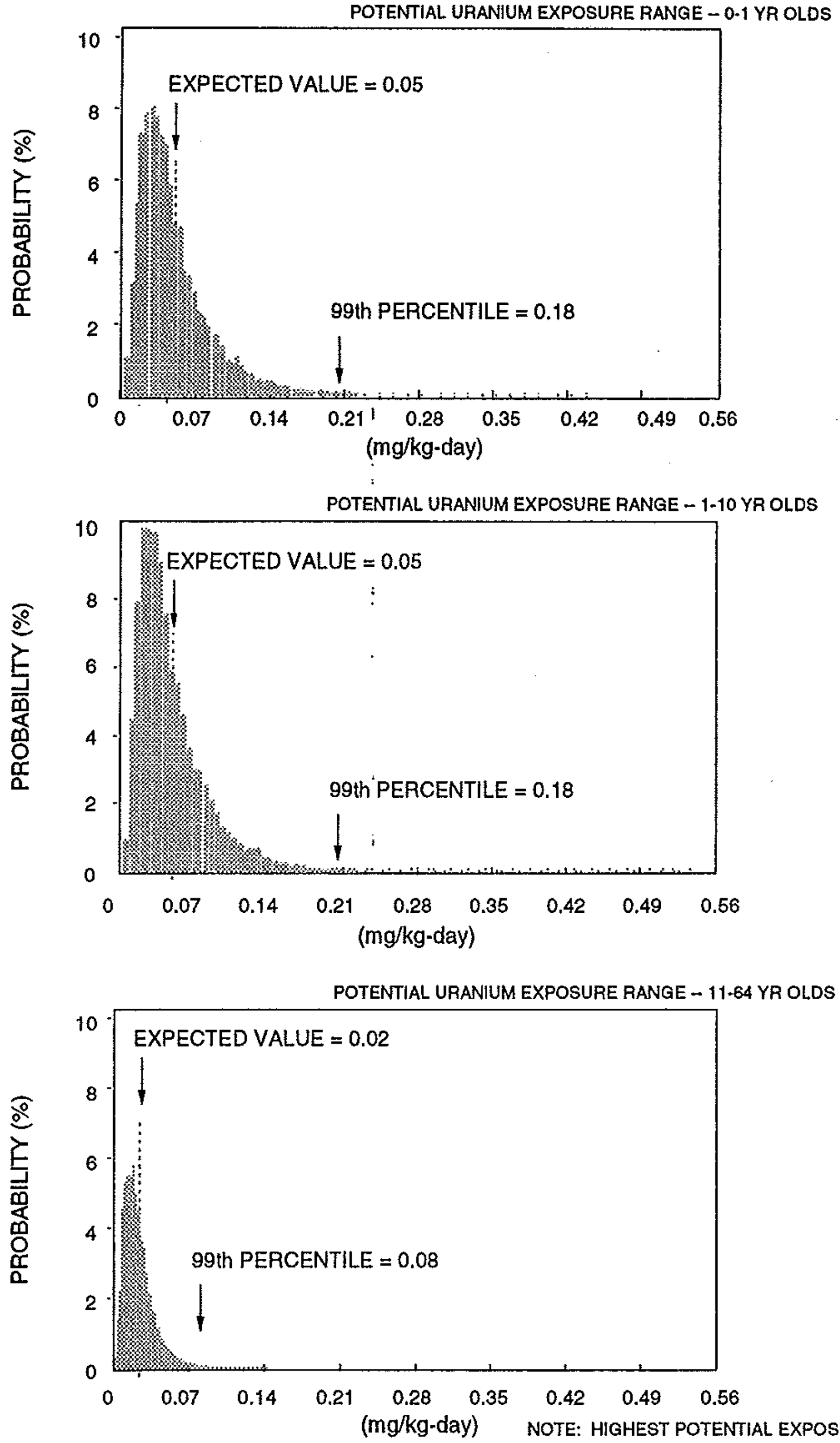


FIGURE 4.9
PROBABILITY DISTRIBUTIONS FOR BODY WEIGHT



NOTE: HIGHEST POTENTIAL EXPOSURE OCCURS IN CHILDREN AGE 1-10 (AVERAGE POTENTIAL EXPOSURE OF 0.05 mg URANIUM/kg BODY WEIGHT/DAY).

FIGURE 4.10
COMPARISON OF DISTRIBUTIONS OF POTENTIAL URANIUM EXPOSURES AS A RESULT OF DRINKING CONTAMINATED GROUNDWATER FOR THE THREE AGE GROUPS EXAMINED
GUNNISON, COLORADO, SITE

the Gunnison site. Therefore, simulated intake distributions for 1- to 10-year old children for the constituents at this site are presented in Figures 4.11 through 4.15.

4.5 EXPOSURE ASSESSMENT UNCERTAINTIES

A number of potential sources of error may arise in all phases of the exposure assessment, including the following more meaningful sources of uncertainty:

- Uncertainties resulting from the lack of thorough environmental sampling data (ground water, surface water, sediment, and biota), which could lead to an underestimate or overestimate in the exposure analysis.

- Uncertainties associated with using filtered ground water samples. The results of the exposure assessment presented in this document are based primarily on filtered (0.45-micrometer [μm]) ground water samples. Therefore, the potential loss of certain ground water constituents as a consequence of filtration, which could not be estimated here, is associated with an additional source of uncertainty.

- Uncertainties arising from the assumption that the ground water constituent source term at the site has reached a steady state and that constituent concentrations at the exposure point will remain constant for chronic periods of exposure (generally greater than 7 years). Because the source of contamination at Gunnison is removed, the assumption of a constant source probably will overestimate exposure.

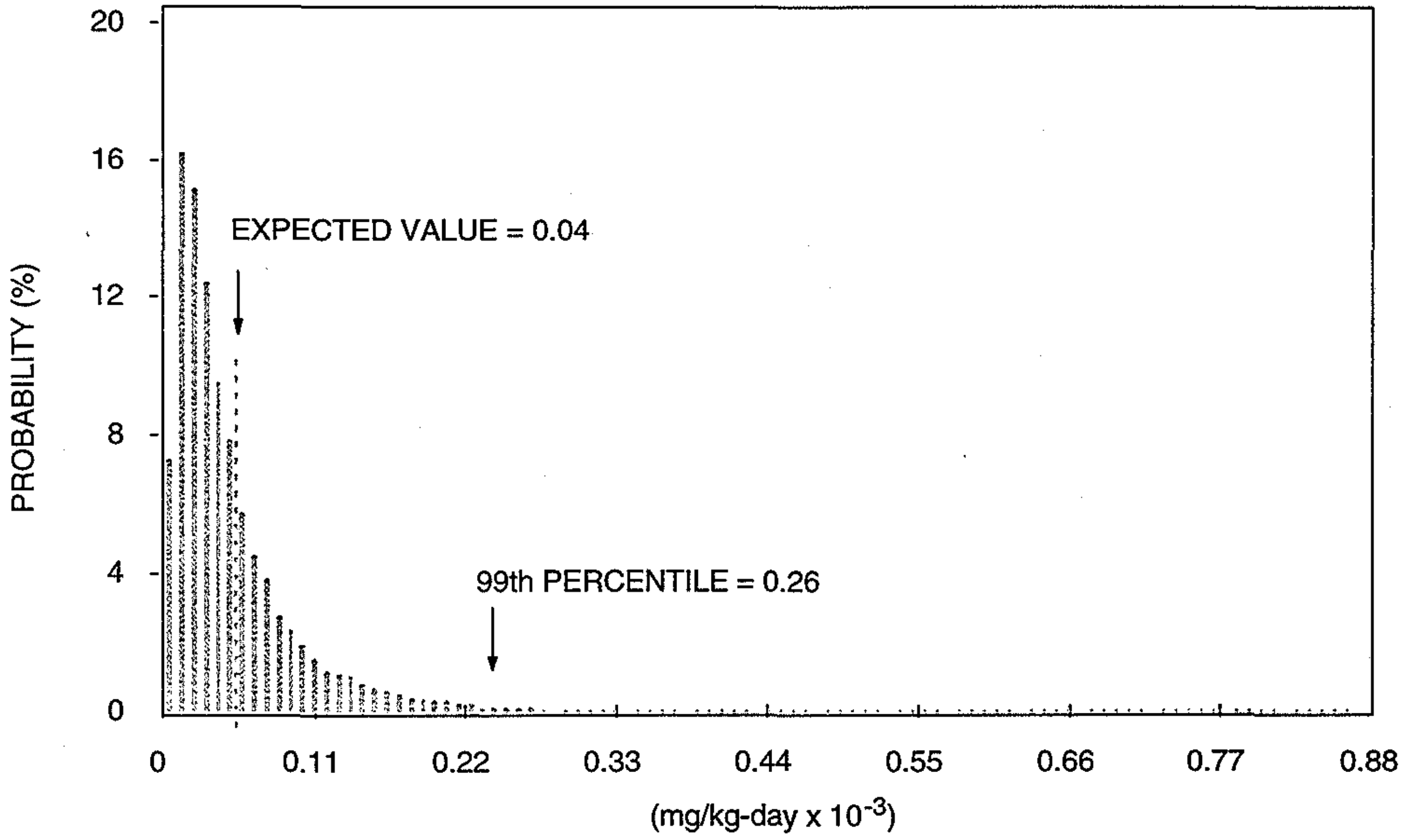
- Uncertainties associated with the model used to estimate constituent uptake into plants for the irrigated garden produce pathway. Site-specific plant uptake factors could vary substantially from the default literature estimates. As with environmental sampling, the net effect on exposure estimates of this uncertainty cannot be predicted.

- Uncertainties with BCFs and meat transfer coefficients for the milk and meat ingestion pathways. Site-specific BCFs and transfer coefficients could vary substantially from the default literature values.

- Uncertainties associated with the relationship of an applied dose (used in this assessment) and absorbed dose or effective toxic dose.

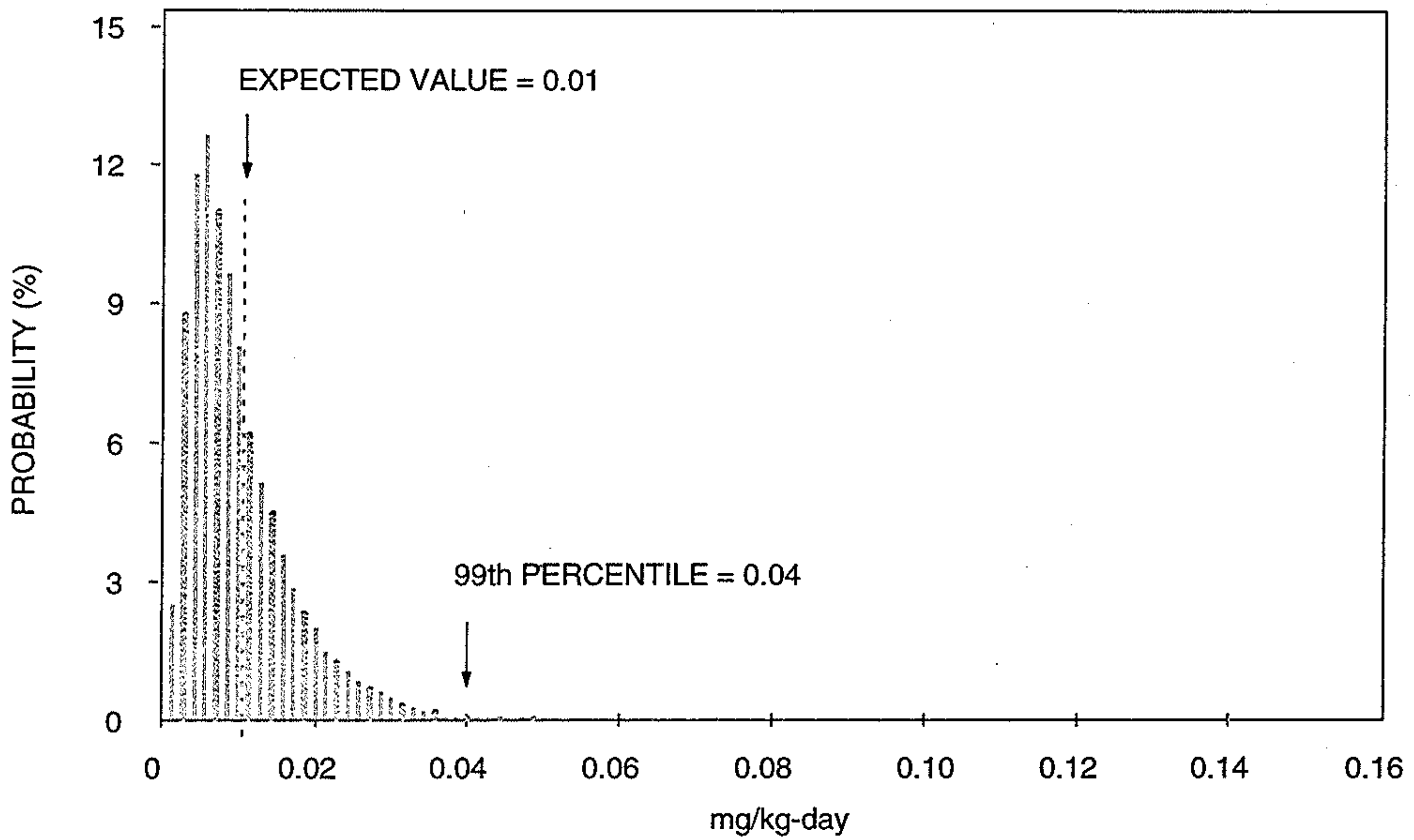
- Uncertainties associated with differing sensitivities of subpopulations, such as individuals with chronic illnesses, that could alter predicted responses to constituents of potential concern.

Despite these uncertainties, the use of probability distributions that incorporate all definable sources of variability provides a representative picture of the potential range of exposure.



NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.04 mg CADMIUM/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.11
DISTRIBUTION OF POTENTIAL CADMIUM EXPOSURES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
GUNNISON, COLORADO, SITE



NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.01 mg COBALT/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.12
DISTRIBUTION OF POTENTIAL COBALT EXPOSURES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION GUNNISON, COLORADO, SITE

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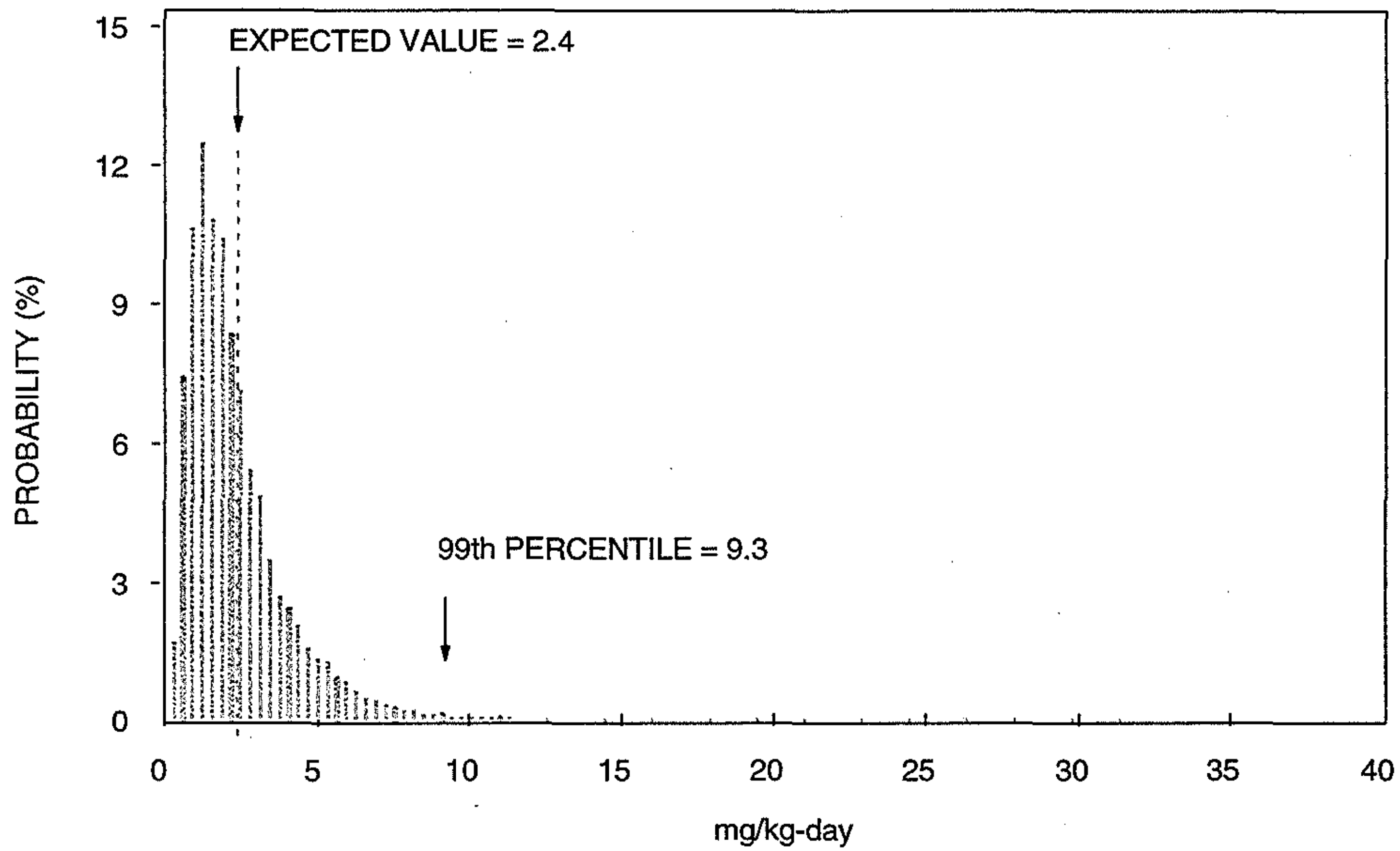
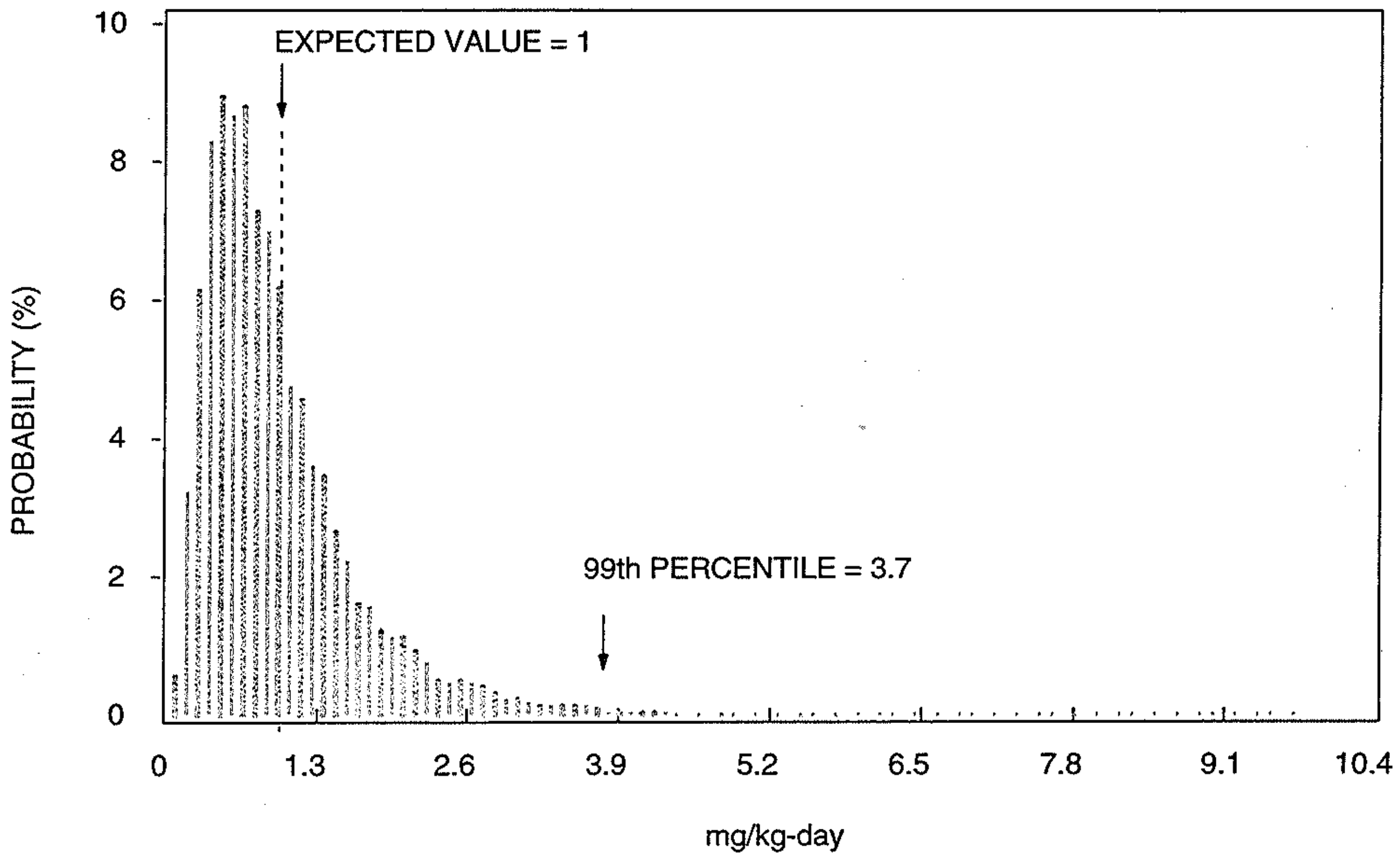


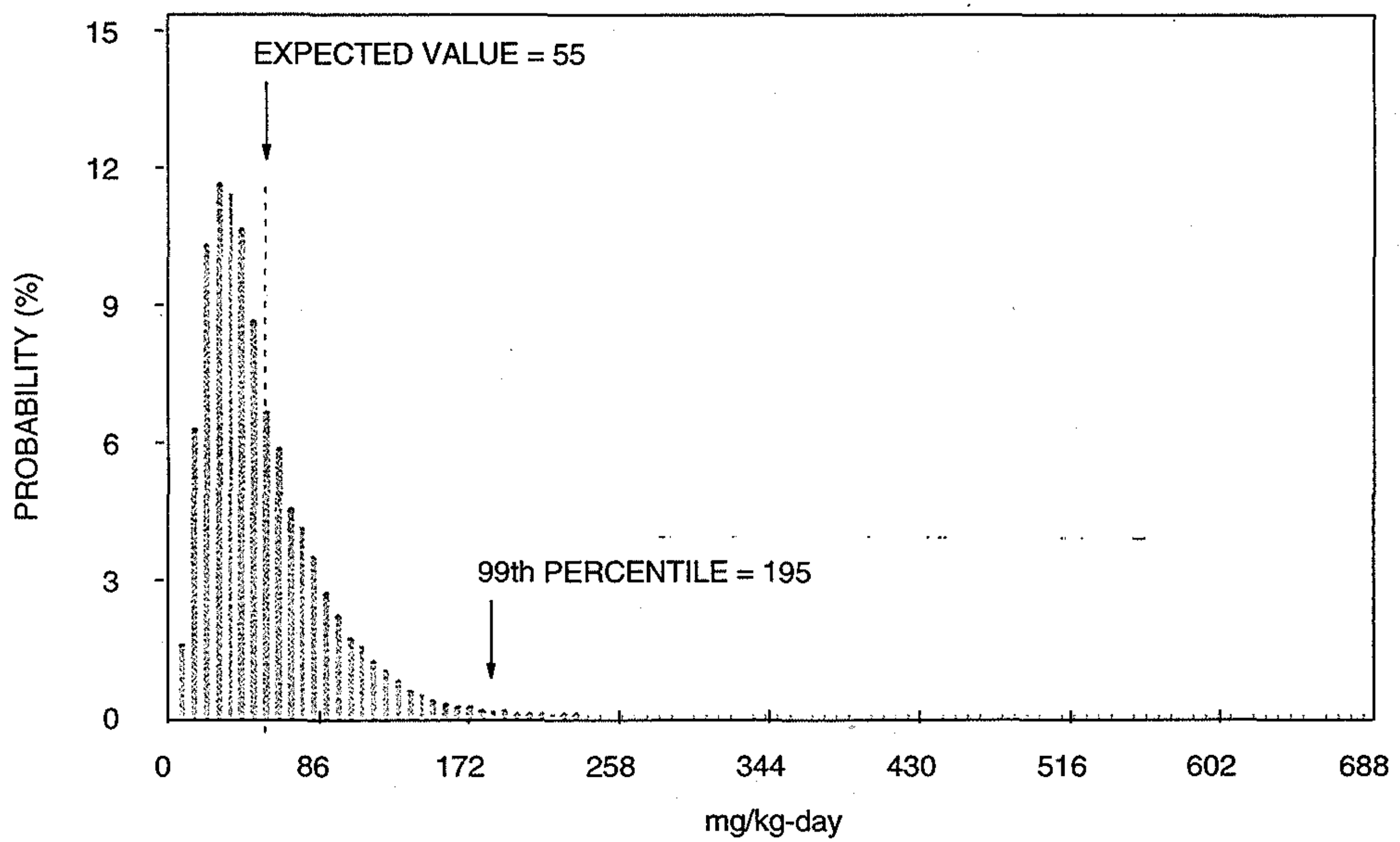
NOTE: AVERAGE POTENTIAL EXPOSURE IS 2.4 mg IRON/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.13
DISTRIBUTION OF POTENTIAL IRON EXPOSURES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION GUNNISON, COLORADO, SITE



NOTE: AVERAGE POTENTIAL EXPOSURE IS 1 mg MANGANESE/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.14
DISTRIBUTION OF POTENTIAL MANGANESE EXPOSURES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION GUNNISON, COLORADO, SITE



NOTE: AVERAGE POTENTIAL EXPOSURE IS 55 mg SULFATE/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.15
DISTRIBUTION OF POTENTIAL SULFATE EXPOSURES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION
GUNNISON, COLORADO, SITE

5.0 TOXICITY ASSESSMENT

Several constituents that have the potential to cause adverse human health effects have been detected in ground water at the site. This section summarizes the toxicological effects of the chemical constituents and carcinogenic potential of the radionuclides. The following source materials were used in developing these toxicological profiles: when available, EPA's Integrated Risk Information System (IRIS) (EPA, 1994a); the Agency for Toxic Substances Disease Registry Toxicological Profiles published by the Department of Health and Human Services (DHHS); and the *Handbook on the Toxicology of Metals* (Friberg et al., 1986). When these review documents were not available, peer-reviewed scientific literature was used. Basing toxicity information on the standardized review documents cited above should make the evaluation of risks at UMTRA Project sites consistent with evaluations at sites regulated under different legislation.

The toxicity profiles presented here focus on drinking water source material in humans when available, including animal data only when human data are not available. Animal data are shown on the toxicity range graphs by widely spaced, dotted lines. Uncertainty about the beginning or ending point of an exposure range that produces specific toxic effects is represented by closely spaced dots.

5.1 CONTAMINANT TOXICITY SUMMARIES

The following summaries address the basic toxicokinetics and toxicity of the six noncarcinogenic constituents of potential concern at Gunnison based on the preliminary screening discussed previously: cadmium, cobalt, iron, manganese, sulfate, and uranium. Wherever possible, data from human studies are reported; animal studies are reported only when human data are unavailable. Although these constituents have a wide range of toxic effects depending on the exposure levels, the discussions below focus on toxic effects observed in the exposure range most relevant to contamination at Gunnison.

5.1.1 Cadmium

Absorption

In humans, approximately 5 percent of ingested cadmium in water is absorbed (Friberg et al., 1986). The amount of cadmium absorbed from food sources is about half the amount absorbed from water. Gastrointestinal absorption is likely to depend on the physiologic status of an individual (age, body stores of iron) and on the presence and levels of divalent and trivalent cations and other dietary components ingested with cadmium. Young individuals appear to absorb more cadmium than older ones, and its absorption is increased in individuals with a diet high in fat and protein (Flanagan et al., 1978; DHHS, 1993). Cadmium gastrointestinal absorption can decrease following exposure to calcium, chromium, magnesium, and zinc. Zinc and iron deficiency may result in an increased absorption and accumulation of cadmium.

Cadmium absorption in the gastrointestinal tract appears to take place in two phases: uptake from lumen into mucosa, then transfer in the blood (DHHS, 1993). Cadmium uptake from lumen into mucosa may involve sequestering of cadmium by metallothionein, but any protective effect is overloaded at moderate doses. Cadmium uptake behaves like a saturable process with fractional absorption decreasing at high concentrations. However, at doses high enough to damage gastrointestinal mucosa, fractional cadmium absorption increases (DHHS, 1993).

Tissue accumulation and clearance

Humans with low-level and long-term exposure to cadmium show 50 percent of the body burden in the kidneys, 15 percent in the liver, and 20 percent in muscle (Kjellström, 1979). The kidney concentration will increase with continued exposure only to about age 50, but the concentration in muscle will increase throughout life. Only 0.01 to 0.02 percent of the total body burden of cadmium is excreted daily, resulting in a continuously increasing body burden with prolonged exposure. The biological half-life of cadmium, or the time necessary to eliminate 50 percent of the cadmium in the body at a given time, is 10 to 30 years in humans (Nordberg et al., 1985).

Environmental sources of cadmium

The average cadmium content of drinking water in nonpolluted areas (0.003 mg/L) results in about 0.00009 mg/kg-day intake of cadmium estimated for a 70-kg adult drinking 2 L of water per day. For a 22-kg child drinking 0.7 L of water per day, the estimated cadmium intake is 0.0001 mg/kg-day (DHHS, 1993). Cadmium occurs naturally with zinc and lead; it is therefore often present as an impurity in products using these metals, such as solders and galvanized metals. These sources lead to contact with water supplies in water heaters and coolers, in some pipes, and in taps.

Toxicity of cadmium

Short-term exposure to high concentrations of cadmium (15 mg/L in water) results in acute gastrointestinal effects, including abdominal cramps, diarrhea, and vomiting (0.48 mg/kg for a 22-kg child drinking 0.7 L of water per day). These gastrointestinal effects have not been reported in any chronic environmental exposure.

In chronic oral human exposure, the kidney is the main target organ of cadmium toxicity (DHHS, 1993). The primary toxic effect is disturbance of reabsorption in the proximal tubules. This effect is first observed by an increase of low molecular-weight proteins in the urine following a daily intake of 0.0075 mg/kg-day. Progressive disruption of kidney function will lead to an increase in amino acids, glucose, phosphate, and protein in the urine. The critical concentration of cadmium in the renal cortex below which no adverse effect would be anticipated has been estimated at about 200 μg/gram fresh weight (Foulkes, 1990; EPA, 1994a). However, the apparent critical level of cadmium in the kidney can vary substantially under different conditions of exposure; it can be affected by age, sex, and other factors that influence individual sensitivity to cadmium toxicity. The EPA-derived RfD is based on the critical concentration of cadmium in the human renal cortex that is not associated with the critical effect manifested as significant proteinuria (EPA, 1994a). Due to their already compromised kidney function, diabetics and the elderly can be more susceptible to cadmium toxicity (Buchet et al., 1990). Long-term exposures also can disturb calcium metabolism, leading to osteoporosis and osteomalacia. A combination of these two effects is referred to as Itai-itai disease and was seen in epidemic proportions in a cadmium-contaminated region in Japan in the 1950s (Friberg et al., 1986). Chronic dietary exposures of humans to cadmium produce no observable adverse effects at exposure levels from 0.001 to 0.002 mg/kg-day (DHHS, 1993). The health effects from exposure to cadmium as a function of dose are summarized in Figure 5.1.

Cadmium has been classified as a probable human carcinogen by the EPA (EPA, 1994a) and International Agency for Research on Cancer (IARC) (IARC, 1987). Although chronic inhalation of cadmium oxide has been related to increased lung and prostate cancers in workers, evidence linking cadmium to cancer in humans is inconclusive at this time because of the presence of other known carcinogens in the workplace and small statistical differences in tumor incidences (DHHS, 1993). There are no data, however, linking oral cadmium ingestion to cancer in humans or animals (DHHS, 1993).

5.1.2 Cobalt

Absorption

Gastrointestinal absorption of soluble cobalt compounds is estimated to be about 25 percent with wide individual variation; the gastrointestinal absorption in individuals reportedly varies from 5 to 45 percent (Friberg et al., 1986).

Cobalt is an integral component of vitamin B_{12}. The total vitamin B_{12} content of the body in a normal (i.e., nondeficient) adult human is about 5 mg, which is equivalent to about 0.2 mg of cobalt (Friberg et al., 1986).

Tissue accumulation and clearance

In humans exposed to cobalt, the liver exhibits the highest concentration, followed by the kidneys. Excretion occurs mainly through the urinary tract. Apparently, most cobalt is eliminated rapidly (within days) for all exposure routes (inhalation, injection, or ingestion). However, a small proportion is eliminated slowly, with a biological half time in the order of years (Friberg et al., 1986). Data are inadequate on the cobalt levels in tissues and fluids of normal populations (persons not occupationally exposed, i.e., background population) in the United States (DHHS, 1992a).

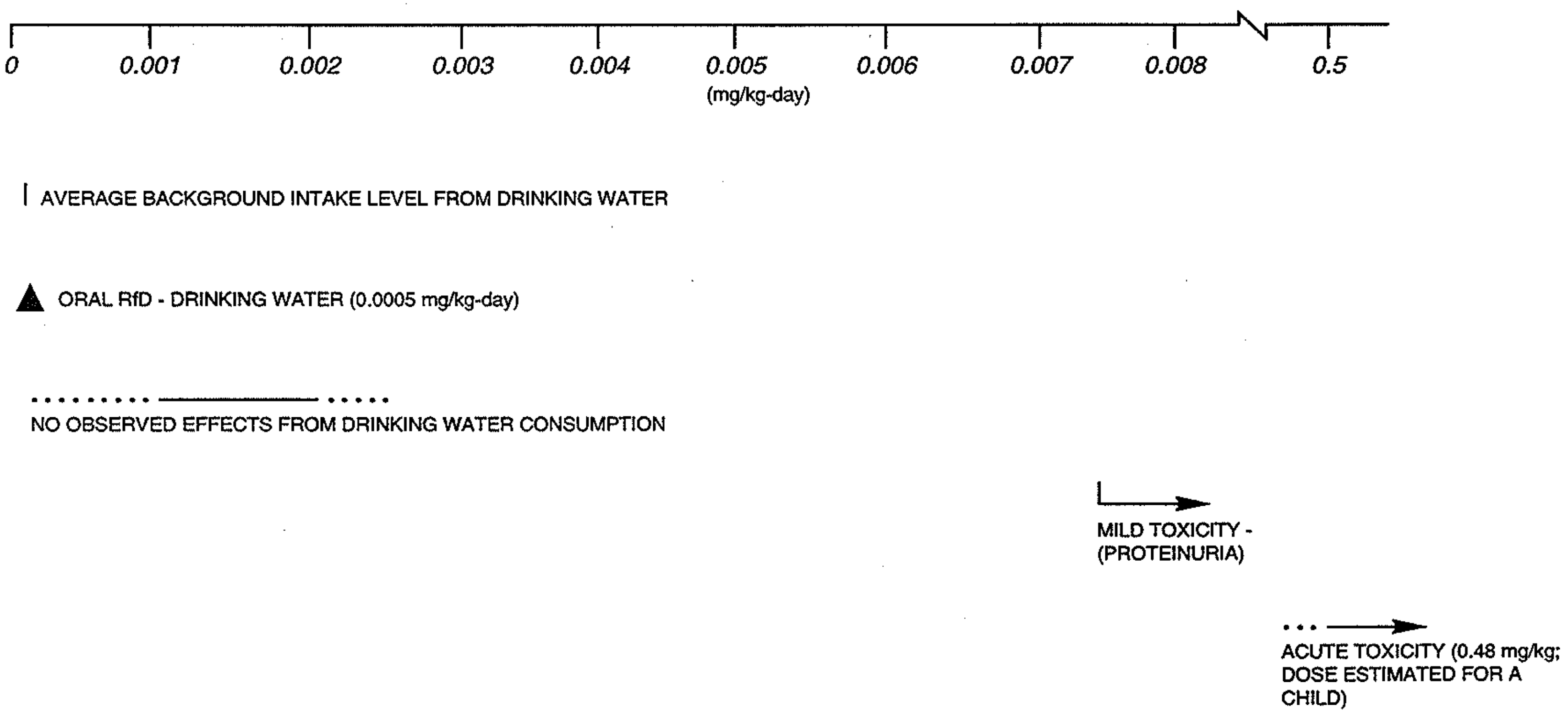


FIGURE 5.1
CADMIUM TOXICITY RANGES

Environmental sources of cobalt

Cobalt occurs naturally in the earth's crust, and as a result, in soil. Cobalt compounds occur naturally in seawater and in some surface, spring, and ground waters. Cobalt also is released into water from industrial and commercial sources, and it is a by-product or coproduct of refining other mined metals (e.g., copper and nickel). Only limited data are available on the levels of cobalt in United States foodstuffs. Therefore, the cobalt intake from food in the United States cannot be determined (DHHS, 1992a).

Toxicity of cobalt

Cobalt is an essential nutrient as an integral component of vitamin B_{12}. No other function for cobalt in human nutrition has been established. Adding cobalt to beer has caused endemic outbreaks of cardiomyopathy (damage to the heart muscle) among heavy beer drinkers, with a 50 percent mortality rate. Similar effects on the heart, including myocardial degeneration and electrocardiographic changes, have been seen in laboratory animals after repeated parenteral or oral exposure to cobalt (Friberg et al., 1986).

The average daily intake of cobalt from food is 5 to 45 µg (about 0.00007 to 0.0006 mg/kg-day for a 70-kg adult). The recommended daily intake of B_{12} for an adult is 3 µg, corresponding to 0.012 µg of cobalt or about 0.0002 µg/kg-day for a 70-kg adult (Friberg et al., 1986)

Cobalt is used in the medical treatment of anemias and has an erythropoietic effect (i.e., it stimulates the production of red blood cells). Duckham and Lee (1976) gave 12 anemic patients daily doses of cobalt chloride orally in amounts corresponding to 6.2 and 12.4 mg cobalt per day for a period of 12 to 30 weeks (approximately 0.09 to 0.13 mg/kg-day). This treatment gave rise to an average increase in the hemoglobin concentration of 46 percent. After cessation of cobalt treatment, the hemoglobin levels decreased. In addition to cardiomyopathy, polycythemia (increased number of red blood cells) was reported in heavy drinkers of cobalt-contaminated beer. It may be assumed that a really heavy beer drinker consuming up to 10 L/day of beer acquires an additional cobalt intake of approximately 10 mg/day (approximately 0.14 mg/kg-day). Although this figure is excessively high compared with nutritional standards, it is not as large as doses given to treat anemias (Friberg et al., 1986).

High levels of chronic cobalt oral exposure may result in the production of goiter. Epidemiologic studies suggest that the incidence of goiter is higher in regions containing increased levels of cobalt in the water and soil. The goitrogenic effect has been elicited by oral administration of 3 to 4 mg/kg to children in the course of sickle cell anemia therapy (Casarett and Doull, 1991). Figure 5.2 summarizes the toxicity of cobalt.

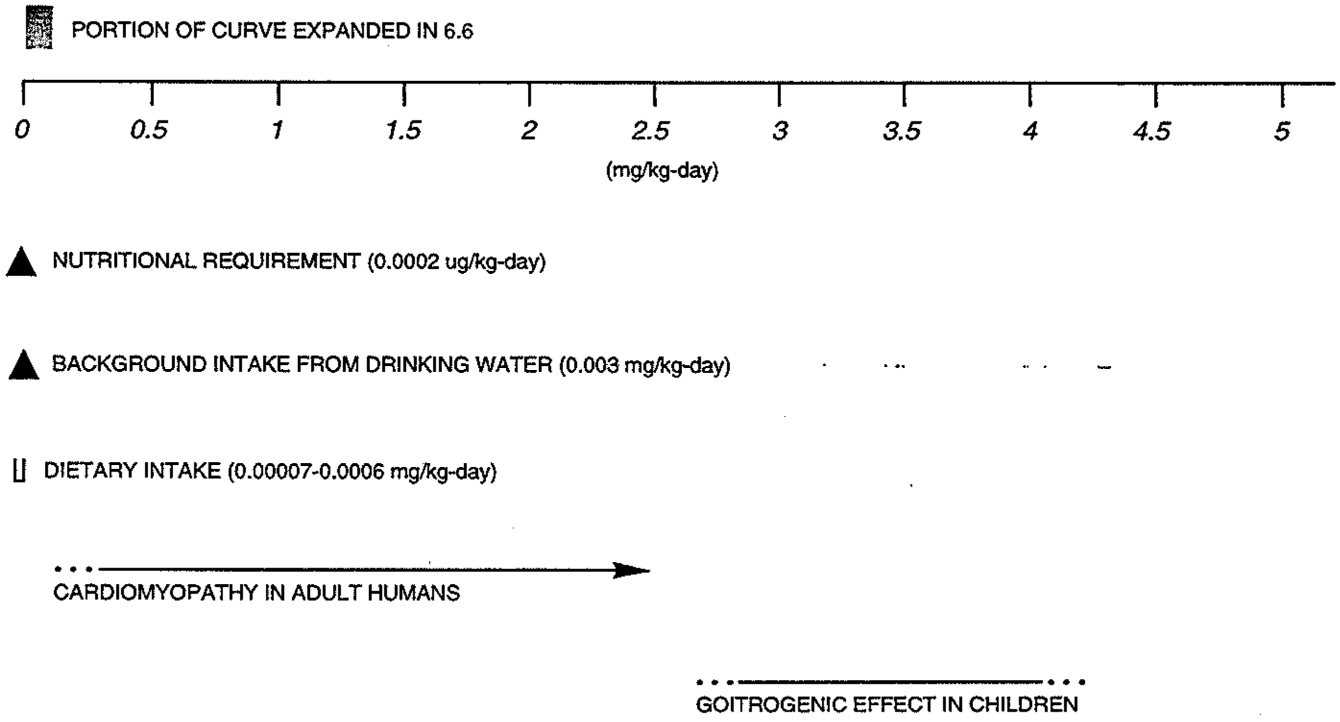


FIGURE 5.2
COBALT TOXICITY RANGES

5.1.3 Iron

Absorption

The percentage of dietary iron that is absorbed ranges from 2 percent in individuals with diseases of the gastrointestinal tract to 35 percent in rapidly growing, healthy children (Goyer, 1991; Whitney et al., 1990). Normally, 10 to 15 percent of dietary iron is absorbed, but this percentage varies to compensate for the level of iron in the body (Elinder, 1986). For example, patients with iron-deficiency anemia can absorb as much as 60 percent of an oral dose of iron (Josephs, 1958).

Iron absorption also is influenced by factors such as the source and chemical form of the ingested iron, other substances in the diet, and the condition of the gastrointestinal tract (Elinder, 1986). Very little is known about the absorption of iron from water and about the chemical species of iron in drinking water from the tap. Although the amount of ferric ion (Fe^{3+}), ferrous ion (Fe^{2+}), and organic complexes of iron in water that are absorbed by humans is unknown, a reducing agent such as ascorbic acid clearly increases the iron absorption in food (NRC, 1980). Ferrous ion appears to have better availability than does ferric ion. Iron from animal sources is absorbed by humans more effectively than iron from vegetables and grains. Soluble forms of iron such as iron sulfate are taken up more readily than insoluble forms such as iron oxide. The presence of other metals also affects iron absorption. Absorption is decreased in the presence of high levels of phosphate, cobalt, copper, and zinc (Elinder, 1986). Excess manganese can substantially decrease iron absorption by impairing hemoglobin regeneration in the blood (NRC, 1980).

Tissue accumulation and clearance

Iron absorption from the gastrointestinal tract occurs in two steps: first, ferrous ions from the intestinal lumen are absorbed into the mucosal cells. Second, they are transferred from the mucosal cells to plasma, where they are bound to transferrin for transfer to storage sites. As ferrous ion is released into plasma, it is oxidized by oxygen in the presence of ferroxidase (Goyer, 1991).

Normally, the adult human body contains about 3 to 5 grams of iron. Two-thirds of this amount is found in the blood, bound to hemoglobin. Less than 10 percent of the body iron is found in myoglobin and iron-requiring enzymes. About 20 to 30 percent of the remaining iron in the body pool is bound to iron-storage proteins in liver, bone marrow, and spleen (Elinder, 1986). Under normal conditions, the total elimination of iron from the body is limited to 0.6 to 1.0 mg/day, or roughly 0.01 percent of the body stores. Not counting iron not absorbed from the gut, about 0.2 to 0.5 mg of elemental iron per day is eliminated through the feces, about 0.1 to 0.3 mg/day in urine, and the remainder through normal dermal losses in sweat, hair, and nails. Based on these rates of elimination, the biological half-life of iron in the body is 10 to 20 years (Elinder, 1986).

Environmental sources of iron

The iron concentrations of liver, kidney, beef, ham, egg yolk, and soybeans are in the order of 30 to 150 mg/kg fresh weight. Grains and fruits are low in iron, usually ranging from 1 to 20 mg/kg. In both human and cow's milk, iron concentration is about 0.5 mg/L (Elinder, 1986).

Iron concentrations in water vary greatly. In the United States, the iron concentrations of freshwater and public water supplies range from 0.01 to 1.0 mg/L (Elinder, 1986). Assuming a 2 L/day consumption of water by a 70-kg adult, this range would result in an intake of 0.0003 to 0.03 mg/kg-day of iron from drinking water.

The average daily iron intake ranges from 9 to 35 mg/day (0.1 to 0.5 mg/kg-day for a 70-kg adult) (Elinder, 1986). Approximately 35 percent of dietary iron comes from meat, fish, and eggs, while 50 percent is supplied by cereals, root vegetables, and other foods of plant origin (NRC, 1980). The recommended dietary allowance (RDA) for iron is 10 mg for adult males (approximately 0.14 mg/kg-day for a 70-kg man) and 18 mg for females of reproductive age (approximately 0.25 mg/kg-day for a 70-kg woman) (NRC, 1980).

Toxicity of iron

Iron intoxication is most frequent in children aged 1 to 3 years due to their eating iron supplements formulated for adults in the form of ferrous sulfate tablets with candy-like coatings. Severe poisoning in children may occur following ingestion of more than 0.5 grams (approximately 22 mg/kg for a 22-kg child) of iron, about 2.5 grams (approximately 110 mg/kg for a 22-kg child) as ferrous sulfate. This acute iron poisoning has occurred in children who ingested as few as 6 iron tablets (Whitney et al., 1990). The iron damages the lining of the gastrointestinal tract, producing vomiting as the first symptom. Bleeding of the damaged gastrointestinal tissue frequently results in blood in the vomit and black stools (Goyer, 1991). Shock and metabolic acidosis can develop. If the patient survives the initial crisis, liver damage with hepatitis and coagulation defects often occur within a couple of days. Renal failure and cirrhosis of the liver may occur as delayed effects (Elinder, 1986).

Long-term intake of iron in a form that is readily absorbed and in doses exceeding 50 to 100 mg of iron per day (0.7 to 1.4 mg/kg-day for a 70-kg adult) results in an increased body burden of iron because iron is removed from the body at a much slower rate than it is absorbed (Elinder, 1986). As the body burden of iron increases to 20 to 40 grams (roughly 10 times the normal level), production of the iron-binding protein hemosiderin increases and results in a condition known as hemochromatosis. This condition starts with increased skin pigmentation and higher iron concentrations in the liver, pancreas, endocrine organs, and heart. This increased tissue iron can produce cirrhosis of the liver, disturbances in endocrine and cardiac function, and diabetes mellitus (Goyer, 1991).

Chronic iron toxicity in adults can be caused by genetic factors, excess dietary iron, excessive ingestion of iron-containing tonics or medicines, or multiple blood transfusions. The pathologic consequences of iron overload are similar regardless of basic cause (Goyer, 1991). Figure 5.3 summarizes the toxic effects and doses for iron.

5.1.4 Manganese

Absorption

Following ingestion, manganese absorption is homeostatically controlled: the rate of absorption depends on both the amount ingested and tissue levels of manganese. Adult humans absorb approximately 3 to 4 percent of dietary manganese (Saric, 1986). Manganese can be absorbed following exposure by inhalation, ingestion, and dermal contact. Available data indicate that humans absorb only 3 percent of an ingested dose of manganese chloride (Mena et al., 1969). Manganese in water appears to be more efficiently absorbed than manganese in foodstuff; the absorption rate is influenced by iron and other metals (EPA, 1994a). In states of iron deficiency, manganese is actively absorbed from the intestine. Individuals with anemia can absorb more than twice the percentage of an ingested dose. However, in states of excess iron, manganese absorption is by diffusion only (Saric, 1986). High levels of dietary calcium and phosphorus have been shown to increase the requirements for manganese in several species (Lönnerdal et al., 1987).

Tissue accumulation and clearance

Absorbed manganese is widely distributed throughout the body. The highest concentrations are found in the liver and kidney. The biological half time in humans is 2 to 5 weeks, depending on body stores. Manganese readily crosses the blood-brain barrier and is more slowly cleared from the brain than from other tissues (Goyer, 1991). Normal concentrations in the brain are low, but the half time in the brain is longer and the metal may accumulate in the brain with excessive absorption (NRC, 1973).

Absorbed manganese is rapidly eliminated from the blood and concentrates in mitochondria. Initial concentrations are greatest in the liver. Manganese penetrates the placental barrier in all species and is more uniformly distributed throughout the fetus than in adult tissues. It is secreted into milk. Absorbed manganese is almost totally secreted in bile and reabsorbed from the intestine as necessary to maintain body levels. At excessive exposure levels, other gastrointestinal routes may participate. Excess manganese is eliminated in the feces; urinary excretion is negligible (Goyer, 1991; Saric, 1986).

Environmental sources of manganese

On the whole, food constitutes the major source of manganese intake for humans. The highest manganese concentrations are found in plants, especially wheat and rice. Drinking water generally contains less than 0.1 mg/L. Manganese levels in soil

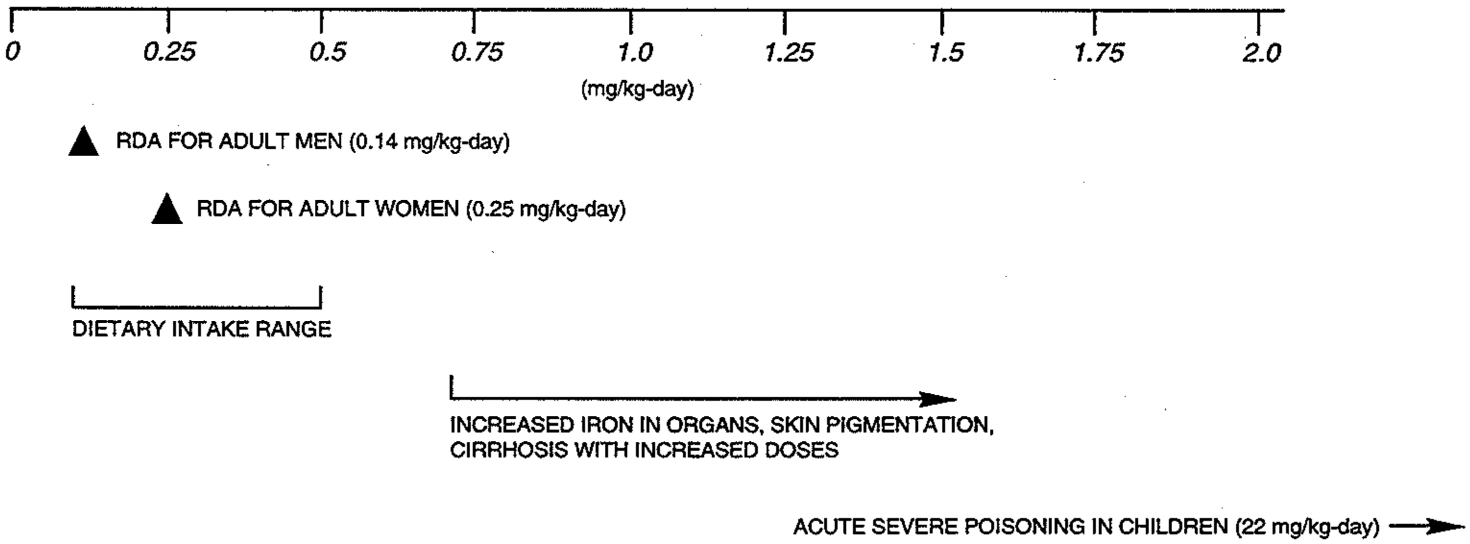


FIGURE 5.3
IRON TOXICITY RANGES

MAC: SITE/GUN/ BRA/IRN-GEN

range from 1 to 7000 mg/kg, with an average of 600 to 900 mg/kg. Mining and natural geological background variations can contribute to this variability. Manganese bioaccumulates in marine mollusks up to 12,000-fold, and there is evidence of toxic effects in plants (phytotoxicity) and plant bioaccumulation. The Illinois Institute for Environmental Quality has recommended a criterion of 1 to 2 mg/kg for manganese in soil and 200 mg/kg in plants (Saric, 1986).

Variations in manganese intake can be explained to a large extent by differences in nutritional habits. In populations that use cereals and rice as main food sources, manganese intake will be higher than in populations for which meat and dairy products make up a larger part of the diet. The average daily intake has been estimated to be between 2.0 to 8.8 mg/day (0.03 to 0.13 mg/kg-day for a 70-kg adult) (EPA, 1994a), but intakes as high as 12.4 mg (about 0.2 mg/kg-day for a 70-kg adult) have been reported in countries with high cereal intake (Saric, 1986).

Drinking water generally results in a daily intake of less than 0.2 mg (0.003 mg/kg-day for a 70-kg adult), although some mineral waters can increase this amount by more than three-fold (Saric, 1986). One study from Greece reported drinking water concentrations of manganese in excess of 2 mg/L, which would result in daily intakes in the range of 0.06 to 0.07 mg/kg-day (EPA, 1994a).

Toxicity of manganese

Manganese is an essential nutrient. Estimated safe and adequate daily dietary intakes for adults range from 0.03 to 0.07 mg/kg-day (Saric, 1986). The EPA no-observed-adverse-effect level (NOAEL) for drinking water is identified at 0.005 mg/kg-day while the lowest-observed-adverse-effect level (LOAEL) for drinking water sources is 0.06 mg/kg-day. The EPA oral RfD for drinking water of 0.005 mg/kg-day is based on human data (Kondakis et al., 1989). The study group was a population of older adults exposed to manganese in drinking water over a lifetime; because this population was considered sensitive, an uncertainty factor of 1 was applied (EPA, 1994a). The RfD for food ingestion is 0.14 mg/kg-day. There is some indication that manganese in drinking water is potentially more bioavailable, i.e., more readily absorbed, than is manganese in dietary food sources. This bioavailability would result in toxic effects at lower ingested doses of manganese in drinking water than in food (EPA, 1994a). However, insufficient data exist to quantify these dose differences.

Inhalation of manganese in industrial settings has provided the largest source of data on chronic manganese toxicity. These data indicate that excess manganese can result in a central nervous system disorder consisting of irritability, difficulty in walking, speech disturbances, and compulsive behavior that may include running, fighting, and singing. With continued exposure, this condition can progress to a mask-like face, retropulsion or propulsion, and a Parkinson-like syndrome. These effects are largely irreversible, although some recovery can be expected when exposure ceases (DHHS, 1992b). Metal chelating agents are ineffective in treatment, but L-dopa has been effective in treatment (Goyer, 1991).

Limited information is available on the effects of manganese ingestion. Because effects from drinking water seem to differ from those from food sources, only studies on water consumption will be considered here. A Japanese study of 25 people drinking well water with manganese concentrations of 14 mg/L (0.4 mg/kg-day estimated intake for a 70-kg adult drinking 2 L of water per day) reported symptoms of intoxication, including a mask-like face, muscle rigidity and tremors, and mental disturbances. Two cases (8 percent) of death were reported among the intoxicated people. A Greek study of over 4000 adults at least 50 years old drinking water with manganese concentrations varying from 0.081 to 2.3 mg/L (estimated intake at 2 L/day for a 70-kg individual range from 0.002 to 0.07 mg manganese/kg-day) showed varying degrees of neurological effects in individuals drinking from 0.007 to 0.07 mg manganese/kg-day; no effects were reported in individuals drinking less than 0.005 mg/kg-day (Kondakis et al., 1989). However, the many limitations to these studies make data interpretation difficult. Among the limitations is uncertainty regarding the exposure levels or whether the effects seen were solely attributable to manganese. Despite these limitations, the similarity of the effects seen in the cases of oral exposure compared with those associated with inhalation exposure suggests that excess manganese intake by humans might lead to neurological injury (DHHS, 1992b).

The chemical form of manganese has complex effects on its toxicity. Although more soluble forms are more readily absorbed from the gastrointestinal tract, they also appear to be more rapidly cleared. Exposure to insoluble forms results in lower manganese absorption, but higher chronic tissue levels and therefore greater toxicity (EPA, 1994a). Only limited information is available on the effects of various forms of manganese.

Few data are available on manganese toxicity in infants, but it is likely that infants will be more susceptible to toxicity due to greater absorption and greater penetration into the central nervous system (EPA, 1994a; Saric, 1986). Figure 5.4 summarizes the toxicity of manganese from drinking water exposure.

5.1.5 Sulfate

Absorption

Following oral ingestion, soluble salts of sulfate are well absorbed from the intestine (about 90 percent at low doses, i.e., less than 50 mg/kg) and distributed throughout the body (EPA, 1992a). At higher doses (50 to 100 mg/kg body weight) sulfate is incompletely absorbed (about 60 to 70 percent), which results in diarrhea.

Tissue accumulation and clearance

In humans, sulfate is a normal component of both extracellular and intracellular fluids (EPA, 1992a).

Ingesting high levels of sulfate results in transient increases in both blood and urine concentrations (EPA, 1992a). Inorganic sulfate is eliminated from the body almost

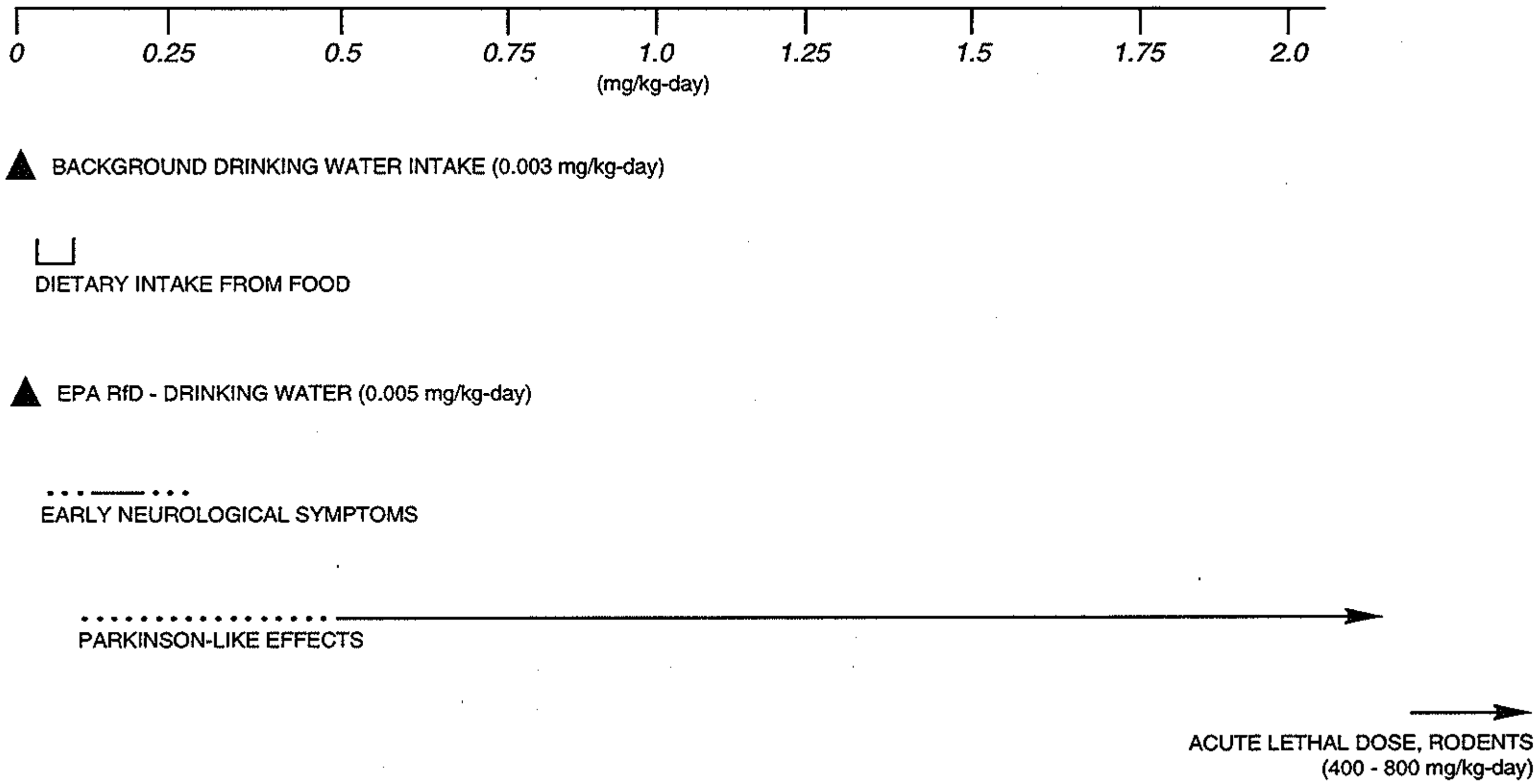


FIGURE 5.4
MANGANESE TOXICITY RANGES

entirely in urine without biotransformation (Morris and Levy, 1983). Approximately 50 percent of a 75-mg/kg dose is excreted over 72 hours (EPA, 1992a). The urinary excretion mechanism is transport-limited and can become saturated at high doses of sulfate. Excess sulfate may be excreted in feces in its inorganic form. To date, no data indicate that sulfate accumulates, even with chronic ingestion of above-normal levels.

Sulfate is used to biosynthesize collagen, cartilage, and dentin and to form sulfate esters of both endogenous compounds (such as lipids and steroids) and exogenous compounds (such as phenols). Sulfation is important in detoxication pathways because it increases the solubility of these compounds, enhancing their excretion in the urine. Exposure to high concentrations of compounds that are conjugated with sulfate and excreted can produce a transient decrease in plasma sulfate concentrations.

Environmental sources of sulfate

In 1978, drinking water sulfate concentrations in the western United States ranged from 0 to 820 mg/L, with a mean concentration of 99 mg/L (which corresponds to sulfate daily intake of up to about 23 mg/kg-day for a 70-kg man drinking 2 L of water per day, with an average of about 3 mg sulfate/kg-day) (EPA, 1992a). The highest sulfate concentrations in drinking water of 1110 mg/L were reported in California (EPA, 1992a). This amount of sulfate in drinking water can result in a daily intake of 32 mg/kg-day (assuming that a 70-kg man ingests 2 L of water daily). The EPA estimates a normal sulfate intake range of 0.00023 to 0.0064 mg/kg-day from air. Estimates on sulfate intake from food are not available.

Toxicity of sulfate

Little information is available on the toxic effects of sulfate on humans (EPA, 1992a). There are no health problems reported following chronic exposure to high concentrations of sulfate. The effects of the sulfate ion are limited to its laxative effect following massive short-term exposure (EPA, 1992a).

Sulfate salts of magnesium and sodium are used medicinally as cathartics. High concentrations of unabsorbed sulfate salts in the gut can pull large amounts of water into the gut, greatly increasing the normal volume of feces. This action is also the basis of sulfate's toxic effects. Ingestion of excessive doses of cathartics without corresponding water ingestion leads to dehydration (EPA, 1992a). Persistent diarrhea may result in severe dehydration and hypovolemic shock, particularly in infants and children (Casarett and Doull, 1991). Extreme dehydration may lead to death.

Toxicity in humans is primarily manifested in diarrhea; the severity of the diarrhea is dose-dependent. The effect is reversible and diarrhea discontinues after cessation of exposure. About 5 grams of sodium sulfate or magnesium sulfate in a single dose produces diarrhea in most adults (Chien et al., 1968). This corresponds to sulfate intake of about 30 mg/kg. It is generally accepted that cathartic effects are commonly experienced by people introduced to drinking water with sulfate concentrations above 600 mg/L (equivalent to above 17 mg/kg-day for a 70-kg man drinking 2 L of water a day) (Chien et al., 1968; EPA, 1992a). Such water is usually a little bitter. If only taste of water is considered, sulfate water concentration should not exceed 400 mg/L (Chien et al., 1968; EPA, 1992a). In regions with high sulfate concentrations in the drinking water, such as Saskatchewan where well water may contain from 400 to 1000 mg/L of sulfate (from about 11 to 29 mg/kg-day for a 70-kg man drinking 2 L of water a day), residents adapt to the taste and find the water palatable (Chien et al., 1968). They also become immune to the laxative action of these levels of sulfate in their drinking water. These results appear to be consistent with data from North Dakota. A survey of 248 private well users in North Dakota indicated that sulfate concentration of 1,000 to 1,500 mg sulfate per liter (equivalent to 29 to 43 mg sulfate/kg-day for a 70-kg man drinking 2 L of water a day) caused diarrhea in 62 percent of respondents, as noted in 1952 by Moore (EPA, 1992a). In those exposed to sulfate concentrations from 200 to 500 mg/L, approximately 20 percent reported laxative effects; however, no dose-response was observed in this group. Infants appear to be the most susceptible population for sulfate-induced diarrhea. Also, some data indicate that diabetic and elderly populations with compromised kidney function may be more sensitive than healthy adults to the effects of sulfates (EPA, 1992a).

Three infants were reported to develop diarrhea from sulfate in drinking water at concentrations from 475 to 680 mg/L (equivalent to 80 to 114 mg/kg-day for a 4-kg infant drinking 0.67 L of water a day) (Chien et al., 1968). These infants recovered from diarrhea when water with high sulfate levels was replaced with "safe water" (sulfate concentration was not reported). Three other cases of severe diarrhea in infants (5, 10, and 12 months old) were reported following ingestion of well water with sulfate content of 630, 720, and 1150 mg/L (Chien et al., 1968). These levels of sulfate correspond to sulfate intakes of about 106, 121, and 193 mg/kg-day, assuming that a 4-kg infant drinks 0.67 L of water a day. In two cases, diarrhea developed promptly; the third infant developed persistent diarrhea several days after the family moved to the area with high sulfate content in drinking water. In all cases, recovery was dependent upon substitution of water low in sulfate for the well water. Also, in all there cases infectious etiology of diarrhea was excluded; however, neither the nature nor the concentrations of the other constituents present in the water were specified. Adults (parents) drinking water with sulfate content of 630 and 720 mg/L (18 and 21 mg/kg-day for a 70-kg man drinking 2 L of water daily) showed no health problems. However, intermittent diarrhea developed in adults and children (age unknown) drinking water with 1150 mg sulfate per liter (equivalent to 33 mg/kg-day for a 70-kg man drinking 2 L of water daily). The onset of the diarrhea was about two weeks after the family moved to the area. Figure 5.5 summarizes these health effects as a function of dose.

Sulfate toxicity data are based primarily on epidemiologic studies of human adults and infants who report to hospitals with symptoms of sulfate exposure. In most

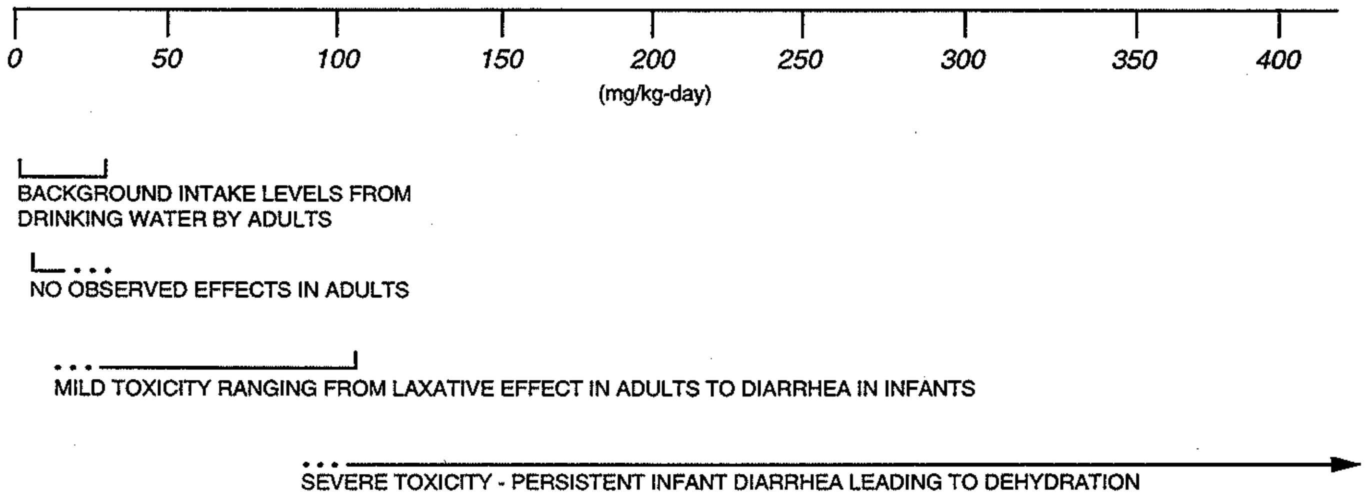


FIGURE 5.5
SULFATE TOXICITY RANGES

MAC: SITE/GUN/BRA/SUL-GEN

cases, exposure doses have been back-calculated from sampling their drinking water. Therefore, these data do not represent well-controlled studies with readily defined dosage ranges.

Based on the study by Chien et al. (1968) and data from North Dakota evaluated by Moore in 1952 (EPA, 1992a), the EPA has proposed the sulfate primary drinking water standard of 500 mg/L (40 CFR Parts 141-143*)*.

5.1.6 Uranium

Naturally occurring uranium, present at UMTRA Project sites, consists of three radioactive isotopes: uranium-234, uranium-235, and uranium-238. More than 99 percent of natural uranium occurs in the form of uranium-238 (Cothern and Lappenbusch, 1983). Uranium-238 undergoes radioactive decay by emitting alpha particles to form uranium-234, thorium-230, radium-226, radon 222, polonium-210, and other radioisotopes. Figure 5.6 summarizes the radioactive decay chain of uranium-238 and uranium-234 is summarized in Figure 5.6. As all uranium isotopes in nature are radioactive, the hazards of a high uranium intake are from both its chemical toxicity and potential radiological damage. This section focuses on the chemical toxicity of natural uranium. Carcinogenic potential associated with exposure to radioactive isotopes of natural uranium is discussed in Section 5.3.

Absorption

Absorption of uranium in the gastrointestinal tract depends on the solubility of the uranium compounds. The hexavalent uranium compounds, especially the uranyl salts, are water soluble, while tetravalent compounds generally are not (Weigel, 1983). However, only a small fraction of the soluble compounds is absorbed. Wrenn et al. (1985) have determined human gastrointestinal absorption rates of 0.76 to 7.8 percent. Uranium may absorb through the skin when applied in concentrated solutions (the concentration level was not reported). The extent of absorption appears to be dose-dependent.

Tissue accumulation and clearance

In humans exposed to background levels of uranium, the highest concentrations of uranium were found in the bones, muscles, lungs, liver, and kidneys (Fisenne et al., 1988). Uranium retention in bone consists of a short retention half time of 20 days, followed by a long retention half time of 5000 days for the remainder (Tracy et al., 1992).

In body fluids, uranium tends to be converted into water-soluble hexavalent uranium (Berlin and Rudell, 1986). Approximately 60 percent of the uranium in plasma complexes with low-molecular-weight anions (e.g., bicarbonates, citrates), while the remaining 40 percent binds to the plasma protein transferrin (Stevens et al., 1980). Following oral exposure to uranium, humans excrete more than 90 percent of the dose in the feces. Of the small percent that is not absorbed (typically less than 5 percent), animal studies show that approximately 60 percent is excreted through the

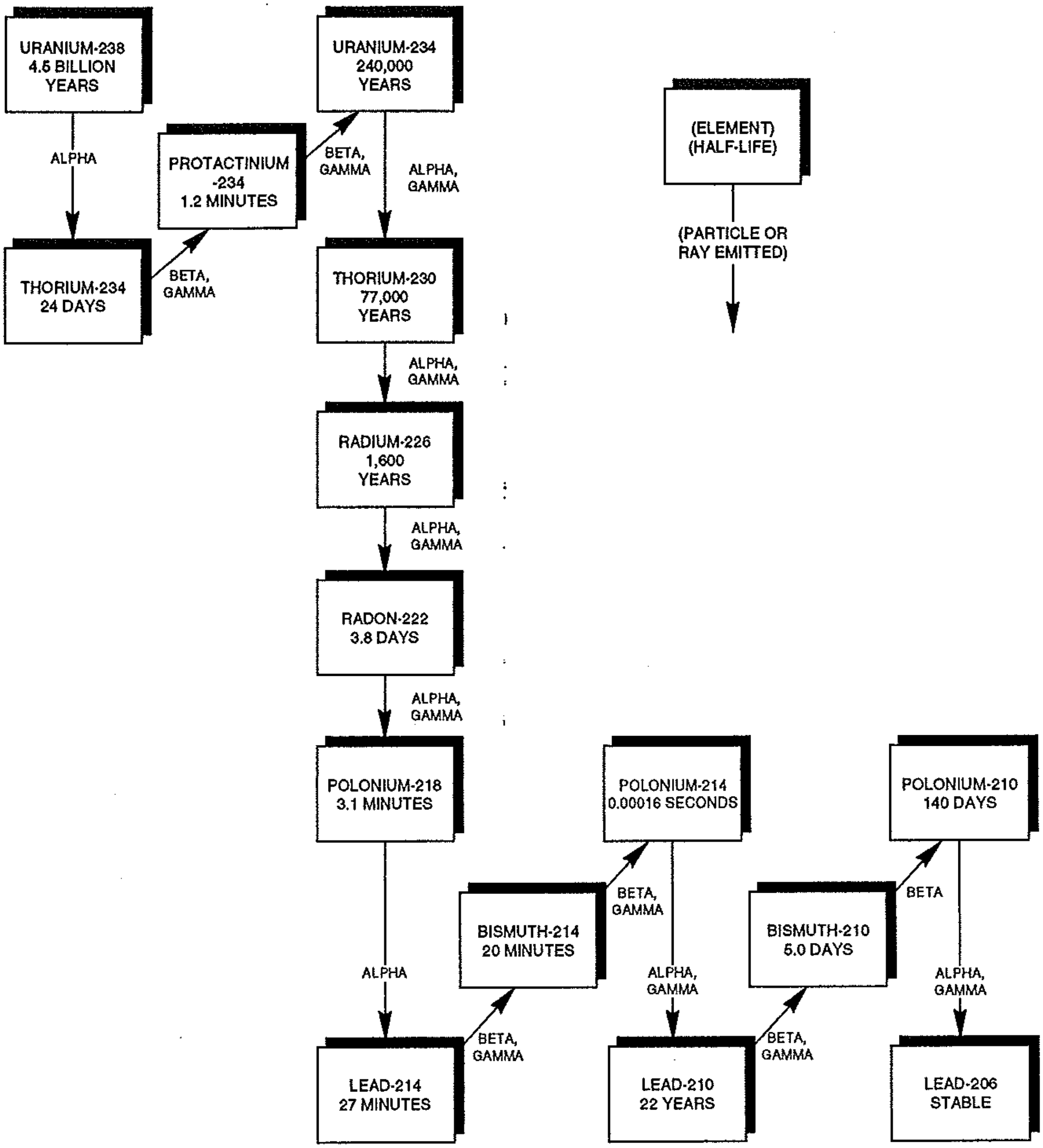


FIGURE 5.6
HALF-LIVES AND EMISSIONS FROM DECAY CHAIN OF URANIUM-238
GUNNISON, COLORADO, SITE

urine within 24 hours, whereas the remainder is distributed to the skeleton and soft tissue; 98 percent of that amount is excreted within 7 days (Ballou et al., 1986; Leach et al., 1984; Sullivan et al., 1986). A small portion of the absorbed uranium is retained for a longer period.

Environmental sources of uranium

Uranium is a ubiquitous element, present in the earth's crust at approximately 4 parts per million. Uranium concentrations in ground water and surface water averaged 1 and 3 pCi/L, respectively (equivalent to 0.001 and 0.004 mg/L, assuming 1 mg of uranium equals 686 pCi) (NCRP, 1984; DOE, 1996). It is absorbed from the soil into plant tissues to an extent that depends on the plant species and the depth of its root system (Berlin and Rudell, 1986). Plant concentrations of uranium averaged 0.075 µg/kg of fresh plant material (Tracy et al., 1983).

The main dietary source of natural uranium for the general population is food products such as potatoes, bakery products, meat, and fresh fish, which may contain uranium concentrations between 10 and 100 µg/kg (Prister, 1969). The total dietary intake of uranium from the consumption of average foods is approximately 1 µg/day; approximately 20 to 50 percent of the total can come from drinking water. Cereals and vegetables, particularly root crops, are likely to contribute most to the daily intake of uranium (Berlin and Rudell, 1986).

Toxicity of uranium

No human deaths have been reported that are definitely attributable to uranium ingestion; therefore, no lethal dose has been determined for humans. Lethal doses of uranium ($LD_{50,23}$) have been reported to be as low as 14 mg/kg-day following 23-day oral exposures, depending on the solubility of the uranium compound tested (higher solubility compounds have greater toxicity), route of exposure, and animal species (Friberg et al., 1986). High doses of uranium cause complete kidney and respiratory failure.

No chronic toxic effects have been reported in humans following oral exposure to uranium. Data available from populations occupationally exposed to high concentrations of uranium compounds through inhalation and information from studies in experimental animals indicate that the critical organ for chronic uranium toxicity is the proximal tubule of the kidney (Friberg et al., 1986). In humans, chemical injury reveals itself by increased catalase excretion in urine and proteinuria. Dose-response data for the toxic effect of uranium on the human kidney are limited.

The LOAEL of uranyl nitrate that caused moderate renal damage was given to rabbits in diet at 2.8 mg/kg-day (Maynard and Hodge, 1949). The EPA oral RfD of 0.003 mg/kg-day was derived based on this study (EPA, 1994a). The EPA applied an uncertainty factor of 1000 to the LOAEL (2.8 mg/kg-day), which reflects intraspecies and interspecies variability and an uncertainty associated with the use of a LOAEL rather than the preferred use of a NOAEL. Figure 5.7 summarizes the health effects for uranium as a function of dose.

5.2 CONTAMINANT INTERACTIONS

The primary concern for interaction between constituents at Gunnison is between manganese and iron. Excess manganese impairs hemoglobin regeneration in the blood and thereby substantially decreases absorption of iron (NRC, 1980). However, excess iron inhibits the absorption of manganese (Saric, 1986). Even though an excess of either constituent can inhibit the absorption of the other, the net result of a substantial excess of the two in combination is not clear.

In addition to interactions between manganese and iron, interactions between several similar metals can alter the predicted absorption, distribution in the body, metabolism, toxicity, or clearance of a metal of interest. Cobalt can decrease iron absorption, and cadmium absorption can be considerably increased under conditions of low intake of calcium, iron, or protein (Nordberg et al., 1985). Low body-iron stores, although unlikely in people drinking potentially contaminated ground water from a hypothetical well from the Gunnison site, can increase cadmium uptake fourfold (Flanagan et al., 1978). Because cadmium binds strongly to metallothionein and its toxicity depends on this binding, metals that increase metallothionein concentrations will increase cadmium binding and potentially its toxicity. Metallothionein can be induced or increased by exposure to a wide range of other metals, including copper and zinc. However, in the continued presence of these other metals, there may be competition for metallothionein binding sites.

In animal studies, iron status affected uranium absorption (EPA, 1989a). The common target organ for uranium suggests interaction with cadmium in the production of kidney toxicity.

Sulfate, although not known to physiologically interact directly with any of the other constituents, induces diarrhea, which can alter the elimination of other toxicants. This might be expected to be an important factor in the elimination and reabsorption of manganese, which occurs almost solely in the intestine.

5.3 CONTAMINANT RISK FACTORS

The EPA Office of Research and Development has calculated acceptable intake values, or RfDs, for long-term (chronic) exposure to noncarcinogens. These values are estimates of route-specific exposure levels that would not be expected to cause adverse health effects when exposure occurs for a considerable portion of a lifetime. The RfDs include safety factors to account for uncertainties associated with limitations of the toxicological database, including extrapolating animal studies to humans and accounting for variability in response from sensitive individuals. These values are updated quarterly and published in the Health Effects Assessment Summary Tables (HEAST) (EPA, 1994b) and are also provided through the EPA's Integrated Risk Information System (IRIS) database (EPA, 1994a). Table 5.1

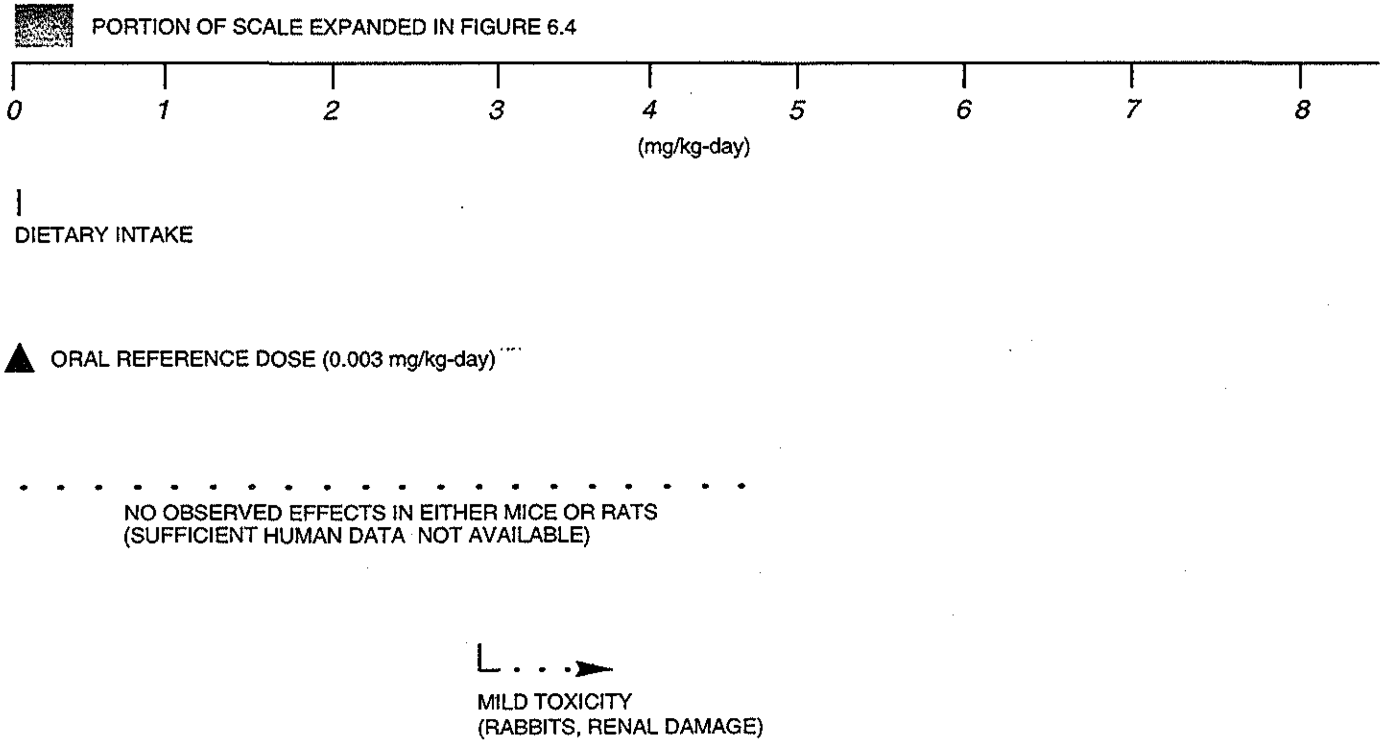


FIGURE 5.7
NONCARCINOGENIC URANIUM TOXICITY RANGES

MAC: SITE/GUN/BRA/URN-GEN

Table 5.1 Toxicity values: potential noncarcinogenic effects

Chemical	Chronic oral RfD[a] (mg/kg-day)	Confidence level	Critical effect/organ	RfD basis/RfD source[b]	Uncertainty factor
Cadmium	5E-4	High	Kidney (proteinuria)	Water/IRIS	10
Cobalt	ND	NA	Cardiomyopathy, goiter	NA	NA
Iron	ND	NA	Increased skin pigmentation, cirrhosis of the liver	NA	NA
Manganese	5E-3	NA	Central nervous system effects	Water/IRIS	1
Sulfate	ND	NA	Diarrhea	NA	NA
Uranium (soluble salts)	3E-3	Medium	Nephtotoxicity, decreased body weight	Diet/IRIS	1000

[a] The chronic oral RfD for manganese is adopted as subchronic oral RfD; subchronic oral RfDs have not been determined for cadmium and uranium.

[b] From EPA, 1994a.

ND - not determined.
NA - not applicable.

summarizes the most recent oral RfDs for the noncarcinogenic constituents of concern. RfDs for iron, cobalt, and sulfate have not been determined.

The EPA currently classifies all radionuclides as Group A, or known human carcinogens, based on their property of emitting ionizing radiation and on the evidence provided by epidemiological studies of radiation-induced cancer in humans. At sufficiently high doses, ionizing radiation acts as a complete carcinogen (both as initiator and promoter), capable of increasing the probability of cancer development. However, the actual risk is difficult to estimate, particularly for the low doses and dose rates encountered in the environment. Most reliable data were obtained under conditions of high doses delivered acutely. It is not clear whether cancer risks at lower doses are dose-proportional (i.e., the linear dose-response hypothesis) or whether the risk is greatly reduced at low doses and rates (the threshold hypothesis). A conservative assumption is that no threshold dose exists below which there is no additional risk of cancer.

Risk factors are published in HEAST and IRIS for correlating intake of carcinogens over a lifetime with the increased excess cancer risk from that exposure. Table 5.2 gives cancer slope factors (SF) for the uranium-234/-238 radioactive decay series.

Table 5.2 Toxicity values: carcinogenic effects

Parameter	Oral SF[a] $(pCi)^{-1}$	Weight of evidence classification	Type of cancer	SF basis/SF source[b]
Lead-210[c]	1.01E-09	A	Bone	HEAST
Polonium-210	3.26E-10	A	Liver, kidneys, spleen	HEAST
Radium-226[c]	2.96E-10	A	Bone	HEAST
Thorium-230	3.75E-11	A	d	HEAST
Uranium-238[c]	6.20E-11	A	e	HEAST
Uranium-234	4.44E-11	A	e	HEAST

[a] For each individual radionuclide listed, oral SFs correspond to the risks per unit intake (risk/pCi) for that radionuclide, except as noted.

[b] From EPA, 1994b.

[c] Oral SF includes the contribution from short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal nuclide in the environment.

[d] Target organ systems have not been identified for oral exposure to thorium.

[e] No human or animal studies have shown a definite association between oral exposure to uranium and development of cancer.

A - known human carcinogen.

6.0 HUMAN HEALTH RISK EVALUATION

To evaluate human health risks to an individual or population, the results of the exposure assessment are combined with the results of the toxicity assessment. As discussed in Section 5.0, potential adverse health effects are a function of how much of the constituent an individual takes into his or her body. Indeed, at lower levels some of the constituents associated with the mill tailings are beneficial to health, since they are essential nutrients. At higher levels, these same elements can cause adverse health effects. In addition, some individuals may be more sensitive to a given constituent at the same level of exposure than others. In this section, the expected intake, if ground water from a hypothetical well within the most contaminated portion of the plume were used as drinking water, is correlated to potential health effects from these levels of exposure.

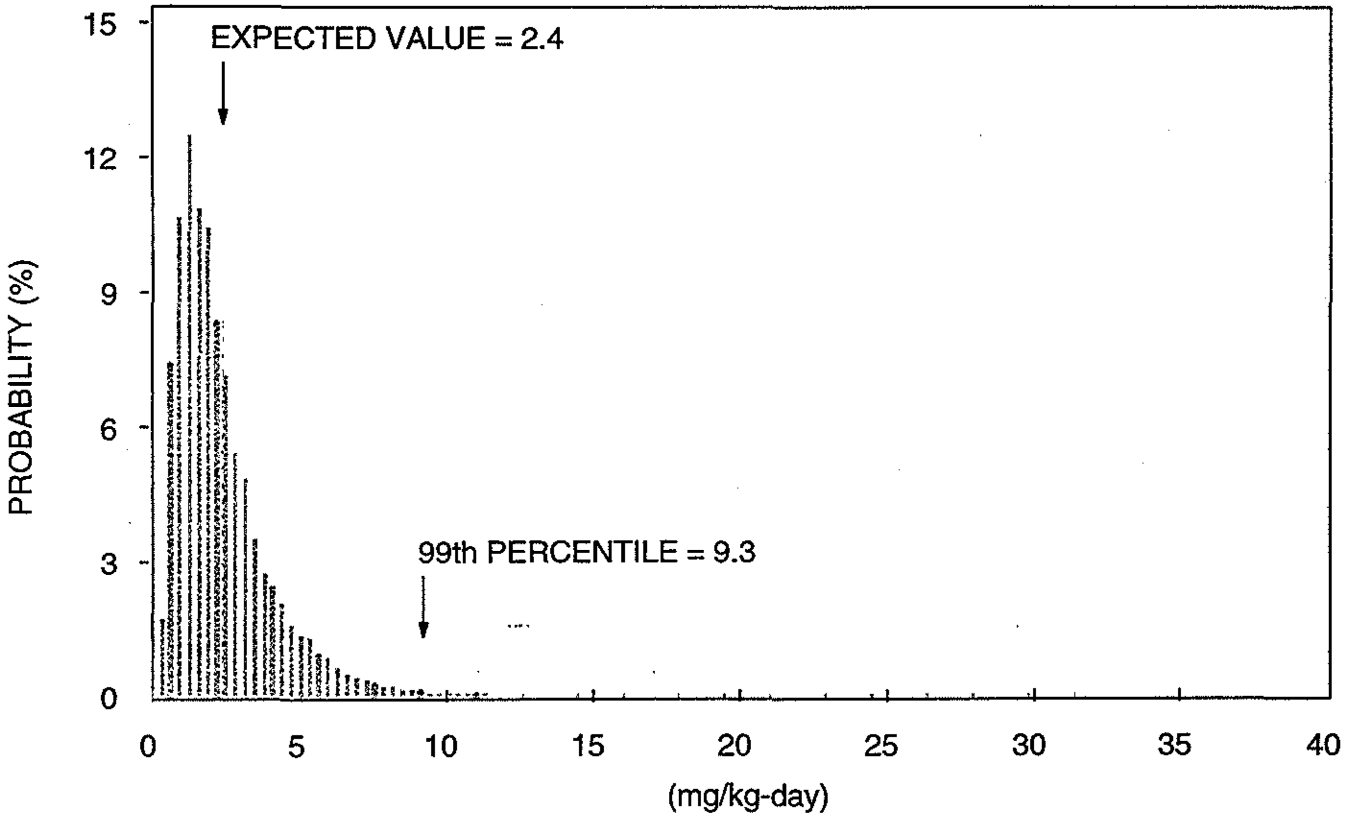
6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

The results from the exposure assessment showing either the highest intake-to-body-weight ratios (or highest doses) or the toxicologically most sensitive group are used to evaluate potential health effects for noncarcinogens. Infant exposures are used to evaluate the health risks of sulfate exposure, because infants are the most toxicologically sensitive population. Although infants appear to be more sensitive to manganese toxicity than other population groups, insufficient data are available to support a quantitative analysis. For cadmium, cobalt, iron, manganese, and uranium, the highest intake-to-body-weight group (children 1 to 10 years old) is evaluated.

The primary concerns for human health from ingesting contaminated ground water at the Gunnison site are from exposure to iron and manganese. Greater than 50 percent of the potential exposure range for iron is above the dose that produces chronic iron toxicity, including increased pigmentation of the skin and potential disruption of liver and endocrine function (Figure 6.1). Long-term exposure to these concentrations of iron could also result in cirrhosis of the liver and/or development of diabetes. A very low percentage of the distribution exceeds the dose that produces acute severe poisoning in children.

The potential exposure range for manganese is greater than the levels reported to produce early neurological signs of manganese toxicity, such as memory loss, irritability, and muscle rigidity. Greater than 50 percent of the distribution falls above levels that have been reported to produce Parkinson's-like effects, including tremors, following chronic exposure through drinking water (Figure 6.2). As discussed in Section 5.1.4 and earlier in this section, infants may be more susceptible to manganese toxicity than children and adults. Therefore, toxic effects may appear in infants at levels lower than those in children.

Iron and manganese are known to interact by inhibiting the absorption of the opposite metal. However, the net effect of this interaction cannot be predicted with both metals in such high concentrations. Although the other pathways screened in Section 4.2 could contribute an additional 4 percent to the potential exposure from



NOTE: THE MAJORITY OF THE POTENTIAL IRON EXPOSURE RANGE FALLS ABOVE THE LEVEL WHERE TOXIC EFFECTS ARE EXPECTED. A VERY SMALL (<1%) PORTION OF THE POTENTIAL DAILY EXPOSURE RANGE FALLS ABOVE THE LEVEL ASSOCIATED WITH ACUTE POISONING IN CHILDREN.

▲ RDA FOR ADULT MEN (0.14 mg/kg-day)

▲ RDA FOR ADULT WOMEN (0.25 mg/kg-day)

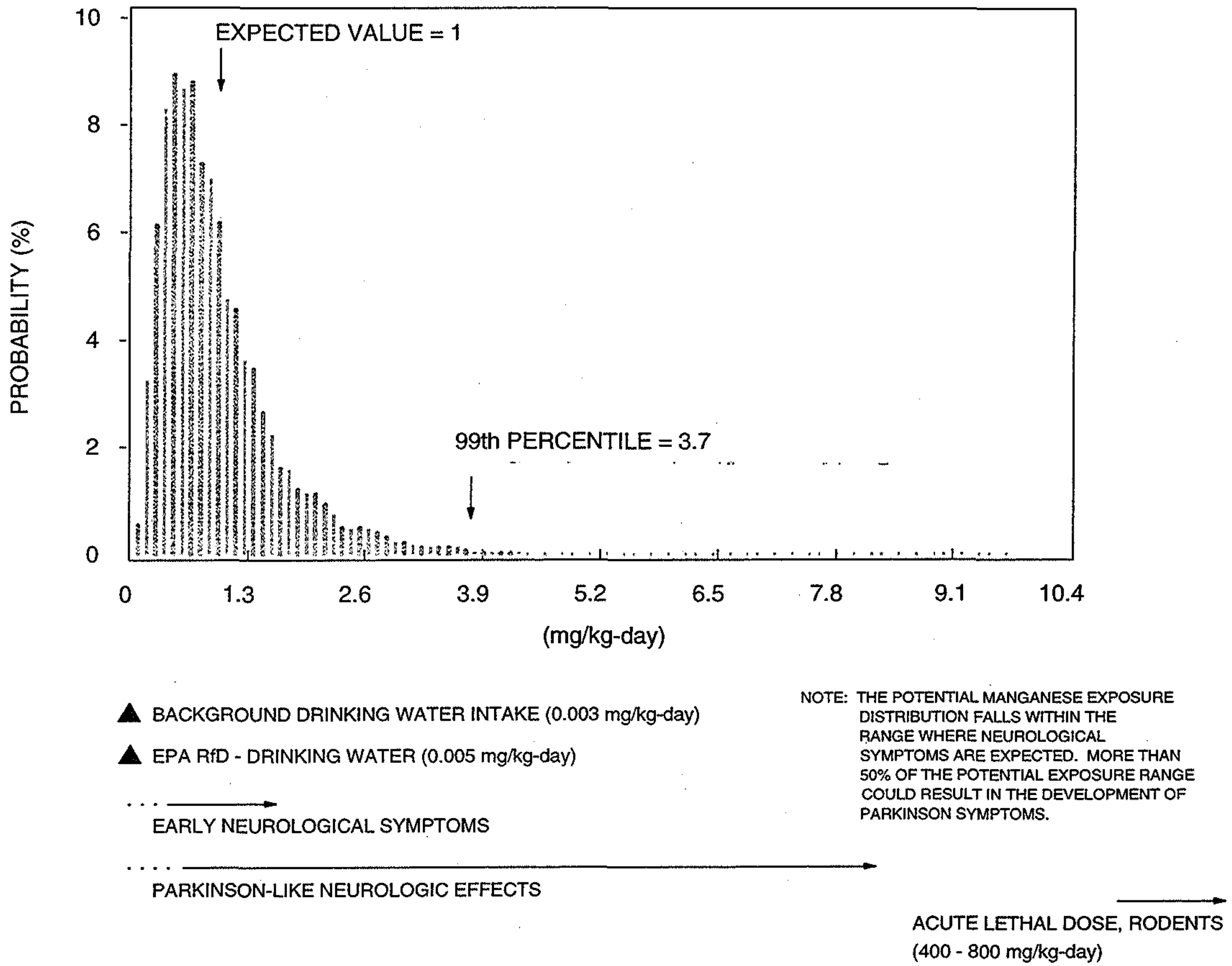
⊔ DIETARY INTAKE RANGE

…⟶

INCREASED IRON IN ORGANS, SKIN PIGMENTATION, CIRRHOSIS WITH INCREASED DOSE

▲ ACUTE SEVERE POISONING IN CHILDREN (22 mg/kg)

FIGURE 6.1
HEALTH EFFECTS OF POTENTIAL IRON EXPOSURE RANGES FOR CHILDREN



NOTE: THE POTENTIAL MANGANESE EXPOSURE DISTRIBUTION FALLS WITHIN THE RANGE WHERE NEUROLOGICAL SYMPTOMS ARE EXPECTED. MORE THAN 50% OF THE POTENTIAL EXPOSURE RANGE COULD RESULT IN THE DEVELOPMENT OF PARKINSON SYMPTOMS.

FIGURE 6.2
HEALTH EFFECTS OF POTENTIAL MANGANESE EXPOSURE RANGES FOR CHILDREN

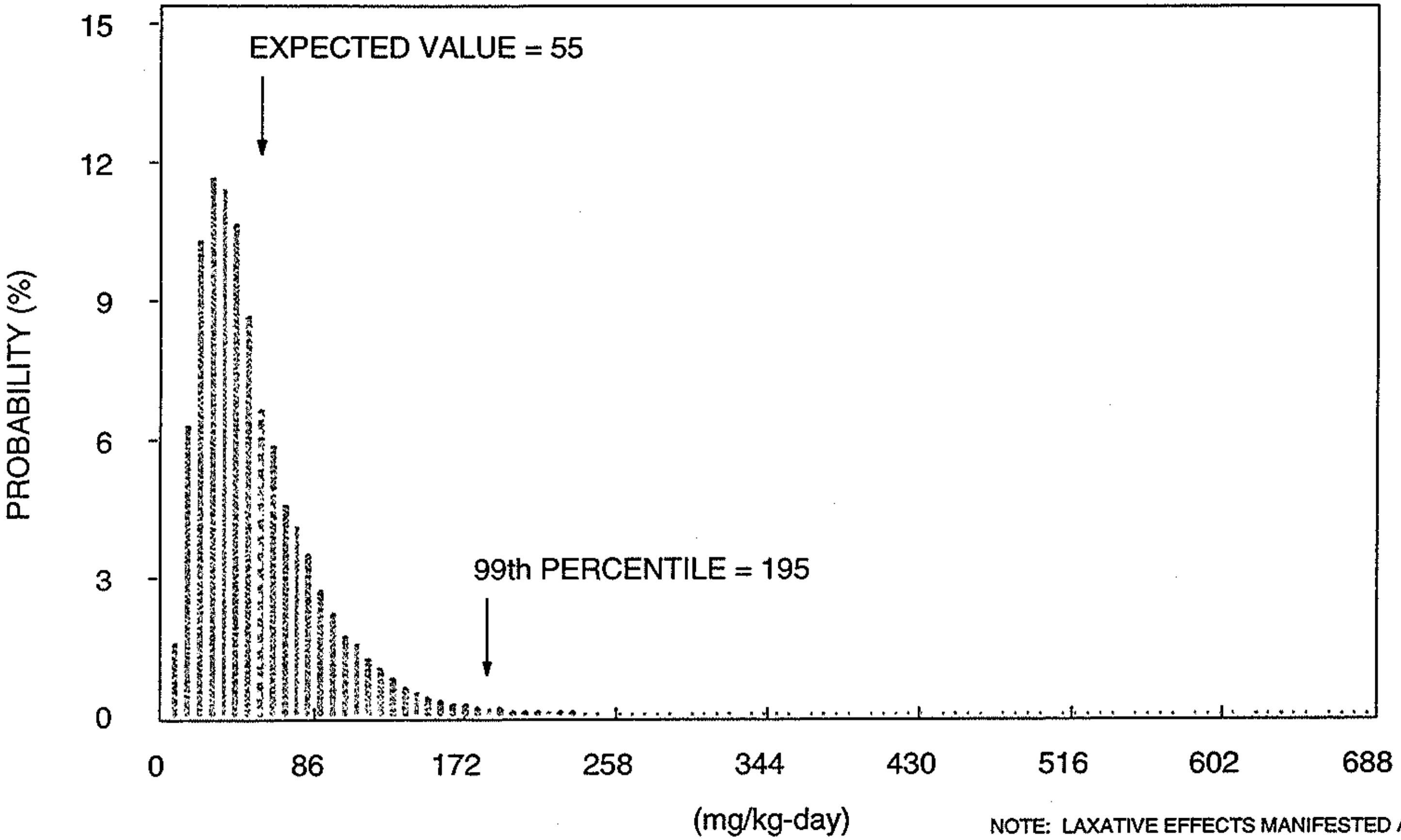
drinking contaminated water, this increase in the potential exposure would not notably increase the risk over that predicted from the drinking water pathway alone.

The potential range of sulfate exposure from drinking contaminated groundwater is mostly above the range expected to produce mild toxicity ranging from laxative effects in adults to diarrhea in infants (Figure 6.3). Nearly 50 percent of the potential exposure range falls within the range of exposures that could result in severe persistent diarrhea in infants. The effect is reversible and diarrhea would terminate on the cessation of exposure. Therefore, drinking this water by infants, children, and adults could preclude long-term exposure. However, because different ground water constituents flush out at different rates and because remedial action strategies may differ for different constituents, the effects from long-term exposures to constituents other than sulfate are evaluated. Sulfate exposure from the other pathways discussed in Section 4.2 is not expected to increase risk over that from groundwater ingestion alone.

For the noncarcinogenic effects of uranium, the entire exposure distribution falls above the oral RfD, but below ranges resulting in adverse effects in animal studies (Figure 6.4). Animal data are not always predictive of human toxicity, however, and the lack of available human data to evaluate oral toxicity of uranium at these concentrations should not be seen as an indication that no toxic effects will occur. RfDs incorporate safety and uncertainty factors and therefore are generally conservative values designed to be protective of human health. However, they are based on careful evaluation of existing databases; therefore, exposures that considerably exceed the RfD should be considered as potential problems. Diabetics and elderly may be more sensitive than healthy individuals to uranium's toxic effects on the kidney, but no information exists to support quantitative analysis. The contribution of uranium exposure from the other pathways would be less than 1 percent of the drinking water pathway and again would not alter the interpretation of risk.

The ranges of potential exposures to cadmium (Figure 6.5) and cobalt (Figure 6.6) are both below any potential toxic effects. The addition of the approximately 7 percent contribution from other sources to the cadmium exposure and approximately 6 percent additional cobalt exposure (Section 4.2) would still not result in potentially toxic effects from these constituents.

To determine whether human ingestion of ground water-irrigated garden produce and milk and meat from cattle watered with contaminated ground water could result in adverse health effects, the total exposure doses for the constituents of potential concern were compared to the EPA-established oral RfDs (Table 6.1). Table 6.1 shows that none of the total exposure doses exceeds the comparison criterion. Therefore, if exposure from these pathways occurred simultaneously and these pathways were the only source of human exposure to constituents of potential concern, no adverse health effects would be expected.



NOTE: LAXATIVE EFFECTS MANIFESTED AS DIARRHEA IN INFANTS COULD OCCUR WITHIN THE LOWER RANGE OF POTENTIAL EXPOSURES, WITH SEVERE DEHYDRATION IN INFANTS OCCURRING AT HIGHER POTENTIAL EXPOSURES.

└─┘ BACKGROUND INTAKE LEVELS FROM DRINKING WATER BY ADULTS

└── . . . NO OBSERVED EFFECTS IN ADULTS

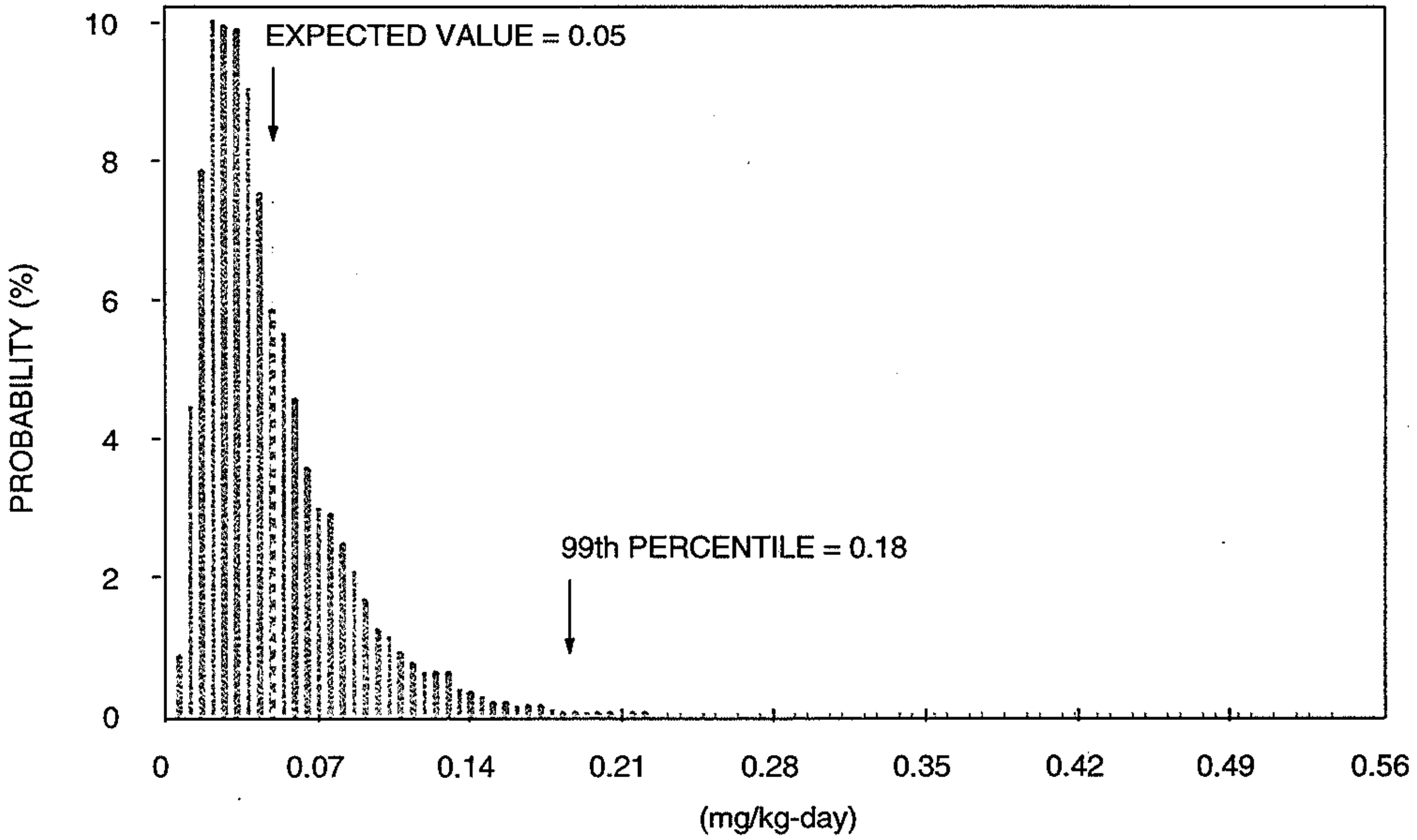
. . . ──────┘

MILD TOXICITY RANGING FROM LAXATIVE EFFECT IN ADULTS TO DIARRHEA IN INFANTS

. . . ──────────────────→

SEVERE TOXICITY - PERSISTENT INFANT DIARRHEA LEADING TO DEHYDRATION

FIGURE 6.3
HEALTH EFFECTS OF POTENTIAL SULFATE EXPOSURE RANGES FOR INFANTS



NOTE: NO OBSERVED HEALTH EFFECTS WOULD BE ASSOCIATED WITH THE SIMULATED EXPOSURES.

▲ ORAL REFERENCE DOSE (0.003 mg/kg-day)

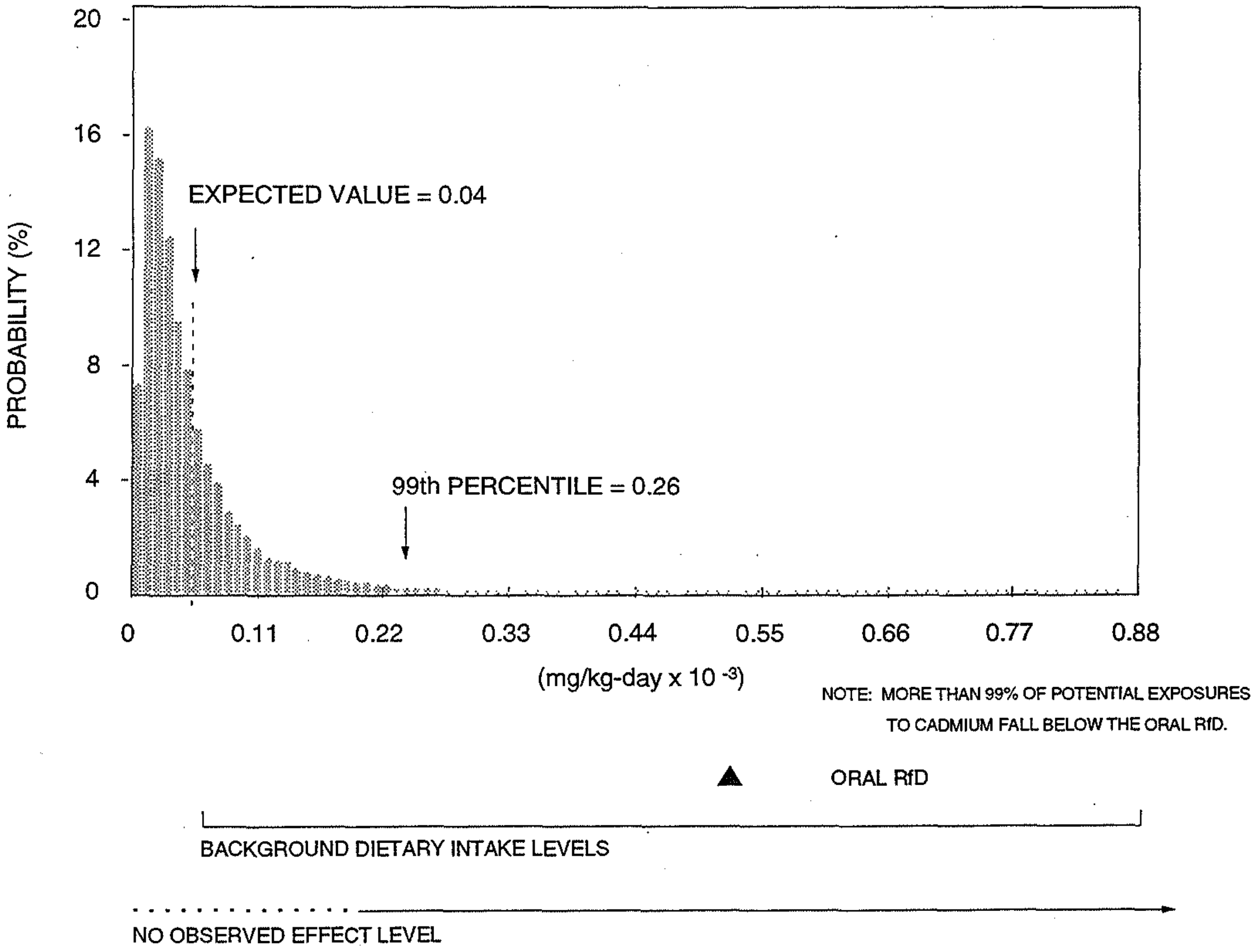
‖ DIETARY INTAKE LEVEL

· ·

NO OBSERVED EFFECTS IN EITHER MICE OR RATS

(SUFFICIENT HUMAN DATA NOT AVAILABLE)

FIGURE 6.4
NONCARCINOGENIC HEALTH EFFECTS OF POTENTIAL URANIUM EXPOSURE RANGES FOR CHILDREN



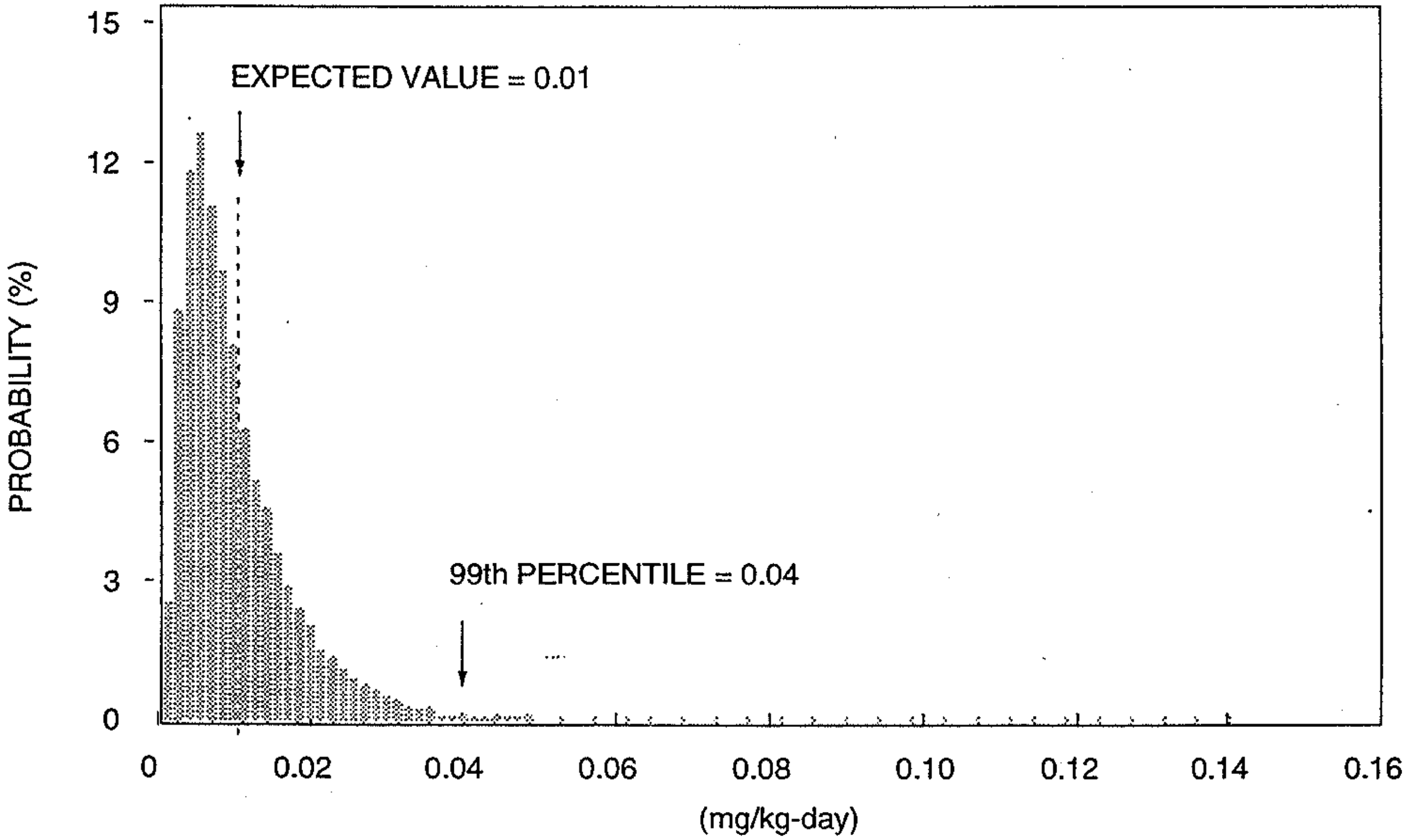
NOTE: MORE THAN 99% OF POTENTIAL EXPOSURES TO CADMIUM FALL BELOW THE ORAL RfD.

FIGURE 6.5

HEALTH EFFECTS OF POTENTIAL CADMIUM EXPOSURE RANGES FOR CHILDREN

PATH I:\FL3.01\GUNBRA

POTENCA1.DRW



▲ NUTRITIONAL REQUIREMENT (0.0002 μg/kg-day)

▲ BACKGROUND INTAKE FROM DRINKING WATER (0.003 mg/kg-day)

U DIETARY INTAKE (0.00007 - 0.0006 mg/kg-day)

NOTE: NO ADVERSE HEALTH EFFECTS HAVE BEEN ASSOCIATED WITH THE POTENTIAL EXPOSURE RANGE FOR COBALT.

→ CARDIOMYOPATHY IN ADULT HUMANS

FIGURE 6.6
HEALTH EFFECTS OF POTENTIAL COBALT EXPOSURE RANGES FOR CHILDREN

Table 6.1 Comparison of exposure doses for potential ingestion of ground water-irrigated garden produce and ingestion of milk and meat from ground water-fed livestock by adults to EPA acceptable intake levels, Gunnison, Colorado, site

Constituent of potential concern	Exposure dose (mg/kg-day)				RfD[a] (mg/kg-day)
	Produce[b]	Milk[b]	Meat[b]	Total exposure dose	
Cadmium	2E-06	4E-06	5E-07	6.5E-06	5E-04
Manganese	2E-03	1E-03	4E-04	3.4E-03	1.4E-01[c]
Uranium	8E-07	2E-04	2E-05	2.2E-04	3E-03

[a] From EPA (1994a).
[b] Exposure doses from Tables 4.2, 4.3, and 4.4, respectively.
[c] Oral RfD developed for ingested food.

Note: Constituents evaluated are those with the EPA-determined oral RfDs.

Another pathway considered feasible for this site is fish ingestion. Based on this evaluation, no adverse health effects would be expected in people eating fish caught in the Gunnison River, Tomichi Creek, or a campground pond near the Gunnison site. However, as indicated in Section 4.2.5, tremendous uncertainty is associated with the fish pathway evaluation. Nevertheless, fish ingestion may cause an increased dietary intake of certain elements that could result in toxicological sensitivity to additional exposure (i.e., ground water ingestion).

6.2 POTENTIAL CARCINOGENIC HEALTH EFFECTS

All uranium isotopes are radioactive and, as such, are considered carcinogens. Table 6.2 presents estimates of the excess lifetime cancer risk predicted to result from ingesting contaminated groundwater at the Gunnison site. These estimates are based on the cancer SFs developed by the EPA (EPA, 1994b); however, natural uranium has not been demonstrated to cause cancer in humans or animals following ingestion.

The potential exposure values could result in an overall excess lifetime cancer risk of about 3 in 1000. This exceeds the EPA's National Contingency Plan (NCP) guidance (developed for Superfund sites) of a maximum increased cancer risk of 1 in 10,000. About 70 percent of this risk could be attributed to the lead-210 exposure. The potential exposure from other pathways combined (produce, milk, and milk ingestion), could result in an excess lifetime cancer risk of about 5 in 100,000, falling within the acceptable range, as defined by the NCP, of 1 in 10,000 and 1 in 1,000,000 (Table 6.3). About 90 percent of this risk could be attributed to lead-210 exposure.

The cancer risk estimates presented here are thought to be conservative because they are based on the worst (highest level) concentrations and a cumulative lifetime (30-year) exposure duration. However, few people (if any) would spend their entire lifetime near the Gunnison site. In addition, ground water concentrations will decline over time because the source term has been removed at the Gunnison site.

Table 6.2 Excess lifetime cancer risk for ground water ingestion by a hypothetical future adult resident, Gunnison, Colorado, site

Contaminant of concern	Maximum concentration in groundwater (pCi/L)	Ground water exposure dose[a] (pCi per lifetime)	Oral Sf[b] $(pCi)^{-1}$	Excess lifetime cancer risk[c]
Lead-210	78	1.6E+06	1.01E-09	2E-03
Polonium-210	1.6	3.4E+04	3.26E-10	1E-05
Thorium-230	2.0	4.2E+04	3.75E-11	2E-06
Uranium 238[d]	550	1.2E+07	6.20E-11	7E-04
Uranium 234[d]	550	1.2E+07	4.44E-11	5E-04
			Total:	3E-03

[a] Calculated using Equation 2 in Section 4.4 and the following exposure assumptions:
Ingestion rate = 2 L of water/day;
Exposure frequency = 350 days/year;
Exposure duration = 30 years.

[b] See Table 5.2.

[c] Excess lifetime cancer risk is calculated by multiplying the exposure dose by the SF.

[d] Maximum concentration in ground water is 1.6 mg/L; 1 mg uranium is assumed to equal 686 pCi; the secular equilibrium between uranium-234 and -238 is assumed to estimate activity concentration of different uranium radioisotopes.

Table 6.3 Excess lifetime cancer risk for ingestion of ground water-irrigated garden produce and milk and meat from ground water-fed livestock by a hypothetical future adult resident, Gunnison, Colorado, site

Radionuclide	Exposure dose (pCi/lifetime)				Oral Sf[a] $(pCi)^{-1}$	Excess lifetime cancer risk[e]
	Produce[b]	Milk[c]	Meat[d]	Total		
Lead-210	3.00E+04	2.00E+04	6.00E+03	5.60E+04	1.01E-09	6E-05
Polonium-210	8.00E+00	1.00E+02	2.00E+01	1.28E+02	3.26E-10	4E-08
Thorium-230	6.00E+00	3.00E+00	9.00E-01	9.90E+00	3.75E-11	4E-10
Uranium-238	2.00E+02	5.00E+04	5.00E+03	5.52E+04	6.20E-11	3E-06
Uranium-234	2.00E+02	5.00E+04	5.00E+03	5.52E+04	4.44E-11	2E-06
Total uranium:						5E-06
Total radionuclides:						**6.5E-05**

[a] The basis for these slope factors is presented in Table 5.2.

[b] From Table 4.2.

[c] From Table 4.3

[d] From Table 4.4.

[e] Estimated by multiplying the total exposure dose by the oral SF.

6.3 LIMITATIONS OF RISK EVALUATION

The following potential limitations apply to interpretations of this risk evaluation:

- This risk assessment evaluates only risks related to inorganic groundwater contamination. Potential contamination with any of the few organic constituents used in uranium processing has not been addressed.
- Subpopulations that might have increased sensitivity, such as individuals with preexisting disease and the elderly, are not specifically addressed on the graphs.
- Some individuals may be more sensitive to the toxic effects of certain constituents for reasons that have not yet been determined.
- Data available to interpret potential adverse health effects are not always sufficient to allow accurate determination of all health effects (i.e., lack of testing in humans or testing of dose ranges other than those expected at this site).
- Although plume movement is evaluated hydrologically and geochemically, the monitoring locations sampled may not be in the most contaminated portion of the plume. However, constituent concentrations may decrease substantially as the plume moves.
- The risk evaluation results presented in this document are largely based on filtered (0.45-µm) ground water samples. Therefore, the potential loss of certain ground water constituents as a consequence of filtration is associated with a source of uncertainty.

The evaluation presented here has considered these limitations and compensated wherever possible by presenting toxicity ranges rather than point estimates to incorporate as much variability as could be reasonably defined. The impacts of these potential limitations are discussed more fully in Section 8.2.

7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION

The objective of the environmental portion of the risk assessment is to determine whether constituents detected at the site have the potential to adversely affect the existing biological community at or surrounding the site. Currently, the EPA has no guidance for quantifying potential impacts to ecological receptors but has developed a qualitative approach generally used for ecological evaluation (EPA, 1989b). With the qualitative approach, the EPA recommends comparing ambient environmental media concentrations to water quality, sediment quality, or other relevant criteria to determine whether any of the concentrations the ecological receptors are expected to encounter exceed these criteria.

An ecosystem is composed of both abiotic and biological components. The abiotic component is called the habitat. Biological components are organized into species, populations, and communities. A population is composed of individuals of a species that occur within a defined area, and a community is a collection of all populations (plant, animal, bacteria, and fungi) that live in a defined area and interact with one another. In practice, it is sometimes difficult to set boundaries for populations and communities. The community plus its habitat make an ecosystem (Moriarty, 1988).

Predicting the ecotoxicological effects of constituents is extremely complicated. Ecosystems are not static; the biological components experience constant fluctuations both in population numbers and relative composition. Abiotic factors (e.g., temperature, precipitation, nutrient availability) are also constantly changing. The stability of an ecosystem therefore is determined to a great extent by the ability to respond to "normal" stresses. The normal or baseline conditions are not well understood or defined for any ecosystem. Therefore, it is difficult to determine whether changes in ecological parameters (e.g., diversity, total biomass, reproductive trends) are associated with a contaminant action or merely reflect normal fluctuations.

It is possible that effects on individual organisms or even populations may not affect the ecosystem at all. If a prey species is affected, predators may be able to feed on other species; predator loss may be compensated for by other predators or by immigration of another predator population. Recognizing when an adverse effect has occurred or is occurring is a challenge. Unless there is a mass killing within a population or community, adverse effects may go unnoticed. Sublethal effects such as behavioral changes, reduced reproductive success, enzyme level changes, or effects on microorganisms can affect the population- or community-level of organization due to an effect on reproductive success in one species that may influence other dependent species. It is often difficult to identify and measure the sublethal effects, and establishing a causal relationship to a specific environmental stressor, such as a specific constituent, is rarely accomplished. Evidence of sublethal effects and gross impacts were not observed during the field survey at the Gunnison site.

The effects of contaminants on ecological receptors are a concern; however, it is difficult to predict whether observed effects on individual populations will damage the ecosystem. Populations are dynamic; therefore, information concerning the normal range of variability within a population is important. Sublethal effects, which may be very important to overall ecosystem health, are difficult to detect, and constituents present at low concentrations may not kill organisms directly but may diminish their ability to survive and reproduce.

7.1 EXPOSURE CHARACTERIZATION

This section identifies the potential exposure pathways associated with the site. For risk to exist, a receptor must be exposed to contaminants. Exposure can occur only if there is both a source of contamination and a mechanism of transport to a receptor population or individual.

The tailings pile and associated contaminated soil were removed from the Gunnison site from 1992 to 1995 and relocated to a disposal cell (see Section 1.0 and 2.1). Thus, direct exposure pathways (such as incidental ingestion of soil, dermal contact with soil, and inhalation of air containing particulates) will not represent a future ecological concern and will not be evaluated in this baseline risk assessment. However, direct exposure pathways (such as ingestion of surface water potentially affected by contaminated ground water and bioconcentration of constituents of potential concern in surface water by aquatic organisms) and indirect exposure pathways (such as consumption of previously exposed organisms, known as bioaccumulation) are possible at the site.

The net accumulation by organisms of a constituent directly from the surrounding environment is known as bioconcentration. Net accumulation by organisms as a result of all routes of exposure, including the diet, is known as bioaccumulation. Generally, bioconcentration is measured for uptake of chemicals from water by aquatic organisms. BCFs for ingestion of and dermal contact with soils are too variable and dependent on site conditions to make identification of generic soil BCFs possible. Freshwater fish BCFs in the scientific literature for the constituents of concern detected in surface waters in the site vicinity range from 30 liters per kilogram (L/kg) for strontium to 100 L/kg for iron (NUREG, 1986; EPA, 1992b). No fish BCFs were found in the available literature for calcium, fluoride, magnesium, manganese, potassium, silica, sodium, and sulfate. Identification of "biologically significant" fish BCF values have ranged from 1000 L/kg down to 300 L/kg (Kenaga, 1980; EPA, 1989b). None of the constituents of concern that have BCFs (including iron) would be identified as "biologically significant."

Surface water bodies in the site vicinity include the Gunnison River, Tomichi Creek, various ponds and ditches, and wetlands. The site lies equidistant (0.4 mi [0.6 km]) between the Gunnison River and Tomichi Creek, with the river west of the site and the creek to the southeast. The confluence of the river and creek is approximately 1.5 mi (2.4 km) southwest of the site (see Figure 2.5). Surface water runoff from the site flows to the south and east toward Tomichi Creek. The site is bounded on the west by drainage ditches and to the south and west by an irrigation ditch. Two ponds are located at the campground, which is approximately 200 ft (60 m) west, across Goodwin Lane from the former tailings pile. The northernmost pond, which is nearest the campground entry driveway, is fed by Gunnison River water from an irrigation ditch. This pond is used for pay fishing by campers and, according to the campground owner, the pond was stocked with rainbow trout during the first week of June 1993. This pond is drained each winter and refilled in the spring. The rainbow trout used for stocking were provided by a state fish hatchery. The other pond adjacent to the pay fishing pond is not stocked and receives surface runoff from the drainage ditch that runs under Goodwin Lane. Wetland areas are located at the western end of the site and along Gold Basin Road in the adjacent windblown area, which has been cleaned up.

All these water bodies are potential exposure points for resident aquatic life and for terrestrial wildlife (including livestock) to surface water and/or sediments. These environmental media and potential exposure pathways to aquatic life, terrestrial wildlife, and livestock were evaluated in this risk assessment.

Another potential current pathway could involve plant uptake of constituents in ground water. Due to the shallow depth to ground water (approximately 5 ft [1.5 m] or less below land surface), plants can reach contaminated ground water. Plant uptake was evaluated in this risk assessment assuming that the plant roots reached soil saturated with ground water containing the mean concentrations for the most contaminated wells for the constituents of potential concern.

Another potential future pathway involves use of ground water from hypothetical domestic well as a water source for livestock or for agricultural activities. For this baseline risk assessment, it was assumed as a conservative measure that a hypothetical domestic well could in the future intercept the most contaminated ground water in the plume. The water from this hypothetical well could be used as a sole source of water for a livestock watering pond (which could also be stocked with fish) or for irrigating agricultural crops.

7.2 ECOLOGICAL RECEPTORS

This section identifies the ecological resources present at the site and vicinity that are potentially exposed to site-related constituents.

The following information on ecological receptors is based primarily on surveys done before the tailings removal process was initiated and is provided as historical perspective. Limited observations of aquatic organisms were conducted at the surface water, sediment, and fish sampling locations during a 23 June 1993 field survey. No observations of terrestrial flora and fauna were conducted during these sampling activities.

Most of the land within the site boundary was disturbed during milling operations. After operations ceased, the tailings pile was covered with soil and reseeded. The unexcavated pile was grass-covered; big sagebrush was scattered over most of the pile.

7.2.1 Flora

Upland plant communities

The former processing site area is within the Great Basin sagebrush habitat of the Southern Rocky Mountain zone, in the floodplains of the Gunnison River and Tomichi Creek. The plant communities within the former processing site area (including the adjacent windblown area) are indicative of the disturbed nature of the area.

Grasses and herbs predominate; an immature stand of cottonwoods grows at the western edge of the former processing site.

Desert shrub and shrub wetlands

The plant communities in the windblown contaminated areas to the north and east of the site are desert shrub and shrub wetlands. Big sagebrush is the most common shrub species in the desert shrub community and grows as scattered individuals or in clumps (TAC, 1989). Rabbitbrush is also present, and grass and herbs are the dominant ground cover. Willow is the most common species in the wetland habitat and occurs in fairly dense stands in some areas. Small (5- to 15-ft [1.5- to 4.6-m]) narrowleaf cottonwood trees are also common in this area. The wetland habitat has dense grass as ground cover (TAC, 1989).

Wetland plant communities

Approximately 8.1 ac (3.3 ha) of U.S. Army Corps of Engineers-designated wetlands are found in the western portion of the processing site and within windblown-contaminated areas east of the processing site (DOE, 1992b). Wetlands at the former processing site consist of wet meadows dominated by grass, sedges, rushes, and herbs. Wetlands in the windblown-contaminated areas are shrub-dominated. A wet meadow-type wetland along the haul road is dominated by grass, sedges, and rushes.

7.2.2 Terrestrial fauna

Brief reconnaissance surveys for wildlife have been conducted in the former processing site area. No reptiles or amphibians were observed; however, seven species, including the short-horned lizard, eastern fence lizard, and bullsnake, would be expected at the site (Hammerson, 1986; CDM, 1981).

Amphibians would be most common in the flooded wetland areas where species such as the leopard frog, boreal chorus frog, and tiger salamander may occur. Lizard species such as the short-horned lizard and sagebrush lizard would be more common in the sagebrush habitat and disturbed tailings area (Hammerson, 1986; CDM, 1981).

A total of 43 species of birds have been observed during various site surveys (TAC, 1990, 1989, 1988, 1986, 1985; CDM, 1981). The western meadowlark, red-wing blackbird, yellow warbler, and robin were common nesting species at and near the tailings pile. Wetland species such as red-wing blackbirds, waterfowl, and shorebirds were common in the flooded hayfields. The sage thrasher, sage grouse, green-tailed towhee, and various species of sparrows were common nesting species in the sagebrush habitat.

A total of 25 species of mammals may occur at the processing site (Bernard and Brown, 1978). Muskrat signs were observed in wetland areas. Other species typical of the disturbed and sagebrush habitats would be the desert cottontail and striped skunk. Mammals typical of the irrigated wetland habitat that would be expected in the area include the masked shrew, western jumping mouse, and muskrat. Surveys in 1990 resulted in the observation of an active prairie dog town at the northern end of the former tailings pile; 20 burrows were observed (TAC, 1990).

Threatened and endangered species

Consultation with the Fish and Wildlife Service (FWS) to determine threatened and endangered (T&E) species and other species of concern began in 1985. This process resulted in six T&E species, one species proposed for listing, and five federal candidate species being identified as potentially occurring in the Gunnison area. Two endangered bird species may occur near the site. The bald eagle occurs in small numbers during the winter along the Gunnison River, while the whooping crane stops to feed in the wetlands along Tomichi Creek during the spring and fall migrations.

The black-footed ferret is closely associated with prairie dog towns. A small prairie dog town was found on the north end of the former tailings pile. However, because of the highly disturbed nature of the area and small size of the town, it is unlikely that any black-footed ferrets would be present.

Of the three endangered fish species (Colorado squawfish, humpback chub, bonytail chub) and one proposed fish species (razorback sucker), only the Colorado squawfish occurs in the Gunnison River. However, this species does not occur in the river in the Gunnison area.

Five federal candidate species occur in the Gunnison area. The white-faced ibis and long-billed curlew occur in the wetland habitat along Tomichi Creek during migration; the snowy plover does not occur or occurs very sporadically in the Gunnison area. All potentially disturbed areas were surveyed for the presence of the skiff milkvetch and Gunnison milkvetch. No skiff milkvetch plants were found in any potentially disturbed areas; however, between 50 and 75 Gunnison milkvetch plants were found growing on the western side of the former tailings pile in 1990. A subsequent survey in 1991, however, identified only two plants present (EES, 1991; TAC, 1990; Carlson, 1989).

Additional details on T&E species are provided in Attachment 2 of the site environmental assessment (DOE, 1992a).

7.2.3 Aquatic organisms

The Technical Assistance Contractor has not conducted quantitative surveys of aquatic organisms occurring in the surface water bodies in the site vicinity (DOE, 1990; 1992b). The observations made during the field survey focused on the water bodies from which samples were collected (i.e., the Gunnison River, Tomichi Creek, and the campground pond); thus, no observations were made in the wetland areas. Extremely high water levels, turbidity, and rapid velocity of the Gunnison River and Tomichi Creek acted to greatly limit visibility and observations of aquatic organisms. Other than some stonefly *(Plecoptera)* nymphs, caddis fly larvae *(Trichoptera)*, and fly larvae *(Diptera)*, no other aquatic organisms were observed in the Gunnison River and Tomichi Creek. In the Gunnison River, an approximately 3/4-pound (340-gram) brook trout was caught at location 775 and an approximately 1 1/4-pound (570-gram) rainbow trout was caught at location 776. One brook trout and one German brown trout, each approximately 1/3 pound (150 grams), were caught in Tomichi Creek at location 777. No fishing was conducted at location 778.

At the campground pond, location 779, water striders *(Gerridae)*, adult water beetles *(Coleoptera)*, and some fly larvae *(Diptera)* were observed. One rainbow trout, approximately 1 pound (450 grams), was collected from the pond during the 23 June 1993 sampling activities.

In addition to the fish species collected during the sampling activities, several other fish species are known to occur in the Gunnison River and Tomichi Creek, including kokonee and cutthroat trout, speckled dace, flannelmouth sucker, western white sucker, bluehead sucker, and bluehead x flannelmouth sucker hybrid (Hebein, 1993).

7.3 CONSTITUENTS OF ECOLOGICAL CONCERN

The complete list of ground water constituent levels that exceed background levels (Table 3.2, column 1) was used as the list of constituents of potential concern for ecological receptors potentially exposed to ground water (e.g., plant uptake). This list is composed of 16 chemicals, plus lead-210, polonium-210, and thorium-230.

The list of constituents of potential concern in the surface water bodies was developed from the list of constituents detected above background levels in ground water (Table 3.2). This list of constituents was then compared with the surface water data. Table 7.1 presents the surface water data for constituents detected in the surface water samples and above background levels in ground water.

If a constituent was never detected in the water body (e.g., ammonium, cadmium, cobalt, nickel, thorium-230, uranium) or the concentration measured downstream of the site was less than or equal to the concentration upstream of the site (the background level), it was not considered a constituent of potential concern for ecological receptors. Additionally, silica in the Gunnison River and calcium and

Table 7.1 Occurrence of constituents detected in surface water bodies in the vicinity of the Gunnison, Colorado, site

	Gunnison River location ID		Tomichi Creek location ID		Campground pond location ID
Constituent	775 (upstream)	776 (downstream)	778 (upstream)	777 (downstream)	779
Calcium	32	32	44	48	44
Fluoride	0.2	0.1	0.3	0.3	0.2
Iron[a]	0.07	0.06	0.3	NA	0.43
Magnesium	6.6	6.5	10.4	11.3	9.6
Manganese	0.01	0.01	0.05	0.05	<0.01
Potassium	0.85	0.82	2.2	2.1	1.7
Silica	9.1	9.3	22	19	10
Sodium	3.0	3.2	8.5	7.9	3.8
Strontium	0.1	0.1	0.2	0.2	0.17
Sulfate	18	16	17	24	12
Zinc	<0.005	<0.005	<0.005	<0.005	0.04

[a] Concentrations presented are from unfiltered samples collected in October 1990.

All concentrations reported in milligrams per liter from filtered samples collected in August 1989, unless specified otherwise.
NA - unfiltered sample data not available.

magnesium in Tomichi Creek are excluded as constituents of potential concern, because the differences between the downstream and upstream concentrations were minimal (<10 percent) and these constituents are not considered site-related constituents. Although no background location was specific to the campground pond, as there was for the Gunnison River and Tomichi Creek, concentrations detected in the pond water were compared to concentrations detected at the upstream locations in the river and creek. If a concentration observed in the pond water was less than or equal to the upstream concentration in the river and/or the creek, it was eliminated as a constituent of potential concern for the pond.

The concentrations used in these comparisons to background were from filtered samples for most of the constituents. Data from filtered samples were used because most of the state of Colorado's water quality standards are stated as dissolved (filtered) metal concentrations. Unfiltered sample data were used for iron because the water quality criterion for this metal is based on the total recoverable (unfiltered) metal concentration.

After these comparisons were made, sulfate was selected as the constituent of potential concern for Tomichi Creek. For the campground pond, the constituents of potential concern are iron and zinc. None of the constituents detected in downstream Gunnison River water are considered constituents of potential concern.

No sediment samples were collected from the surface water bodies prior to the June 1993 sampling. Thus, the list of constituents of potential concern includes metals for which the sediment samples were analyzed: manganese, molybdenum, uranium, and zinc.

7.4 POTENTIAL IMPACTS TO WILDLIFE AND PLANTS

7.4.1 Terrestrial risk

A potential exposure pathway that was evaluated involves terrestrial vegetation. Terrestrial vegetation can be directly exposed to constituents in ground water through uptake by the roots. Constituents may bioaccumulate in various plant parts and exert a wide range of influences, depending on the specific constituent. Plant uptake rates vary greatly among species and are affected by factors such as soil characteristics (pH, moisture, redox potential, organic matter), plant sensitivity, input-output balance, and cumulative effects. Foraging wildlife can be indirectly exposed to constituents in ground water by ingesting plants that may have bioconcentrated certain constituents. Terrestrial wildlife can be directly exposed to constituents in surface water bodies by ingesting the surface water, aquatic organisms, and sediments. However, good information on generic BCFs for terrestrial wildlife is not available in the scientific literature. Based on the shallow depth to contaminated ground water at the site, it is possible that some plants could have rooting zones in soils that intercept contaminated ground water.

Concentrations of the constituents of potential concern in plant tissue were estimated using soil-to-plant BCFs. No soil data are available for the site, nor are water-to-plant BCFs available. However, because plants require nutrients to be in an aqueous form for root uptake, it was considered appropriate to use the soil-to-plant BCFs for estimating potential plant uptake at the site. Soil concentrations in the saturated zone were estimated by multiplying the ground water concentration by the soil-water distribution coefficient, Kd. The methodology and parameters used to estimate root uptake and plant tissue concentrations for the constituents of potential concern are presented in Table 7.2. This methodology is described in detail elsewhere in the literature (Baes et al., 1984) and therefore will not be presented here.

The estimated tissue concentrations for the constituents of potential concern in the vegetative portions (e.g., stems, leaves) and in the nonvegetative portions (e.g., fruits, tubers) were compared to approximate concentrations (in mature leaf tissue) that have been reported to be toxic to plants (phytotoxic) (Table 7.2); few available data relate tissue concentrations to phytotoxicity. The reported phytotoxic concentrations are not representative of very sensitive or highly tolerant plant species. The estimated tissue concentrations for the constituents of potential concern in plants that may reach soil saturated with contaminated ground water do not exceed the available phytotoxicity data. No comparison data were available for calcium, iron, lead-210, magnesium, polonium-210, potassium, silica, sodium,

Table 7.2 Comparison of estimated plant concentrations to phytotoxic concentrations, Gunnison, Colorado, site

Constituents of potential concern	Mean concentration in ground water[a] (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factors Bv	Br	Estimated concentration in vegetative growth[b] (mg/kg DW)	Estimated concentration in fruits/tubers[c] (mg/kg DW)	Approximate concentration in mature leaf tissue that is toxic[d] (mg/kg DW)
Cadmium	0.0014	15	0.021	0.55	0.15	0.012	0.0032	5-30
Calcium	602	4	2400	3.5	0.35	8400	840	NA
Cobalt	0.29	1.9	0.55	0.02	0.007	0.011	0.0039	15-50
Fluoride	0.9	0	NC	0.06	0.006	NC	NC	50-500
Iron	67	15	1000	0.004	0.001	4.0	1.0	NA
Magnesium	31	70	2200	1	0.55	2200	1200	NA
Manganese	28	17	480	0.25	0.05	120	24	200[e]-1000
Nickel	0.15	12	1.8	0.06	0.06	0.11	0.11	10-100
Potassium	5.6	0	NC	1.0	0.55	NC	NC	NA
Silica	17	5	85	0.35	0.07	30	6.0	NA
Sodium	15	0	NC	0.075	0.055	NC	NC	NA
Strontium	0.7	24	17	2.5	0.25	43	4.3	NA
Sulfate[f]	1530	0	NC	0.5	0.5	NC	NC	NA
Uranium	1.4	1	1.4	0.0085	0.004	0.012	0.0056	NA
Zinc	0.58	13	7.5	1.5	0.9	11	6.8	100-400
Radionuclides								
Lead-210[g]	1.6E-10	230	3.7E-08	0.045	0.009	1.7E-09	3.3E-10	NA
Polonium-210[g]	1.1E-13	5.9	6.5E-13	0.0025	0.0004	1.6E-15	2.6E-16	NA
Thorium-230[g]	4.5E-08	100	4.5E-06	0.00085	0.000085	3.8E-09	3.8E-10	NA

[a] Mean concentration in ground water from the most contaminated wells at the site.
[b] Estimated concentration in vegetative portions, calculated as estimated soil concentration multiplied by Bv.
[c] Estimated concentration in nonvegetative portions, calculated as estimated soil concentration multiplied by Br.
[d] Concentrations are not presented for very sensitive or for highly tolerant plant species (Kabata-Pendias and Pendias, 1992).
[e] The value of 200 mg/kg is a recommended criterion of the Illinois Institute of Environmental Quality (Saric, 1986).
[f] Bv and Br factors available for elemental sulfur only; thus, these factors were reduced by a factor of 3 for sulfate.
[g] The ground water concentration in pCi/L was converted to mg/L.

Kd - soil-water distribution coefficient; (PNL, 1989), except for uranium and calcium. The Kd for uranium is a site-specific value. No Kd for calcium is available in PNL (1989); Kd shown is from Baes et al. (1984).
Bv - soil-to-plant elemental transfer factor for vegetative portions of food crops and feed plants (Baes et al., 1984).
Br - soil-to-plant elemental transfer factor for nonvegetative portions (e.g., fruits, tubers), of food crops and feed plants (Baes et al., 1984).
DW - dry weight.
mg/kg - milligrams per kilogram.
mg/L - milligrams per liter.
L/kg - liters per kilogram.
NA - not available.
NC - value cannot be calculated because Kd is zero.
No Kd, Bv, or Br values available for ammonium.

strontium, sulfate, thorium-230, and uranium. Thus, based on existing data, it is not possible to evaluate whether the estimated tissue concentrations could adversely affect plants.

Bioaccumulation in terrestrial organisms as a function of constituents of potential concern in ingested plants or animals (e.g., birds eating fish) is a potential exposure pathway at the site. Birds and other vertebrates consuming these plants and animals can bioaccumulate some of the constituents of potential concern from this diet if the amount ingested exceeded the amount eliminated. This is often a function of the areal extent of contamination versus the areal extent of the animals' feeding range. In small contaminated areas, the amount of food in the diet usually exceeds the impacted food, and bioaccumulation is not a concern. Therefore, exposure via the diet for all trophic-level species is possible in certain areas (e.g., wetland areas), but the potential for bioaccumulation is not always a concern. While predators of fish would be exposed to concentrations present in the entire body (including bones), as opposed to concentrations in the muscle tissue, the primary purpose of the fish sampling conducted in June 1993 was to evaluate potential human health risk of fish ingestion.

Biomagnification is a more severe situation in which the concentration of a constituent increases in higher levels of the food chain because the constituent concentrations accumulate through each successive trophic level. Of particular concern for biomagnification effects are the top predators, especially the carnivorous birds and mammals. Only a limited number of constituents have the potential to magnify in the food chain. Because most constituents are metabolized in organisms and eliminated at each level of the food chain, the constituent concentration does not increase up the food chain. Based on available information, the potential that the detected constituents of potential concern represent a hazard via food chain transfer is probably low.

To evaluate the potential impact on wildlife of using contaminated ground water in a livestock pond (i.e., animals drinking from the pond or fish stocked in the pond), the mean ground water concentrations for the constituents of potential concern were compared to available water quality criteria (Table 7.3). There are no available federal or state criteria or standards established for the protection of terrestrial wildlife via water exposure. Therefore, it is difficult to evaluate the potential hazards to terrestrial receptors without additional information. However, available surface water quality values for the protection of freshwater aquatic life do exist and include the state of Colorado standards (CDH, 1991).

The mean ground water concentrations for iron and manganese exceeded the comparison water quality values (Table 7.3), while the ground water concentrations for cadmium, nickel, thorium-230, and zinc were below the comparison values. The concentrations for iron and manganese exceed the state standards, indicating this water would be unacceptable for aquatic organisms. No comparison water quality values are available for ammonium, calcium, cobalt, fluoride, lead-210, magnesium, polonium-210, potassium, silica, sodium, strontium, sulfate, and uranium.

Table 7.3 Comparison of constituents of potential concern in ground water with available water quality values, Gunnison, Colorado, site

Constituent of potential concern	Mean concentration in ground water	Aquatic life water quality value[a]	Water concentration protective of livestock[b]	Concentration in irrigation water protective of plants[b]
Ammonium	0.5	NA	NA	NA
Cadmium	0.0014	0.010[c]	0.05	0.01
Calcium	602	NA	NA	NA
Cobalt	0.29	NA	1.0	0.05
Fluoride	0.9	NA	2.0	1.0
Iron	67	1.0	NA	5.0
Lead-210	13 pCi/L	NA	NA	NA
Magnesium	31	NA	NA	NA
Manganese	28	1.0	NA	0.20
Nickel	0.15	0.80[c]	NA	0.20
Polonium-210	0.5 pCi/L	NA	NA	NA
Potassium	5.6	NA	NA	NA
Silica	17	NA	NA	NA
Sodium	15	NA	NA	NA
Strontium	0.7	NA	NA	NA
Sulfate	1530	NA	1000[d]	NA
Thorium-230	0.9 pCi/L	60 pCi/L	NA	NA
Uranium	1.4	32[c]	NA	NA
Zinc	0.58	1.1[c]	25	2.0

[a] Value obtained from the Basic Standards and Methodologies for Surface Water, 3.1.0 (5 CCR 1002-8), Colorado Department of Health, Water Quality Control Commission (CDH, 1991), unless specified otherwise. These values are standards protective of aquatic life via chronic exposure.

[b] From EPA (1972), unless specified otherwise. Irrigation water values shown are for water used continuously on all soils.

[c] Water hardness-related state standard (CDH, 1991). Criterion presented was calculated using the mean hardness (1630 mg/L) determined from concentrations of calcium and magnesium in plume wells 006, 133, and 134.

[d] From Church (1984).

Note: Concentrations reported in milligrams per liter unless otherwise noted. The mean constituent concentrations in ground water are from the most contaminated wells at the site.

NA - not available.

Another future hypothetical use of the ground water in the area is irrigating agricultural crops. Table 7.3 compares the approximate constituent concentrations in water used for irrigation that should be protective of plants with the ground water concentrations (EPA, 1972). Seven of the constituents of potential concern — cadmium, cobalt, fluoride, iron, manganese, nickel, and zinc — have comparison criteria. The mean ground water concentrations for cobalt, iron, and manganese exceed the comparison criteria, while the concentrations of cadmium, fluoride, nickel, and zinc are below the comparison criteria. No comparison criteria are available for the remainder of the constituents of potential concern. Thus, it is not possible to evaluate the potential for these constituents to adversely affect plants when applied in irrigation water.

Based on the available information, use of the alluvial ground water near the site (containing the mean constituent concentrations) as a continuous source of irrigation water could result in deleterious effects to crops, primarily due to the elevated concentrations of cobalt, iron, and manganese.

7.4.2 Aquatic risk

Surface water (unfiltered) and surficial sediment (0 to 4 in [0 to 10 cm]) samples were collected from the Gunnison River upstream (location 775) and downstream (location 776) of the site, from Tomichi Creek upstream (location 778) and downstream (location 777) of the site, and from the campground pond (location 779) (Figure 3.12) on 23 June 1993. The surface water samples were analyzed for calcium, magnesium, manganese, molybdenum, uranium, and zinc; the sediment samples were analyzed for manganese, molybdenum, uranium, and zinc.

No samples of surface water or sediment have been collected to date from the wetland areas in the site vicinity.

Gunnison River water

A potential exposure point is the Gunnison River in the site vicinity. The plume of contaminated alluvial ground water is believed to be discharging to the river, Tomichi Creek, and possibly the campground pond. The comparison of the surface water data collected from the river at the upstream location (location 775) to the downstream location (location 776) that was conducted as part of the selection process for the constituents of ecological concern (Section 7.3) indicated that most of the constituents did not exceed background concentrations. This suggests that ground water discharge to the river has not affected the water quality, though the limited sampling is not conclusive.

Gunnison River sediments

There are no established state or federal sediment quality criteria (SQC) for the protection of aquatic life for the constituents of potential concern at this site (EPA, 1988).

The EPA is evaluating a methodology based on the three-phase sorption model for free metal ion activity and is assessing its applicability for determining the bioavailable fraction within sediments (EPA, 1989c). Currently, a number of other predictive models and methods are being investigated for metals, but no single approach has been accepted to adequately develop sediment-based metals criteria (Shea, 1988; Chapman, 1989; EPA, 1989c; NOAA, 1990; Di Toro et al., 1991; Burton, 1991). Therefore, only a qualitative hazard assessment of the metals detected in sediments is presented in this risk assessment.

National Oceanic and Atmospheric Administration (NOAA) effects-based sediment quality values are available for evaluating the potential for constituents in sediment to cause adverse biological effects. These values are not standards or criteria. Effects range-low (ER-L) values are concentrations equivalent to the lower 10th percentile of available data screened by the NOAA and indicate the low end of the range of concentrations in specific sediments at which adverse biological effects were observed or predicted in sensitive species and/or life stages. The effects range-median (ER-M) values are concentrations based on the NOAA screened data at which effects were observed or predicted in 50 percent of the test organisms evaluated. The NOAA ER-L and ER-M values were compared with the concentrations of the constituents of potential concern detected in sediment. One limitation of the ER-L and ER-M is that the concentration at which toxicity was observed could not be readily extrapolated from one sediment location to another. Sediment characteristics (e.g., organic carbon content, grain/particle size) greatly influence constituent toxicity; thus, the ER-L and ER-M cannot be used as direct indicators of adverse effects to aquatic organisms under the Gunnison site conditions.

The NOAA sediment quality value is available for only one of the detected constituents of potential concern, zinc (Table 7.4). The zinc concentrations upstream of the site (105 mg/kg) and downstream of the site (110 mg/kg) are below the NOAA ER-L value (120 mg/kg). This suggests that the potential for zinc to represent a hazard to aquatic life is low. Molybdenum was not detected in the river, in Tomichi Creek, or in the campground pond. Because there are no sediment quality values for manganese and uranium, it is not possible, with available information, to evaluate whether the detected sediment concentrations could adversely affect biota. However, the concentrations of these two metals were slightly higher at the upstream location than downstream of the site. Although the sediment database is limited, these data suggest that the site is not a notable release source to the river for sediment-bound metals.

Tomichi Creek water

One constituent of potential concern was identified for Tomichi Creek: sulfate (refer to Table 7.5). The sulfate concentration was approximately 30 percent higher downstream of the site. A statistical evaluation of the data could not be conducted because of the small sample size. Therefore, it is not known whether this higher sulfate concentration is related to site contamination or to other factors (e.g.,

Table 7.4 Comparison of constituents of potential concern in sediment from surface water bodies in the site vicinity with available sediment quality values, Gunnison, Colorado, site

Constituent of potential concern	Gunnison River location ID		Tomichi Creek location ID		Campground pond location ID	NOAA values	
	775 (upstream)	776 (downstream)	778 (upstream)	777 (downstream)	779	ER-L	ER-M
Manganese	445	397	291	503	234	NA	NA
Molybdenum	<1.0	<1.0	<1.0	<1.0	<1.0	NA	NA
Uranium	2.45	1.38	2.17	3.99	1.96	NA	NA
Zinc	105	110	31.7	84.4	65.6	120	270

All concentrations reported in milligrams per kilogram.
ER-L - effects range-low (NOAA, 1990).
ER-M - effects range-median (NOAA, 1990).
NA - not available.

Table 7.5 Comparison of constituents of potential concern in surface water bodies in the site vicinity with available water quality values, Gunnison, Colorado, site

Constituent of potential concern	Gunnison River location ID		Tomichi Creek location ID		Campground pond location ID	Aquatic life water quality value[a]	Water concentration protective of livestock[b]
	775 (upstream)	776 (downstream)	778 (upstream)	777 (downstream)	779		
Iron[c]	0.07	0.06	0.3 (0.1)	NA (0.05)	0.43	1.0	NA
Sulfate	18	16	17	24	12	NS	1000[d]
Zinc	<0.005	<0.005	<0.005	<0.005	0.04	0.16[e]	25

[a] Value obtained from the Basic Standards and Methodologies for Surface Water, 3.1.0 (5 CCR 1002-8), Colorado Department of Health, Water Quality Control Commission (CDH, 1991), unless specified otherwise. These values are standards protective of aquatic life via chronic exposure.
[b] From EPA (1972), unless specified otherwise.
[c] Concentrations shown are from unfiltered samples, except those shown in parentheses ( ), which are from filtered samples.
[d] From Church (1984).
[e] Water hardness-related state standard (CDH, 1991), calculated using a constituent-specific equation and the average hardness (164 mg/L) determined from the concentrations of calcium and magnesium measured in the campground pond.

All concentrations reported in milligrams per liter (mg/L) for filtered samples, unless specified otherwise.
NA - not available.
NS - no state or federal water quality standard or criterion available.

variability in laboratory analyses). Based on the limited data available, the significance of this increase above background is not known.

Tomichi Creek sediments

Surficial sediment samples were also collected from Tomichi Creek from the same locations where the surface water was sampled (Figure 3.12). Molybdenum was not detected at either location. The concentrations for the other analyzed constituents of potential concern (manganese, uranium, and zinc) were all higher at the downstream location than at the upstream location (Table 7.4). In addition to contaminated ground water discharge, another potential release source that exists between the upstream and downstream locations on the creek is the water discharge pipe from the gravel pit. Water that has accumulated in the gravel pits is periodically pumped out and into the creek. Therefore, the downstream concentration increases may be associated with site-related contamination and/or releases not associated with the site; however, insufficient data are available to determine if sediment quality is affected by the site.

The detected concentration of zinc at both the upstream location (31.7 mg/kg) and the downstream location (84.4 mg/kg) are below the NOAA ER-L value of 120 mg/kg. Without further study, it is not possible to evaluate whether these concentrations represent potential hazards to ecological receptors because there are no sediment quality values for manganese and uranium.

Campground pond water

Two constituents of potential concern were identified in water collected from the campground pond: iron and zinc (Table 7.5).

A comparison of the surface water data with available water quality values indicates the concentrations of iron and zinc are below the state standards (Table 7.5). This suggests that the concentrations of iron and zinc would not represent a hazard to aquatic life and that the site is not affecting the water quality in the campground pond. However, concentrations may increase during low-flow conditions in the Gunnison River.

Campground pond sediment

The concentrations of manganese, uranium, and zinc detected in sediment from the campground pond were less than the concentrations detected at the upstream locations in both the Gunnison River and Tomichi Creek. This provides strong evidence that site-related contamination has not affected the sediment quality in this pond.

Fish tissue

As described previously in Section 4.2.5 and earlier in this section, fish were collected from two locations in the Gunnison River and from one location in both

Tomichi Creek and the campground pond. The results from the muscle tissue analyses are presented in Table 4.5. The concentrations are presented in units of mg/kg dry weight. Manganese was detected at the highest concentration (3.8 mg/kg) in the fish collected from the upstream Gunnison River location. The manganese concentrations in fish from the other locations ranged from 1.1 mg/kg (campground pond) to 1.9 mg/kg (downstream locations in both the river and creek). Molybdenum was detected only in the fish collected at the upstream Gunnison River location. Uranium was detected at a concentration of 0.46 mg/kg in fish tissue from the downstream Gunnison River location. Uranium was not detected (<0.20 mg/kg) in fish from the other sampling locations. Note, however, that these fish concentrations represent only the muscle tissue uranium levels; because uranium is a bone-seeking element, whole-body uranium concentrations could be considerably higher. Because only muscle tissue was analyzed for the site-related constituents, fish whole-body uranium concentrations could not be determined. Zinc was detected at similar concentrations in fish tissue from all the sampling locations; the concentrations ranged from 24 to 28.4 mg/kg.

There is little information concerning the relationships between tissue residue levels of constituents and biological effects in aquatic organisms. In a study involving rainbow trout, no adverse effects were noted in fish having a muscle tissue molybdenum concentration of 7 mg/kg fresh weight (Short et al., 1971). For comparative purposes, the dry weight concentration of molybdenum detected in the fish muscle tissue sample from the Gunnison River (0.26 mg/kg) was converted to fresh weight by using the average moisture content of 77 percent from the tissue samples (0.26 multiplied by [1-0.77] = 0.06 mg/kg). The converted concentration of 0.06 mg/kg is less than the concentration of 7 mg/kg detected in muscle tissue from the literature study (Short et al., 1971).

Manganese concentrations have been reported to range from 0.66 to 3.16 mg/kg fresh weight in prepared samples (headless, dressed, and homogenized) of northern pike and lake whitefish collected from several Canadian lakes (Uthe and Bligh, 1971). Similarly prepared lake trout collected from a New York lake were reported to contain manganese in concentrations ranging from 0.013 to 0.052 mg/kg fresh weight (Tong et al., 1974). Converting the detected muscle tissue concentrations from dry weight to fresh weight, the concentrations range from 0.25 mg/kg fresh weight (campground pond) to 0.87 mg/kg fresh weight (upstream in Gunnison River). Although these muscle tissue concentrations cannot be compared directly to the available literature data because the literature includes bone and skin, this finding indicates the concentrations are within the ranges observed in background populations from the literature.

The baseline zinc concentration in muscle tissue of rainbow trout used as control fish in a laboratory study was reported as 20 mg/kg dry weight (Goettl et al., 1972). The average concentration of zinc in muscle tissue of rainbow trout collected from several lakes used as background sites was approximately 18 mg/kg dry weight (Densinger et al., 1990). These concentrations are slightly lower than the concentrations reported in muscle tissue samples collected in the site vicinity.

The uranium concentration in whole-body samples of rainbow trout collected from a background area in a uranium mining district in Washington state ranged from 0.09 to 0.79 mg/kg dry weight (Nichols and Scholz, 1989). The concentration of uranium detected in the fish muscle tissue sample from the Gunnison River (0.46 mg/kg dry weight) is within the background range for whole-body samples reported in the literature. Although a direct comparison cannot be made between the muscle tissue concentration and the whole-body concentration (uranium is a known bone-seeker), this finding suggests that the uranium concentration in fish from the river may be within background ranges from the literature.

Based on the available site-specific data and literature information, no evidence suggests that bioaccumulation is a concern or that the constituents of potential concern would cause adverse effects to the fish at the levels observed. However, because of the habitat range of fish over a lifespan, the importance of these data with respect to the Gunnison site cannot be determined.

7.5 POTENTIAL IMPACTS TO LIVESTOCK

Contaminated ground water may discharge to the downgradient surface water bodies. The potential exists for livestock to drink water from the Gunnison River and/or Tomichi Creek. It is unlikely that livestock currently have access to the campground pond due to its present use as a designated camping area. However, if ownership of this property changes in the future, the pond could be used as a livestock watering pond.

Ingestion by livestock of vegetation that may have bioconcentrated constituents from alluvial ground water is a potential pathway. However, without additional data (e.g., actual plant tissue concentrations) this exposure pathway is difficult to evaluate.

To evaluate the potential impact to livestock that might drink from the surface water bodies, the detected concentrations were compared to approximate drinking water concentrations considered protective of livestock (Table 7.5). Although criteria are available for only three of the constituents of potential concern, comparing them to the surface water concentrations suggests livestock could safely use all the surface water bodies as a source of drinking water.

Based on past and current agricultural activities in the area surrounding the site, the possibility exists that ground water could be used in the future to provide a sole water source for a livestock watering pond. To evaluate the potential impact to livestock in this future hypothetical scenario, the mean ground water concentrations for the constituents of potential concern were compared to approximate drinking water concentrations considered protective of livestock (EPA, 1972) (Table 7.3). The comparison water quality criterion for sulfate is exceeded by the mean ground water concentration, while the mean concentrations for cadmium, cobalt, fluoride, and zinc are below the comparison criteria. Using this ground water as the sole source of drinking water for livestock could result in diarrhea in exposed animals (Church, 1984). No comparison water quality criteria have been reported for the remaining constituents of concern. However, available data suggest the sulfate levels would preclude the use of ground water as a sole source of drinking water for livestock.

7.6 LIMITATIONS OF THE ECOLOGICAL RISK ASSESSMENT

The qualitative evaluation of potential ecological risks presented here is a screening level assessment of the risks associated with potential exposure of plants and animals to contaminated ground water, surface water, and sediment at the Gunnison site. Sources of uncertainty in any ecological assessment arise from the monitoring data, exposure assessments, toxicological information, and the inherent complexities of the ecosystem. In addition, methods of predicting nonchemical stresses (e.g., drought), biotic interactions, behavior patterns, biological variability (i.e., differences in physical conditions, nutrient availability), and resiliency and recovery capacities are often unavailable. In general, limitations for the Gunnison ecological risk assessment include the following:

- Only a small amount of ecological data were collected during this screening.
- Little is known about site-specific amounts of constituents taken up by plants. General literature values were used to estimate this uptake in all cases.
- Only limited ecotoxicological reference data are available.
- Considerable uncertainty is associated with the toxicity of mixtures of constituents.

7.7 SUMMARY

Surface water data from the water bodies in the site vicinity indicate the presence of slightly higher concentrations for one constituent (sulfate) in Tomichi Creek at the downstream location. However, this sulfate level would not be associated with adverse ecological effects. None of the constituents in the Gunnison River were detected downstream of the site at concentrations above background levels. Two constituents (iron and zinc) were detected in water from the campground pond at concentrations slightly above background concentrations in the river or creek. However, there is no trend suggesting that site-related constituents have adversely affected the water quality of the creek, river, or pond.

A limited data set currently exists to characterize the sediment quality in the surface water bodies in the vicinity of the site. The data from the Gunnison River suggest that the site is not acting as a source of sediment-bound metals. Sediment concentrations in Tomichi Creek, for the metals which were analyzed, were higher downstream of the site than upstream. Insufficient data are available to determine whether this is due to releases from the site and/or from other sources unrelated to the site. Sediment constituent concentrations in the campground pond are less than those detected at the upstream locations in both the Gunnison River and Tomichi Creek.. Zinc concentrations detected in the river and creek sediments (both upstream and downstream of the site) and in the campground pond are below the available sediment quality values. Sediment quality values are not available for several of the constituents of concern; thus, it is not possible to evaluate the potential for these concentrations to represent an ecological concern.

Potential exposure to livestock drinking from the water bodies sampled in the site vicinity was evaluated. A comparison of available livestock drinking water quality values with concentrations detected in the surface water bodies suggests that livestock could use these water bodies as their sole drinking water source without adverse health effects. However, the quantity of data is limited; additional monitoring, expanded to include the wetland areas, could provide information on possible temporal and seasonal variations in water quality, as well as sediment quality.

Based on available data and criteria, no ecological threat exists to plants that may have roots in contact with soil saturated with the most contaminated ground water in the alluvial aquifer. This ground water would not be suitable for continuous use as irrigation water for crops due to cobalt, iron, and manganese. Water from the most contaminated wells in this aquifer would not be suitable as a sole source of water for fish habitat. This ground water also may not be suitable as a sole source of drinking water for livestock due to the laxative effects of sulfate.

The potential for the constituents of potential concern detected in media at the site to represent a food chain hazard (via bioaccumulation and biomagnification) is considered low, based on available surface water, sediment, and fish muscle tissue data. However, other than fish muscle tissue, no tissue analysis from other food chain organisms (e.g., invertebrates and plants) has been conducted.

Insufficient water quality and sediment quality values were available to allow a comprehensive evaluation of the impact of surface water, sediments, and contaminated ground water on ecological receptors. However, based on available data there is no evidence that the surface water and sediments in the vicinity of the site have been impacted by the former milling activities.

8.0 INTERPRETATION AND RECOMMENDATIONS

8.1 RISK SUMMARY

The UMTRCA requires the UMTRA Project to protect public health and the environment from radiological and nonradiological hazards associated with the uranium mill sites. This baseline risk assessment was conducted on the Gunnison site to evaluate the presence of these hazards. This risk assessment is conservative in that it evaluates residential exposure that could be associated with contamination from the highest exposure level by drinking water from a hypothetical well at the site. Because contaminated ground water currently is not used for drinking water by area residents, human health is not at risk.

Although some domestic wells downgradient of the processing site have been affected by site-related constituents, these residences were connected to a new water supply system in July 1994. Therefore, good quality drinking water is available to the residents. This favorable risk situation should continue if water uses at or near the site do not change; changes may or may not create future risks.

Health risks could be associated with potential future use of contaminated ground water from a hypothetical well placed in the alluvial aquifer for drinking; however, the likelihood of such use is considered very low because a new water supply system is available to the area residents. Note also that in the future residential scenario, only the people who drill a well in the most contaminated portion of the aquifer (a fraction of contamination) could experience the health problems discussed below. Drinking water from a future well drilled farther downgradient from the site could result in risks lower than estimated here. Furthermore, the ground water constituent concentrations will decline over time.

Adverse health effects from possible future ingestion of ground water at the former processing site could result from nearly the entire range of potential exposures to manganese and iron, as well as from the upper range of potential exposures to sulfate. Short-term exposures to sulfate levels in ground water could result in persistent diarrhea in infants (the most sensitive population group). The effect is reversible and diarrhea would terminate after substituting water low in sulfate for contaminated water. These sulfate levels also would make the water unpalatable, thus reducing its potential for consumption. Although sulfate short-term toxicity could preclude chronic exposures, the health effects from long-term exposures to constituents other than sulfate are discussed because different ground water constituents may flush out at different rates and because remedial action strategies may differ for different constituents. Exposure to iron could occur at concentrations that have been reported in literature to cause serious effects in children following short-term exposures. However, the likelihood of occurrence of such high exposures is considered extremely low for the Gunnison site. Manganese levels could produce neurological symptoms including memory loss, irritability, and muscle rigidity. Although adverse health effects have not yet been reported to occur in people as a result of chemical (noncarcinogenic) uranium toxicity following ingestion exposure to the levels found at the Gunnison site, uranium exposure should be considered a potential problem because its level exceeds the EPA acceptable intake level, which is derived based on animal studies. If this ground water were used for drinking water, possible lifetime exposures to radioactive uranium and its progeny could be associated with an overall lifetime cancer risk of about 3 in 1000; this exceeds the maximum excess cancer risk of 1 in 10,000 recommended as acceptable by the EPA for Superfund sites (40 CFR Part 300).

Use of contaminated ground water from the current or potential future wells downgradient of the site for irrigating crops or gardens and watering livestock is not anticipated to result in human health risks. Adverse human health effects would not be expected following ingestion of milk and meat from animals grazed and watered on the pastureland downgradient of the site or from ingestion of garden produce watered with the contaminated ground water. In addition, the excess lifetime cancer risk associated with human consumption of this produce, milk, and meat is estimated to be within the EPA-accepted range.

Based on available data, ingestion of fish collected from the water bodies in the site vicinity is not expected to result in adverse human health effects.

Assuming that a hypothetical pond were fed exclusively with the most contaminated ground water, some constituents in the water could adversely affect aquatic organisms such as fish. This water would not be acceptable as a sole source of continuous irrigation water for agricultural crops. In addition, the ground water may not be suitable as a sole source of drinking water for livestock due to the laxative effects of sulfate. However, a pond fed exclusively with ground water is unlikely because rainfall and surface water runoff probably would dilute constituent concentrations to safe levels.

The available surface water and sediment data from the water bodies in the site vicinity suggest that contaminated ground water from the site has not adversely affected the surface water and sediment quality. None of the constituent levels detected in the surface water and sediments exceeded the available water quality criteria or sediment quality values.

8.2 LIMITATIONS OF THIS RISK ASSESSMENT

The following limitations to this evaluation of health risks should be noted:

- This document evaluates risks associated with exposures only to inorganic ground water constituents at the UMTRA site near Gunnison. As discussed in Section 3.0, potential organic constituents (those few related to uranium processing) have not been considered.

- Uncertainties arise from the use of filtered versus unfiltered water samples. In general, the results presented in this document are based on filtered (0.45 μm) water samples. The effect of filtration differs for different elements. Although the difference on UMTRA sites is usually not large, filtered samples can have somewhat lower or equal concentrations than unfiltered samples for some constituents. Constituents in suspension may be lost with filtration, but still can produce toxic effects if ingested and broken down in the acidic environment of the stomach.

- The toxicity of any constituent varies from person to person. For example, normal variability in biochemical factors between individuals, differences in medical history, previous exposure to toxicants, and dietary and exercise habits can all affect susceptibility to chemical toxicity. In presenting exposure ranges that can produce toxic effects, this assessment tries to emphasize that variability. However, it is not possible to account for all sources of variability and still present useful and meaningful analyses. Specific subpopulations of individuals known to be more sensitive to toxic effects of given constituents have been noted. Using ranges for expected toxic effects provides a better understanding of the likelihood that toxicity might occur.

- To assess toxicity, standardized reference values developed by agencies such as the EPA are used to determine plant uptake, tissue concentrations in livestock, and toxic effects in humans. These reference values themselves have limitations, including the following:

    - Toxicity, uptake, and bioconcentration data are not available for all constituents elevated above background at the site.

    - In some cases, data obtained from laboratory animal testing at exposure doses different from those expected at the site were used to determine toxicity. The relationship between dose and response is not always linear and humans do not always exhibit the same response as animals.

    - Data used to determine toxicity generally are based on exposure to only the constituent of concern. In reality, exposures generally occur simultaneously to several chemicals. The interactive effects of multiple constituents and the impact of these interactions on expected toxicity generally cannot be accurately assessed from existing data.

- Although considerable effort has been directed at determining the areal extent of contaminated ground water and its movement and placing monitor wells in locations that capture maximal contamination, variability in physical systems and models used to determine plume migration could still result in well placements that do not measure the highest constituent concentrations or determine the fullest extent of plume impact. On the other hand, because the major source of contamination (the tailings and contaminated soils) at the Gunnison site has been removed, the assumption of a constant source used in this risk assessment may overestimate future risks. Because constituent concentrations could be higher or lower than used in the analysis, the net effect of these uncertainties on future risk estimates cannot be predicted.

- Some variability may have been introduced through sampling and analytical processes. However, the data at UMTRA Project sites have been collected over many years and subjected to rigorous quality assurance procedures to minimize this source of uncertainty. Testing multiple samples introduces high confidence in the reliability and validity of the collected data.

- The drinking water pathway is considered the major determinant of exposure in this assessment. Other pathways were screened and determined not to contribute substantially to the total exposure, however, the additivity of exposure from these pathways should be considered. When a measurable contribution from other pathways could increase expected exposure enough to alter the predicted toxicity, the alternate source contribution is noted in Section 6.0.

- Limited ecological data were collected for this screening. In addition, little is known about site-specific intake amounts of constituents taken up by plants. Limited ecotoxicological reference data and considerable uncertainty associated with the toxicity of mixed constituents add to the limitations of this risk assessment.

By presenting ranges of toxic effects, summaries of available data on health effects and interactions, and outlines of potential limitations, this document provides a reasonable interpretation of potential health risks associated with ground water contamination at this site. This assessment presents both contamination and risk as accurately as possible, based on available data, and conveys areas of uncertainty.

8.3 GROUND WATER CRITERIA

In 1983, the EPA established health and environmental protection standards for the UMTRA Project (40 CFR Part 192). The standards were revised and the final rule was published on 11 January 1995 (60 FR 2854). The ground water standards consist of ground water protection standards to evaluate disposal cell performance and ground water cleanup standards for existing contamination at processing sites. Table 8.1 summarizes concentration limits for constituents at the site. Because maximum concentration limits (MCL) are not established for every constituent, background levels should be met for constituents without MCLs. The standards also allow for supplemental standards or alternate concentration limits (ACL) where appropriate.

While these standards apply specifically to the UMTRA Project, the EPA has also published drinking water health advisory levels (EPA, 1995) for both long- and short-term exposures. Table 8.1 shows 10-day and lifetime health advisories.

8.4 RISK MITIGATION MEASURES

Because health risks could be associated with using the contaminated ground water from the alluvial aquifer below the former tailings site, the use of this ground water should be limited. This section presents possible ways to restrict access to ground water so as to mitigate risks.

Table 8.1 Concentration limits of constituents

Constituent	EPA MCL for UMTRA (mg/L)	Health advisories 10-kg child, 10-day[a] (mg/L)	Health advisories 70-kg adult lifetime[a] (mg/L)
Chemicals (inorganic)			
Antimony	NA	0.01	0.003
Arsenic	0.05	NA	NA
Barium	1.0	NA	2
Boron	NA	0.9	0.6
Cadmium	0.01	0.04	0.005
Chromium	0.05	1.0	0.1
Lead	0.05	NA	0.015[b]
Mercury	0.002	NA	0.002
Molybdenum	0.1	0.04	0.04
Nickel	NA	1.0	0.1
Nitrate	44[c,d]	44[e]	NA
Selenium	0.01[c]	NA	NA
Silver	0.05	0.2	0.1
Strontium	NA	25	17
Thallium	NA	0.007	0.0004
Zinc	NA	6.0	2
Radionuclides			
Radium-226/-228	5 pCi/L	NA	NA
Uranium (U-234/-238)	30 pCi/L[c] (0.044 mg/L)	0.03 mg/L[c,f,g]	0.1 mg/L[c,f]

[a] From EPA, 1995. The 10-day health advisory represents a chemical concentration in drinking water that is not expected to cause adverse noncarcinogenic effects up to 14 consecutive days of exposure, with a margin of safety; the lifetime health advisory represents the concentration of a chemical in drinking water that is not expected to cause adverse noncarcinogenic effects over a lifetime exposure, with a margin of safety.
[b] Action level.
[c] Exceeded in plume wells.
[d] Equal 10 mg/L nitrate as nitrogen.
[e] Under review.
[f] Proposed value (EPA, 1989a); under review.
[g] Longer-term health advisory.

NA - not available.

Institutional controls are defined in the EPA ground water standards as mechanisms that can be effectively used to protect human health and the environment by controlling access to contaminated ground water.

Although the proposed standards refer to institutional controls for long periods of time (e.g., up to 100 years during natural flushing), this concept can also be applied to short-term or interim restriction of access to ground water. Since not all 24 UMTRA Project sites can be evaluated simultaneously, interim institutional controls may be needed before remedial action decisions are made for individual sites.

At the Gunnison site, the residents living on developed land downgradient from the contaminant plume were provided bottled water, on an interim basis. In July 1994, as a permanent solution, a new water supply system was constructed to provide water for domestic use to the residents and the nearby Valco cement/concrete plant, eliminating the bottled water supply. At present, some of the existing wells are still being used to water residential lawns and gardens. The DOE has developed a ground water monitoring plan for residences downgradient from the contaminant plume that are not connected to this water supply system (DOE, 1995).

A portion of undeveloped land downgradient of the contaminated plume does not have an alternate water supply (Figure 2.7). This area is zoned for agricultural use, and water is provided by flood irrigation from Tomichi Creek. Any new development in this area would be required by the county to tie onto the alternate water system. The capacity of the water supply system would have to be evaluated should this occur. Any future attempt at development would require a change in zoning through application to the county. The placement of any wells would be regulated through either the county or the state, depending on well depth. The state of Colorado water law and land use regulations are discussed below.

Well permits

All Colorado UMTRA Project sites are on the Colorado west slope, outside the designated ground water basins. Construction of a well in Colorado outside the designated basins requires a written application to the state engineer for a permit. Designated basins are isolated hydrogeologic areas where ground water use is stringently evaluated based on the demands for water rights. The state engineer is required to act on applications for new well permits within 45 days after their receipt. If a well would affect existing water rights or if an applicant wanted to establish a legal right, adjudication would likely be required before the permit was granted. The state engineer is authorized to enforce the state ground water quality standards. However, the state engineer does not have jurisdiction to deny a permit for drilling a new well based on water quality.

Ground water quality

The Colorado Department of Public Health and Environment (CDPHE) is the state agency responsible for setting water quality standards. Within the CDPHE, the State Water Quality Control Commission is responsible for adopting the water quality standards and classifications for state waters in Colorado. The state of Colorado proposed ground water must quality standards require that ground water must be free of substances in concentrations shown to be "carcinogenic, mutagenic, teratogenic or toxic to human beings and/or a danger to public health, safety, or welfare" (CDH, 1990).

The state engineer's office can issue a warning to well users if the well is placed in a known contaminated aquifer. Private domestic well water quality is not regulated by the state. Well water that will be consumed by 25 or more people must meet water quality standards, and its use can be restricted by the CDPHE, Water Quality Control Division, Drinking Water Section.

Land use regulations

Any change in land use in Gunnison County would require a land use change permit, issued by the Gunnison County Board of Commissioners. Since the processing site and the surrounding land are outside the Gunnison city limits, city zoning restrictions do not apply.

Gunnison County's land use policies are set forth in a 1984 land use resolution. One policy of the resolution is "to ensure that adequate water in terms of quality, quantity, and dependability is both legally and actually available to development approved within Gunnison County" (Gunnison County Planning Department, 1984). The County Planning Department can designate an area a hazard zone if the criteria for a hazard zone listed in the land use resolution are met.

Establishing interim institutional controls to protect human health and the environment would require a consensus among the state of Colorado governing agencies and the Gunnison County Planning Commission. Governing authorities would need to be informed of ground water monitoring results and the anticipated duration of contamination problems. The presence of a public water supply system largely reduces the likelihood of ground water being used for domestic consumption at this site.

8.5 FUTURE SITE ACTIVITIES

Water quality and water level monitoring at the Gunnison processing site and vicinity involves site characterization and health-protection monitoring at the former processing site and in the vicinity. Ground water monitoring of private domestic and irrigation wells is being conducted (and will continue) in the Dos Rios subdivision area for the following reasons:

- Although the DOE has provided an alternate water supply system, a provision has been made to monitor a "buffer zone" of representative domestic wells in Unit 2 of the Dos Rios subdivision. The buffer zone is downgradient from the area covered by the water distribution system. Details of this monitoring plan are provided as the buffer zone monitoring plan, an attachment to the Gunnison UMTRA Project water sampling and analysis plan (DOE, 1995).

- The water distribution system was connected to households for potable water consumption and washing. Water needed for outside use may be withdrawn from private irrigation wells, which will be monitored to ensure the water quality remains suitable for irrigation and livestock watering.

If a notable increase in the concentration of a constituent is detected, several activities will be conducted to determine whether the increase indicates a significant change in water quality. Increases in analyte concentrations can occur for various reasons, including 1) artifacts of sample collection and laboratory analysis procedures, 2) changes in background water quality, 3) changes in the environment that are unrelated to uranium processing activities, 4) a single localized "pulse" in the geochemical environment or, 5) an actual increase in concentration due to the former uranium processing activities. Activities to determine the nature and extent of an increase in concentration may include reanalysis, resampling, comparison to other geochemical parameters, and trend analysis. If the increase is significant and perceived to be a health threat, the CDPHE and DOE will consult to evaluate potential actions.

As additional water quality and water level data are collected and interpreted, the ground water sampling plan for the Gunnison site will be updated annually to provide ongoing protection for public health and the environment, including future ground water monitoring of private domestic and irrigation wells in the Dos Rios subdivision area.

Most of the site characterization activities at the Gunnison processing site were conducted to acquire sufficient preliminary site characterization information for designing and implementing a surface remediation plan (for relocating the residual radioactive materials to the Gunnison disposal site). The results of the hydrogeologic testing were not intended to provide a conclusive hydrogeologic characterization of the Gunnison processing site and are considered estimated values. Additional site characterization activities will be conducted during the UMTRA Ground Water Project. These site characterization activities may include additional aquifer performance testing, installation and sampling of additional monitor wells, and installation of surface water level recorders and staff gages at the Gunnison River.

8.6 RECOMMENDATIONS

Additional data should be collected so that the interactions between surface water and ground water and the vertical and horizontal alluvial ground water flow system at and in the vicinity of the Dos Rios subdivision can be better characterized.

Because of the potential for contaminated ground water migration, an institutional control that advises against installation of high-capacity production wells on the west side of the Gunnison River may be warranted.

Additional characterization is recommended to further evaluate conditions of the surface water bodies and the potential for ecological impact in the site vicinity. Such additional characterization will be discussed in future Gunnison ground water documents and work plans.

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The following individuals contributed to the preparation of this report.

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40 CFR Part 192, *Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings*, U.S. Environmental Protection Agency.

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42 USC §7922 *et seq., Uranium Mill Tailings Remedial Action Amendments Act*, 5 November 1988.