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REV. 1

BASELINE RISK ASSESSMENT OF GROUND WATER CONTAMINATION AT THE URANIUM MILL TAILINGS SITE NEAR LAKEVIEW, OREGON

December 1995

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BASELINE RISK ASSESSMENT OF GROUND WATER CONTAMINATION AT THE URANIUM MILL TAILINGS SITE NEAR LAKEVIEW, OREGON

December 1995

Work performed under DOE Contract No. DEAC04-91AL62350

Prepared for
U.S. Department of Energy
Albuquerque Operations Office
Grand Junction Projects Office

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CITIZENS' SUMMARY

The U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project consists of the Surface Project (Phase I) and the Ground Water Project (Phase II). Under the UMTRA Surface Project, tailings, radioactive contaminated soil, equipment, and materials associated with the former uranium ore processing at UMTRA Project sites are placed into disposal cells. The cells are designed to reduce radon and other radiation emissions and to minimize further contamination of ground water. Surface cleanup at the UMTRA Project site near Lakeview, Oregon, was completed in 1989. The Ground Water Project evaluates the nature and extent of ground water contamination that resulted from the uranium ore processing activities. The Ground Water Project is in its beginning stages.

Human health may be at risk from exposure to ground water contaminated by uranium ore processing. Exposure could occur by drinking water pumped out of a hypothetical well drilled in the contaminated areas. Ecological risks to plants or animals may result from exposure to surface water and sediment that have received contaminated ground water.

A risk assessment describes a source of contamination, how that contamination reaches people and the environment, the amount of contamination to which people or the ecological environment may be exposed, and the health or ecological effects that could result from that exposure. This risk assessment is a site-specific document that will be used to evaluate 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 determine whether any action is needed to protect human health or the ecological environment.

RISK SUMMARY

In the area around the Lakeview site, ground water is used for domestic purposes such as drinking, bathing, and cooking, and for agricultural purposes. However, current users of the site-related contaminated ground water (for domestic or agricultural purposes) have not been identified at the Lakeview site. Therefore, people are not exposed to contaminants and there is no risk to human health. This situation will continue if land and water use at the site does not change. Access to the site-related contaminated ground water is not restricted. Since different land use may or may not create future risks, when specific land and water uses are determined they should be evaluated to identify any potential human health risks that could occur because of exposure to contaminated ground water.

This risk assessment evaluates possible future health problems associated with exposure to site-related contaminated ground water; the results indicate some health problems could occur if contaminated ground water were used as drinking water. Consequently, it is recommended that the site-related contaminated ground water should not be used as drinking water.

Evaluation of the potential impacts of site-related contaminated ground water on the ecological environment indicates the ground water would not be suitable for continuous irrigation of agricultural crops, as a long-term source of drinking water for livestock, or as a habitat for aquatic life (such as a ground water-fed pond).

Approximately 3000 feet south of the site are several private wells with some of the same constituents detected in them as the site-related contaminated ground water. Although it is unlikely that the constituents in these wells are from the site, the source has not been identified. Evaluation of the ground water from the private wells indicates the manganese levels could cause adverse health effects.

GROUND WATER QUALITY AND USES

Background ground water quality

Background ground water quality is the natural quality of ground water if uranium milling activities had not taken place. Ground water was sampled in two ground water zones in the immediate vicinity of the Lakeview site: a shallow zone and a deeper zone. There are also two types of background ground water in the area of the Lakeview site. One type is geothermal ground water, which comes from hot springs in the area. The geothermal water is characterized by high levels of total dissolved solids (TDS), sodium, sulfate, and chloride, and proportionately low amounts of calcium and magnesium. The nongeothermal background ground water, on the other hand, is characterized by a higher proportion of calcium, magnesium, and bicarbonate, and low levels of sulfate, chloride, and TDS. The quality of both types of ground water is naturally poor because of iron and manganese in the nongeothermal ground water, and because of high TDS and arsenic in the geothermal ground water.

Site-related ground water quality

There are two types of contaminated ground water at the Lakeview site. One type is associated with contamination that seeped from the former tailings pile and has high levels of sodium, calcium, magnesium, and sulfate, and low levels of chloride. The other type of contaminated ground water is associated with contamination that seeped from the former evaporation ponds and has high levels of sodium, sulfate, and chloride, and low levels of calcium and magnesium. The contaminated ground water associated with the tailings pile is chemically similar to the nongeothermal background ground water, and the contaminated ground water associated with the evaporation ponds is chemically similar to the geothermal background ground water. However, some chemicals are in both types of contaminated ground water. One explanation for this is that prior to construction of the tailings pile and evaporation ponds, the soil contained salts from background ground water. After the tailings pile and evaporation ponds were built, water passed through the tailings and ponds and through the soil, leaching the salts from the soil into the ground water. As a result, the contaminated ground water contains chemicals from the processing fluids as well as from natural salts. Therefore, although the composition of site-related contaminated ground water is similar to that of the background ground water, the concentrations of the constituents in the contaminated ground water are 10 to 20 times higher than background concentrations.

Elevated sulfate levels appear in both types of contaminated ground water, and the distribution of sulfate has been used to determine how far the contamination extends from the site. Based on the sulfate distribution, it appears that the shallow zone contains two areas of contamination. One area is associated with the former tailings pile and the other is associated with the former evaporation ponds.

Private well ground water quality

Located approximately 3000 feet south of the site are several private wells. Sulfate, manganese, chloride, sodium, calcium, and iron have been detected in some of these wells as well as some constituents that are detected in the site-related contaminated water at the site. Water quality analyses of ground water from these wells suggest that these constituents are not present because of site-related contaminated ground water, primarily because their pattern of occurrence in the private wells is inconsistent with site-related contamination. That is, wells having the highest constituent levels are farthest from the site, whereas levels in wells closer to the site are similar to background. Thus, based on limited coverage of the on-site and private wells, there appears to be no connection between site-related contamination and elevated constituents in private wells. Based on a water use survey, none of the owners of the wells with high constituent levels use the ground water for drinking water, although they do use it for bathing, cooking, livestock watering, irrigation, and miscellaneous household activities.

Drinking water is supplied by the city of Lakeview for residents within city limits; people living outside the city limits have private wells. The city obtains its water from springs in the mountains part of the year and from very deep wells located at least 1 mile (1.6 kilometers) south of the Lakeview site. These wells have not been impacted by contaminated ground water from the site. Although surface water is the primary source of irrigation water for crops in the site area, ground water is sometimes used for irrigation.

When contaminated ground water feeds a surface water body, it can contaminate the surface water and sediment. Limited analytical data are available for surface water and sediment. These data suggest surface water is not impacted by site-related contamination, and elevated concentrations of iron, manganese, and uranium detected in sediment may be due to natural variations in sediment composition. However, neither observation for surface water or sediment is conclusive because of the lack of data for these two media.

HUMAN HEALTH RISK ASSESSMENT

Methods

This risk assessment identifies contaminants that are present in ground water because of past uranium ore processing activities. This is done by comparing water quality data from wells drilled on the site to water quality in wells from background areas. This ground water analysis probably will overestimate the potential risk because the comparison uses ground water from the most contaminated part of the shallow zone. Contaminants detected in the ground water that could cause adverse human health effects if taken into the body are called contaminants of potential concern. Arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium are the contaminants of potential concern detected in the shallow zone at the Lakeview site.

Next, this risk assessment examines potential human health problems that could be caused by exposure to contaminated ground water. In a future scenario, a person in the Lakeview site area could be exposed to contaminated ground water by drinking or bathing in ground water, eating meat or drinking milk from livestock watered with contaminated ground water, or eating produce irrigated with contaminated ground water. All these potential exposure routes initially are screened for their potential contribution to high exposure levels. Of all of the exposure routes evaluated, drinking the contaminated ground water was determined to contribute the greatest exposure dose. Therefore, this document evaluates this exposure route in detail.

Health risks are evaluated for the age group most likely to experience health problems from drinking contaminated ground water. Children (aged 1 to 10 years) are most likely to be affected by drinking contaminated ground water because they are small and they tend to drink more water compared to their body weight than adults and consequently take in a higher contaminant dose than adults. Infants (aged 0 to 1 year) are especially sensitive to sulfate. Cancer risks are evaluated for adults (11+ years) assuming lifetime exposure.

The seriousness of health effects varies for several reasons. The levels of contaminants in ground water can vary over time. Also, people vary in body weight, the amounts of water they drink each day, and their reactions to chemical exposures. This risk assessment considers these differences whenever possible.

The risk assessment provides graphs showing the various exposure levels that might occur and the most current scientific information on the health effects that could result from this hypothetical exposure.

Results

Some possible future health risks are associated with drinking the ground water at the Lakeview site. Table CS.1 summarizes the nature of these health problems. Note that only people who drink all their water from a well placed in the most contaminated part of the ground water could experience the health problems listed in Table CS.1. Therefore, the table presents the upper limit of possible risks; real risks probably would be lower.

Severe health effects could develop due to the water's arsenic, iron, manganese, and sulfate content. Sulfate could affect infants in particular by causing severe diarrhea, quickly leading to dehydration. Manganese exposure could cause memory loss, irritability, and muscle rigidity. Long-term iron exposure could lead to cirrhosis of the liver. Arsenic exposure could cause adverse effects to the liver, arteries, and skin. Additionally, arsenic exposure could increase the risk of developing cancer.

The exposure levels from the other routes of exposure (skin contact with ground water while bathing, eating meat and drinking milk from livestock watered with contaminated ground water, and eating produce irrigated with contaminated ground water) would not be expected to cause adverse health effects if they would be the only source of exposure.

Table CS.1 Potential adverse human health effects of drinking site-related contaminated ground water, Lakeview, Oregon, site

Contaminant of potential concern	Potential health problems
Arsenic	Mild to severe toxicity including neurological effects, cirrhosis of the liver, thickening of the arteries, and skin discoloration. Individual excess lifetime cancer risk above the acceptable EPA risk range (skin cancer).
Boron	Adverse effects not expected to occur.
Iron	Mild to severe effects following long-term exposure: ranging from iron buildup in body organs and skin discoloration to cirrhosis of the liver.
Manganese	Mild to severe effects following long-term exposure: neurological symptoms such as memory loss, irritability, and muscle rigidity.
Molybdenum	Mild to severe effects following long-term exposure: copper deficiency anemia.
Nickel	Allergic dermatitis in sensitized humans.
Sodium and chloride	Hypertension following long-term exposure.
Sulfate	Short-term mild to severe effects: diarrhea leading to dehydration in infants.
Uranium	Health problems not expected from chemical toxicity; individual excess lifetime cancer risk within the acceptable EPA risk range.
Polonium-210	Individual excess lifetime cancer risk within the acceptable EPA risk range.

EPA - U.S. Environmental Protection Agency

Note: These effects could vary from person to person depending on the amount of water each person drinks and individual body weight, dietary habits, and sensitivities (for example, preexisting kidney or heart disease).

Furthermore, if the additional exposure levels would be added to the drinking water pathway, the exposure levels would not be expected to substantially increase the potential for adverse health effects.

Private well risk evaluation

Evaluation of the constituents in the private wells indicates that if the ground water is ingested over a long period of time, manganese could potentially cause adverse health effects in people. However, the assumptions used to evaluate the ground water in the private wells are very conservative and likely overestimate the potential risks.

ECOLOGICAL ASSESSMENT

Methods

The ecological risk assessment presented in this document is a qualitative screening level study that relies almost completely on existing site data. This assessment identifies habitats, aquatic life, terrestrial wildlife, livestock, and vegetation resources that could be affected by contamination from site activities, as well as potential exposure pathways.

The field of ecological risk assessment has many uncertainties and limitations including limited data from media of interest; limited information on how some contaminants affect plants, animals, and aquatic life; and the inherent complexities of the ecosystem. In addition, methods of predicting nonchemical stresses (for example, drought), biologic interactions, behavior patterns, biological variability (differences in physical conditions, nutrient availability), and resiliency and recovery capacities are often unavailable. Therefore, it is often difficult to determine if contaminants can affect the biological component of an ecosystem and to predict whether observed effects will adversely affect the ecosystem.

This risk assessment evaluates surface water and sediment from Hunters Hot Springs, Warner Creek, Thomas Creek, and Hammersley Creek. Concentrations of the contaminants of potential concern identified in surface water, sediment, and ground water are compared to applicable guidelines from regulatory agencies and literature values to determine their possible effects on aquatic life, terrestrial wildlife, livestock, and vegetation.

RESULTS

Surface water does not appear to be impacted and no contaminants of potential concern were identified. Three contaminants (iron, manganese, and uranium) were detected above background concentrations in sediments and were retained as contaminants of potential concern. However, because sediment guidelines are not available, it is not possible to evaluate the potential hazard to the ecological environment from these contaminants.

Ground water at the Lakeview site would not be suitable as a long-term source of drinking water for livestock, due to elevated levels of boron and sulfate. If the ground water were used in a pond that could be stocked with fish, the ground water concentrations of arsenic, boron, chloride, and iron would exceed the water quality values considered protective of aquatic life. Therefore, the contaminated ground water would not be suitable as a habitat for aquatic life.

The concentrations of boron, manganese, and molybdenum observed in the contaminated ground water could result in phytotoxicity to plants rooted in the contaminated ground water. Additionally, the concentrations of arsenic, boron, iron, manganese, molybdenum, and nickel exceed levels considered protective of plants in water used for irrigation.

CONCLUSIONS

Currently, no users of the site-related contaminated ground water (for domestic or agricultural purposes) at the Lakeview site have been identified. Therefore, people are not exposed to contaminants and human health is not at risk. This situation will continue if land and water use at the site does not change. However, the source of the constituents detected in the private wells south of the site has not been identified. Although it is unlikely that the constituents in these wells are from the site, it should be verified that the constituents are not site-related.

The results of this risk assessment indicate that if the site-related contaminated ground water is used for drinking water in the future, some health problems could occur. Consequently, it is recommended that the contaminated ground water should not be used as drinking water. However, this risk assessment is a conservative estimate of potential future risks at the Lakeview site because data from the most contaminated wells on the sites were used in this evaluation. Furthermore, the uranium mill tailings (the source of the contamination) have been removed, and contaminated ground water at the site will tend to disperse and dilute (move away from the site and decrease in concentration, although slowly, at this site), thereby reducing potential exposure concentrations over time. For these reasons, this risk assessment is a conservative estimate of risk to human health at the Lakeview site.

This risk assessment was performed without complete characterization of the contaminated ground water at the Lakeview site. To fully characterize the potential risks at the site, ground water contamination needs to be investigated further. Recommendations concerning the type of data needed are presented in this document. This risk assessment and future investigations will be used to determine compliance strategies for the UMTRA ground water standards.

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LIST OF ACRONYMS

Acronym	Definition
BCF	bioconcentration factor
DHHS	Department of Health and Human Services
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
FWS	U.S. Fish and Wildlife Service
HEAST	Health Effects Assessment Summary Tables
IRIS	Integrated Risk Information System
LOAEL	lowest observed adverse effect level
MCL	maximum concentration limit
NCP	National Contingency Plan
NEPA	National Environmental Policy Act
NOAEL	no observed adverse effect level
NRC	U.S. Nuclear Regulatory Commission
ODEQ	Oregon Department of Environmental Quality
PEIS	Programmatic Environmental Impact Statement
RDA	recommended dietary allowance
RfD	reference dose
RRM	residual radioactive material
SAR	sodium adsorption ratio
SF	slope factor
SQC	sediment quality criteria
TDS	total dissolved solids
UCL	upper confidence limit
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 if ground water contamination at the former uranium mill processing site near Lakeview, Oregon, could adversely affect public health or the environment. The Lakeview site is one of 24 abandoned uranium processing sites that are undergoing remediation in accordance with the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA) of 1978 (42 USC §7901 *et seq.*) under the oversight of the U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project. The 1988 amendments to the UMTRCA authorize the DOE to conduct ground water restoration activities. Under the UMTRA Ground Water Project, site-related ground water contamination is being evaluated and an appropriate ground water compliance strategy will be selected. Results of this risk assessment will be one of the documents used to develop a ground water compliance strategy for the Lakeview site.

Under the UMTRA Surface Project, the source of ground water contamination, residual radioactive material (RRM), was stabilized in an off-site disposal cell at Collins Ranch, Oregon. The disposal cell is engineered to prevent radon and other radiation emissions, and to prevent ground water contamination. Surface remedial action at the Lakeview site was completed in October 1989.

The programmatic environmental impact statement (PEIS) for the UMTRA Ground Water Project (DOE, 1995a) proposes a framework for selecting a ground water compliance strategy to meet the U.S. Environmental Protection Agency (EPA) ground water standards for the UMTRA Project (40 CFR Part 192). This baseline risk assessment evaluates potential current and future human health and environmental risks from ground water contamination at the Lakeview site, and provides information to help determine an appropriate ground water compliance strategy. Remaining data gaps in ground water characterization will be addressed in the Lakeview site observational work plan. Based on the PEIS, the site observational work plan, and this risk assessment, site-specific National Environmental Policy Act (NEPA) documentation will be prepared to identify the impacts of the proposed ground water compliance strategy for the site.

This risk assessment is a baseline assessment in that it describes preremediation ground water conditions at the site, with ground water quality only partially characterized. This document evaluates potential risks that may need attention before the site is fully characterized, and examines risk to human health and the ecological environment from the most contaminated wells at the site.

This risk assessment follows the human health risk assessment methodology for the UMTRA Ground Water Project (DOE, 1995b). The methodology is designed to evaluate human health risks associated with ground water contamination at inactive uranium processing sites. It allows the development of probabilistic distributions for exposure variables when data are sufficient and the simulation of exposure dose distributions, using Monte Carlo techniques rather than a single exposure dose calculation. Risk interpretation is based on a comparison between the predicted exposure dose distribution and observed toxicity ranges of contaminants of potential concern. Graphic presentations (figures) are designed to increase understanding of potential risks based on relative toxicity, likelihood of effect, and severity of effect.

The methodology used for the ecological portion of this risk assessment generally follows the EPA's *Risk Assessment Guidance for Superfund* (EPA, 1989a). Ecological risk assessment at the Lakeview site identifies potentially exposed habitats and receptors and qualitatively evaluates analytical data to describe how contamination could affect the ecological environment. Thus, this qualitative approach is a screening level assessment of the ecological risks associated with potential exposure to contaminated media at the site.

2.0 SITE DESCRIPTION

2.1 SITE BACKGROUND

The Lakeview site is approximately 1.5 mile (mi) (2.4 kilometer [km]) north-northwest of the town of Lakeview in Lake County, Oregon (Figure 2.1). Klamath Falls is about 96 mi (150 km) west of the site and the California-Oregon state line is approximately 16 mi (26 km) south of the site. The Lakeview mill was active from February 1958 until November 1960. It was constructed to produce uranium ore, and 171 tons (155 metric tons) of uranium oxide were sold to the Atomic Energy Commission (predecessor to the DOE). Ore was trucked to the mill from the White King and Lucky Lass mines of southern Lake County, Oregon. The ore was ground to a minus 65 mesh size and leached with sodium chlorate and sulfuric acid. 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 3 years of operation, the mill processed about 130,000 dry tons (120,000 metric tons) of ore with an average grade of 0.15 percent uranium oxide.

The designated site covers 258 acres (ac) (104 hectares [ha]); approximately 30 ac (12 ha) were occupied by the almost square-shaped tailings pile, approximately 64 ac (26 ha) were the evaporation ponds, and the remaining area included the former mill structures, ore storage area, and acres of windblown contamination from tailings. Figure 2.2 provides information on the site and land use in the vicinity of the site.

Surface remedial action at the Lakeview site consisted of cleanup of contaminated buildings, excavation of all contaminated soils, and transport to the disposal site. Approximately 926,000 cubic yards (yd^3) (708,000 cubic meters [m^3]) of contaminated materials from the tailings pile, evaporation ponds, windblown and waterborne deposit areas, and windblown-contaminated areas dispersed around the site were relocated to the Collins Ranch disposal site approximately 7 mi (11 km) north of Lakeview. Approximately 736,000 yd^3 (563,000 m^3) of these contaminated materials are classified as tailings (DOE, 1994). The tailings were moved because seismic and geothermal activity in the area prevented on-site stabilization and disposal. The disposal site was selected based on public input, environmental considerations, and design opportunities acceptable to the state of Oregon and the U.S. Nuclear Regulatory Commission (NRC).

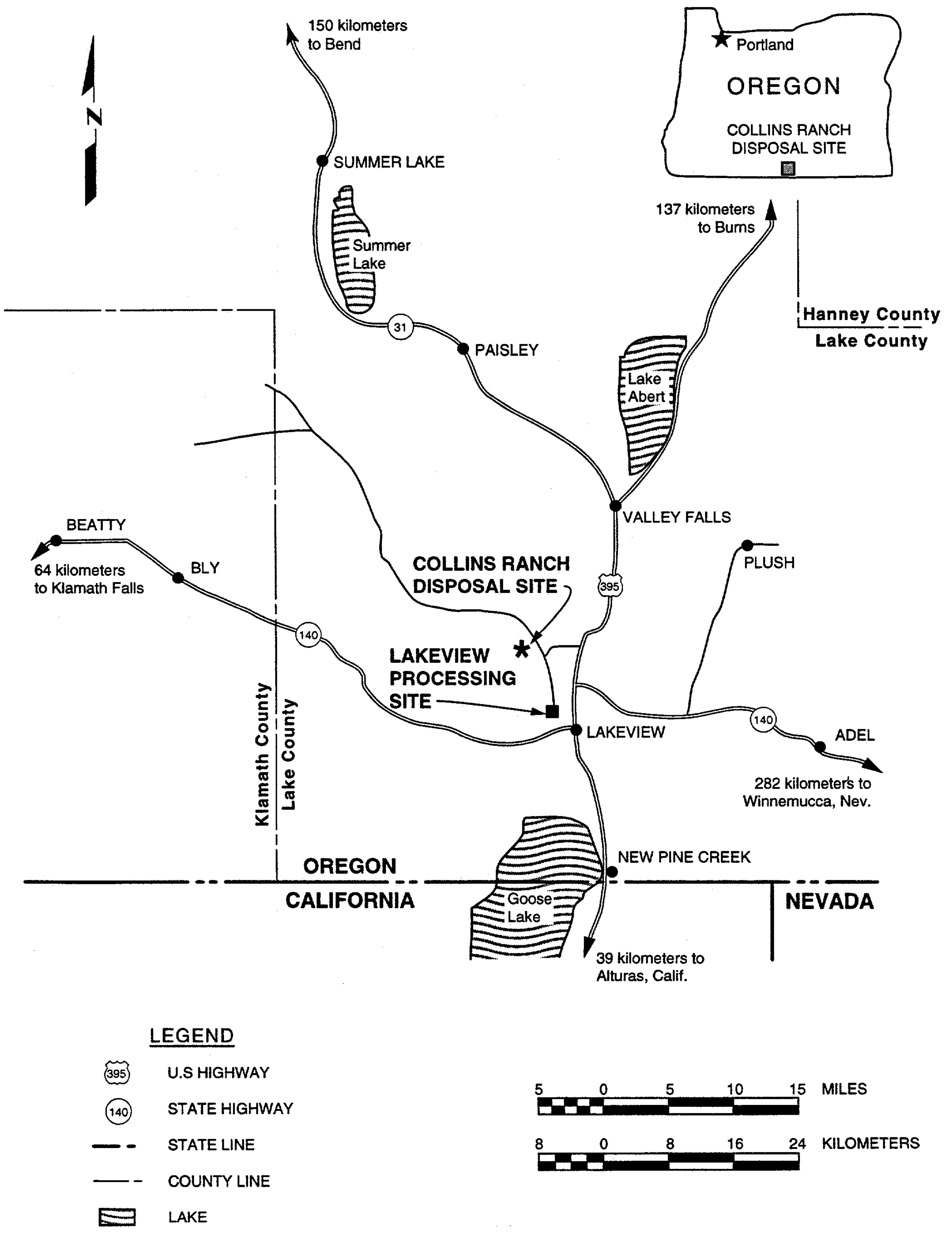


FIGURE 2.1
LOCATION MAP
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/SITELOC

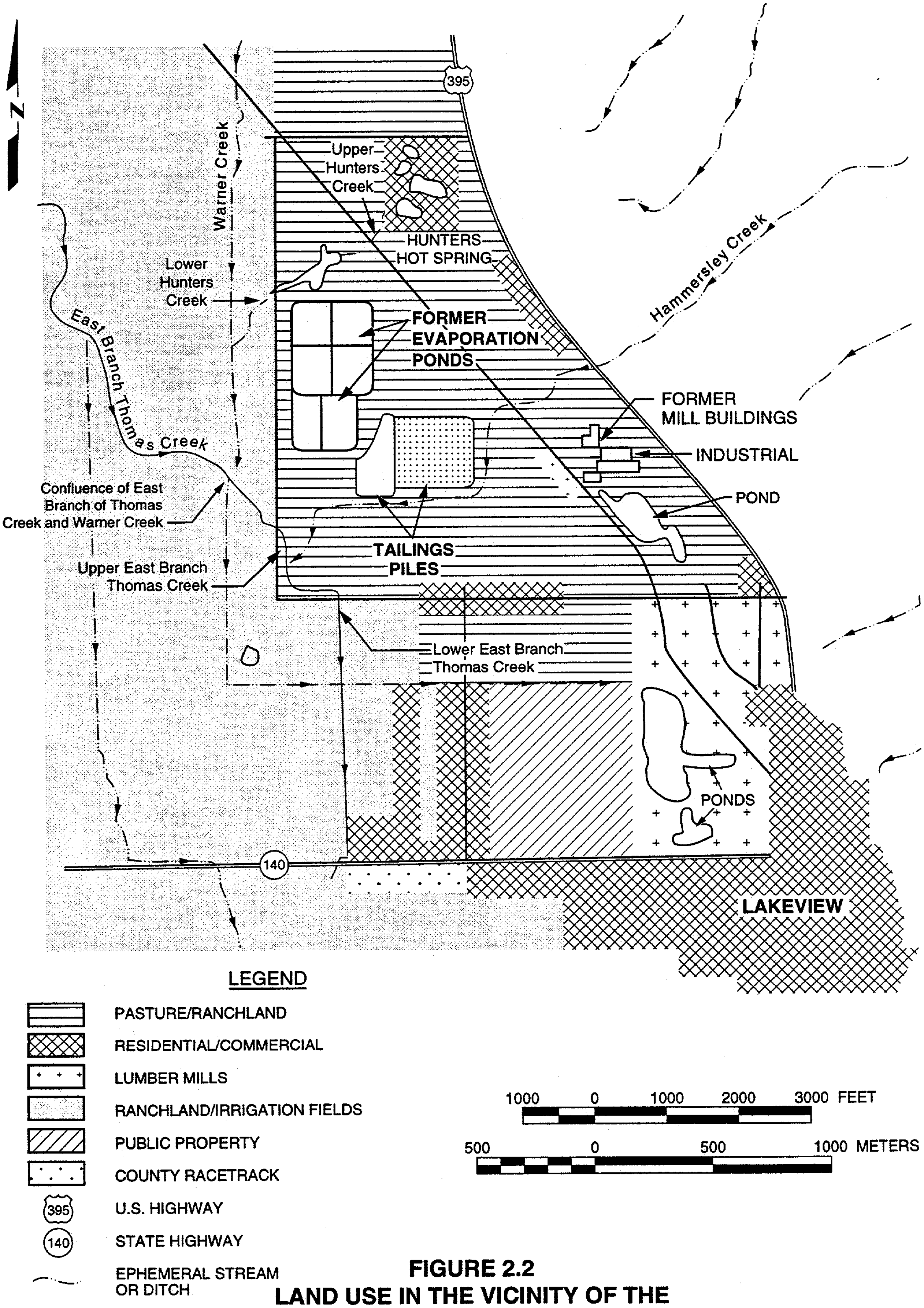


FIGURE 2.2
LAND USE IN THE VICINITY OF THE LAKEVIEW, OREGON, SITE

2.2 CLIMATE

The Lakeview area is characterized by low humidity, frequent sunny days, and moderate seasonal temperature ranges. The average annual temperature is 46 degrees Fahrenheit (°F) (7.8 degrees Celsius [°C]) and ranges from an average temperature of 27°F (-2.8°C) in January to 67°F (19°C) in July. The average annual precipitation is 14 inches (36 centimeters [cm]) (NOAA, 1974). Maximum rainfall occurs during January and February, while the least rainfall occurs from June through October. Thunderstorms are common during the summer. The average annual snowfall accumulation is 48 inches (122 cm), with the largest amount (15.6 inches [40 cm]) falling during January (NOAA, 1974).

2.3 HYDROGEOLOGY

The Goose Lake Valley area, which includes the city of Lakeview and the processing site vicinity, is underlain by more than 2000 feet (ft) (600 meters [m]) of heterogeneous alluvial and lacustrine deposits (DOE, 1992). Unconsolidated alluvial deposits consisting of silt and sandy clay extend from the ground surface to an approximate depth of 20 ft (6 m). Underlying the alluvial deposits are discontinuous lenses of unconsolidated to consolidated water-bearing gravels, sands, and silts derived from recent fluvial and ancient lacustrine processes. A known geothermal resource area, Hunters Hot Springs, is approximately 2000 ft (600 m) upgradient of the former mill site. Hunters Hot Springs provide an upgradient source of thermal water northeast of the site that recharges surface water and ground water sources in the site vicinity.

Two alluvial/lacustrine ground water zones have been characterized at the former processing site (DOE, 1992). Monitor wells are screened and filter-packed in a shallow zone from about 16 to 30 ft (5 to 9 m) deep and a deeper zone between depths of 60 to 90 ft (18 to 27 m). DOE monitor well locations at the processing site are shown in Figure 2.3.

Regional ground water recharge occurs from mountains on the west (Fremont Mountains) and east (Warner Mountains) sides of Goose Lake valley, precipitation, snowmelt, and upward movement from the underlying geothermal system. Ground water discharges to Goose Lake, surface drainages, springs, and wells.

A water table contour map of the shallow ground water zone at the processing site is shown in Figure 2.4. Shallow ground water beneath the site generally moves from northeast to southwest under a hydraulic gradient ranging from 0.003 to 0.010. The water table beneath the Lakeview site generally occurs at a depth of 5 to 15 ft (1.5 to 4.6 m).

Vertical gradients have been noted between the shallow and deep monitor well clusters (DOE, 1993). The background well cluster (501 and 502) shows head

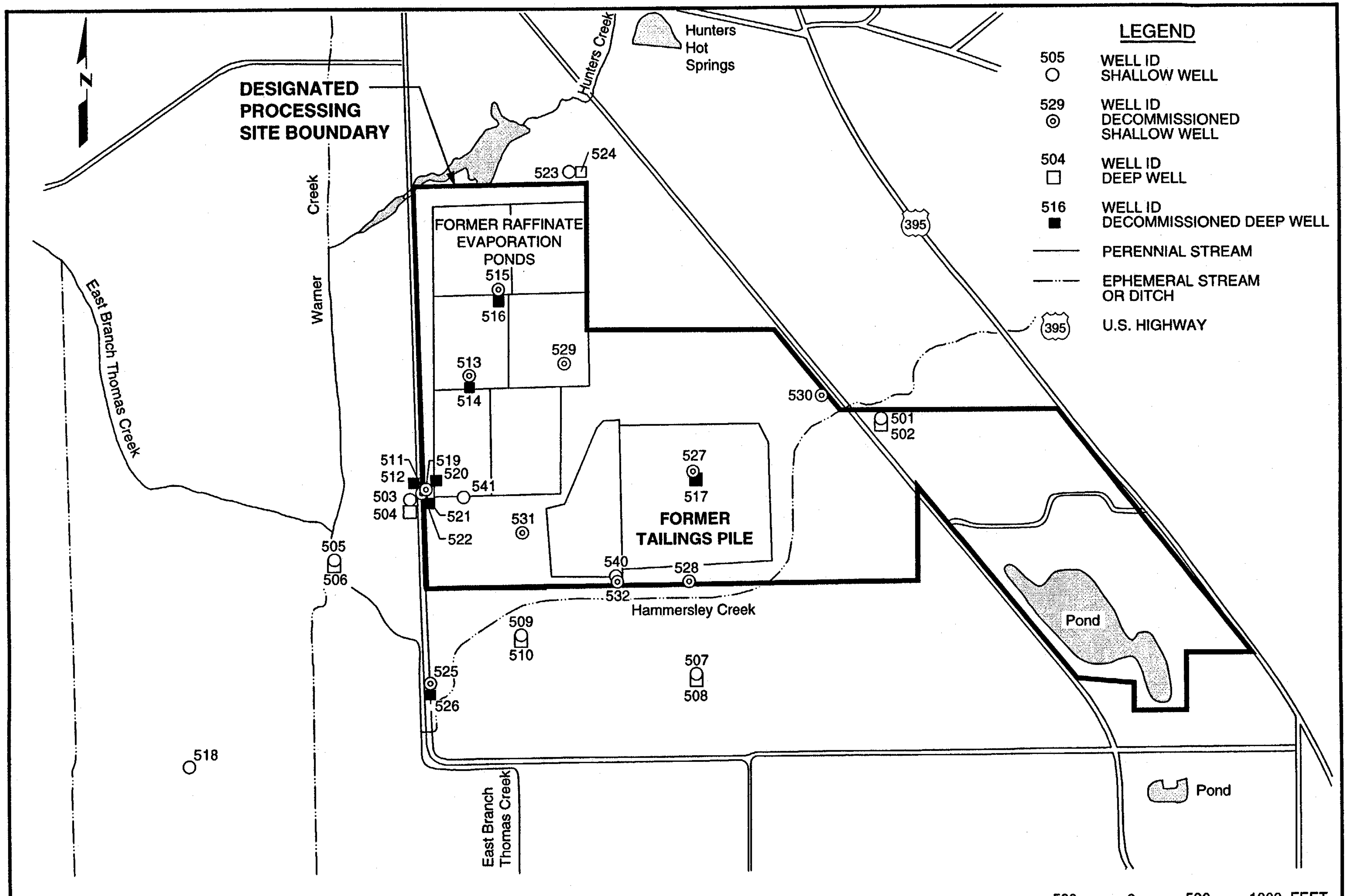


FIGURE 2.3
LOCATION OF SELECTED DOE MONITOR WELLS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE-DOEMONWELLOC

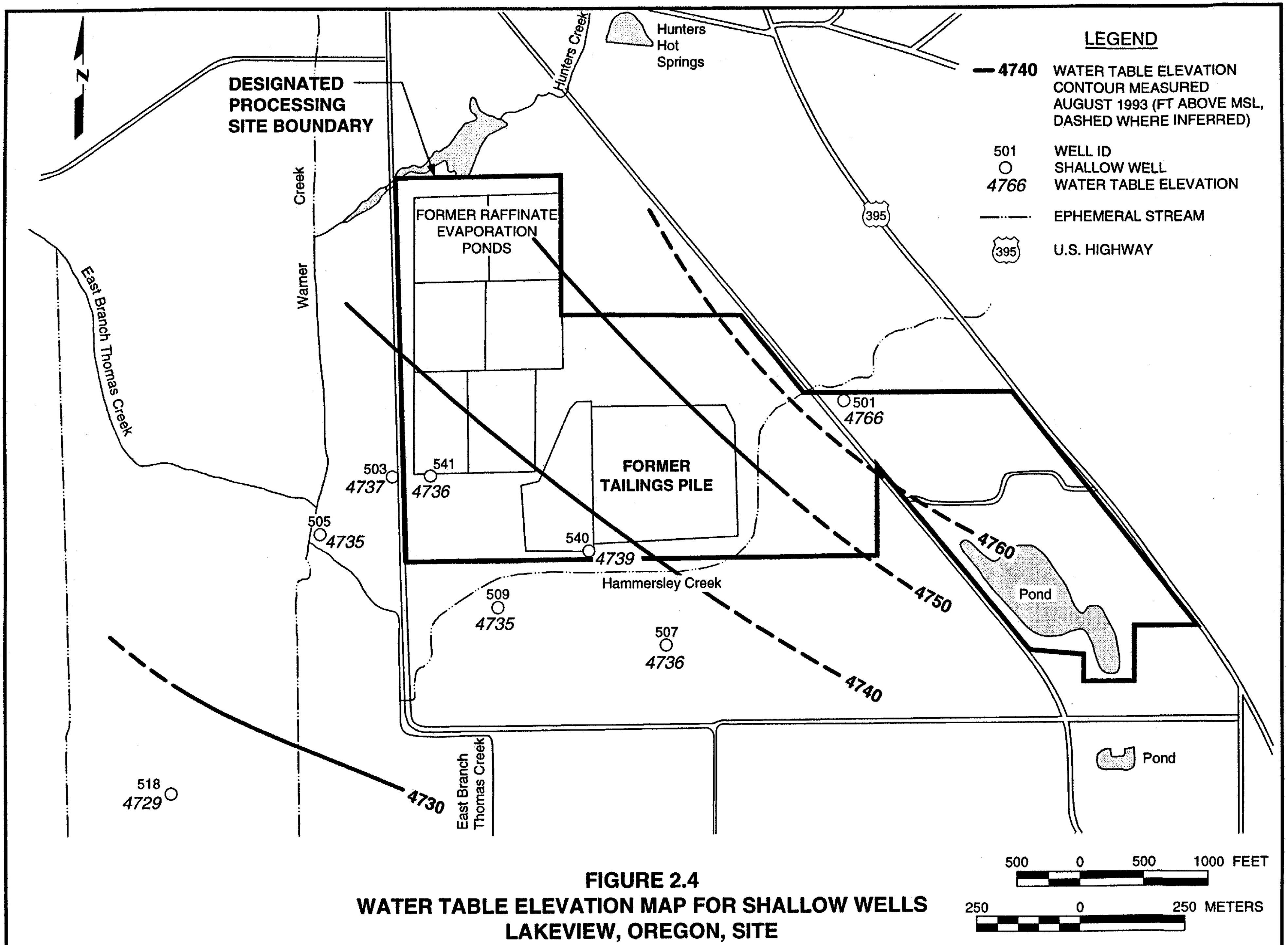


FIGURE 2.4
WATER TABLE ELEVATION MAP FOR SHALLOW WELLS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE-WATERTABLE

variations between the shallow (501) and deep (502) wells that range from 13 to 17 ft (4 to 5 m), indicating a downward vertical gradient. Water level elevations in geothermal monitor well cluster 523 and 524 are similiar, indicating no clear vertical gradient. Static water level elevations in the downgradient monitor well clusters range from similiar constant measurements (within ± 1 ft [0.3 m]) to downward gradients with static head differences of 1 to 12 ft (0.3 to 4 m) (Table 2.1). Water levels are expected to fluctuate due to a combination of natural discharge, evapotranspiration, and localized pumpage in private wells.

Hydraulic properties at the Lakeview site were estimated; the results are presented in the surface remedial action plan (DOE, 1992) and the Lakeview site characterization report (DOE, 1985a). These site characterization efforts included two 48-hour aquifer performance tests (with 24 hours of continuous pumping followed by 24 hours of recovery) to determine the hydraulic properties of the shallow and deep lacustrine zones. In one test, a shallow monitor well (519) was pumped at a rate of 0.60 gallons per minute (gpm) (38 cubic centimeters [cm^3] per second) and water levels were measured in three shallow observation wells (503, 511, and 521). Hydraulic conductivity values were calculated to range from approximately 0.92 to 11 ft per day (3.2×10^{-4} to 39×10^{-4} cm per second), using the Jacob-Cooper, Theis, and Boulton methods of analysis. In the other aquifer performance test, a deep monitor well (520) was pumped at a rate of approximately 12 gpm (760 cm^3 per second) and the water levels in one shallow (522) and three deep (504, 506, and 512) observation wells were measured. Hydraulic conductivities for the three deep monitor wells were estimated to range from approximately 0.78 to 8.9 ft per day (2.7×10^{-4} to 31×10^{-4} cm per second), using the Boulton, Hantush, and Jacob-Cooper methods of analysis. Hydraulic conductivity of the shallow monitor well ranged from 5.7 to 6.0 ft per day (20×10^{-4} to 21×10^{-4} cm per second), using the Jacob-Cooper and Boulton methods of analysis, respectively. These aquifer tests indicated that the two zones are hydraulically connected.

The calculated average linear ground water velocities range from 50 ft per year (8×10^{-5} cm per second) in the shallow zone to 160 ft per year (2.5×10^{-4} cm per second) in the deep zone. These values were determined using the hydraulic conductivities (ranging from 0.78 to 11 ft per day [2.8×10^{-4} to 39×10^{-4} cm per second]) and an assumed effective porosity of 0.10 (DOE, 1985a).

2.4 SURFACE WATER

The Lakeview site is in the drainage basin of Thomas Creek, a perennial tributary of Goose Lake. Goose Lake is a closed-basin lake with an overflow elevation of approximately 4720 ft (1440 m). This elevation is approximately 35 ft (11 m) below the base of the former tailings pile. The lake is approximately 8 mi (13 km) south of the site (DOE, 1985b).

Surface water features at and adjacent to the former processing site include three creeks, hot springs, ponds, and drainage ditches. An unnamed

Table 2.1 Static ground water levels, Lakeview, Oregon, site

Location ID	North coordinate (ft)	East coordinate (ft)	Formation of completion	Flow relationship	Casing elevation (ft MSL)	Ground elevation (ft MSL)	Log date	Log time	Depth from top of casing (ft)	Depth from ground (ft)	Ground water elevation (ft MSL)
LKV-01-0501	8523.5	7525.3	SP	Upgradient	4771.05	4769.32	08/25/93	15:29	5.28	3.55	4765.77
LKV-01-0503	7931.9	3921.1	SP	Downgradient	4747.73	4745.89	08/25/93	15:54	10.99	9.15	4736.74
LKV-01-0505	7476.2	3342.9	SP	Downgradient	4744.64	4742.74	08/25/93	15:59	9.88	7.98	4734.76
LKV-01-0507	6599.3	6102.0	SP	Crossgradient	4742.77	4741.00	08/25/93	16:35	6.76	4.99	4736.01
LKV-01-0509	6897.1	4762.6	SP	Downgradient	4742.14	4740.06	08/25/93	16:47	7.36	5.28	4734.78
LKV-01-0518	5914.7	2231.0	SP	Downgradient	4739.79	4737.55	08/25/93	16:18	10.31	8.07	4729.48
LKV-01-0540	7340.20	5486.57	AL	Downgradient	4747.89	4745.93	08/25/93	15:20	9.09	7.13	4738.80
LKV-01-0541	7938.71	4223.85	AL	Downgradient	4747.56	4745.55	08/25/93	15:10	11.34	9.33	4736.22

Report date: 1 July 1994.

SP – sand or gravelly sand, poorly graded.
AL – alluvium.
MSL – mean sea level.

intermittent stream issuing from Hammersley Canyon, northeast of the site, flows across the site adjacent to the north side of the pile. This creek is referred to as Hammersley Creek. Thomas Creek flows approximately 1 mi (2 km) west of the site; the East Branch of Thomas Creek flows 1000 to 2500 ft (300 m to 760 m) west of the site. An ephemeral stream, Warner Creek, flows approximately 1000 ft (300 m) west of the site where it joins the East Branch of Thomas Creek. The confluence of the East Branch of Thomas Creek with Hammersley Creek is approximately 1800 ft (550 m) southwest of the site. Surface water flows are also encountered approximately 0.5 mi (0.8 km) north of the tailings in the Hunters Hot Springs area. All drainage flows northeast to southwest across the site.

The drainage area for Hammersley Creek is about 3.3 square miles (mi^2) (8 square kilometers [km^2]) with an approximate slope of 0.06. Surface water in an area of approximately 30 mi^2 (80 km^2) drains into Thomas Creek above the Lakeview site. Most of this drainage basin is in the wooded area northwest of Lakeview. The drainage basin for Goose Lake is approximately 1140 mi^2 (2920 km^2), including the lake area.

2.5 LAND USE

Livestock grazing is the most extensive land use in Lake County, with approximately 69 percent of the county's total surface area used for this purpose (Figure 2.2) (DOE, 1985b). Timber production is another major land use, with about 23 percent of the county's total surface used for this purpose, primarily in the mountains. Approximately 4 percent of the county's land area, primarily in the intermountain basin area, is used to irrigate crops; this area includes the former processing site. The city of Lakeview is zoned mostly for single-family residential development. Demand for residential land in the site vicinity is very low (DOE, 1985b) and is expected to remain low because population over the last several years has not increased.

The former processing site, approximately 1 mi (2 km) north-northwest of the city of Lakeview, is zoned for heavy industry. However, current land use in the area immediately surrounding the site is primarily agricultural, i.e., grazing or growing forage crops such as alfalfa. The fields are marshy to the northeast, north, and west of the site because of surface discharge from the geothermal springs 800 to 1800 ft (240 to 550 m) north of the site. These marshy areas, including several small ponds, are associated with Hunters Creek. At the headwaters of Hunters Creek, about 1800 ft (550 m) north of the site, a lodge serves tourists visiting the hot springs. Farther north is ranch land, including three residences about 0.5 mi (0.8 km) north of the site.

The area immediately east of the northern portion of the site is pastureland with about seven associated residences and a trailer park on Highway 395, about 1600 ft (480 m) east of the site. East of the southern portion of the site are pastureland, an auto repair shop, and a small lumber mill (about 1000 ft [300 m] east of the site) set among larger areas of undeveloped land. Farther east, along

U.S. Highway 395, is the former uranium mill site, which is privately owned and used by local companies to store lumber. Land east of Highway 395 is primarily residences. The area southeast of the former mill site is primarily heavy industry (lumber mills). Immediately south of the site is pastureland.

The nearest residences are about 1000 ft (300 m) south and southeast of the site. These are single family homes separated from the site by open fields. To the south of these are county fairgrounds and a few residences mixed with light industry. The area southwest and west of the site is pastureland and forage crop land irrigated by canals.

2.6 WATER USE

Information on private, municipal, and county wells in the Lakeview area is presented in Table 2.2. Of nearly 100 wells on file at the Oregon State Engineer's Office, approximately 23 private wells are approximately within 1 mi (2 km) of the Lakeview site (Figure 2.5). These private wells are south of the site. Because of the lack of monitor well coverage between the site and the private wells, it has not been determined if these wells are crossgradient or downgradient from the site.

Surface water, primarily from Thomas and Warner Creeks, is used for irrigation through a system of irrigation canals in areas west, southwest, and south of the site. Ground water is also used for irrigation.

Drinking water is supplied by the city of Lakeview within city limits; residents outside the city limits have private wells. Lakeview obtains its water from two sources. In the winter and spring, water is obtained from springs in the mountains east of the city. In summer and fall, as these springs dry up, water is obtained from six water wells within the city limits. These municipal water supply wells are in sections 16 and 17. All these wells are more than 1 mi (1.6 km) south of the site and are completed at depths significantly greater (approximately 250 to 350 ft [76 to 110 km] below land surface) than the known extent of site-related contamination. The municipal wells are analyzed routinely for water quality by the state of Oregon and to date have met all EPA primary drinking water quality standards. Iron and manganese have exceeded secondary drinking water quality levels.

Approximately 10 residences are along Missouri Avenue, about 1100 ft (340 m) south of the site. Another group of up to 15 residences is south of Missouri Avenue along Roberta Avenue, between 2700 and 5100 ft (820 and 1600 m) south of the site. A telephone survey of these two groups of residents indicates that while a few homes on Missouri Avenue are connected to city water, most use shallow wells as their only source of domestic water, including drinking water. Due to poor water quality in some Roberta Avenue residences, private well water (wells 546, 547, 548, 549, and 550) is used only for domestic purposes other than drinking (cooking, laundry, livestock watering, etc.). Bottled water is used for drinking.

Table 2.2 Private and public wells, Lakeview, Oregon, site vicinity

Location	Number of wells	Depth range (ft)	Depth range (m)	Screen range (ft)	Screen range (m)
East of processing site					
Upgradient Section 3					
	1	685	209	None	None
	3	105-317	32-97	50-280	15-85
Crossgradient Section 10					
	4	155-310	47-94	70-310	21-94
	5	58-603	18-184	38-603	12-184
Downgradient Section 15					
City well	1	595	181	NA	NA
City well	2	1820-2380	555-725	NA	NA
City well	3	50-405	15-123	24-340	7-103
North/south of processing site					
Upgradient Section 4					
County well	4	8-509	2-155	8-505	2-154
Public well	16	70-415	21-126	32-360	10-110
	1	320	98	160-240	49-73
	5	100-300	30-91	50-300	15-91
	4	150-560	46-171	30-400	9-122
Downgradient Section 9					
	2	No information other than "drilled"			
	5	90-800	27-244	35-515	11-157
County well	4	17-60	5-18	NA	NA
County well	6	35-330	11-101	60-320	18-98
BLM well	2	46-115	14-35	40-100	12-30
Far downgradient Section 16					
County well	1	6	2	NA	NA
County well	5	101-140	31-43	NA	NA
	2	81-123	25-38	81-123	25-37

Table 2.2 Private and public wells, Lakeview, Oregon, site vicinity (Concluded)

Location	Number of wells	Depth range (ft)	Depth range (m)	Screen range (ft)	Screen range (m)
Far downgradient Section 16 (continued)					
City well	1	371	113	71-371	22-113
	3	81-132	25-40	59-131	18-40
	3	230-365	70-111	NA	NA
City well	3	80-360	24-110	22-340	7-104
	1	300	91	280-300	85-91
West of processing site					
Crossgradient Section 5					
	1	165	50	40-95	12-29
	1	130	40	NA	NA
	3	48-223	15-68	48-210	15-64
Downgradient Section 8					
County well	2	5-300	2-91	NA	NA
	2	43-?	13-?	NA	NA

Note: This information is from well logs on file at the Oregon Water Resources Department in Salem, Oregon. Data on depth and screen interval are not always provided. Screen interval includes perforation depths. Wells are private unless noted otherwise. Section numbers refer to sections in Township 39 South, Range 20 East.

NA – no information available.
? – unknown well depth.

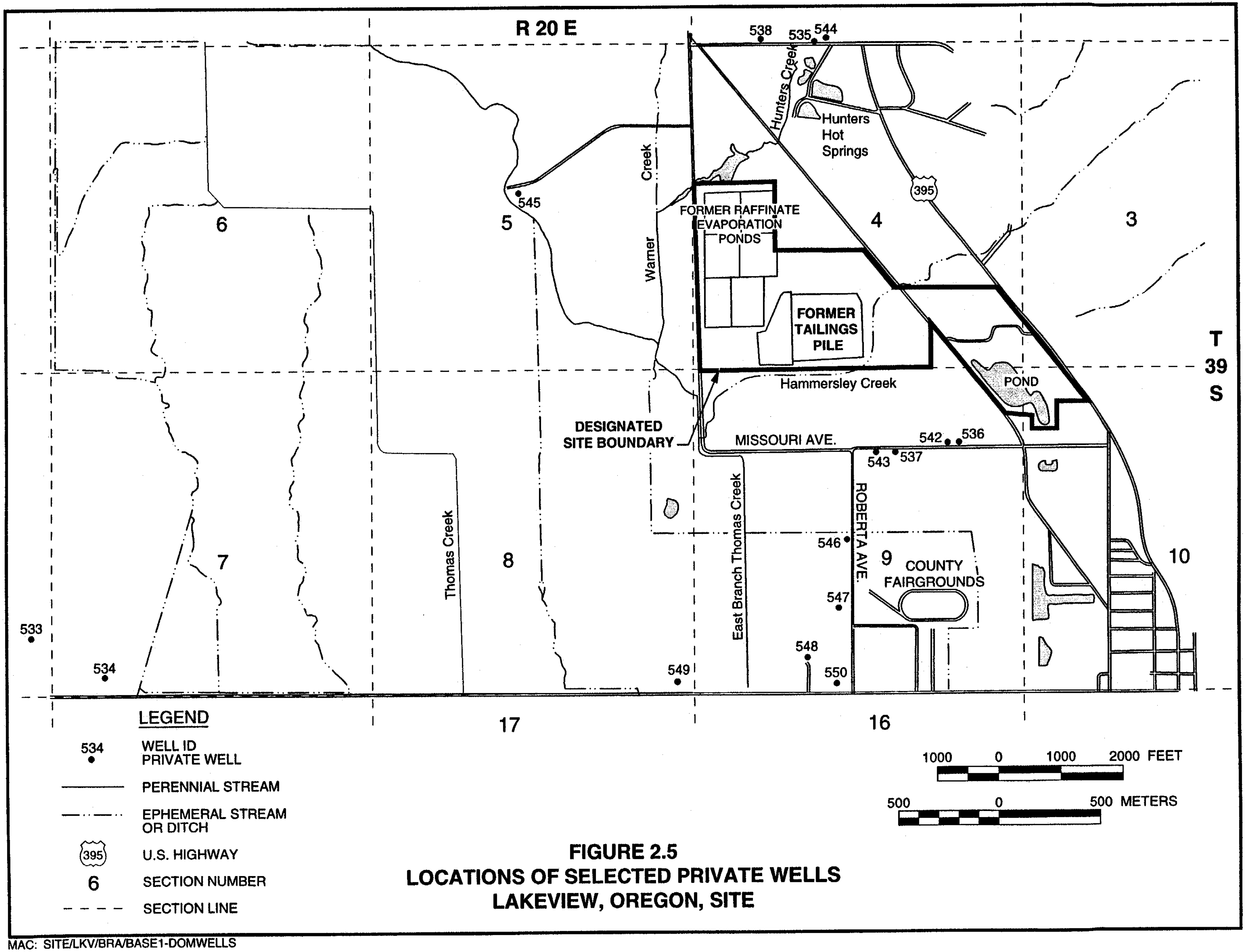


FIGURE 2.5
LOCATIONS OF SELECTED PRIVATE WELLS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE1-DOMWELLS

3.0 EXTENT OF CONTAMINATION

Water quality data were collected from 1982 to 1993 from 34 monitor wells, located both on and off the site, and from 14 private wells (discussed in Section 3.3). The locations of monitor wells and private wells are shown in Figures 2.3 and 2.5, respectively. Well completion data and sampling information from the DOE monitor wells are provided in Table 3.1 (DOE, 1995b). Eighteen of the monitor wells were decommissioned during surface remediation of the site, while two additional wells were completed after remediation. Two monitor wells were decommissioned in 1993 after they were noted in the field to have been seriously damaged since the September 1992 water quality sampling event. At present, 15 monitor wells exist at the Lakeview site.

Ground water quality data from 1984 through 1993 were used in this risk assessment (DOE, 1995b). Data from the 1982 sampling round were not incorporated because the reported concentrations of several key constituents were inconsistent with results of all subsequent sampling rounds.

Most monitor well water quality data were obtained from filtered water samples (0.45-micron filter). These data were used to evaluate the extent of contamination at the Lakeview site. For a few sampling rounds, both filtered and unfiltered water samples were collected and analyzed for most constituents. All available unfiltered data were examined and incorporated as needed for exposure assessment, as explained below (DOE, 1995b). In contrast to monitor well data, private well data were not filtered prior to laboratory analysis.

The samples were analyzed for the following inorganic chemicals: aluminum, ammonium, antimony, arsenic, barium, beryllium, boron, bromide, cadmium, calcium, chloride, chromium, cobalt, copper, cyanide, fluoride, iron, lead, magnesium, manganese, mercury, molybdenum, nitrate, nickel, phosphate, potassium, selenium, silica, silver, sodium, strontium, sulfate, sulfide, thallium, tin, uranium, vanadium, and zinc, and the following radionuclides: lead-210, polonium-230, radium-226, and thorium-230. No data on organic chemical contaminants were collected from ground water at the site during this period.

3.1 BACKGROUND GROUND WATER QUALITY SUMMARY

Background ground water quality is defined as the quality of water that would exist if uranium milling activities had not occurred. At the Lakeview site, there are two distinct types of background ground water upgradient from the site: geothermal and nongeothermal. Geothermal ground water issues from at least two hot springs (Hunters Hot Springs) immediately upgradient from the evaporation pond area of the site (Figure 2.2). The second type of natural ground water (nongeothermal) is found upgradient of the tailings pile area. Monitor well 523, upgradient and off the site, is considered the geothermal background well. Monitor well 501, also upgradient, is representative of nongeothermal background water.

Table 3.1 Summary of DOE monitor wells, Lakeview, Oregon, site

	Depth		Screen			Number of rounds
Location ID	(ft)	(m)	(ft)	(m)	Sampling date	
Upgradient						
LKV-01-0501	25	8	20-25	6-8	1984-93	20
LKV-01-0502	85	26	80-85	24-26	1984-92	18
LKV-01-0530	31	9	16-15	5-8	1982-85	3
LKV-01-0523	25	8	20-25	6-8	1984-85	3
LKV-01-0524	80	24	70-75	21-23	1984-85	3
Downgradient, shallow evaporation pond area						
LKV-01-0503	25	8	20-25	6-8	1984-93	21
LKV-01-0505	30	9	23-28	7-8.5	1984-93	21
LKV-01-0511	34.5	10.5	27-32	8-10	1984-85	2
LKV-01-0513	29.5	9.0	17-22	5-7	1984-85	3
LKV-01-0515	24.5	7.5	20-25	6-8	1984-85	2
LKV-01-0518	29.5	9.0	24-29	7-9	1984-93	14
LKV-01-0519	34.5	10.5	30-40	9-12	1984	1
LKV-01-0521	24.5	7.5	16-21	5-6	1984-85	2
LKV-01-0525	29.5	9.0	24-29	7-9	1984-92	21
LKV-01-0529	26	8	23-37	7-11	1982, 1984, 1985	3
LKV-01-0541	30	9	28-33	8.5-10	1990-93	6
Downgradient, shallow tailings area						
LKV-01-0507	25	8	18-23	5.5-7	1984-93	21
LKV-01-0527	26.5	8	25-37	8-11	1982, 1984, 1985	3
LKV-01-0528	28	8.5	23-29	7-9	1982, 1984, 1985	3
LKV-01-0532	28	8.5	22-38	7-11.5	1982, 1984, 1985	4
LKV-01-0540	30	9	27-32	8-10	1990-93	6
Downgradient, shallow between tailings and evaporation ponds						
LKV-01-0509	35	11	29-34	9-10	1984-85, 1993	4
LKV-01-0531	43	13	38-51	11.5-15.5	1982, 1984, 1985	3
Downgradient, deep evaporation ponds						
LKV-01-0504	80	24	73-78	22-24	1984-92	20
LKV-01-0506	85	26	77-82	23.5-25	1984-92	20
LKV-01-0512	125	38	82-87	25-26.5	1984-85	2
LKV-01-0514	80	24	70-75	21-23	1984-85, 1991	4
LKV-01-0516	81	25	72-77	22-23.5	1984-85	2
LKV-01-0520	90	27	70-85	21-26	1984	1
LKV-01-0522	59	18	52-57	16-17	1984-85	2
LKV-01-0526	80	24	63-68	19-21	1984-92	19
Downgradient, deep tailings						
LKV-01-0508	80	24	72-77	22-23.5	1984-92	19
LKV-01-0517	80	24	70-75	21-23	1984-85	3
LKV-01-0510	75	23	70-75	21-23	1984-85	2

Shallow monitor well 530 also was upgradient from the site within the area of nongeothermal ground water. However, data from this well were not used for the statistical evaluation of background ground water quality because only three rounds of older data were available and because the water quality was similar to that of the nearby background monitor well 501, for which several sampling rounds, including recent data, were available. Thus, only data from monitor well 501 were used to establish background water quality in the shallow, nongeothermal ground water.

The chemistry of the two types of shallow background ground water is summarized in Table 3.2. The nongeothermal ground water has relatively low total dissolved solids (TDS), ranging from 98 to 256 milligrams per liter (mg/L). The water is sodium-calcium-magnesium-bicarbonate water, with notably low concentrations of chloride (1 to 5 mg/L) and sulfate (<1 to 14 mg/L). In contrast, the geothermal ground water has relatively high TDS, ranging from 657 to 714 mg/L, and is sodium-sulfate-chloride water, notably depleted in calcium and magnesium. Several elements commonly associated with geothermal water are found in elevated concentrations in the geothermal water at Lakeview, including silica, arsenic, boron, and fluoride. The differences in major ion proportions in the geothermal and nongeothermal water are illustrated by trilinear plots of major elements (Figure 3.1).

Background ground water quality in the deeper zone is summarized in Table 3.3. Analysis of changes in background ground water quality with depth indicates that in a given area (geothermal or nongeothermal), water quality remains relatively constant with depth (Figure 3.1). This risk assessment focuses on background and downgradient ground water in the shallow portion of the aquifer because it has the greatest magnitude of contamination.

The hydraulic gradient in the shallow portion of the aquifer (discussed in Section 2.3) indicates the evaporation pond area is downgradient from the area of geothermal ground water and the tailings pile area is downgradient from the area of nongeothermal ground water. The geochemistry of contaminated water supports this, demonstrating that the appropriate background ground water for the pond area is geothermal (monitor well 523), while the appropriate background ground water for the tailings area is nongeothermal (monitor well 501). However, there are too few geothermal background data to make a rigorous statistical comparison to contaminated ground water. Therefore, all downgradient ground water data initially were compared to the nongeothermal ground water to determine contaminants of potential concern (discussed more fully below). The nongeothermal background ground water is a conservative benchmark because, with the exception of calcium and magnesium, the nongeothermal ground water has equal or lower concentrations of major and minor ions than geothermal ground water.

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a]	Maximum
Inorganic constituents				
Aluminum				
Nongeothermal background	0/7	<0.05	<0.05	<0.10
Geothermal background	0/2	<0.10	<0.10	<0.10
Plume (well LKV-01-0540)	4/5	<0.10	0.45	0.71
Ammonium				
Nongeothermal background	8/18	<0.1	<0.1	4.1
Geothermal background	2/3	<0.1	0.3	0.4
Plume (well LKV-01-0521)	2/2	1.6	1.7	1.8
Antimony				
Nongeothermal background	1/18	<0.001	<0.003	0.004
Geothermal background	2/2	0.006	0.008	0.010
Plume	7/23	<0.003	<0.004	0.027
Arsenic				
Nongeothermal background	3/20	<0.001	<0.01	0.003
Geothermal background	0/3	<0.01	<0.01	<0.01
Plume (well LKV-01-0503)	20/21	0.002	0.06	0.10
Barium				
Nongeothermal background	0/8	<0.01	<0.06	<0.10
Geothermal background	0/1	—	<0.10	—
Plume	8/14	0.02	0.04	<0.10
Beryllium				
Nongeothermal background	0/6	<0.005	<0.008	<0.01
Geothermal background		No data available		
Plume	0/11	<0.005	<0.01	<0.01
Boron				
Nongeothermal background	4/17	<0.01	<0.10	0.19
Geothermal background	3/3	4.6	4.8	5.2
Plume (well LKV-01-0521)	2/2	55	56	57
Bromide				
Nongeothermal background	1/3	<0.1	<0.1	0.3
Geothermal background		No data available		
Plume	5/6	<0.1	1.1	3.0

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a]	Maximum
Cadmium				
Nongeothermal background	2/19	<0.001	<0.001	0.002
Geothermal background	1/3	<0.001	<0.002	0.003
Plume	5/25	<0.001	<0.001	0.031
Calcium				
Nongeothermal background	20/20	15	21	25
Geothermal background	3/3	7	9	10
Plume (well LKV-01-0540)	6/6	525	563	642
Chloride				
Nongeothermal background	20/20	1	2	5
Geothermal background	3/3	12	94	100
Plume (well LKV-01-0513)	3/3	2200	2300	3400
Chromium				
Nongeothermal background	1/18	<0.01	<0.01	0.02
Geothermal background	0/2	<0.01	<0.01	<0.01
Plume	6/24	<0.01	<0.01	0.03
Cobalt				
Nongeothermal background	0/8	<0.01	<0.05	<0.05
Geothermal background	0/2	<0.05	<0.05	<0.05
Plume	4/13	<0.01	<0.05	0.23
Copper				
Nongeothermal background	0/8	<0.01	<0.02	<0.02
Geothermal background	0/2	<0.02	<0.02	<0.02
Plume	3/13	<0.01	<0.02	0.03
Cyanide				
Nongeothermal background	1/5	<0.01	<0.02	0.03
Geothermal background		No data available		
Plume	2/9	<0.01	<0.01	0.11
Fluoride				
Nongeothermal background	11/17	<0.1	0.1	0.2
Geothermal background	3/3	2.0	3.2	4.2
Plume (well LKV-01-0503)	19/19	0.4	3.6	7.0

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Observed concentration Median[a] (mg/L)	Observed concentration Maximum (mg/L)
Iron				
Nongeothermal background	4/20	<0.02	<0.03	0.07
Geothermal background	1/3	<0.03	<0.03	0.03
Plume (well LKV-01-0513)	3/3	15	23	27
Lead				
Nongeothermal background	1/9	<0.001	<0.01	0.01
Geothermal background	0/2	<0.01	<0.01	<0.01
Plume	0/15	<0.001	<0.01	<0.02
Magnesium				
Nongeothermal background	20/20	6.8	8.2	9.6
Geothermal background	3/3	0.1	0.2	0.6
Plume (well LKV-01-0540)	6/6	200	222	245
Manganese				
Nongeothermal background	4/19	<0.01	<0.01	0.08
Geothermal background	3/3	0.01	0.02	0.03
Plume (well LKV-01-0540)	5/5	14	34	39
Mercury				
Nongeothermal background	0/8	<0.0002	<0.0002	<0.000
Geothermal background	0/1	—	<0.0002	—
Plume	1/14	<0.0002	<0.0002	0.000
Molybdenum				
Nongeothermal background	0/9	<0.01	<0.01	<0.01
Geothermal background	1/1	—	0.03	—
Plume (well LKV-01-0521)	1/1	0.44	0.44	0.44
Nitrate				
Nongeothermal background	19/19	1.2	3.5	13
Geothermal background	1/3	<1.0	<1.0	5
Plume	7/25	<0.1	<1.0	10
Nickel				
Nongeothermal background	1/8	<0.01	<0.04	0.01
Geothermal background	0/1	—	<0.04	—
Plume (well LKV-01-0540)	5/5	0.04	0.09	0.13

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 (Continued)

		Observed concentration		
	Frequency of	Minimum	Median[a]	Maximum
Parameter[b]	detection[c]		(mg/L)	
Phosphate				
Nongeothermal background	7/8	<0.1	0.3	1.4
Geothermal background	1/3	<0.1	<0.1	0.4
Plume	9/13	<0.1	0.2	0.8
Potassium				
Nongeothermal background	20/20	0.9	1.1	2.4
Geothermal background	3/3	5.3	6.8	7.4
Plume (well LKV-01-0513)	3/3	31	33	43
Selenium				
Nongeothermal background	0/9	<0.002	<0.005	<0.03
Geothermal background	0/2	<0.005	<0.005	<0.005
Plume	0/15	<0.002	<0.02	<0.05
Silica				
Nongeothermal background	6/6	38	44	52
Geothermal background	1/1	—	66	—
Plume	11/11	36	58	73
Silver				
Nongeothermal background	0/8	<0.01	<0.01	<0.01
Geothermal background	0/1	—	<0.01	—
Plume	1/14	<0.01	<0.01	0.01
Sodium				
Nongeothermal background	20/20	11	15	23
Geothermal background	3/3	160	179	184
Plume (well LKV-01-0529)	3/3	3370	4310	5760
Strontium				
Nongeothermal background	5/8	0.05	0.05	<0.1
Geothermal background	0/3	<0.1	<0.1	<0.1
Plume (well LKV-01-0513)	3/3	4.0	5.0	6.5
Sulfate				
Nongeothermal background	19/20	0.2	3.0	14
Geothermal background	3/3	160	200	210
Plume (well LKV-01-0529)	3/3	6100	6700	7300

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a] (mg/L)	Maximum (mg/L)
Sulfide				
Nongeothermal background	0/5	<0.1	<0.1	<1.0
Geothermal background		No data available		
Plume	0/9	<0.1	<0.1	<1.0
Thallium				
Nongeothermal background	0/6	<0.01	<0.01	<0.1
Geothermal background		No data available		
Plume	0/11	<0.01	<0.01	<0.1
Tin				
Nongeothermal background	0/7	<0.005	<0.005	<0.03
Geothermal background	0/1	—	<0.005	—
Plume	5/15	<0.005	<0.01	0.062
Uranium				
Nongeothermal background	7/18	<0.001	<0.003	0.002
Geothermal background	0/1	—	<0.003	—
Plume (well LKV-01-0540)[d]	6/6	0.003	0.009	0.014
Vanadium				
Nongeothermal background	3/8	<0.01	<0.01	0.03
Geothermal background	0/1	—	<0.01	—
Plume	6/16	<0.01	<0.01	0.13
Zinc				
Nongeothermal background	9/19	<0.005	<0.005	0.25
Geothermal background	1/2	<0.005	<0.008	0.01
Plume (well LKV-01-0540)	6/6	0.078	0.30	0.40
Radionuclides			(pCi/L)	
Lead-210				
Nongeothermal background	3/4	0.1	0.5	1.0
Geothermal background	0/1	—	<1.0	—
Plume	6/7	0.0	0.3	1.6
Polonium-230				
Nongeothermal background	3/4	0.0	0.1	0.8
Geothermal background	0/1	—	<0.5	—
Plume (well LKV-01-0503)	5/5	3.8	6.8	9.1

Table 3.2 Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 (Concluded)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (pCi/L)	Median[a] (pCi/L)	Maximum (pCi/L)
Radium-226				
Nongeothermal background	16/17	0.0	0.0	1.6
Geothermal background	0/1	—	<1.0	—
Plume	22/23	0.0	0.1	0.7
Thorium-230				
Nongeothermal background	13/14	0.0	0.3	1.4
Geothermal background	0/1	—	<1.0	—
Plume	17/18	0.0	0.2	1.2

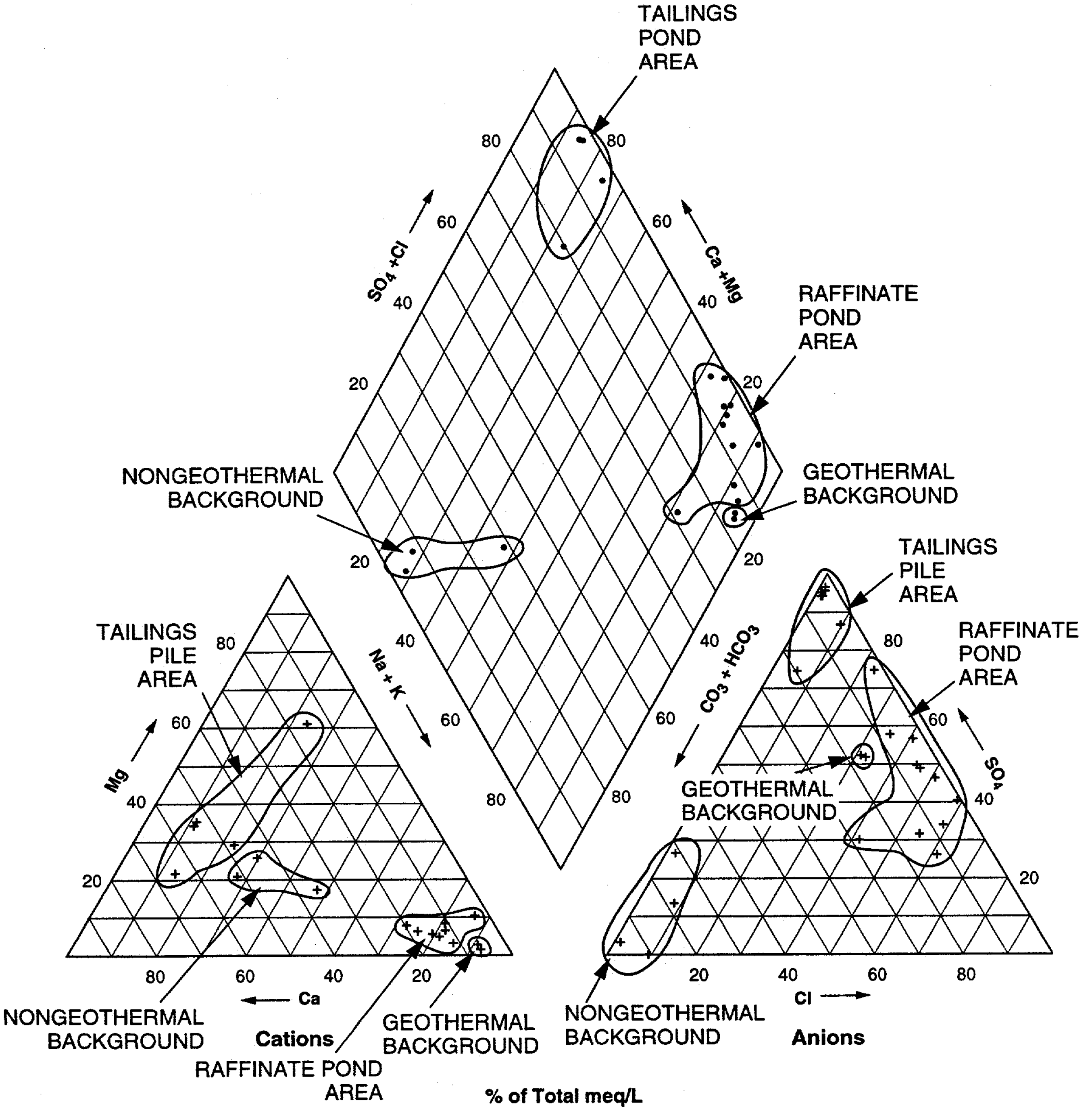
[a] The 50th percentile of the data. When only two data are available, the reported median is the arithmetic average of the two.

[b] Nongeothermal background water quality represented by monitor well 501. Geothermal background water quality represented by monitor well 523, sampled during 1984-1985 only. Plume water quality represented by monitor well 503 and monitor well 540, except as noted.

[c] Number of samples with concentrations reported above detection limit/total number of samples.

[d] Uranium concentrations in monitor well 540 have a statistically significant (p = 0.03) upward trend with time, from a low of 0.003 mg/L in 1990 to 0.014 mg/L in 1993.

Dash indicates only one measurement.
pCi/L – picocuries per liter.



NOTE: BACKGROUND INCLUDES BOTH DEEP AND SHALLOW MONITOR WELLS. TAILINGS AND RAFFINATE POND AREAS INCLUDE DATA FOR MONITOR WELLS COMPLETED ONLY IN THE SHALLOW PORTION OF THE AQUIFER.

FIGURE 3.1
TRILINEAR DIAGRAM OF MAJOR CHEMISTRIES FOR GROUND WATER LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/TRILINEAR PLOT

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992

Parameter[b]	Frequency of detection[c]	Observed concentration		
		Minimum	Median[a]	Maximum
			(mg/L)	
Inorganic constituents				
Aluminum				
Nongeothermal background	0/6	<0.05	<0.05	<0.10
Geothermal background	0/2	<0.10	<0.10	<0.10
Plume	0/14	<0.05	<0.05	<0.10
Ammonium				
Nongeothermal background	6/17	<0.1	<0.1	0.2
Geothermal background	3/3	0.2	0.4	0.4
Plume (well LKV-01-0506)	18/19	0.2	0.5	0.7
Antimony				
Nongeothermal background	0/15	<0.001	<0.003	<0.003
Geothermal background	2/2	0.008	0.008	0.008
Plume (well LKV-01-0506)	5/16	<0.003	<0.003	0.008
Arsenic				
Nongeothermal background	4/18	0.001	<0.01	<0.01
Geothermal background	3/3	0.04	0.08	0.11
Plume (well LKV-01-0506)	15/20	0.001	0.01	0.02
Barium				
Nongeothermal background	0/6	<0.01	<0.01	<0.10
Geothermal background	0/1	—	<0.10	—
Plume	4/12	<0.01	<0.01	<0.10
Beryllium				
Nongeothermal background	0/5	<0.005	<0.01	<0.01
Geothermal background		No data available		
Plume	0/12	<0.005	<0.008	<0.01
Boron				
Nongeothermal background	3/16	<0.01	<0.1	0.2
Geothermal background	3/3	4.5	4.6	4.6
Plume (well LKV-01-0506)	19/19	0.4	6.0	6.7
Bromide				
Nongeothermal background	2/3	<0.1	0.4	0.4
Geothermal background		No data available		
Plume	5/6	<0.1	0.5	1.1

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a] (mg/L)	Maximum (mg/L)
Cadmium				
Nongeothermal background	1/18	<0.0001	<0.001	0.002
Geothermal background	0/3	<0.001	<0.001	<0.001
Plume	2/39	<0.0001	<0.001	0.006
Calcium				
Nongeothermal background	18/18	14	19	23
Geothermal background	3/3	6	8	8
Plume (well LKV-01-0508)	19/19	10	30	34
Chloride				
Nongeothermal background	13/18	0.5	1.1	13
Geothermal background	3/3	89	94	97
Plume (well LKV-01-0506)	20/20	140	160	180
Chromium				
Nongeothermal background	1/17	<0.01	<0.01	0.02
Geothermal background	0/2	<0.01	<0.01	<0.01
Plume	2/38	<0.01	<0.01	0.05
Cobalt				
Nongeothermal background	0/5	<0.01	<0.03	<0.05
Geothermal background	0/2	<0.05	<0.05	<0.05
Plume	0/12	<0.01	<0.04	<0.05
Copper				
Nongeothermal background	0/5	<0.01	<0.01	<0.02
Geothermal background	0/2	<0.02	<0.02	<0.02
Plume	0/12	<0.01	<0.02	<0.02
Cyanide				
Nongeothermal background	1/3	<0.01	<0.01	0.04
Geothermal background		No data available		
Plume	0/6	<0.01	<0.01	<0.01
Fluoride				
Nongeothermal background	15/16	<0.10	0.20	0.32
Geothermal background	3/3	3.4	3.7	4.7
Plume (well LKV-01-0506)	19/19	2.0	2.5	3.3

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a] (mg/L)	Maximum (mg/L)
Iron				
Nongeothermal background	3/18	<0.02	<0.03	0.10
Geothermal background	1/3	<0.03	<0.03	0.04
Plume (well LKV-01-0506)	19/20	<0.03	0.06	0.13
Lead				
Nongeothermal background	1/6	<0.001	<0.007	<0.01
Geothermal background	0/2	<0.01	<0.01	<0.01
Plume	0/14	<0.001	<0.01	<0.01
Magnesium				
Nongeothermal background	18/18	5.3	6.0	6.6
Geothermal background	3/3	0.16	0.19	0.64
Plume (well LKV-01-0508)	19/19	5.7	8.8	10.1
Manganese				
Nongeothermal background	17/17	0.14	0.30	0.59
Geothermal background	2/3	<0.01	<0.01	0.02
Plume (well LKV-01-0508)	18/18	0.10	0.86	0.93
Mercury				
Nongeothermal background	0/6	<0.0002	<0.0002	<0.0002
Geothermal background	0/1	—	<0.0002	—
Plume	0/12	<0.0002	<0.0002	<0.0002
Molybdenum				
Nongeothermal background	0/6	<0.01	<0.01	<0.01
Geothermal background	1/1	—	0.11	—
Plume (well LKV-01-0506)	6/7	<0.01	0.03	0.04
Nitrate				
Nongeothermal background	8/18	<0.1	<1.0	5.0
Geothermal background	1/3	<1.0	<1.0	6.0
Plume	16/39	<0.1	<1.0	11.2
Nickel				
Nongeothermal background	0/5	<0.01	<0.04	<0.04
Geothermal background	0/1	—	<0.04	—
Plume	0/12	<0.01	<0.04	<0.04

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (mg/L)	Median[a]	Maximum
Phosphate				
Nongeothermal background	7/7	0.4	1.0	1.3
Geothermal background	2/3	<0.1	0.2	0.75
Plume	15/15	0.2	0.6	1.1
Potassium				
Nongeothermal background	18/18	0.75	1.0	4.9
Geothermal background	3/3	7.2	9.9	14.4
Plume (well LKV-01-0506)	20/20	4.8	5.2	10.9
Selenium				
Nongeothermal background	0/6	<0.002	<0.005	<0.03
Geothermal background	0/2	<0.005	<0.005	<0.005
Plume	0/14	<0.002	<0.005	<0.03
Silica				
Nongeothermal background	6/6	6.0	32	38
Geothermal background	1/1	74	74	74
Plume (well LKV-01-0506)	6/6	55	57	66
Silver				
Nongeothermal background	0/6	<0.01	<0.01	<0.01
Geothermal background	0/1	<0.01	<0.01	<0.01
Plume	0/12	<0.01	<0.01	<0.01
Sodium				
Nongeothermal background	18/18	16	18	39
Geothermal background	3/3	160	173	178
Plume (well LKV-01-0506)	20/20	250	285	335
Strontium				
Nongeothermal background	7/7	0.07	0.07	0.10
Geothermal background	0/3	<0.10	<0.10	<0.10
Plume (well LKV-01-0506)	8/8	0.11	0.14	0.30
Sulfate				
Nongeothermal background	17/18	0.2	2.3	31
Geothermal background	3/3	130	190	220
Plume (well LKV-01-0506)	16/16	240	280	336

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 (Continued)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum	 Median[a]	 Maximum
			(mg/L)	
Sulfide				
Nongeothermal background	1/3	<0.1	<1.0	2.8
Geothermal background		No data available		
Plume	1/6	<0.1	<0.2	<1.0
Thallium				
Nongeothermal background	0/5	<0.01	<0.01	<0.10
Geothermal background		No data available		
Plume	0/12	<0.01	<0.01	<0.10
Tin				
Nongeothermal background	0/4	<0.005	<0.005	<0.01
Geothermal background	0/1	<0.005	<0.005	<0.005
Plume	1/9	<0.005	<0.005	<0.05
Uranium				
Nongeothermal background	7/16	0.0003	<0.003	0.012
Geothermal background	0/1	<0.003	<0.003	<0.003
Plume	6/36	<0.0003	<0.003	0.008
Vanadium				
Nongeothermal background	1/6	<0.01	<0.01	0.02
Geothermal background	0/1	<0.01	<0.01	<0.01
Plume	4/12	<0.01	<0.01	0.02
Zinc				
Nongeothermal background	4/15	<0.005	<0.005	0.038
Geothermal background	1/2	<0.005	<0.11	0.21
Plume	9/32	<0.005	<0.005	0.024
Radionuclides			(pCi/L)	
Lead-210				
Nongeothermal background	4	0.0	0.3	<1.0
Geothermal background	1	—	<1.0	—
Plume	8	0.0	0.7	20.3
Polonium-210				
Nongeothermal background	4	0.0	0.7	1.6
Geothermal background	1	—	<0.5	—
Plume	9	0.1	0.5	0.8

Table 3.3 Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 (Concluded)

Parameter[b]	Frequency of detection[c]	Observed concentration Minimum (pCi/L)	 Median[a] (pCi/L)	 Maximum (pCi/L)
Radium-226				
Nongeothermal background	17	0.0	0.1	1.0
Geothermal background	1	—	<1.0	—
Plume	35	0.0	0.0	<1.0
Thorium-230				
Nongeothermal background	14	0.0	0.2	1.5
Geothermal background	1	—	<1.0	—
Plume	27	0.0	0.5	1.7

[a] The 50th percentile of the data. When only two data are available, the reported median is the arithmetic average of the two.

[b] Nongeothermal background water quality is represented by monitor well 502. Geothermal background water quality is represented by monitor well 524, sampled during 1984-1985 only. Plume water quality is represented by monitor wells 506 (ponds area) and 508 (tailings area), except as noted.

[c] Number of samples with reported concentrations above detection limit/ total number of samples.

Dash indicates only one measurement.
pCi/L – picocuries per liter.

3.2 MAGNITUDE AND EXTENT OF SITE-RELATED CONTAMINATION

3.2.1 Tailings and raffinate chemistry

Uranium ores were obtained from the White King and Lucky Lass mines about 15 mi (24 km) northwest of the mill and hauled to the mill by truck for processing. Ores from the White King Mine contained uraninite (UO_2), coffinite $U(Si,H_4)O_4$, and various secondary uranium minerals in a silicified volcanic breccia. This ore was associated with molybdenum, arsenic, mercury, lead, and antimony (BFEC, 1984). The Lucky Lass Mine primarily contained secondary uranium minerals with some uraninite or coffinite in tuff and vesicular basalt.

The ore was leached using sodium chlorate and sulfuric acid. The uranium-bearing leachate then was separated from the solids (tails), which were sent to the tailings pile. Uranium was extracted from the leachate by amine solvent extraction, and the remaining leachate (raffinate) was then sent to the evaporation ponds. Sodium carbonate was used to strip uranium from the amine solvent, and additional sulfuric acid and magnesia were used to complete the precipitation of yellow cake (Merritt, 1971). Thus, the tailings and raffinates would have contained sodium, chloride, sulfate, and possibly magnesium. Also, the dissolution of the ore would have contributed uranium, thorium, radium, molybdenum, arsenic, mercury, antimony, and several other elements commonly found in rocks, such as aluminum, iron, manganese, and silica.

Analysis of water and acid leachable compounds in the tailings identified several of these components, including uranium, strontium, chloride, aluminum, calcium, manganese, iron, and arsenic (DOE, 1985a). Chemical profiles of solids, water-soluble extracts, and acid-soluble extracts from tailings and soils beneath the tailings revealed that large amounts of material dissolved in the tailings solutions (including uranium, strontium, arsenic, iron, and sulfate) were precipitated or adsorbed from solution within a few feet of the tailings-subsoil interface. Precipitation and sorption were primarily in response to neutralization of the low pH tailings solutions (pH of 2 to 3) by carbonate minerals in the soils, which caused the pH to increase to values of about 5 within 3 ft (1 m) below the base of the tailings.

3.2.2 Contaminated ground water chemistry

Contaminated ground water on the site is recognized by high TDS and sulfate concentrations relative to background levels. Elevated concentrations of radium, thorium, and heavy metals occur at relatively low levels in the contaminated ground water because they were adsorbed or coprecipitated in the subsoils beneath the source areas of contamination (DOE, 1985a). As a result, molybdenum and nickel concentrations in contaminated ground water are only 3 to 4 times those of background, and maximum uranium concentrations are below the maximum concentration limit (MCL) of 0.044 mg/L. These conclusions are discussed below.

Analyses of the major-element water chemistry of the contaminated shallow ground water beneath the site indicate two discrete types of contaminated ground water. The first, associated with the former tailings pile, is a sodium-calcium-magnesium-sulfate water relatively depleted in chloride. The second, associated with the former evaporation ponds, is a sodium-sulfate-chloride water relatively depleted in calcium and magnesium. When the relative proportions of major anions and cations are graphed on a trilinear diagram, these two types of contaminated water show relative ion proportions similar to their respective background water types (Figure 3.1). The exception is bicarbonate, which accounts for a larger proportion of the anions in the more dilute background water. Bicarbonate is held at near-constant levels in all ground water by equilibrium with calcite. However, while the relative proportions of major ions in the contaminated ground water are similar to their respective background water types, the absolute concentrations of ions in the contaminated water are 10 to 20 times greater than those in the background water.

The following hypothesis explains the similarity of ground water contaminated from tailings-seepage water to nongeothermal water and the similarity of evaporation pond seepage water to geothermal water. Prior to construction of the tailings piles and evaporation ponds, the soil on the site contained evaporite salts derived from the background water. These salts were leached from the soil by seepage from the tailings piles and evaporation ponds, producing contaminated ground water that is a mixture of constituents from both processing fluids and natural salts. This hypotheseis explains why both types of contaminated ground water contain sulfate derived from ore processing, while only the evaporation pond water contains elevated chloride concentrations; the major source of this chloride may have been natural salts in soils formed by the evaporation of the geothermal water. This hypothesis also explains why contaminated ground water associated with the evaporation ponds contains high concentrations of sodium, boron, and fluoride (constituents that are relatively high in the geothermal ground water) and relatively low concentrations of calcium and magnesium (constituents that are depleted in the background geothermal waters, averaging 9 and 0.2 mg/L, respectively). Furthermore, it explains why the contaminated ground water from the tailings pile area contains relatively high calcium and magnesium concentrations, which are also relatively high in the nongeothermal background water upgradient of the tailings pile area (averaging 21 and 8 mg/L).

The amount of salt in a saline soil required to produce the observed concentrations of chloride, boron, and fluoride was determined using the following values and assumptions: contamination extends to a depth of 40 ft (12 m) in an aquifer with 20 percent porosity; chloride, boron, and fluoride concentrations are approximately 1200, 42, and 3.6 mg/L, respectively, in the contaminated water; and natural chloride, borate, and fluoride salts were concentrated in the upper 39 inches (1 m) of the soil prior to leaching. The computation indicates that the soil, prior to leaching, would have contained 0.2 percent sodium chloride, 0.04 percent borate salt (as borax), and

0.0009 percent calcium fluoride, by weight (DOE, 1995b). Assuming these are present in the saline soil at the site is reasonable. The computation demonstrates that a small amount of salt can significantly affect water chemistry.

This model gains support from observations of natural salt deposits at the surface south and west of the site, though the composition of these salts has not been analyzed. Additional support comes from U.S. Department of Agriculture Soil Conservation Service soils maps for the area in the site vicinity. Soils in the area of the evaporation ponds and upgradient geothermal springs are described as "sodic and saline" and unfit for lawns, topsoil, and embankments, due to excess salt. Thus, it is possible that several constituents of the contaminated ground water, including sodium, calcium, magnesium, some sulfate, and especially chloride, boron, and fluoride, were derived at least in part from the dissolution of natural salts by tailings and evaporation pond seepage. Dissolution of natural salts elsewhere in the region also may explain the presence of high dissolved solids in some private wells, as discussed in Section 3.3.

Sulfate distribution was used to determine the extent of contamination at the site, because this mobile major ion is common to both types of contaminated ground water. Data from March 1985 were used to delineate the extent of contamination, since this database provides the greatest areal coverage. The 1985 ground water data appear to be pertinent to current conditions, because trend analyses with time indicate virtually no change in the chemistry of contaminated ground water in 12 wells sampled from 1984 through 1993.

The sulfate distribution indicates two contaminant plumes in the shallow ground water zone above 40 ft (12 m): one associated with the tailings pile, and a second associated with the evaporation ponds (Figure 3.2). This finding is consistent with the chemical identification of two types of contaminated ground water. Sulfate contamination does not extend upgradient from either of these sources but does extend at least 2300 ft (700 m) downgradient from the evaporation ponds, and at least 690 ft (210 m) downgradient from the tailings ponds in directions that are consistent with the hydraulic gradient. Thus, contamination in the shallow portion of the aquifer (0 to 40 ft [0 to 10 m]) extends beyond existing monitor wells and off the site (Figure 3.2). However, sulfate and other major contaminant concentrations systematically decrease downgradient, and approach but do not reach background concentrations.

Table 3.3 summarizes filtered ground water quality in the deeper portion of the aquifer. The vertical extent of contamination appears to be limited to depths of less than 80 ft (20 m). One well (522) screened between 52 and 57 ft (16 and 17 m) has elevated sulfate concentrations (880 mg/L), indicating contamination. The chemistries of the deeper wells, screened from about 63 to 87 ft (19 to 27 m), are similar to the geothermal background wells downgradient of the evaporation ponds, or to the nongeothermal wells downgradient of the tailings pile. Slightly elevated sulfate concentrations (220 to 300 mg/L) relative to

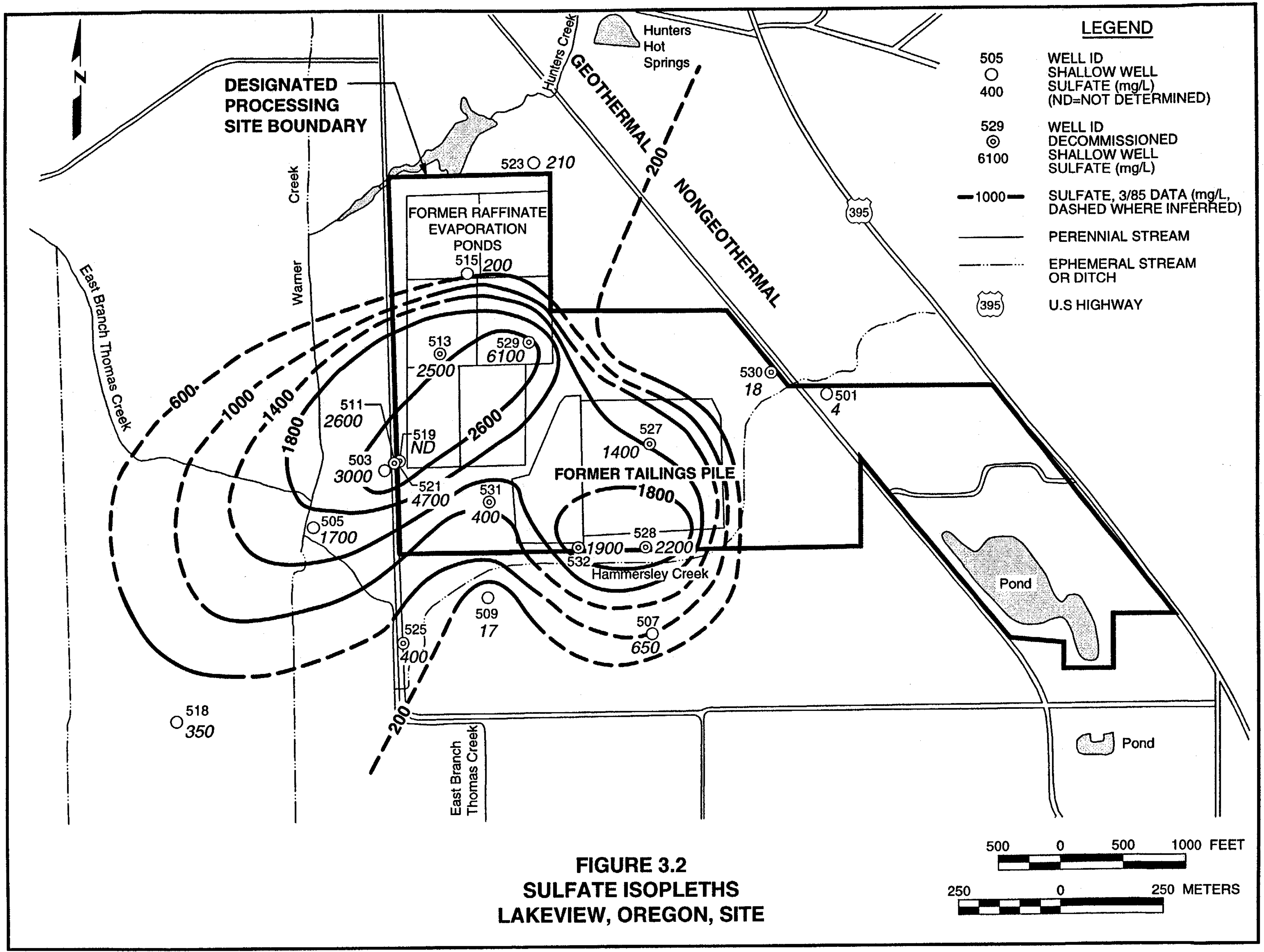


FIGURE 3.2
SULFATE ISOPLETHS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE-ISOPLETHS

geothermal background (200 mg/L) may indicate that a small amount of contamination has reached a depth of 80 ft (24 m) in monitor wells 504, 506, and 520. The downward movement of contaminants is consistent with the downward vertical hydraulic gradient on the site. However, the low concentration of possible contaminants at 80 ft (24 m) suggests a very slow downward movement of contaminants (if it is occurring).

3.2.3 Statistical comparison of background and plume water quality

After determining the extent of contamination (using sulfate as an indicator), water quality data collected between 1984 and 1993 were used to statistically identify the full suite of contaminants elevated above background levels in contaminated shallow portions of the aquifer (DOE, 1995b). For the statistical evaluation, the 1984-1993 filtered water chemistry data from nongeothermal background well 501 were compared to concurrent data from well 503 in the vicinity of the evaporation ponds, and then to well 540 downgradient of the former tailings pile. Well 501 was chosen as background because it has the greatest number of sampling rounds and because it is within the shallow portion of the aquifer, the portion most affected by contamination. It was necessary to use filtered water samples because very few unfiltered data were available. Wells 503 and 540 were selected for analysis because they consistently have the highest concentrations of key contaminants among the current wells on or downgradient from the site.

The statistical comparison method used was the Wilcoxon signed-rank test, a nonparametric blocked comparison method that incorporates sampling dates as random blocks. Individual tests were conducted at the 0.05 level of significance. A contaminant was identified as above background levels if a significant test result was obtained for either the evaporation pond monitor well (503) or the tailings area monitor well (540). Rigorous statistical comparison of plume water quality to the geothermal background was not done because of inadequate data on background geothermal water quality (monitor well 523 had only three rounds of data, collected between 1984 and 1985, and many of the chemicals were monitored only once). Contaminants statistically above nongeothermal background levels include aluminum, antimony, ammonium, arsenic, boron, calcium, chloride, fluoride, iron, magnesium, manganese, molybdenum, nickel, polonium-210, potassium, silica, sodium, strontium, sulfate, uranium, and zinc.

After determining which constituents in the plume were above nongeothermal background levels, each identified constituent was compared to geothermal background levels to see if the average value for the contaminated ground water from either monitor well 503 or 540 was within the range of geothermal background water quality. This was done to identify constituents that might have a geothermal source rather than a contaminant source. Three constituents in the plume identified as above nongeothermal background levels were found to be within the range of geothermal background water quality: antimony, fluoride, and silica. On average, all remaining constituents identified as above nongeothermal background levels were present at levels above the range of geothermal background levels and therefore were identified as contaminants.

Antimony, fluoride, and silica were screened further to determine if they could be reasonably associated with a geothermal ground water source rather than the result of contamination. Antimony concentrations exceeded nongeothermal background levels only in the evaporation pond area monitor well (503). Antimony in the tailings area monitor well (540) was not statistically elevated above nongeothermal background levels. Because the pond area is downgradient from the geothermal water source, the antimony in the plume in this area likely is from a natural source, rather than from a contaminant. Therefore, antimony is not considered a contaminant.

Fluoride in monitor wells 503 and 540 (up to 3.7 mg/L) was close to the range of geothermal background levels (up to 3.1 mg/L). Although fluoride in the tailings area ground water is statistically higher than its appropriate background levels, there is evidence that nongeothermal ground water near the site is naturally high in fluoride. The highest fluoride concentration observed at the site (9 mg/L) was found in monitor well 509, a downgradient well at the very edge of the area of contamination (Figure 3.2). Water from this well has very low sulfate (30 mg/L or less), chloride (13 mg/L or less), and TDS (344 mg/L or less), indicating that it has received little or no contamination and that the ground water is from a predominantly nongeothermal source. Thus, the fluoride in this well and the plumes is probably natural, derived from minerals in the aquifer matrix. For this reason, fluoride was not considered a contaminant.

Finally, silica concentrations in both the tailings and pond areas were found to be at levels near those in the geothermal ground water. However, in the tailings area, the appropriate background water is the nongeothermal ground water, which has relatively low silica concentrations. Therefore, silica is considered a contaminant in the tailings area, even though silica concentrations in contaminated ground water across the site are on average less than those in the natural geothermal water.

Table 3.2 summarizes the water quality measurements from filtered water samples collected between 1984 and 1993 from the contaminated shallow aquifer and compares ground water affected by the former milling-related activities to geothermal and nongeothermal background ground water quality. Most contaminated ground water concentrations presented were observed in well 503 (evaporation pond area) and/or well 540 (tailings area). In several cases, however, well 503 water quality was supplanted by higher 1982-1985 levels recorded for one or more of the abandoned wells in the evaporation pond area prior to surface remediation activities. Because there is no evidence that ground water contamination has decreased significantly during the last 10 years, it is possible that these early high levels may still be present under the site but not accessible to current wells. Contaminants for which early data are reported in Table 3.2 include boron, chloride, iron, molybdenum, potassium, sodium, strontium, and sulfate.

For a limited number of sampling rounds, both filtered and unfiltered water samples were collected and analyzed for antimony, arsenic, barium, beryllium, cadmium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, potassium, selenium, silica, silver, sodium, thallium, tin, uranium, vanadium, and zinc. These data pairs of filtered and unfiltered results were analyzed to assess the impact that filtering may have had on the constituent concentrations in monitor well samples. Arsenic, iron, manganese, and zinc concentrations were typically higher in unfiltered than in filtered samples. Conversely, calcium, magnesium, potassium, and sodium concentrations do not appear to be systematically affected by filtration. The remaining constituents are present at the Lakeview site at low concentrations (near or below the detection level); thus, the impacts of filtration are not discernible from the available data. The higher unfiltered arsenic levels were considered in evaluating potential risks from exposure to this constituent.

3.3 PRIVATE WELL WATER QUALITY

Water quality data were collected in 1982, 1984, and 1985 from 5 private wells north and southeast of the site and in 1993 from 14 private wells north, southeast, south, southwest, and west of the site, including wells on Missouri and Roberta Avenues (Table 3.4). Locations of the wells are shown in Figure 2.5. Unfiltered samples were collected and analyzed for major elements and trace elements indicative of the uranium milling process. Analytical data from private wells north of the site (upgradient), southeast of the site (crossgradient), and west of the site (crossgradient) were collected to determine variations in natural water chemistry in the area. Analytical data from wells south and southwest of the site were collected to determine if those wells were receiving contaminated ground water from the site. The depths of these wells range from 50 to 280 ft (15 to 85 m), but most are in the 50- to 100-ft (15- to 30-m) range (Table 3.4). Table 3.5 presents results of the latest water quality sampling tests for each well.

Ground water geochemistry and temperatures in private wells north of the site (upgradient wells 535, 538, and 544) indicate they are geothermal waters having relatively high concentrations of sodium, sulfate, chloride, and TDS. These geothermal waters also contain elevated concentrations of zinc and arsenic. Hunters Hot Springs are the source of these waters; the water quality is similar to that in DOE monitor wells used to establish upgradient geothermal background water quality (discussed in Section 3.1).

Private wells west, southwest, and southeast of the site (crossgradient to downgradient wells 545, 533, 534, 536, 537, 542, and 543) have relatively low TDS (180 to 320 mg/L) and sulfate (3 to 18 mg/L) concentrations. Both site-related contamination and natural geothermal water are associated with much higher sulfate and TDS; thus, the low-sulfate private wells are believed to contain uncontaminated, nongeothermal. Several of these nongeothermal water wells have relatively high concentrations of manganese (up to 2.6 mg/L), iron (up to 1.4 mg/L), and zinc (up to 0.5 mg/L). The water quality in these private

Table 3.4 Summary of DOE domestic wells sampled, 1984 through 1994, Lakeview, Oregon, site vicinity

Location ID	Depth (ft)	Depth (m)	Sampling dates	Number of sampling rounds
LKV-01-0533	Unknown		1984, 1985, 1993	3
LKV-01-0534	92	28	1984, 1993	2
LKV-01-0535	150	46	1984, 1985, 1993	3
LKV-01-0536	Unknown		1982, 1984, 1985, 1993	4
LKV-01-0537	50	15	1984, 1993, 1994	3
LKV-01-0538	100	30	1982, 1984, 1993	3
LKV-01-0542	90	27	1993, 1994	2
LKV-01-0543	280	85	1993	1
LKV-01-0544	Unknown		1993	1
LKV-01-0545	71	22	1993	1
LKV-01-0546	50	15	1993, 1994	2
LKV-01-0547	68	21	1993, 1994	2
LKV-01-0548	85	26	1993, 1994	2
LKV-01-0549	80	24	1993, 1994	2
LKV-01-0550	Unknown		1994	1

Table 3.5 Summary of unfiltered water quality for private wells, Lakeview, Oregon, site vicinity

Location ID	Date	Alk	As	Ca	Cl	Fe	Mg	Mn	Mo	Ni	pH	K	SiO_2	Se	Na	SO_4	Temp	TDS	U	V	Zn
North of site (geothermal)																					
535	8/93	138	0.009	76	40	<0.03	12	<0.01	<0.01	<0.04	7.7	3.2	44	<0.005	27	74	20	380	0.001	<0.01	0.057
538	8/93	NA	0.013	5	96	<0.03	0.2	0.01	0.04	<0.04	NA	6.7	111	NA	196	222	NA	1010	<0.001	<0.01	0.066
544	8/93	NA	0.157	13	99	<0.03	<0.1	<0.01	0.04	<0.04	NA	8.6	145	NA	203	258	95	820	<0.001	<0.01	0.007
West of site																					
545	8/93	68	<0.005	<0.5	5	0.21	0.1	0.03	<0.01	<0.04	8.1	1.2	68	<0.005	39	9	16	180	0.003	<0.01	0.022
Southeast of site																					
536	8/93	138	<0.005	29	2	0.7	9.5	0.18	<0.01	<0.04	7.6	1.2	52	<0.005	17	3	13	200	<0.001	0.02	0.011
537	8/93	210	0.005	41	4	0.53	14	2.4	<0.01	<0.04	7.4	0.9	50	<0.005	30	9	13	270	<0.001	<0.01	0.014
542	8/93	138	<0.005	29	1	0.11	8.9	0.46	<0.01	<0.04	7.3	1.2	57	<0.005	18	2	15	190	<0.001	<0.01	0.074
543	8/93	232	<0.005	51	5	1.43	16	2.57	<0.01	<0.04	7.3	0.9	50	NA	33	18	11	320	0.001	<0.01	0.009
Southwest of site																					
533	8/93	158	<0.005	32	2	<0.03	12	0.04	<0.01	<0.04	7.0	4.3	70	<0.005	22	13	14	250	0.002	<0.01	0.137
534	8/93	137	0.006	29	1	<0.03	10	<0.01	<0.01	<0.04	6.9	4.3	70	<0.005	18	11	19	230	<0.001	<0.01	0.277
South of site																					
546	5/94	223	0.011	39	12	1.01	11.4	1.34	<0.01	<0.04	7.6	0.6	53	NA	36	7	11	308	<0.001	<0.01	0.11
547	5/94	152	0.007	61	538	1.44	65	8.6	<0.01	<0.04	7.2	5.2	61	NA	88	152	11	1440	<0.001	<0.01	<0.05
548	5/94	153	0.007	48	1484	4.04	153	28.7	<0.01	<0.04	7.2	9.2	55	NA	462	1030	11	4380	<0.001	<0.01	0.08
549	5/94	99	0.015	4	59	0.80	21.7	0.31	0.01	<0.04	7.8	3.4	59	NA	111	94	11	452	<0.001	<0.01	0.05
550	5/94	117	0.006	91	829	1.73	61	9.0	<0.01	<0.04	7.3	8.2	58	NA	301	483	12	2160	<0.001	<0.01	<0.05

Note: All data are in milligrams per liter except for pH (standard units) and temperature (degrees Celsius [°C]).

Alk	Alkalinity	Se	Selenium
As	Arsenic	SiO_2	Silica
Ca	Calcium	Na	Sodium
Cl	Chloride	SO_4	Sulfate
Fe	Iron	Temp	Temperature (°C)
Mg	Magnesium	TDS	Total dissolved solids
Mn	Manganese	U	Uranium
Mo	Molybdenum	V	Vanadium
Ni	Nickel	Zn	Zinc
K	Potassium	NA	Not analyzed

wells is similar to that in DOE monitor wells used to establish nongeothermal background water quality.

Four of five private wells due south of the site (including locations 546, 547, 548, 549, and 550) have elevated sulfate concentrations, which could indicate contamination from the site has reached these wells. However, the areal pattern of sulfate distribution in these wells is inconsistent with site-related contamination. The wells with the highest sulfate (locations 548 and 550, having 1030 and 483 mg/L sulfate, respectively) are farthest from the site, whereas the well closest to the site (location 546) has the lowest sulfate concentration (7 mg/L) (Figure 3.3). Sulfate contamination from the site in excess of 300 mg/L appears to be limited to an area within 2400 ft (730 m) of the site, whereas the private wells with high sulfate concentrations are about 5000 ft (1500 m) south of the site, placing them outside the apparent area of site-related contamination. Therefore, the private wells with high sulfate concentrations apparently are not affected by site-related contaminants. However, few ground water quality data are available for the relatively large area between the site and the private wells. Therefore, the conceptual model of the extent of contaminated ground water downgradient of the site is not completely substantiated. While it seems unlikely, based on limited existing information, that private wells south of the site are affected by site-related contamination, this conclusion is not certain. More monitoring wells are needed to confirm the site-conceptual model.

The private wells south of the site with high sulfate levels also have elevated concentrations of chloride, sodium, calcium, manganese, and iron. The concentrations of these components are greater than those observed in geothermal waters, so these waters probably are not geothermal. Field observations and discussions with area residents affected by the high sulfate ground water revealed that the originally low-lying land surface was filled during the last 12 years with about 1.5 ft (0.46 m) of wood chips, boiler ash, and dredgings from log ponds. Also, a drainage has been partially filled and dammed, providing a source of local recharge. The one resident who has lived in the area for more than 12 years stated that water quality began to deteriorate after the fill was added. A plausible explanation (other than site-related contamination) for the high salinity is that both rain water captured by the very permeable wood chip fill and surface water from the dammed drainage have leached natural salts in the soils and that this leachate is the source of saline ground water in this area. The mechanism of leaching salts from soils in disturbed areas also appears to have occurred on the former processing site. Like the conceptual model of the distribution of site-related ground water contamination near the site, the conceptual model of saline salts in soil as a source of saline ground water has not been substantiated by direct tests for soluble salts in soil near the private wells south of the site.

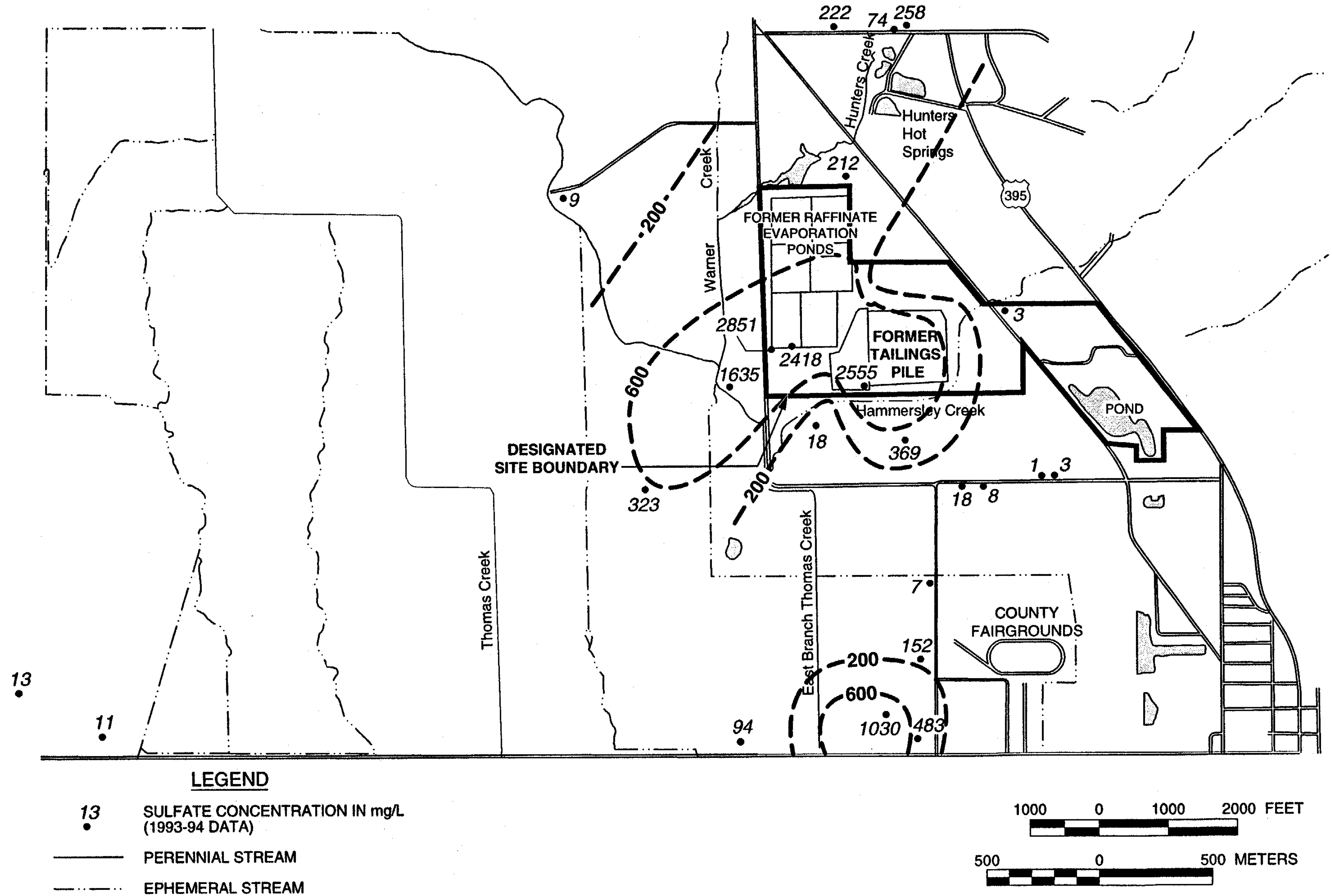


FIGURE 3.3
DISTRIBUTION OF SULFATE IN SHALLOW DOE MONITOR WELLS AND PRIVATE WELLS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE1-SULFATE

3.4 CONTAMINANTS OF POTENTIAL CONCERN

The data presented in Sections 3.1 and 3.2 were used to compile a list of contaminants of potential concern for this assessment of human health risks at the Lakeview site. A constituent was placed on the list of contaminants of potential concern (Table 3.6) if reported concentrations in the shallow on-site or downgradient monitor wells were on average higher than background levels. The contaminants in column 1 of Table 3.6 (contaminants that exceed background levels) were screened for their impact on human health using the criteria discussed below. Column 4 of Table 3.6 is a final list of contaminants of potential concern for human health (DOE, 1995b).

Two chemicals (potassium and zinc) detected above background levels were screened out because they are essential nutrients and the levels at which they occur are within nutritional ranges.

A second screening of the remaining contaminants was made of the constituents that exhibited very low toxicity and relatively high dietary intake in comparison to the detected concentrations on the site. For human health, these screening criteria eliminated ammonium, aluminum, calcium, magnesium, silica, and strontium. The contaminants of potential concern that remain after screening are arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium (Table 3.6). The human health portion of this risk assessment focuses on these contaminants.

Because ecological impacts differ from effects on human health, the list of contaminants that will be considered for the ecological assessment is presented in Section 7.0.

3.5 CONTAMINANT FATE AND TRANSPORT

Contaminant mobility, uptake, and toxicity depend on the species of ions in the aqueous environment. The types of ion species and complexes depend on the availability of various anions and cations to form complex ions and pH and Eh conditions. Species of the contaminants of potential concern were computed using the geochemical speciation code PHREEQE (Parkhurst et al., 1980). Table 3.7 summarizes the predominant species and their molar percentages (DOE, 1995b).

In the following discussion, contaminant dilution and dispersion in conjunction with ground water flow are cited as a major control on the fate of contaminants in ground water at the site. The current rate of ground water transport of contaminants appears to be slow, as indicated by little or no change in contaminant concentrations downgradient of the site over the last 12 years. During that period, for example, sulfate concentrations remained relatively constant in downgradient well 528 near the edge of the contaminated ground water. Nonetheless, the presence of site-related contaminants in this downgradient well indicates ground water flow has occurred at the site. The

Table 3.6 Contaminants of potential concern for human health evaluation, Lakeview, Oregon, site

Contaminants that exceed background levels	Contaminants in nutritional range	Contaminants of low toxicity and/or high dietary range	Contaminants of potential concern
Ammonium		Ammonium	
Aluminum		Aluminum	
Arsenic			Arsenic
Boron			Boron
Calcium		Calcium	
Chloride			Chloride
Iron			Iron
Magnesium		Magnesium	
Manganese			Manganese
Molybdenum			Molybdenum
Nickel			Nickel
Polonium-210			Polonium-210
Potassium	Potassium		
Silica		Silica[a]	
Sodium			Sodium
Strontium		Strontium	
Sulfate			Sulfate
Uranium			Uranium
Zinc	Zinc		

[a] Although silica causes toxicity when inhaled, ingesting dissolved silica has not been demonstrated to produce adverse health effects in humans or animals.

Table 3.7 Predominant ionic species of contaminants of potential concern, Lakeview, Oregon, site

Constituent	Common name	Dominant species	Molar percent	Valence
Arsenic	Arsenate	$HAsO_4^{2-}$	70	As(V)
	Arsenate	$H_2AsO_4^-$	30	As(V)
Boron	Borate	$H_3BO_{3\,aq}$	100	B(III)
Chloride	Chloride	Cl^-	100	Cl(I)
Iron	Ferric hydroxide	$FeOH^{2+}$	96	Fe(III)
	Ferric hydroxide	$Fe(OH)_{3\,aq}$	2	Fe(III)
Manganese	Manganese	Mn^{2+}	68	Mn(II)
	Manganese sulfate	$MnSO_{4\,aq}$	26	Mn(II)
Molybdenum	Molybdate	MoO_4^{2-}	100	Mo(VI)
Nickel	Nickel carbonate	$NiCO_{3\,aq}$	64	Ni(II)
	Nickel	Ni^{2+}	19	Ni(II)
Polonium-210		Not determined		
Sodium	Sodium	Na^+	97	Na(I)
Sulfate	Sulfate	SO_4^{2-}	79	S(VI)
	Sodium sulfate	$NaSO_4^-$	10	S(VI)
	Calcium sulfate	$CaSO_{4\,aq}$	8	S(VI)
Uranium	Uranyl hypophosphate	$UO_2(HPO_4)_2^{2-}$	99	U(VI)

apparent contradiction between the extent of contamination and the lack of time-dependent trends in contaminant concentrations in individual wells represents a data gap in both the understanding of the hydrologic regime at the site and the prediction of contaminant fate and transport. Without contaminant movement, dispersion and dilution occur very slowly.

Arsenic

Arsenic concentrations in the contaminated ground water are about 60 times the detection limit of 0.001 mg/L, but they are actually lower than the arsenic concentrations measured in the deep background geothermal ground water and background geothermal surface water (with up to 0.24 mg/L arsenic reported in the samples collected). However, the shallow background geothermal ground water does not contain arsenic at concentrations above the detection limit, and these were used to determine background levels since data indicate that the most contaminated ground water is in the shallow portion of the aquifer.

Arsenic occurs as arsenate species $HAsO_4^{2-}$ in the ground water. Reaction of ferric hydroxides with arsenate may be an effective control on the concentration of arsenic (Hem, 1970), especially because the contaminated ground water is oversaturated with respect to iron hydroxides. Thus, concentrations may decrease by coprecipitation of arsenate with iron hydroxides over time. A similar mechanism may explain the lack of arsenic in the shallow geothermal ground water when this water is compared to both geothermal surface water and deep geothermal ground water.

Boron

Boron concentrations in the contaminated ground water are about 9 times those of background. Boron was not used to process the ores, and usually is not associated with uranium ores. It is commonly associated with geothermal water, including that upgradient of the site. Along with chloride, its presence at concentrations above background levels is due to its having been leached from natural soils as a result of uranium processing activities.

Boron occurs as borate in the contaminated and natural waters near the site. Borate forms very soluble salts, and precipitation of those salts will not occur at depth. Thus, dilution probably will play the greatest role in decreasing borate concentrations.

Chloride

Chloride concentrations in the contaminated ground water ranges up to about 12 times concentrations reported in background samples. Much of the chloride in the contaminated ground water may have been leached from natural soils beneath the tailings and ponds by site-related seepage. Chloride does not form insoluble salts in low-salinity ground waters (less than 177,000 mg/L chloride concentrations), and concentrations are too great to be significantly affected by adsorption. Therefore, decreases in chloride concentrations within the contaminated ground water will be due to dilution.

Iron

Iron concentrations in the contaminated ground water are several hundred times those reported in background ground water samples. Iron is soluble under acid or reducing conditions but forms insoluble hydrated oxides under oxidizing conditions at a pH greater than 6. All water at the site is oversaturated with respect to these iron oxides, and it is expected that with time, as oxygen diffuses into the ground water, iron concentrations will decrease to background levels (0.12 mg/L) due to precipitation and dilution.

Manganese

Manganese concentrations in the contaminated water are about 1700 times background levels. Like iron, manganese is most soluble under acid or reducing conditions, though manganese can persist at greater concentrations than iron in oxidizing, slightly alkaline ground water. Several ground water samples taken at that site are saturated with respect to manganese carbonate (rhodochrosite), and the natural occurrence of this phase may explain the prevalence of manganese in ground water throughout the region. However, none of the contaminated ground water samples are oversaturated with respect to rhodochrosite; thus, the major decrease in concentration will be due to dilution, perhaps accompanied by coprecipitation with iron oxides and sorption.

Molybdenum

Molybdenum concentrations in contaminated ground water are four times those of the geothermal background ground water. Molybdenum occurs in both geothermal water and the contaminated ground water as molybdate (MoO_4^{2-}), a negatively charged ligand. As with most negatively charged ligands, molybdenum sorption is most effective under acidic conditions (pH in the 3 to 4 range). Thus, molybdenum sorption in the contaminated ground water will be less important than dilution as a mechanism for decreasing concentrations.

Nickel

Nickel concentrations are 3 times the detection limit for the analyses (0.01 and 0.04 mg/L) in ground water associated with the tailings pile. Nickel also was reported in unfiltered samples collected from monitor well 540. It is strongly sorbed by iron and manganese oxides (Hem, 1970), and sorption onto these hydroxides will be the most effective mechanism for removing the nickel from solution with time.

Polonium-210

Polonium-210 is a relatively short-lived daughter product in the uranium-238 decay chain, with a half-life of 138 days. Thus, its presence in the contaminated ground water indicates the additional presence of parent isotopes such as lead-210, radon-222, radium-226, thorium-230, uranium-234, or uranium-238. Lead-210, radium-226, thorium-230, and uranium were not detected in the contaminated ground water in amounts sufficient to support the observed activity of polonium-210. Therefore, the source of the polonium-210 is either dissolved radon-222, which was not monitored in ground water, or one of the other parent isotopes occurring in solid form within the aquifer. Radon-222 is commonly associated with geothermal water in amounts more than adequate to explain the observed activity of polonium-210; this could be the source of the polonium-210 at the Lakeview site (Davis and DeWiest, 1966). However, because polonium-210 was not observed in the geothermal background water, the geothermal radon-222 probably is not the source of the polonium-210. Thus, the most likely source for the polonium-210 is a solid parent isotope (lead-210, radium-226, thorium-230, uranium-234, or uranium-238). Of these, uranium is the most mobile in the ground water environment, and thus the parent most likely to have migrated into the aquifer in aqueous solution before being adsorbed onto soils.

Because the source of polonium-210 appears to be a solid that will not migrate with ground water flow and because polonium-210 has a short half-life, the distribution of dissolved polonium-210 will be restricted to the area near its source. The size of this area is dictated by the rate of ground water flow and the rate of polonium-210 decay. Assuming the rate of shallow horizontal ground water movement is approximately 50 ft per year (8×10^{-5} cm per second) (see Section 2.3) and the polonium-210 will decay almost entirely within 10 half-lives (3.8 years), the polonium will be restricted to a distance of about 190 ft (58 m) from its source. The daughter product of its decay is the stable isotope lead-208. This isotope probably will be sorbed by iron oxides, especially since the amount of stable lead produced by the polonium-210 decay is less than 5.8×10^{-12} mg/L per year.

Sodium

Sodium concentrations are about 10 times background levels in the most contaminated ground water. Sodium forms very soluble bicarbonate, chloride, and sulfate salts; much of the sodium in the contaminated ground water may have been leached from natural soils beneath the tailings and ponds by site-related seepage. Precipitation of those salts will not occur at depth, and dilution probably will play the greatest role in decreasing sodium concentration.

Sulfate

Sulfate concentrations in contaminated ground water vary across the site from 215 to 3380 mg/L. None of the water is oversaturated with respect to gypsum, the phase that most likely will control the concentration of sulfate in this ground water. Therefore, decreased sulfate downgradient of the tailings pile and evaporation ponds (matched by decreased chloride) is due to dilution of the tailings solutions by natural waters. Further sulfate decreases will occur by dilution and dispersion during migration of the contaminated ground water.

Uranium

Uranium in the contaminated water near the tailings is at relatively low levels (0.003 to 0.014 mg/L). The uranium occurs in the hexavalent state (+6 state) in the site ground water as a uranyl phosphate complex ($UO_2(HPO_4)_2^{2-}$). Low uranium levels in the contaminated ground water attest to the ability of the soil at the site to strongly adsorb uranium species, and further attenuation by adsorption probably will occur.

3.6 SURFACE WATER AND SEDIMENT

Surface water quality has been monitored in the water bodies in the vicinity of the Lakeview site. Surface water bodies at and adjacent to the site include ponds, ditches, and three creeks (Figure 2.2). Hunters Hot Springs are important to surface water hydrology and geochemistry; they feed geothermal water into a creek (Hunters Hot Springs Creek) and associated ponds north of the site.

Surface water has been sampled at six locations (Figure 3.4). Unfiltered surface water samples were collected in October 1993 from two locations along Hunters Hot Springs Creek (608 and 609), one location just downstream of the confluence of Warner Creek and Thomas Creek (605), and one location in Thomas Creek just upstream of the confluence with Hammersley Creek (606). Historic sampling location 607, in Thomas Creek downstream of its confluence with Hammersley Creek, was dry during the October 1993 sampling event. One to three sampling rounds were conducted at these five locations (605 through 609) from 1982 through 1985. In addition, one sample (613) was taken from a pond northwest of a cemetery and southeast of the former mill buildings. This pond may have received runoff from the area where the former mill buildings were located.

The samples collected in October 1993 were analyzed for a select list of analytes (arsenic, calcium, iron, magnesium, manganese, molybdenum, nickel, sulfate, tin, uranium, vanadium, and zinc). These constituents were selected prior to the development of this risk assessment. Sediment samples were collected in October 1993 from all six locations.

Table 3.8 presents surface water data for contaminants analyzed in the surface water samples that were above background levels in ground water.

Surface water data from these locations show that most of the constituent levels found at the adjacent locations were no greater than their respective

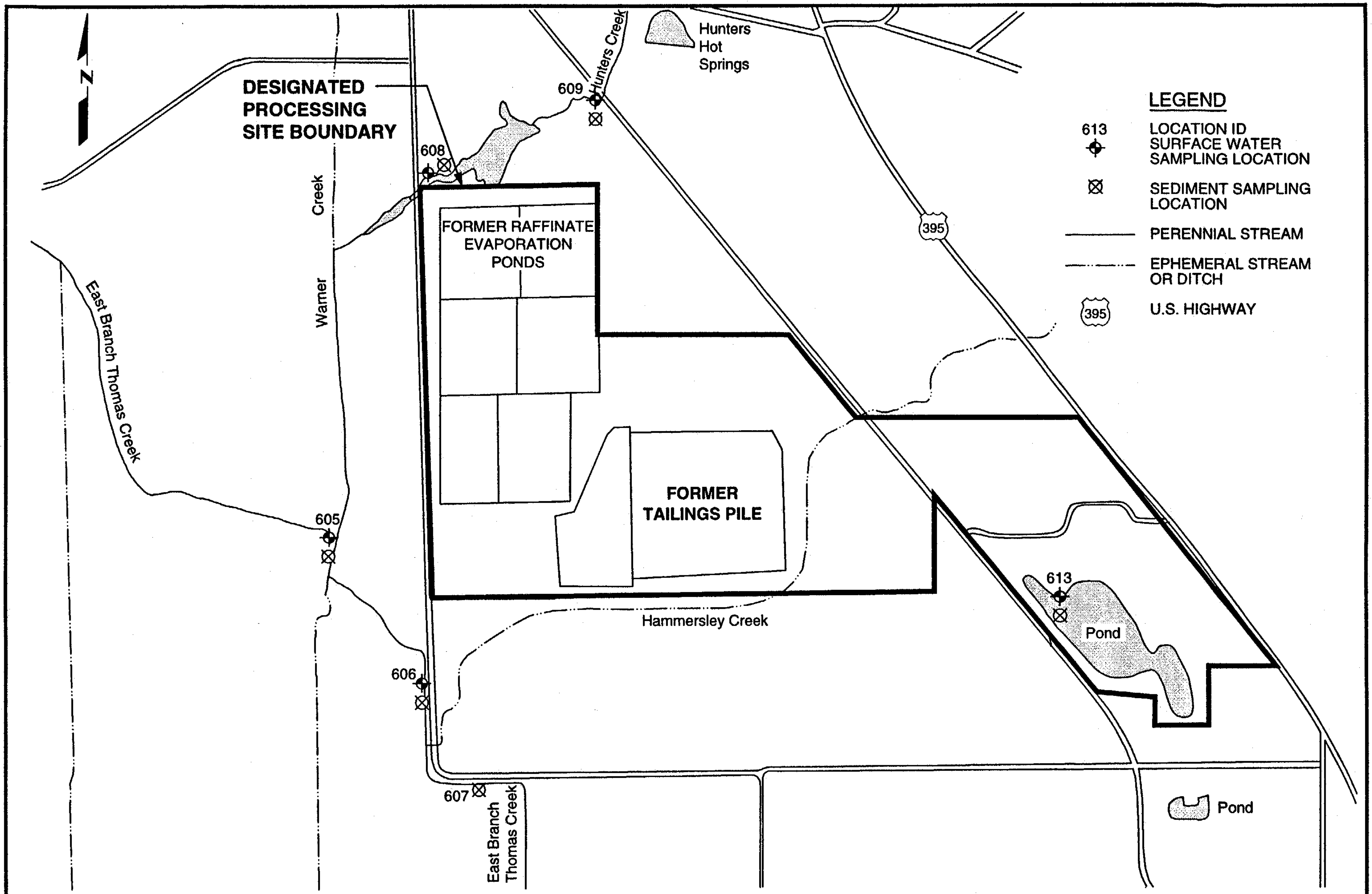


FIGURE 3.4
SURFACE WATER AND SEDIMENT SAMPLING LOCATIONS
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/BASE-SURF&SED

Table 3.8 Constituents monitored in surface water collected from water bodies in the Lakeview, Oregon, site vicinity

	Upstream of site location ID	Adjacent to site location ID			Pond location ID
Constituent	**LKV-01-0609**	**LKV-01-0608**	**LKV-01-0605**	**LKV-01-0606**	**LKV-01-0613**
Arsenic	0.24	0.16	0.16	0.16	0.005
Iron	0.63	0.39	0.66	0.41	0.16
Manganese	0.11	0.04	0.06	0.05	0.03
Molybdenum	0.03	0.04	0.04	0.04	<0.01
Nickel	<0.04	<0.04	<0.04	<0.04	<0.04
Sulfate	300	260	260	260	4.5
Uranium	0.001	<0.001	<0.001	<0.001	<0.001

All results reported in milligrams per liter for unfiltered samples collected 26 October 1993.

background (upstream) concentrations. From the list of ground water constituents that exceed background ground water quality (Table 3.6), only iron and molybdenum were detected at concentrations slightly above background levels at the adjacent locations. However, the differences in concentrations are negligible.

Sediments are an integral part of the aquatic environment, providing habitat, feeding, and rearing areas for many aquatic organisms (Hull and Suter, 1994). The sediment layer includes solid particles (inorganic and organic) settling on the bottom of a body of water such as a river or a pond, and interstitial water (pore water), which fills the spaces between the sediment particles (Power and Chapman, 1992). In the environment, these particles are derived both from material originally suspended in the water and minerals that precipitate from the water. Pore water usually accounts for more than 50 percent (by volume) of the sediment layer. Results presented in this risk assessment represent the chemistry of both the solid and pore-water components of the sediment layer.

Surficial sediment samples (0 to 4 inches [0 to 10 cm]) were collected on 26 October 1993, from surface water sampling locations (Figure 3.3). Sediment samples had not been collected near the site before this sampling event. The sediment samples were analyzed for arsenic, iron, manganese, molybdenum, nickel, sulfate, tin, uranium, vanadium, and zinc. Table 3.9 presents data for the site-related contaminants analyzed in the sediment samples (tin, vanadium, and zinc are not considered site-related contaminants). The differences between the adjacent and downstream concentrations of arsenic, nickel, and sulfate were negligible. The differences between the adjacent and downstream concentrations of iron, manganese, and uranium were greater. However, the observed levels of these constituents (uranium up to 1.4 mg per gram, iron up to 9010 mg per gram and manganese up to 270 mg per gram are less than those typical of soils in the western United States where uranium averages 2.3 mg per gram, iron averages 18,000 mg per gram, and manganese averages 330 mg per gram (Shacklette and Boerngen, 1984). The observed increases downstream and adjacent to the site may reflect natural variation associated with different drainage areas such as Warner Creek, rather than site-related contamination.

Precipitation and snowmelt can carry both dissolved and suspended constituents along surface drainages. Metal constituents transported as dissolved species can become diluted after discharging to water bodies. Alternatively, dissolved species can precipitate, becoming adsorbed to sediments or absorbed into biota with varying biochemical and geochemical conditions. Variations in geochemical conditions or biological action can release constituents adsorbed onto sediments into surface water. Thus, deposited sediments can act as a source of site-related surface water contamination.

Table 3.9 Occurrence of constituents monitored in sediment collected from surface water bodies in the Lakeview, Oregon, site vicinity

	Upstream of site location ID	Adjacent to site location ID			Downstream of site location ID	Pond location ID
Constituent	**LKV-01-0609**	**LKV-01-0608**	**LKV-01-0605**	**LKV-01-0606**	**LKV-01-0607**	**LKV-01-0613**
Arsenic	17	12	11	25	12	4.8
Iron	1190	965	5350	8810	9010	8930
Manganese	25	16	115	192	195	270
Molybdenum	2.0	1.0	1.0	<1.0	1.0	<1.0
Nickel	<4.0	<4.0	4.0	<4.0	4.0	5.0
Sulfate	154	78	158	74	168	13
Uranium	0.1	0.3	1.1	0.8	1.4	1.3

All results reported in milligrams per kilogram dry weight. The samples were collected 26 October 1993.

Based on the limited data for surface water and sediment, it appears that surface water is not impacted and the elevated sediment concentrations (compared to background) may reflect natural variations in different drainage areas. However, these observations are not conclusive due to the small amount of data for these two media.

4.0 EXPOSURE ASSESSMENT

This section quantifies the potential exposures that could be incurred by current or future residents who use site-related contaminated ground water in the vicinity of the former Lakeview site. The methodology is consistent with the EPA guidance on exposure assessments, which recommends analysis based on a reasonable maximum exposure under both current and future land-use conditions (EPA, 1989b). Reasonable maximum exposure is defined as the greatest exposure that can reasonably be expected, based on the current evaluation of on- and off-site conditions. Ground water-use scenarios are based on current domestic ground water use in the region. This exposure assessment uses ground water quality data that were collected before complete site characterization.

4.1 POTENTIALLY EXPOSED POPULATIONS AND EXPOSURE PATHWAYS

An exposure pathway describes the course a contaminant takes from the source to the exposed individual. To complete a pathway there must be a source of contamination, a release mechanism for the contaminant (e.g., contaminants leaching from tailings into the ground water), a point of contact with a population or individual, and a route of exposure (e.g., water ingestion). Evaluating potential exposure pathways involves determining not only the most likely pathways but also the major contributing pathways to risk as a function of exposure dose. Both direct and indirect exposures to contaminated ground water are assessed below. Given the chemical properties of the inorganic contaminants of potential concern at UMTRA Project sites, direct exposure through drinking water ingestion is expected to result in the most significant contaminant exposure dose. The main contributions to risk are identified by a screening process that compares exposures through the drinking water pathway to exposures through other pathways. For the screening exposure pathways the adult population was evaluated. Currently, no uses of site-related contaminated ground water by human receptors have been identified at the Lakeview site. However, shallow ground water is used in the Lakeview area for domestic purposes such as drinking, bathing, and cooking. Additionally, ground water in this area is used for agricultural purposes such as crop irrigation and livestock watering. Using contaminated ground water to water crops and livestock could lead to indirect exposure to humans who eat contaminated produce and meat and drink contaminated milk. Access to the ground water at and downgradient of the former processing site is not restricted. Therefore, it is possible people could come into contact with site-related contaminated ground water and be exposed to contaminants. Figure 4.1 presents the conceptual site model of potential exposure pathways at the Lakeview site.

Several of the contaminants of potential concern for the Lakeview site (see Table 3.6) were detected in private wells south of the site (wells 546, 547, 548, 549, and 550). These constituents included arsenic, chloride, iron, manganese, molybdenum, sodium, and sulfate. As discussed in Section 3.3, it is unlikely that the presence of these constituents in the private wells is associated with the Lakeview site even though the source of these constituents

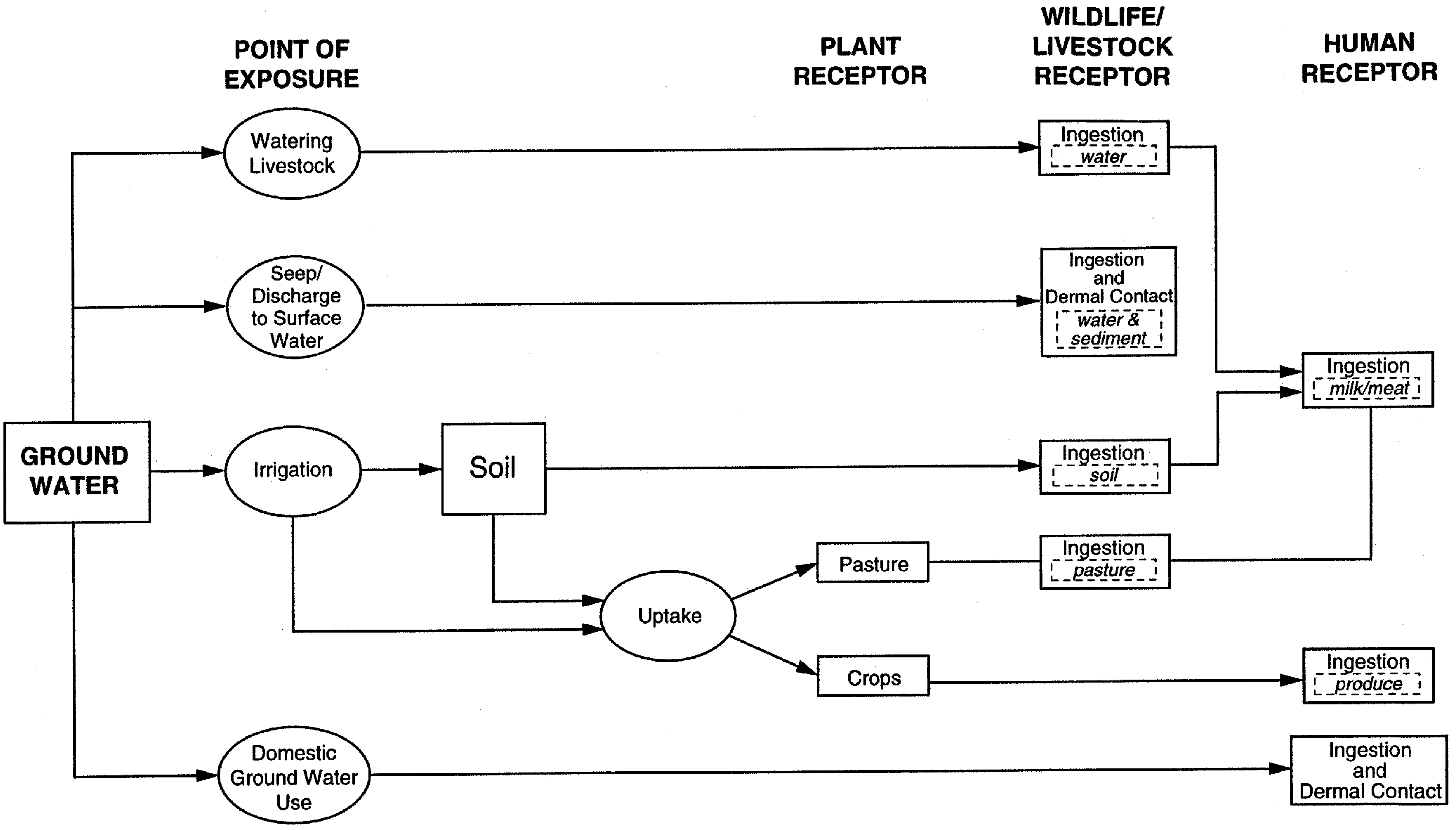


FIGURE 4.1
CONCEPTUAL MODEL
LAKEVIEW, OREGON, SITE

MAC: SITE/LKV/BRA/CONCEPTMOD

has not been determined. Regardless of the source, a discussion concerning the potential of these constituents to cause adverse human health effects is included in this document (Section 4.2). The inhabitants of this group of residences do not use the ground water for drinking water. However, the ground water is used for other domestic purposes such as cooking, bathing, watering livestock, and irrigating gardens.

4.1.1 Drinking water ingestion

Drinking water ingestion is generally the most significant exposure pathway for ground water contaminated with metals and other nonvolatile compounds. In this evaluation, drinking water consumption includes water consumed by drinking and water used directly in food preparation (e.g., reconstituted juices, soup, rice, and beans). To show 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 95 percent upper confidence limit (UCL) on the mean contaminant concentration from the most contaminated plume wells.

4.1.2 Dermal absorption

Dermal absorption is the process by which chemicals coming into contact with the skin are absorbed into the blood vessels near the skin surface. 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 significant compared to the drinking water pathway for the contaminants of potential concern. The contaminants were assumed to absorb across the skin at the same rate as water. This assumption probably will overestimate any potential contribution from dermal absorption. Additionally, the concentration in water is assumed to be the 95 percent UCL, which also could overestimate exposure.

The results of the screening are given in Table 4.1. Based on these results, dermal absorption was eliminated from more detailed evaluation because it contributed less than 1 percent of the total intake from drinking water for all constituents. However, any potential toxic effects from dermal contact exposure are discussed in Section 6.0.

4.1.3 Ingestion of ground-water-irrigated produce

The exposure pathway of ingested ground water-irrigated produce also was evaluated for its relative significance compared to the drinking water ingestion route. Table 4.2 shows the results of the screening calculation. The assumptions for this evaluation probably will overestimate the exposure potential from this route also, because it is assumed that a domestic garden would be the source of all garden produce in the diet. This screening shows that for the contaminants of potential concern at this site, ingesting garden

Table 4.1 Exposure dose calculations for ground water ingestion and dermal contact, Lakeview, Oregon, site

Contaminant of potential concern	Cw (mg/L)	Ground water exposure doses (mg/kg-day)		Ratio of dermal: ingestion[a]
		Ingestion	Dermal contact	
Carcinogenic effects				
Arsenic	0.11	1E-03	3E-06	0.002
Polonium-210	8.5[b]	2E+05[b]	3E+02[b]	0.002
Uranium	14[b]	4E+02[b]	6E+02[b]	0.002
Noncarcinogenic effects				
Arsenic	0.11	3E-03	6E-06	0.002
Boron	58	2E+00	3E-03	0.002
Chloride	3000	8E+01	2E-01	0.002
Iron	27	7E-01	1E-03	0.002
Manganese	40	1E+00	2E-03	0.002
Molybdenum	0.46	1E-02	2E-05	0.002
Nickel	0.13	4E-03	7E-06	0.002
Sodium	4000	1E+02	2E-01	0.002
Sulfate	6900	2E+02	4E-01	0.002
Uranium[c]	0.02	3E-05	1E-06	0.002

Equation definitions for exposure dose calculations

Ingestion of ground water

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

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

Dermal contact with ground water

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

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

Where:
Cw = Contaminant concentration in ground water (95 percent UCL).
IRw = Ingestion rate for water (L per day) (2 L per day for an adult).
EF = Exposure frequency (350 days per 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; AT = 365 days x 70 years for carcinogens).
SA = Skin surface area (19,400 square centimeters).
Pc = Dermal permeability constant (0.001 cm per hour).
Cf = Conversion factor (0.001 L/cm^3).
ET = Exposure time (0.2 hour per day).

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[a] Calculated by dividing the dermal contact exposure dose by the ground water ingestion exposure dose.
[b] Measured in picocuries per lifetime.
[c] Uranium-234 and uranium-238 combined.

UCL - 95th upper confidence limit on the mean contaminant concentration from the most contaminated plume wells.

mg/kg-day - milligrams per kilograms per day.

Table 4.2 Exposure dose calculations for ground water-irrigated garden produce ingestion, Lakeview, Oregon, site

Contaminant 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]
Carcinogenic effects						
Arsenic	0.11	19.4	0.04	0.006	2E-06	0.002
Polonium-210	8.5[c]	14.9	0.0025	0.0004	1E+01[c]	0.00006
Uranium	14[c]	50	0.0085	0.004	3E+02[c]	0.0009
Noncarcinogenic effects						
Arsenic	0.11	19.4	0.04	0.006	4E-06	0.001
Boron	58	1.3	4	2	2E-02	0.009
Iron	27	15	0.004	0.001	8E-05	0.0001
Manganese	40	25.3	0.25	0.05	1E-02	0.01
Molybdenum	0.46	120	0.25	0.06	7E-04	0.07
Nickel	0.13	58.6	0.06	0.06	3E-05	0.008
Sodium	4000	0.2	0.075	0.055	4E-03	0.00004
Sulfate	6900	0	0.5	0.5	d	d
Uranium	0.02	50	0.0085	0.004	5E-07	0.001

Equation definitions for exposure dose calculations

Ingestion of garden produce irrigated with ground water

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

Radionuclides: Lifetime intake (pCi/lifetime) $= Cw \times Kd \times Bv \text{ or } Br^{e} \times Df \times IRp \times FI \times EF \times ED$

Where:

Cw = Contaminant concentration in ground water (95th UCL).
Kd = Soil-water partition coefficient (L/kg); from Pacific Northwest Laboratory (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).
Br = Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless).
DF = Dry weight fraction of plant (0.066 unitless).
IRp = Ingestion rate for garden produce (0.05 kg per day for vegetative parts; 0.03 kg per day for reproductive parts).
FI = Fraction of garden produce ingested from contaminated source (1.0 unitless).
EF = Exposure frequency (350 days per 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; AT = 365 days x 70 years for carcinogens).

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

[b] Calculated by dividing the garden produce ingestion exposure dose by the ground water ingestion exposure dose.

[c] Measured in picocuries per lifetime.

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

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

UCL - 95th upper confidence limit on the mean contaminant concentration from the most contaminated plume wells.
mg/kg-day - milligrams per kilogram per day.

produce irrigated with contaminated ground water would represent potential exposures of 7 percent or less than would be associated with ingesting contaminated drinking water. Any potential toxic effects from exposure to contaminants through ingesting garden produce are discussed in Section 6.0.

4.1.4 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 except molybdenum, which contributes 20 percent and sodium which contributes 30 percent (Table 4.3). Ingesting meat from these animals would also contribute 7 percent or less of the exposure anticipated from drinking water for all constituents except molybdenum, which contributes 20 percent (Table 4.4). This pathway may be important, but current data are insufficient to produce meaningful quantitative estimates. The contribution from these sources is not included in the exposure simulations, but the impact of these additional contributions is discussed in Sections 6.1 and 6.2.

4.1.5 Ingestion of fish

This pathway was not evaluated because the surface water bodies in the vicinity of the Lakeview site provide very limited fish habitat and would not support a fish population that humans would consume (i.e., game fish). No fish were observed in the streams and drainages near the site.

4.2 PRIVATE WELLS

Arsenic, chloride, iron, manganese, molybdenum, sodium, and sulfate are evaluated for the potential to cause adverse health effects to people using the ground water from the private wells south of the site. Of wells 546, 547, 548, 549, and 550, the highest concentrations of the constituents detected in the latest round of sampling (Table 3.5) are used to evaluate potential adverse health effects.

Exposure is evaluated by calculating exposure doses from dermal absorption, ingesting garden produce watered with ground water from the private wells, and eating meat and drinking milk from livestock watered with ground water from the private wells. Since this water is used for cooking, ground water ingestion is also evaluated. It is assumed that the water would be used for making soups and for other cooking purposes where the water would be directly ingested. The same equations and input parameters as presented on Tables 4.1 through 4.4 were used except for the ingestion rate for the drinking water route. For this route of exposure, 0.5 L per day was used. The exposure dose for each exposure route and each contaminant is calculated and summed (Table 4.5). Also on this table are the carcinogenic risk estimates for arsenic. The total exposure doses are evaluated for the potential to cause adverse health effects. The exception is manganese, because this constituent is more bioavailable in water than in food; only the exposure doses for dermal contact and water used

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

Contaminant of potential concern	Cw (mg/L)	Kd[a] (L/kg)	Bv	Fm	Milk ingestion exposure doses (mg/kg-day)	Ratio of milk ingestion: water ingestion[b]
Carcinogenic effects						
Arsenic	0.11	19	0.04	0.00006	9E-07	0.0009
Polonium-210	8.5[c]	15	0.0025	0.00035	6E+02[c]	0.003
Uranium	14[c]	50	0.0085	0.0006	2E+03[c]	0.007
Noncarcinogenic effects						
Arsenic	0.11	19	0.04	0.00006	2E-06	0.0007
Boron	58	1.3	4	0.0015	6E-02	0.03
Iron	27	15	0.004	0.00025	2E-03	0.003
Manganese	40	25	0.25	0.00035	1E-02	0.01
Molybdenum	0.46	120	0.25	0.0015	2E-03	0.2
Nickel	0.13	59	0.06	0.001	8E-05	0.02
Sodium	4000	0.2	0.075	0.035	3E+01	0.3
Sulfate	6900	0	0.5	0.005	d	d
Uranium	0.02	50	0.0085	0.0006	4E-06	0.008

Equation definitions for exposure dose calculations

Ingestion of milk from ground water fed livestock

Chemicals: Chronic daily intake (mg/kg-day)

$$= \frac{Cm \times IRm \times FI \times EF \times ED}{BW \times AT}$$

Radionuclides: Lifetime intake (pCi/lifetime)

$$= Cm \times IRm \times FI \times EF \times ED$$

Where:

Cm = Contaminant 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 (kilograms per day).
Qp = The quantity of pasture eaten by cattle per day (19 kg per day).
Qs = The quantity of soil eaten by cattle per day (0.38 kg per day).
Qw = The quantity of water consumed by cattle per day (56 L per day).
Cp = Contaminant concentration in pasture (mg/kg); Cp = Kd x Cw x Bv.
Cs = Contaminant concentration in soil (mg/kg); Cs = Kd x Cw.
Cw = Contaminant concentration in ground water (95th UCL).
Kd = Soil-water partition coefficient (L/kg); from PNL (1989), except uranium. The Kd for uranium is a site-specific value.
Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
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; AT = 365 days x 70 years for carcinogens).
IRm = Ingestion rate of milk (0.3 kg per day).
FI = Fraction of milk ingested from contaminated source (1.0 unitless).

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[a] Kd value was not available for chloride; therefore, it has been omitted.

[b] Calculated by dividing the milk ingestion exposure dose by the ground water ingestion exposure dose.

[c] Measured in picocuries per lifetime.

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

UCL - 95th upper confidence limit on the mean contaminant concentration from the most contaminated plume wells.

mg/kg-day - milligrams per kilogram per day.

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

Contaminant of potential concern	Cw (mg/L)	Kd (L/kg)	Bv	Ff	Meat ingestion exposure dose	Ratio of meat ingestion: water ingestion[a]
Carcinogenic effects						
Arsenic	0.11	19.4	0.04	0.002	8E-06	0.008
Polonium-210	8.5[b]	14.9	0.0025	0.00025	1E+02[b]	0.005
Uranium	0.14[b]	50	0.0085	0.0002	2E+00[b]	0.0005
Noncarcinogenic effects						
Arsenic	0.11	19.4	0.04	0.002	2E-05	0.006
Boron	58	1.3	4	0.00055	5E-03	0.003
Iron	27	15	0.004	0.02	3E-02	0.05
Manganese	40	25.3	0.25	0.0004	3E-03	0.003
Molybdenum	0.46	120	0.25	0.006	2E-03	0.2
Nickel	0.13	58.6	0.06	0.006	1E-04	0.03
Sodium	4000	0.2	0.075	0.015	3E+00	0.04
Sulfate	6900	0	0.5	0.033	c	c
Uranium	0.14	50	0.0085	0.0002	2E-06	0.0006

Equation definitions for exposure dose calculations

Ingestion of meat from ground water fed livestock

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

Radionuclides: Lifetime intake (pCi/lifetime) $= Cb \times IRb \times FI \times EF \times ED$

Where:

Cb = Contaminant 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 (kilograms per day).
Qp = The quantity of pasture eaten by cattle per day (19 kg per day).
Qs = The quantity of soil eaten by cattle per day (0.38 kg per day).
Qw = The quantity of water consumed by cattle per day (56 L per day).
Cp = Contaminant concentration in pasture (mg/kg). $Cp = Kd \times Cw \times Bv$.
Cs = Contaminant concentration in soil (mg/kg). $Cs = Kd \times Cw$.
Cw = Contaminant concentration in ground water (95th UCL).
Kd = Soil-water partition coefficient (L/kg); from PNL (1989), except uranium. The Kd for uranium is a site-specific value.
Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
EF = Exposure frequency (350 days per 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; AT = 365 days x 70 years for carcinogens).
IRb = Ingestion rate of meat (0.075 kg per day).
FI = Fraction of meat ingested from contaminated source (1.0 unitless).

[a] Calculated by dividing the meat ingestion exposure dose by the ground water ingestion exposure dose.
[b] Measured in picocuries per lifetime.
[c] Value cannot be calculated because Kd is equal to zero.

UCL - 95th upper confidence limit on the mean contaminant concentration from the most contaminated plume wells.
mg/kg-day - milligrams per kilogram per day.

Table 4.5 Exposure dose results for the private wells south of the Lakeview, Oregon, site

	Produce ingestion	Dermal contact	Meat ingestion	Milk ingestion	Ground water ingestion through cooking	Total
Carcinogenic effects						
Arsenic	3E-07	3E-07	1E-06	9E-08	4E-05	5E-05
Risk[a]	4E-07	6E-07	2E-06	2E-07	8E-05	8E-05
Noncarcinogenic effects						
Arsenic	6E-07	8E-07	2E-06	3E-07	1E-04	1E-04
Chloride	2E+00	8E-02	5E+01	4E+01	1E+01	10E+01
Iron	1E-05	2E-04	5E-03	3E-04	3E-02	3E-02
Manganese	9E-03	2E-03	2E-03	8E-03	2E-01	2E-01
Molybdenum	2E-05	5E-07	4E-05	4E-05	7E-05	2E-04
Sodium	5E-04	3E-02	2E+00	4E+00	3E+00	8E+00
Sulfate	—	6E-02	2E+00	1E+00	7E+00	1E+01

[a] Calculated by multiplying the exposure dose by the slope factor (SF). The SF is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen. For example, the estimated risk for the ingestion of produce is an increased probability of 4 chances in 10,000,000 of developing cancer. Total risk is calculated by combining all risk values over all exposure pathways.

Values are reported in milligrams per kilogram per day.
Dash indicates constituent was not evaluated for this parameter.

for cooking are combined. The exposure dose evaluation is presented in Section 6.3.

It should be noted that the evaluation of the private wells is conservative because of the conservative assumptions. It was assumed that the people would get all of their garden produce, meat, and milk from their own garden or livestock and ingest the constituents for 350 days per year for 30 years, concurrently. Using these assumptions likely overestimates the true risks associated with using the ground water from these wells.

4.3 EXPOSURE CONCENTRATIONS

The exposure concentration of a contaminant in ground water is defined as the concentration an individual is assumed to take in over a specific period. In this evaluation, the contaminant concentrations are assumed to be in a steady state, although actual contaminant concentrations (and therefore exposures) are expected to decrease with time because the tailings have been removed. However, no seasonal variations have been noted in the last 12 years of data. Nonetheless, these estimates are reasonable for chronic exposure soon after surface remediation. Chronic exposure for noncarcinogens is considered exposure for any period longer than 7 years.

For noncarcinogens, exposure concentrations are expressed as probability distributions. Distributions were derived from actual recorded water quality measurements of the well or wells currently in existence that consistently recorded the highest concentrations of each contaminant. For example, the manganese distribution reflects concentrations in well 540, whereas the sulfate distribution reflects well 503 levels. The theoretical distribution was selected to have, to the extent possible, the same average concentration, standard deviation (spread), and pattern of occurrences (shape) as occurred in actual water quality data. The normal distribution provided an adequate fit of the data for all constituents.

Recent analytical data were used except in cases where 1982-1985 contaminated concentrations from wells in the vicinity of the former evaporation ponds clearly and grossly exceeded more recent levels. In those cases, early data were used instead of more recent data, and the selected distribution was shifted upward to reflect the higher values. This approach was used because ground water contaminant levels appear to be decreasing only slowly over time, and worst-case exposure may occur in areas of the former ponds where recent analytical data are not available. This shift was used for boron, chloride, molybdenum, sodium, and sulfate. The shift was multiplicative, so that the resulting mean of the distribution was centered on the higher values, but the spread about the mean was also proportionately larger. This procedure was necessary because early data were inadequate to provide information on distributional shape.

For example, the theoretical probability distribution developed to model concentrations of sulfate in well 503 was a normal distribution with a mean of 2985 mg/L and standard deviation of 189 mg/L. To reflect the 1984-1985 concentrations observed in monitor well 529 (measurements of 6100 and 7300 mg/L), the new distribution was shifted up by a factor of 2.24 (6700/2985 = 2.24) to become a normal distribution with a mean of 6700 mg/L and standard deviation of 425 mg/L.

The data used for distribution development were from filtered samples, except for arsenic. Several rounds of unfiltered arsenic measurements indicated the unfiltered levels were about 35 percent higher than levels measured in filtered samples. This information was incorporated into the arsenic distribution. For the other contaminants of potential concern, unfiltered results were too limited in quantity to incorporate into the risk assessment.

Uranium concentrations from well 540 in the former tailings area appear to have increased with time between 1990 and 1994. This apparent trend may or may not be real, and if real, may or may not continue to rise in the future. However, the trend was significant at the 0.05 level of significance. Therefore, linear regression methods were used to estimate the current uranium concentration as 0.016 mg/L, and the theoretical distribution is centered on this estimated value.

The distributions are all truncated: values below 0 mg/L or above the 99th percentile are not used. The lower limit of 0 mg/L reflects the impossibility of negative exposures. The truncation of the upper end of the distribution places a reasonable limit on possible exposures. For every contaminant, this highest allowable concentration was higher than the maximum observed concentration in the historical water quality data. The original distributions derived from existing wells were used to calculate the 95 percent UCL for the true mean concentration of that constituent. The software package @RISK was used to generate the probability curves for the noncarcinogenic contaminants of potential concern (Palisade Corporation, 1992). The results are shown in Figures 4.2 through 4.11.

Radionuclide concentrations were represented by the same wells that represented the inorganic constituents. Potential carcinogenicity of ground water contaminants is discussed in Section 5.0.

4.4 ESTIMATION OF INTAKE

Future residents are expected to vary with respect to water consumption habits, 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 this group, 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 nationwide public health and census documents. All distributions were truncated at the

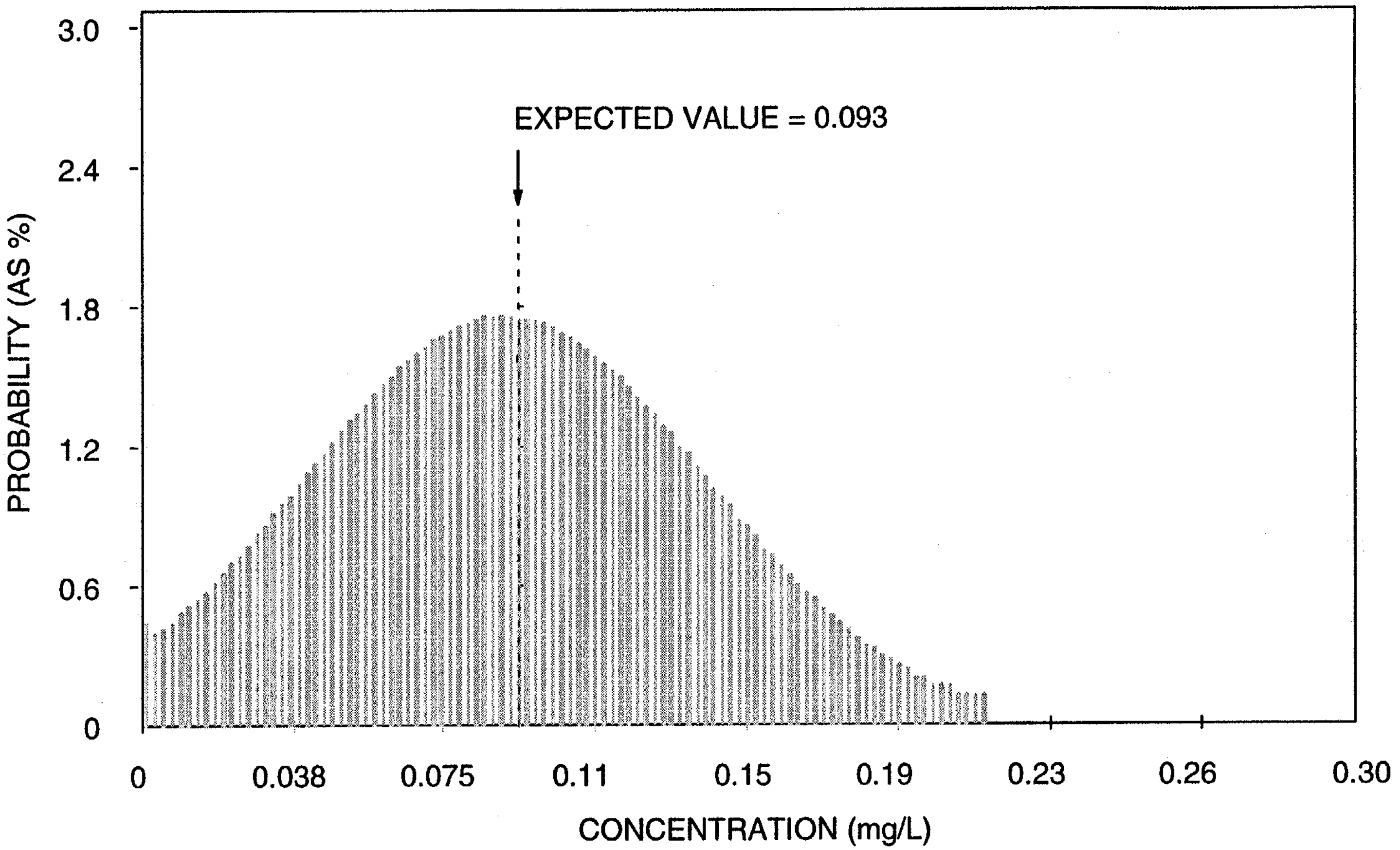

FIGURE 4.2
SIMULATED DISTRIBUTION OF ARSENIC CONCENTRATIONS
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

ARSCON.DRW

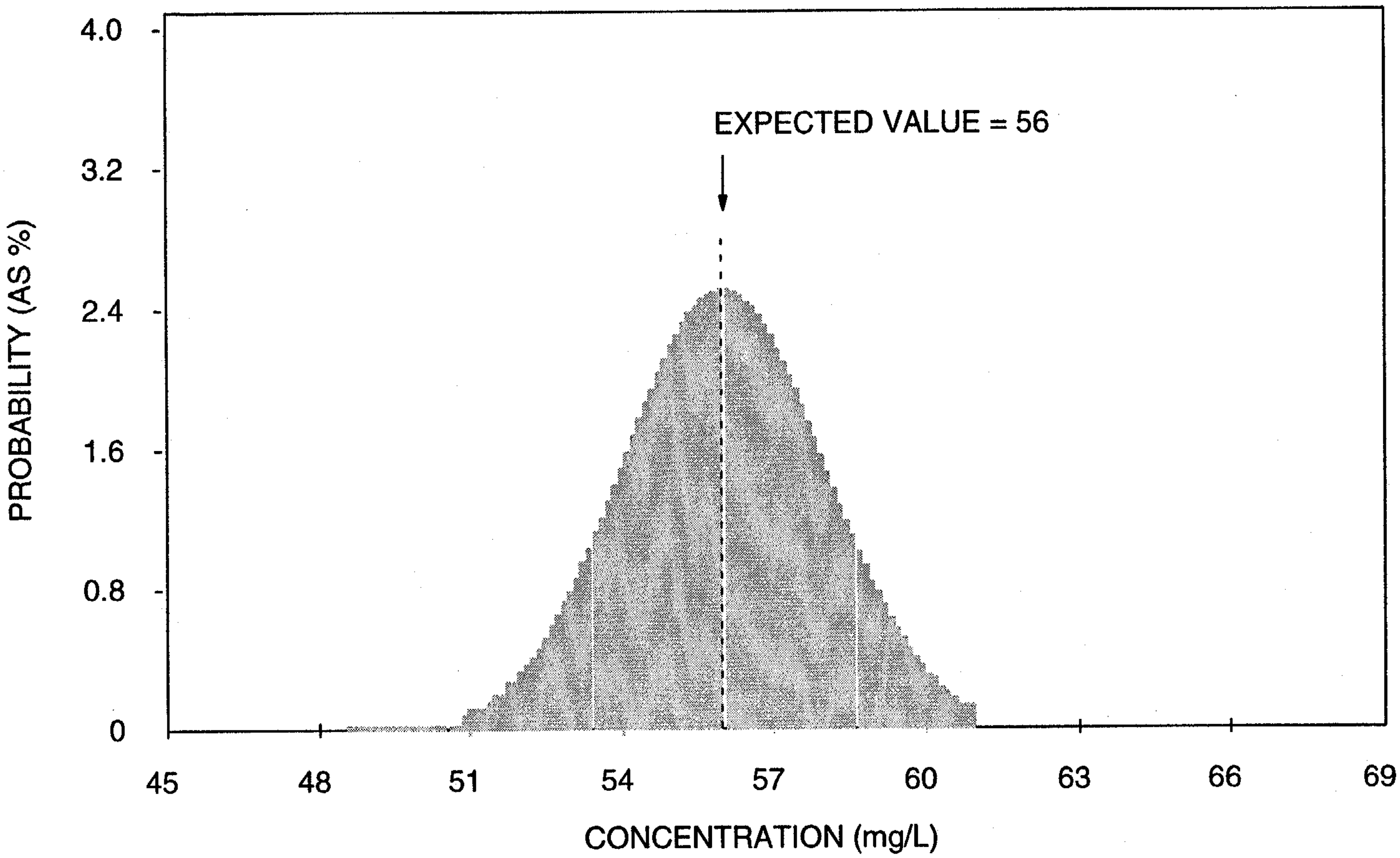

FIGURE 4.3
SIMULATED DISTRIBUTION OF BORON CONCENTRATIONS
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

BORCON.DRW

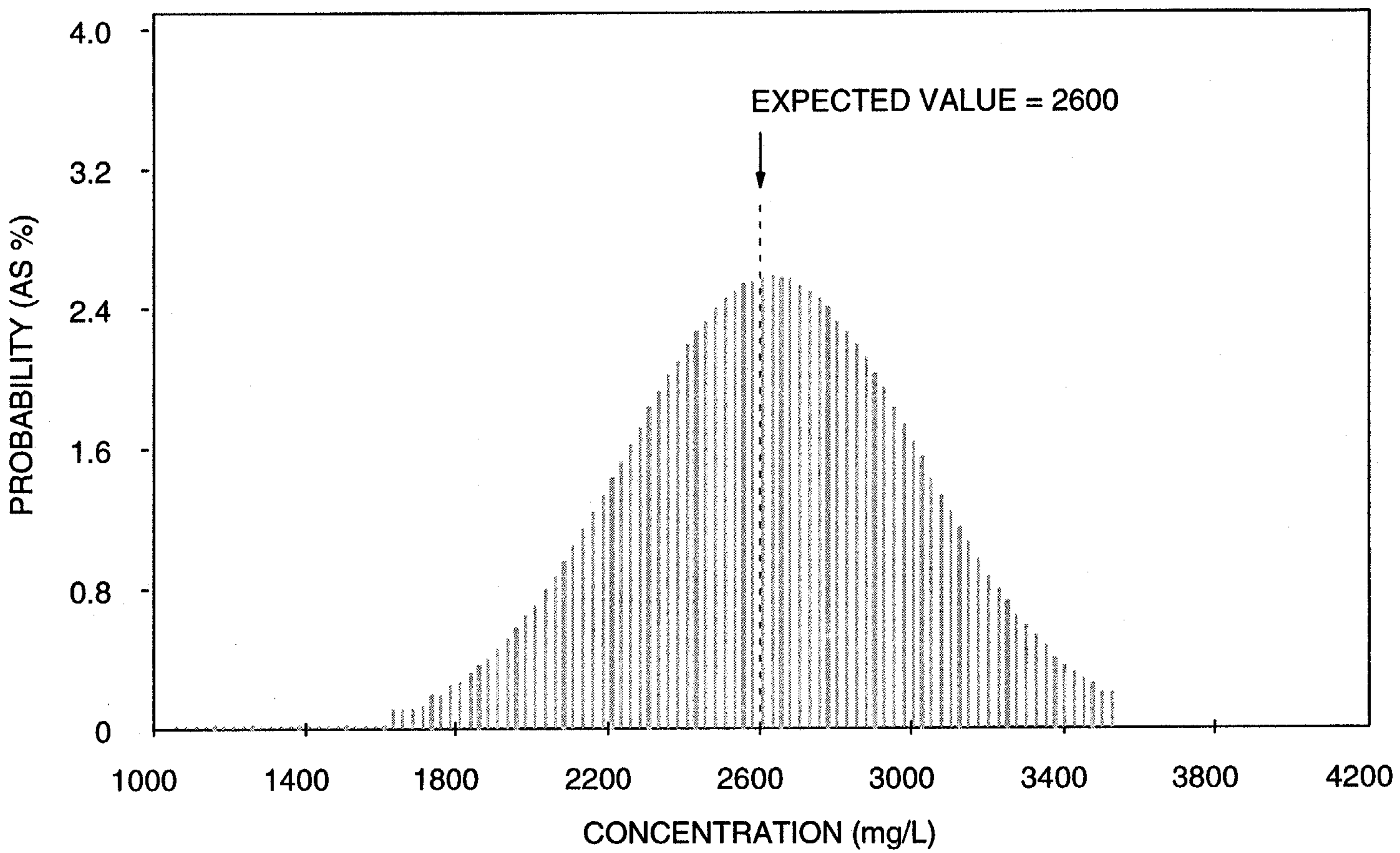

FIGURE 4.4
SIMULATED DISTRIBUTION OF CHLORIDE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

CHLCON.DRW

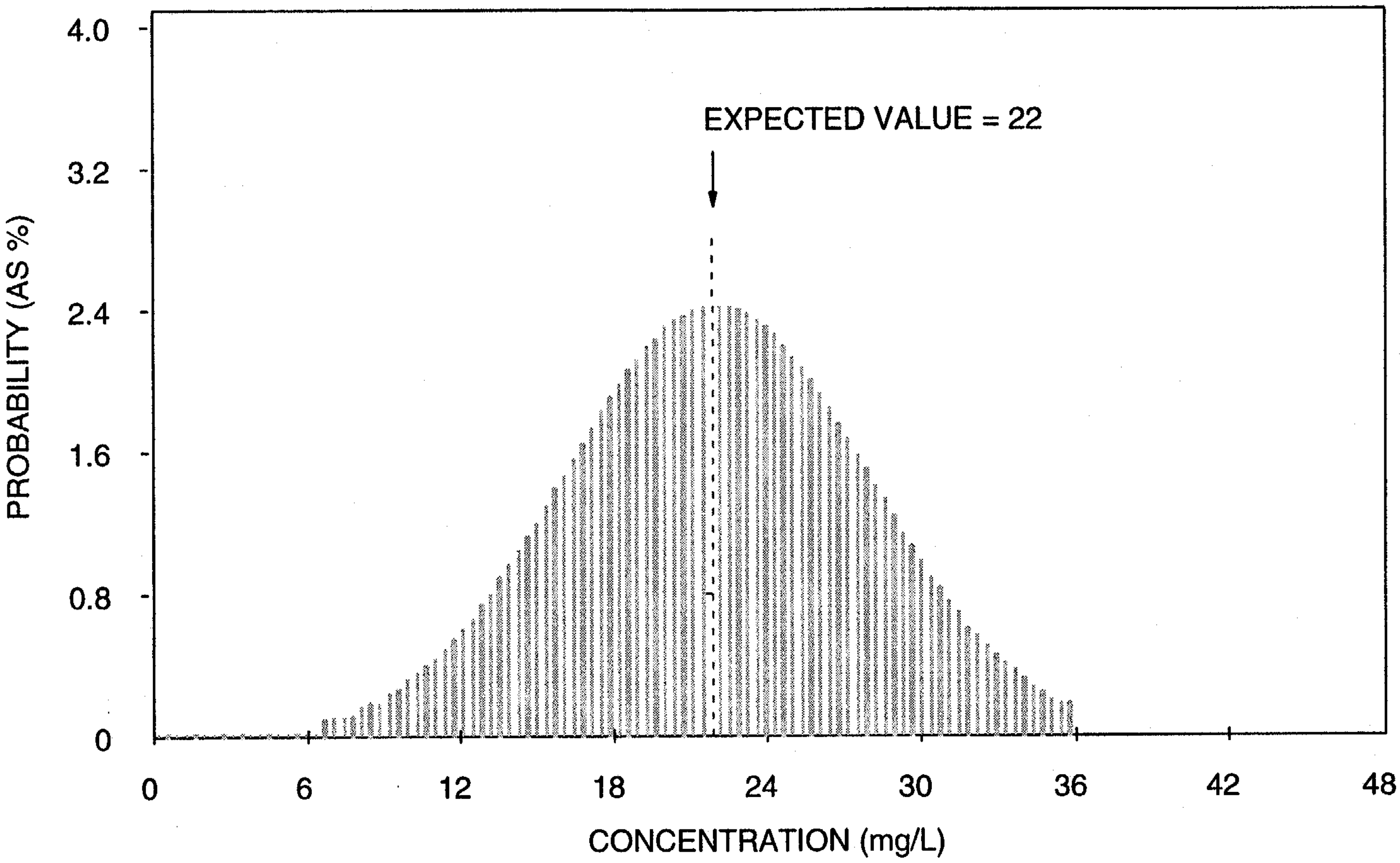

FIGURE 4.5
SIMULATED DISTRIBUTION OF IRON CONCENTRATIONS
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

IROCON.DRW

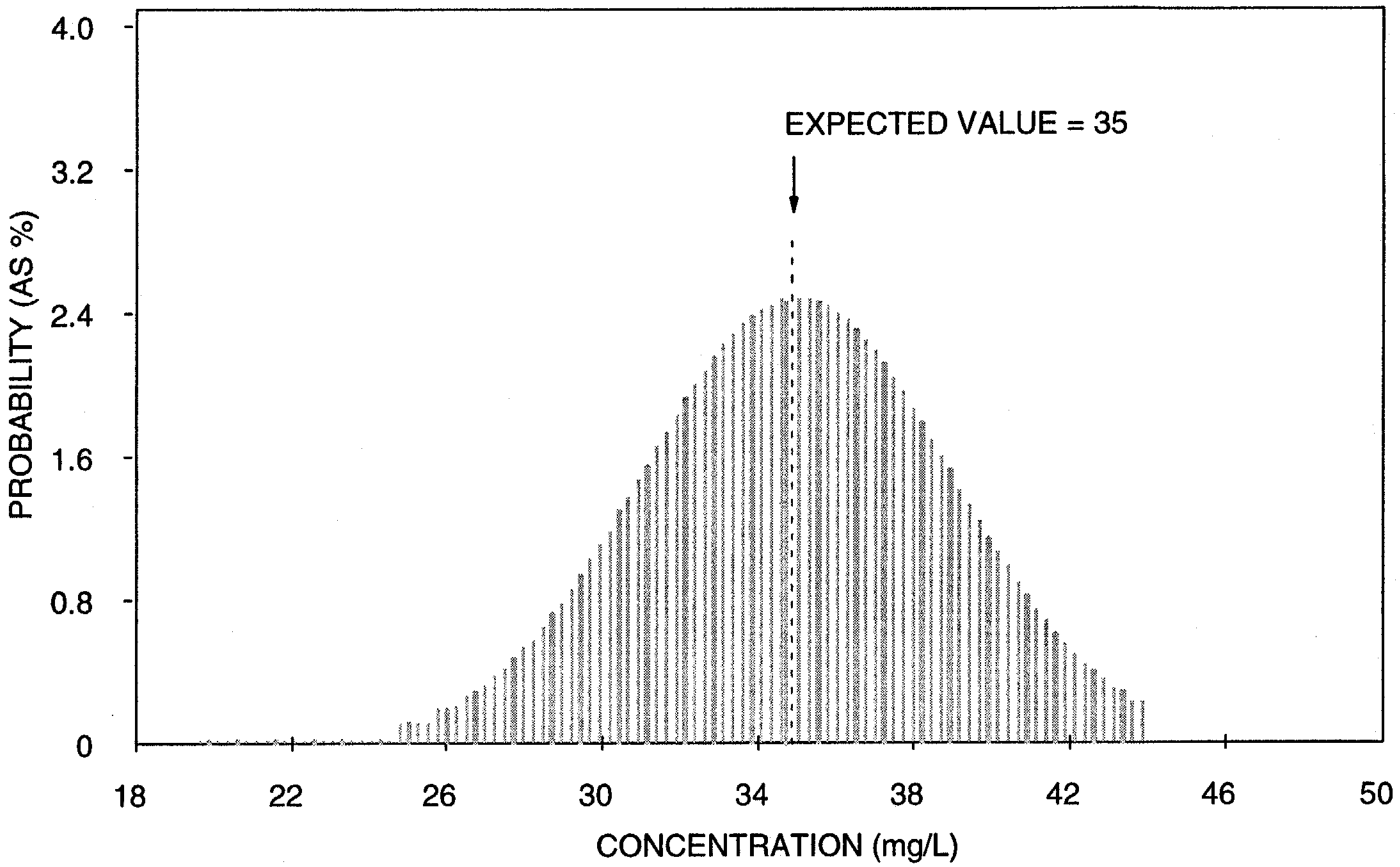

FIGURE 4.6
SIMULATED DISTRIBUTION OF MANGANESE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

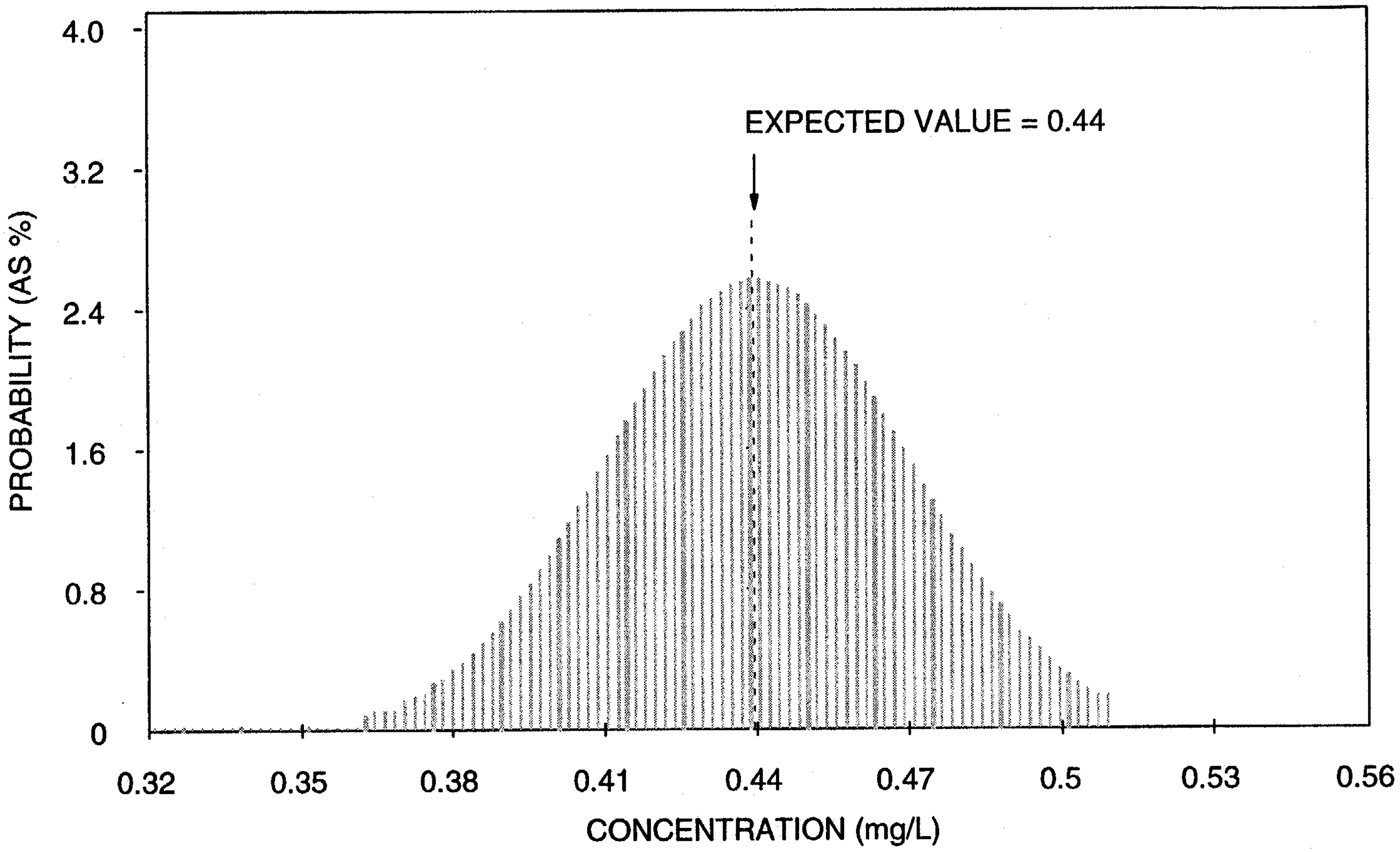

FIGURE 4.7
SIMULATED DISTRIBUTION OF MOLYBDENUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

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MOLYDON.DRW

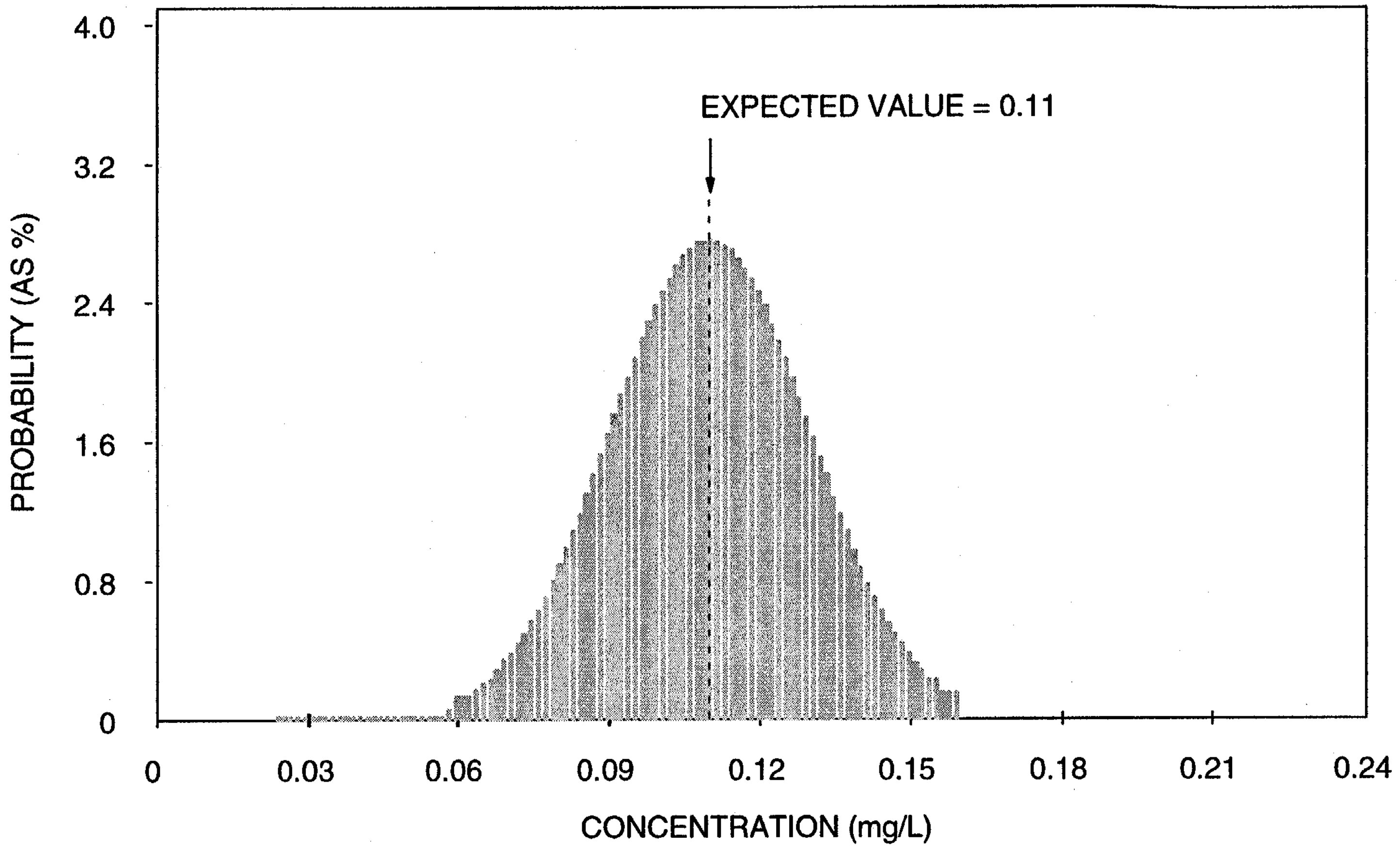

FIGURE 4.8
SIMULATED DISTRIBUTION OF NICKEL CONCENTRATIONS
LAKEVIEW, OREGON, SITE

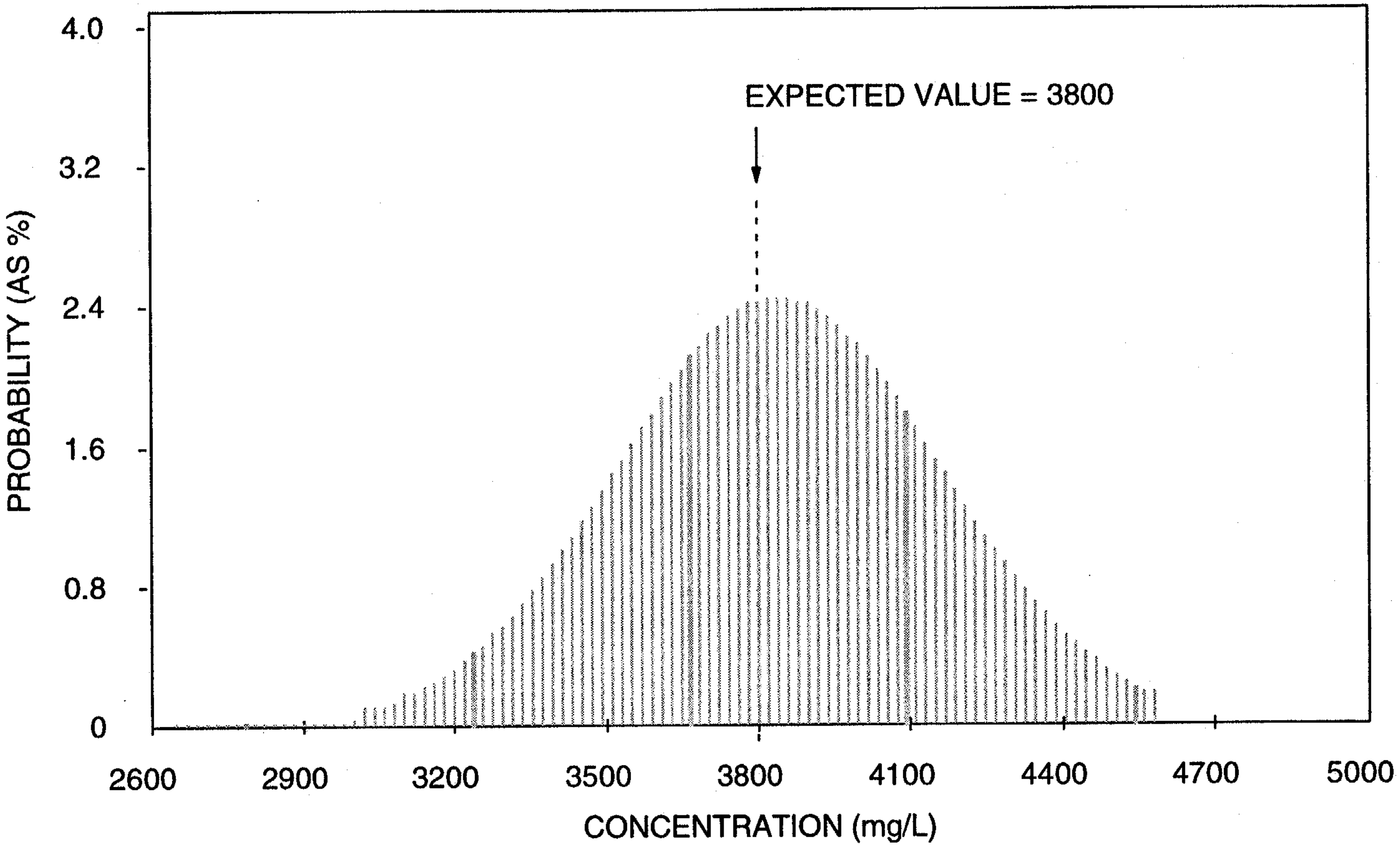

FIGURE 4.9
SIMULATED DISTRIBUTION OF SODIUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

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SODICON.DRW

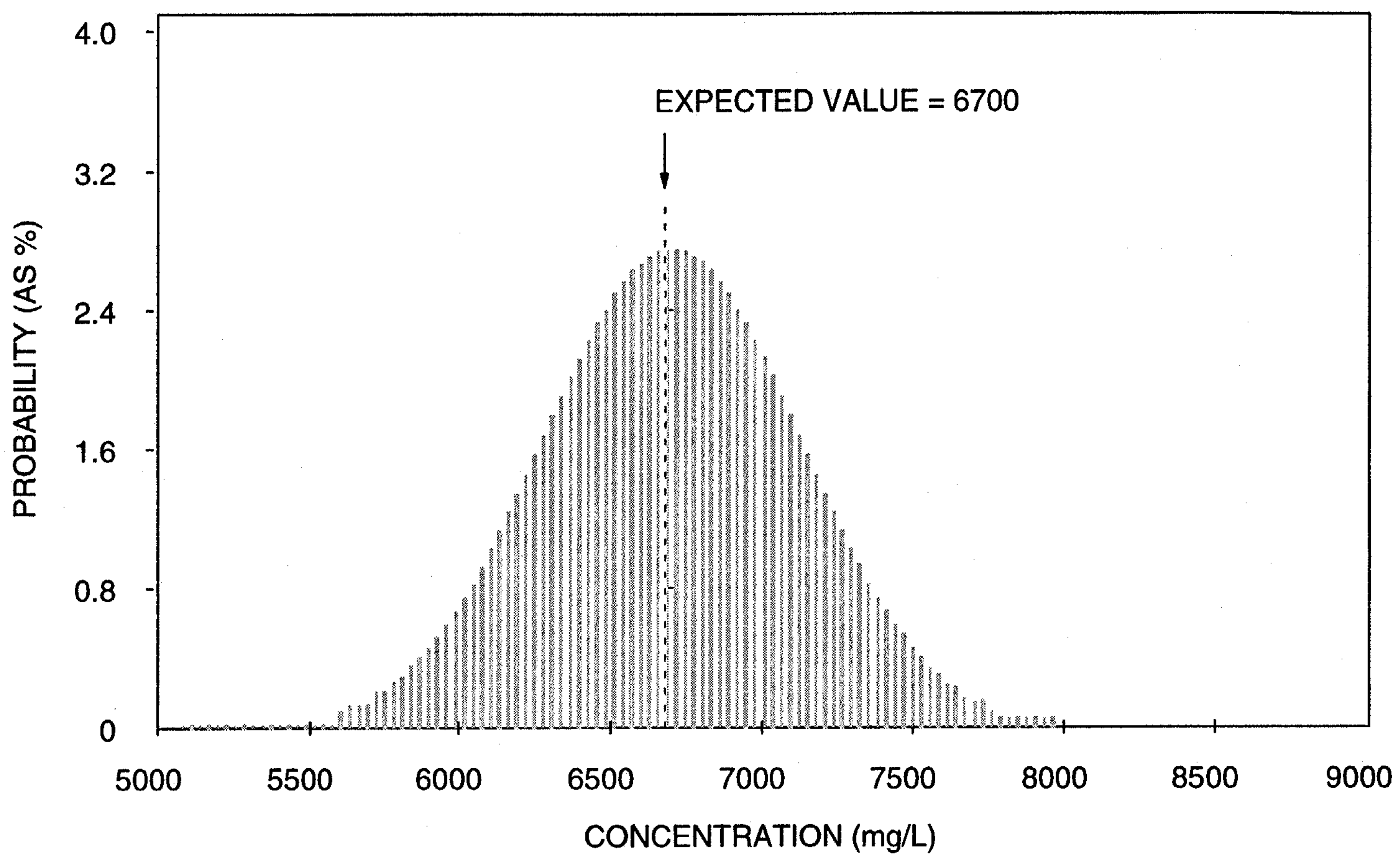


FIGURE 4.10
SIMULATED DISTRIBUTION OF SULFATE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

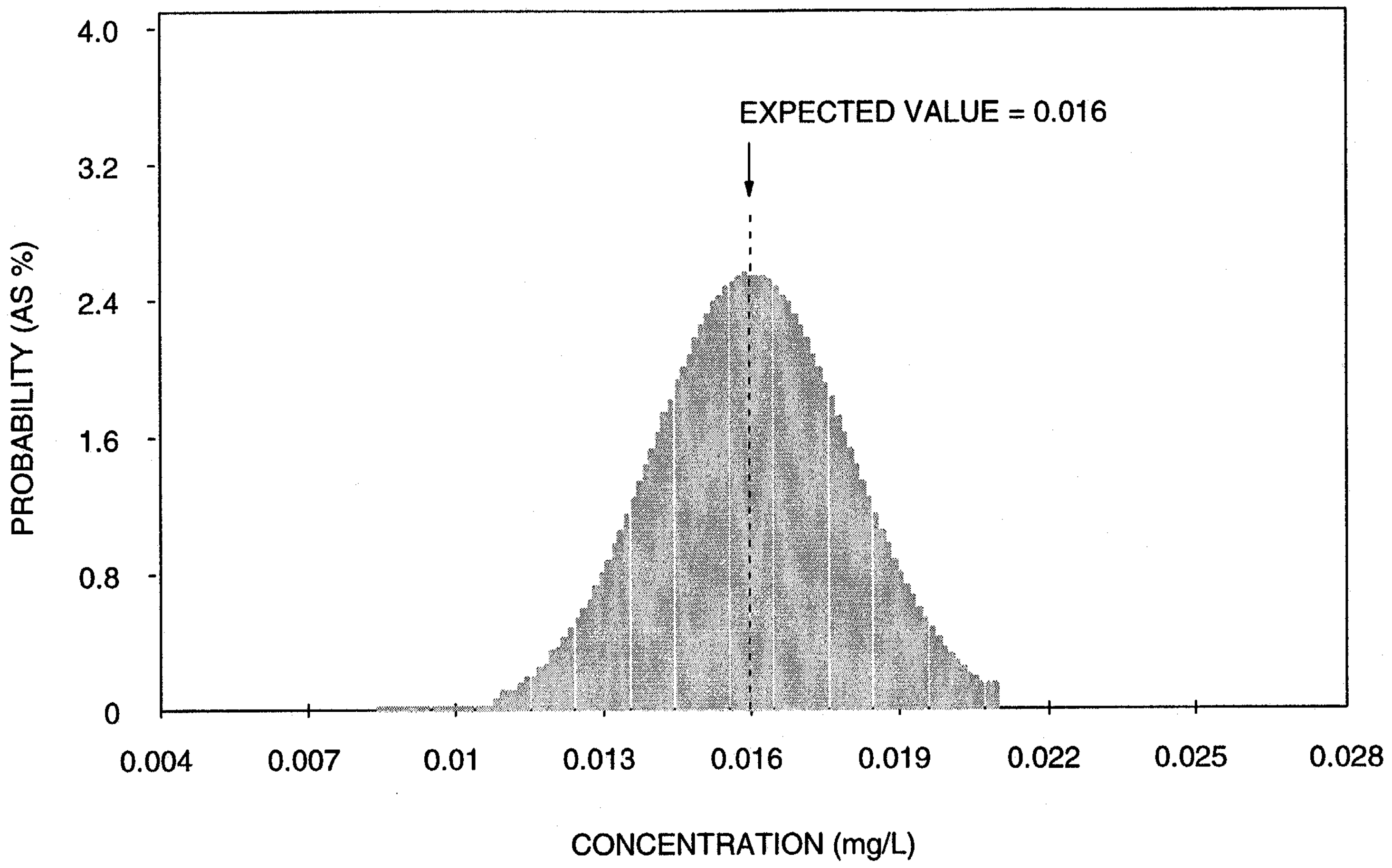


FIGURE 4.11
SIMULATED DISTRIBUTION OF URANIUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

URACON.DRW

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 contaminants in drinking water depends primarily on long-term average daily consumption of the contaminant per kilogram of body weight. For probability distributions of noncarcinogens, the chronic daily intake is calculated as follows:

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

Potential carcinogenicity is thought to increase with *total* intake over time, instead of with *average daily* intake as for noncarcinogens. The only carcinogens elevated above background levels at the Lakeview site are arsenic and the radionuclides polonium-210 and uranium. Intake of a radioactive carcinogen therefore is quantified as total exposure to radioactivity throughout the residency period of an individual:

$$\underset{\text{(pCi)}}{\text{Intake}} = \underset{\text{(pCi/L)}}{\text{Concentration}} \times \underset{\text{(L per day)}}{\text{ingestion rate}} \times \underset{\text{(days per year)}}{\text{exposure frequency}} \times \underset{\text{(years)}}{\text{exposure duration}}$$

Intake of a chemical carcinogen such as arsenic is quantified using the following equation:

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

Populations

The potentially exposed populations evaluated in this risk assessment include 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:

- Survey data for population variables such as age, weight, and daily water intake are available for these age groups.

- Toxicological variables are similar within these age groups, including responsiveness of sensitive subgroups (infants and children) to the contaminants of potential concern, toxicant intake-to-body-weight ratios, and toxicokinetics.

Average daily intake (L per day)

Lognormal probability distributions were used to describe variations in average daily tap water intake among members of the population (Roseberry and Burmaster, 1992). These distributions were developed from data collected during a 1977-1978 food consumption survey conducted nationwide by the U.S. Department of Agriculture. During the survey, total tap water consumption during a 3-day period was recorded for 26,081 survey participants nationwide (Figure 4.12).

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. 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 4.13).

Exposure duration (years)

Distributions of total residence time (or exposure duration) for the populations described above were developed by Israeli and Nelson (1992) using data collected by the U.S. Department of Commerce, the Bureau of the Census, and the U.S. Department of Housing and Urban Development in 1985 and 1987. However, a fixed exposure duration of 30 years for adults, 10 years for children, and 1 year for infants was assumed for the populations in the Lakeview site risk assessment.

Using exposure concentration distributions discussed in Section 4.3 and the intake parameter distributions described in this section, total intake distributions were simulated for the three age groups. 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. Therefore, simulated intake distributions for 1- to 10-year old children for the contaminants at this site (with the exception of sulfate, where intake by infants was assessed) are presented in Figures 4.14 through 4.23.

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 significant sources of uncertainty:

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

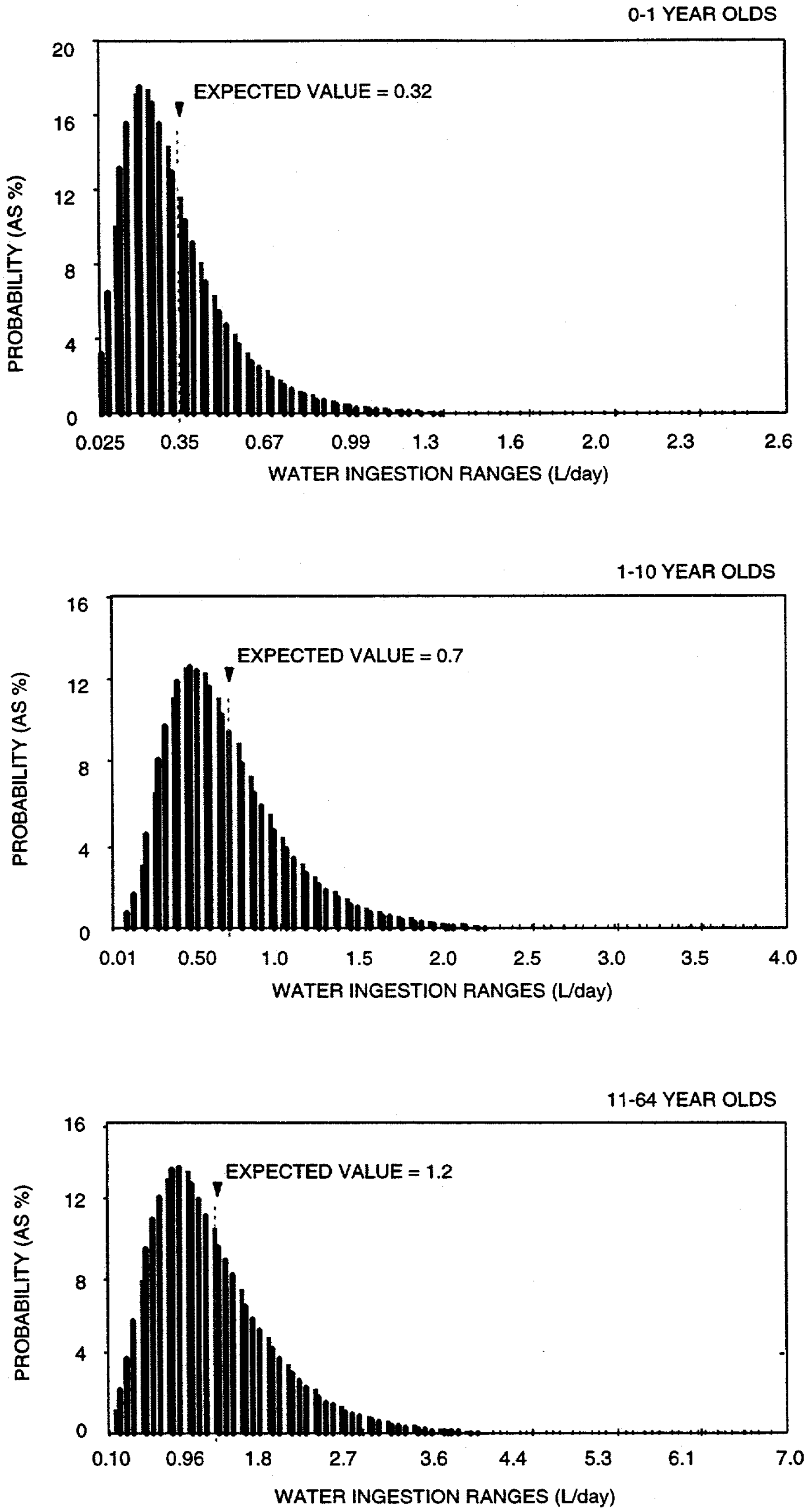
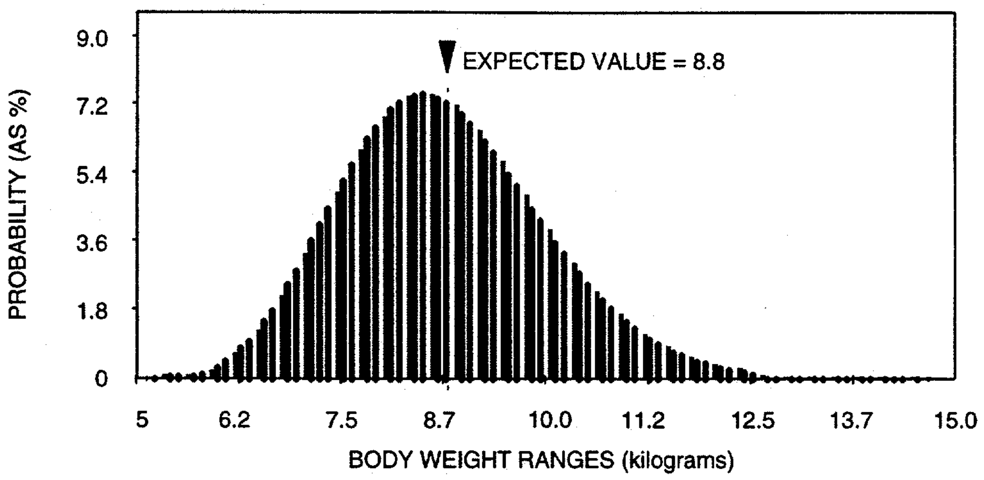
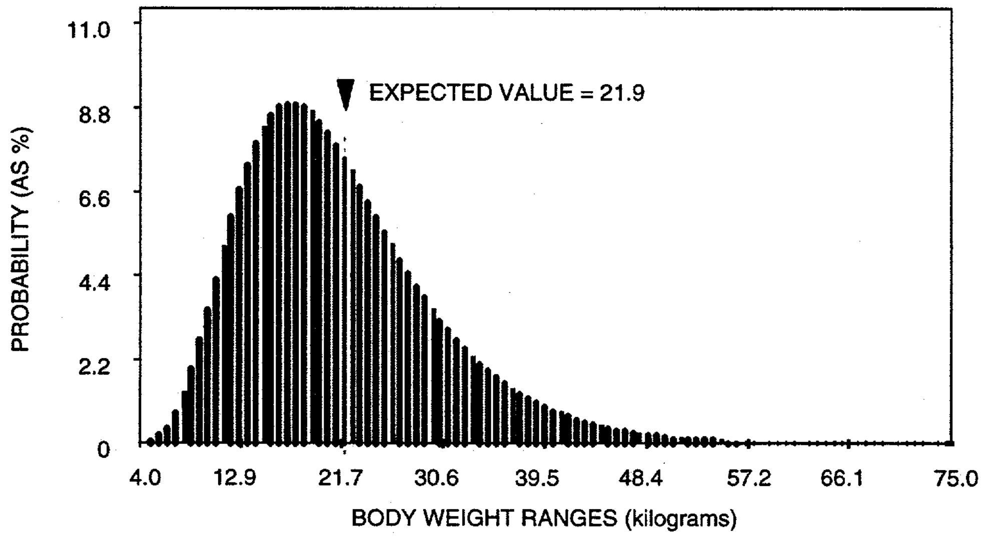


FIGURE 4.12
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES

0-1 YEAR OLDS



1-10 YEAR OLDS



11-64 YEAR OLDS

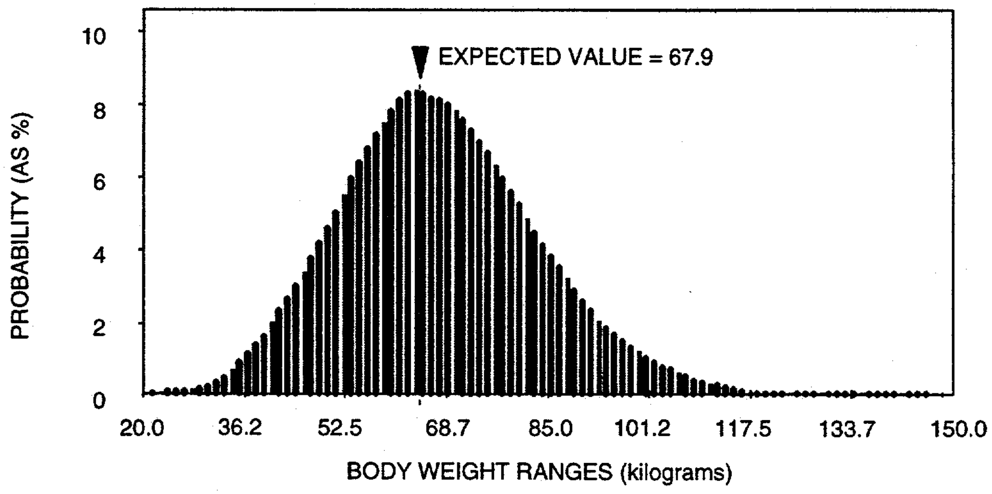
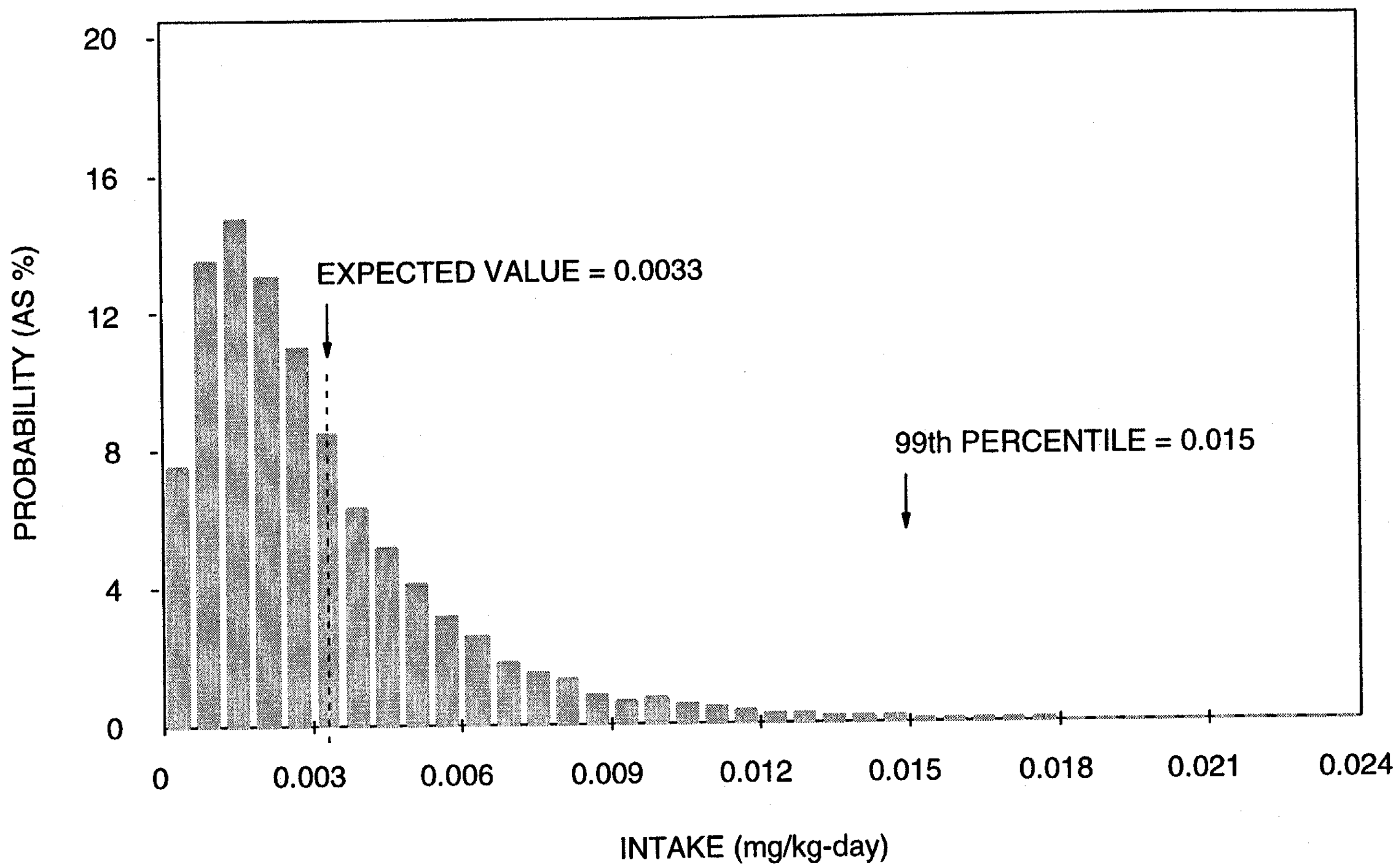
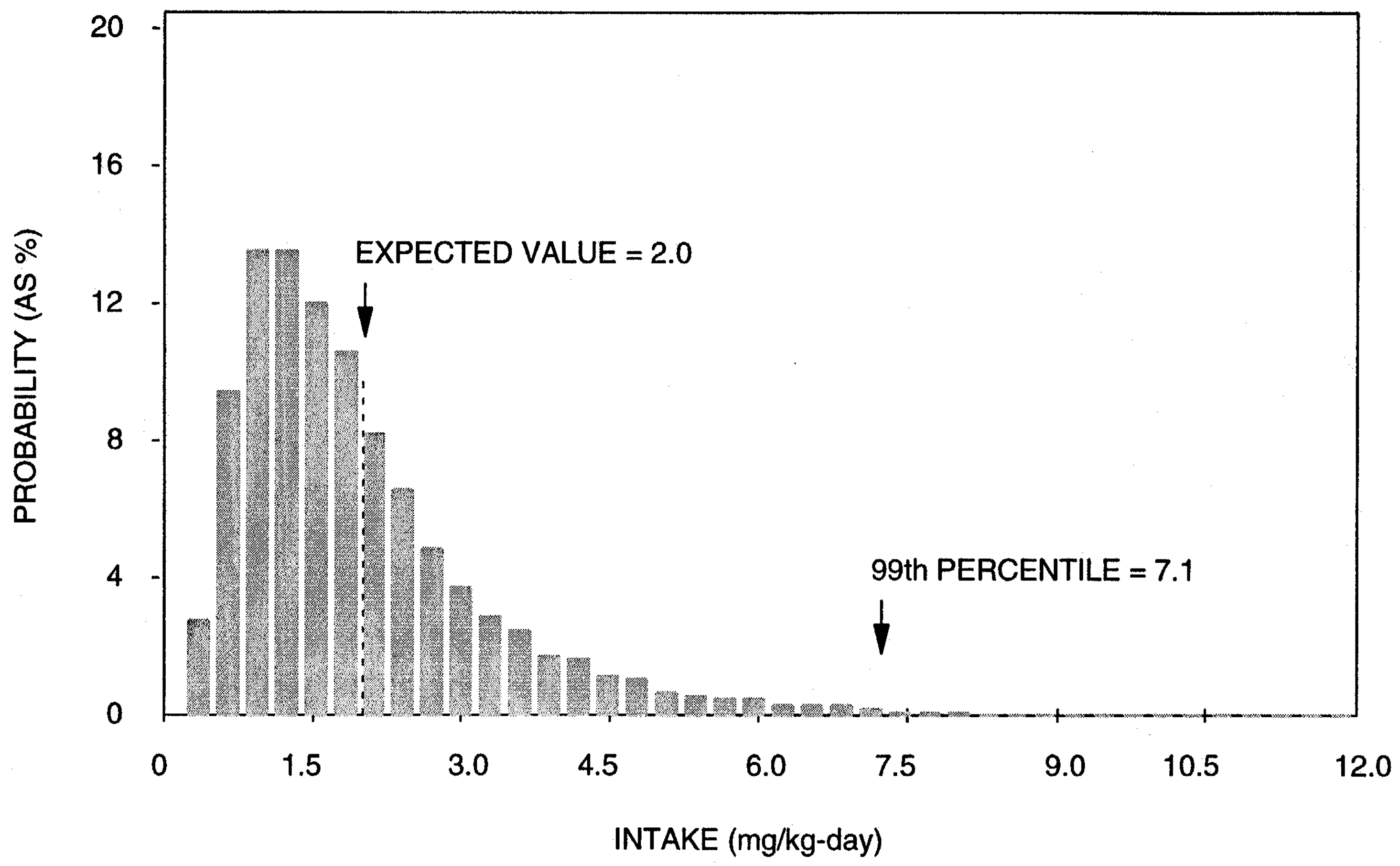


FIGURE 4.13
PROBABILITY DISTRIBUTIONS FOR BODY WEIGHT



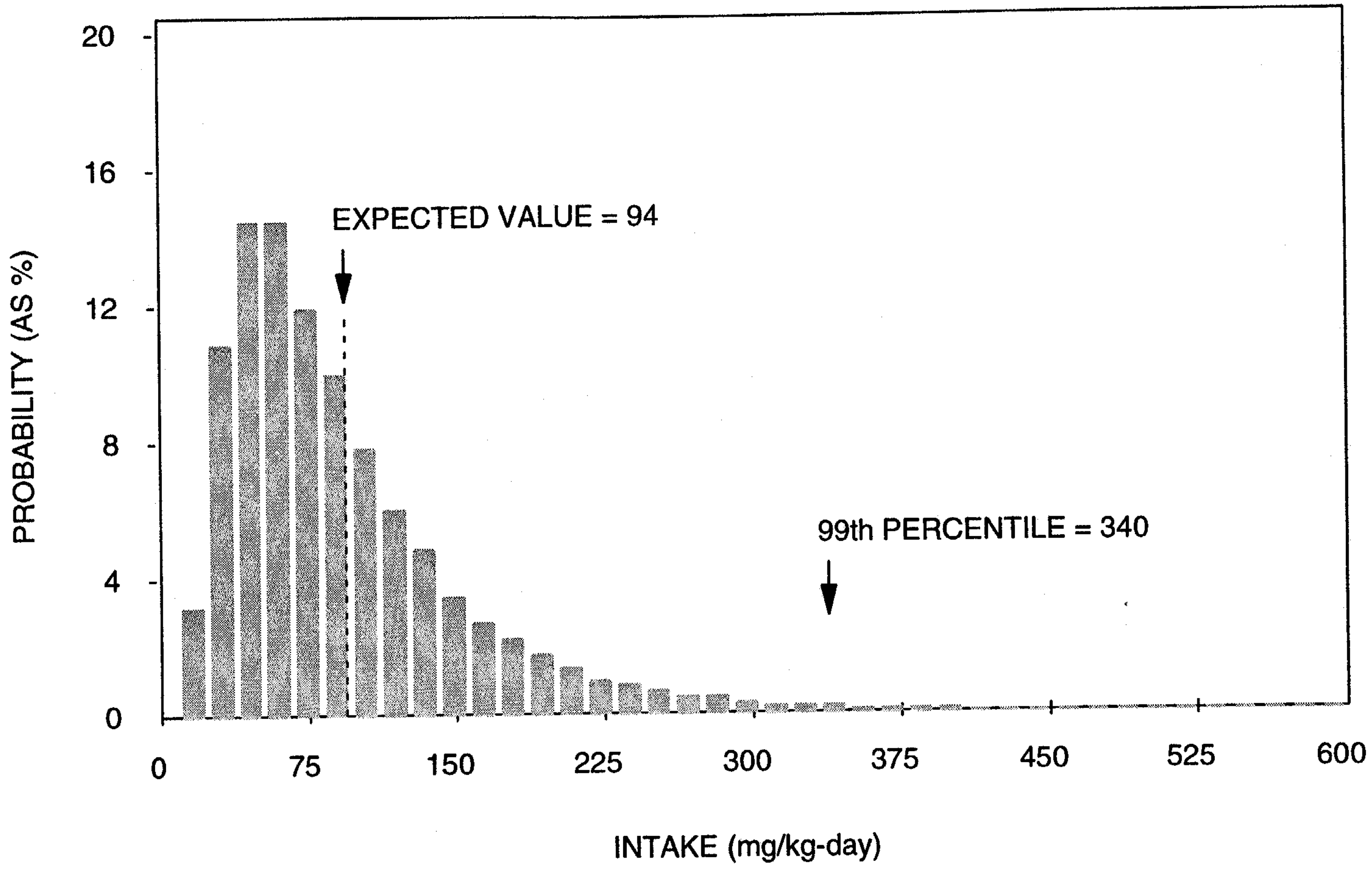
NOTE: AVERAGE POTENTIAL INTAKE IS 0.0033 ARSENIC/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.14
DISTRIBUTION OF POTENTIAL ARSENIC INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION LAKEVIEW, OREGON, SITE



NOTE: AVERAGE POTENTIAL INTAKE IS 2.0 mg BORON/kg BODY WEIGHT/DAY. INTAKES 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 BORON INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

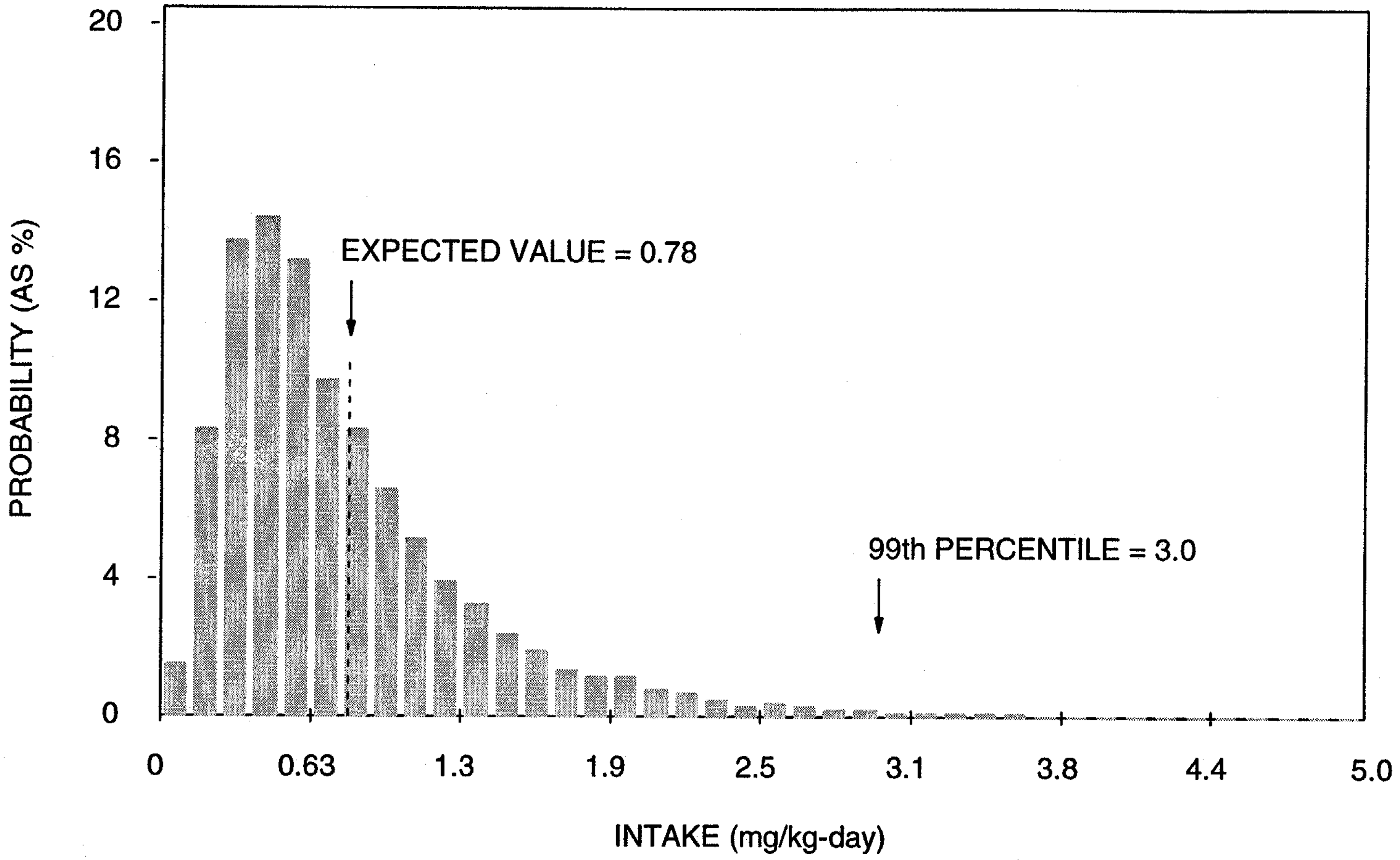


NOTE: AVERAGE POTENTIAL INTAKE IS 94 mg CHLORIDE/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.16
DISTRIBUTION OF POTENTIAL CHLORIDE INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

CHLINT.DRW

PATH I:\FL3.01\LKVBRA

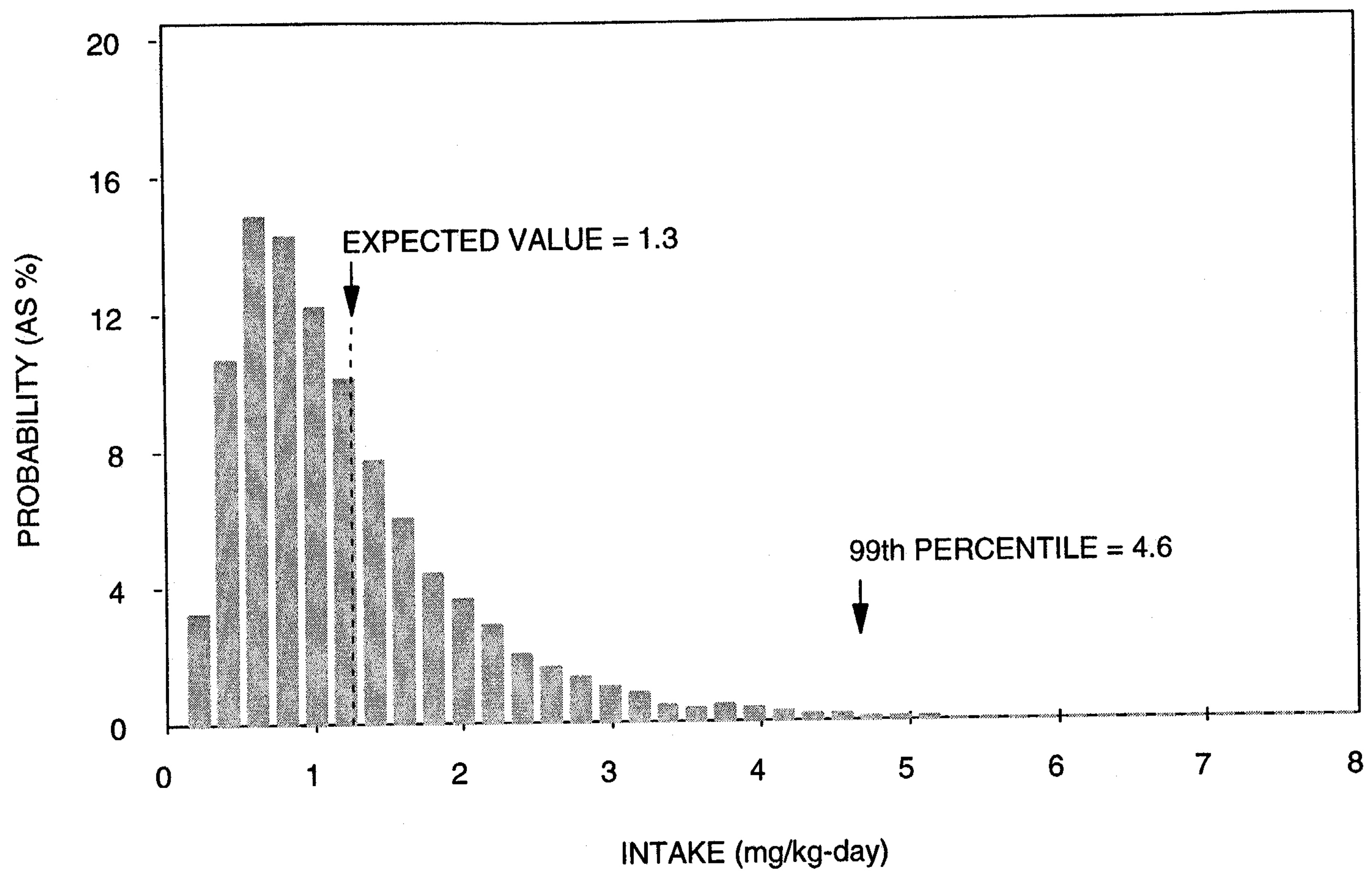


NOTE: AVERAGE POTENTIAL INTAKE IS 0.78 mg IRON/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.17
DISTRIBUTION OF POTENTIAL IRON INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

IRONINT.DRW

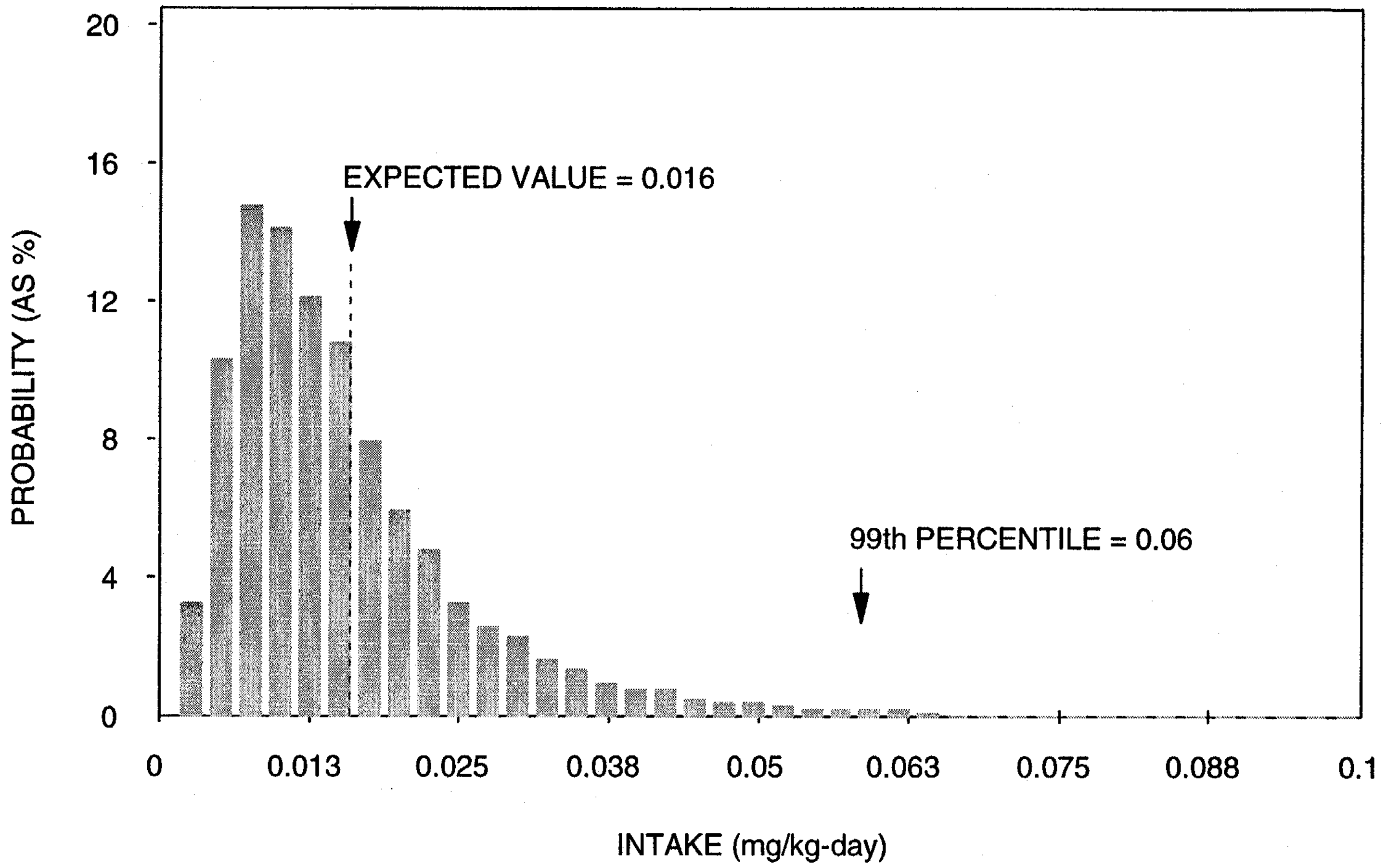


NOTE: AVERAGE POTENTIAL INTAKE IS 1.3 mg MANGANESE/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.18
DISTRIBUTION OF POTENTIAL MANGANESE INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

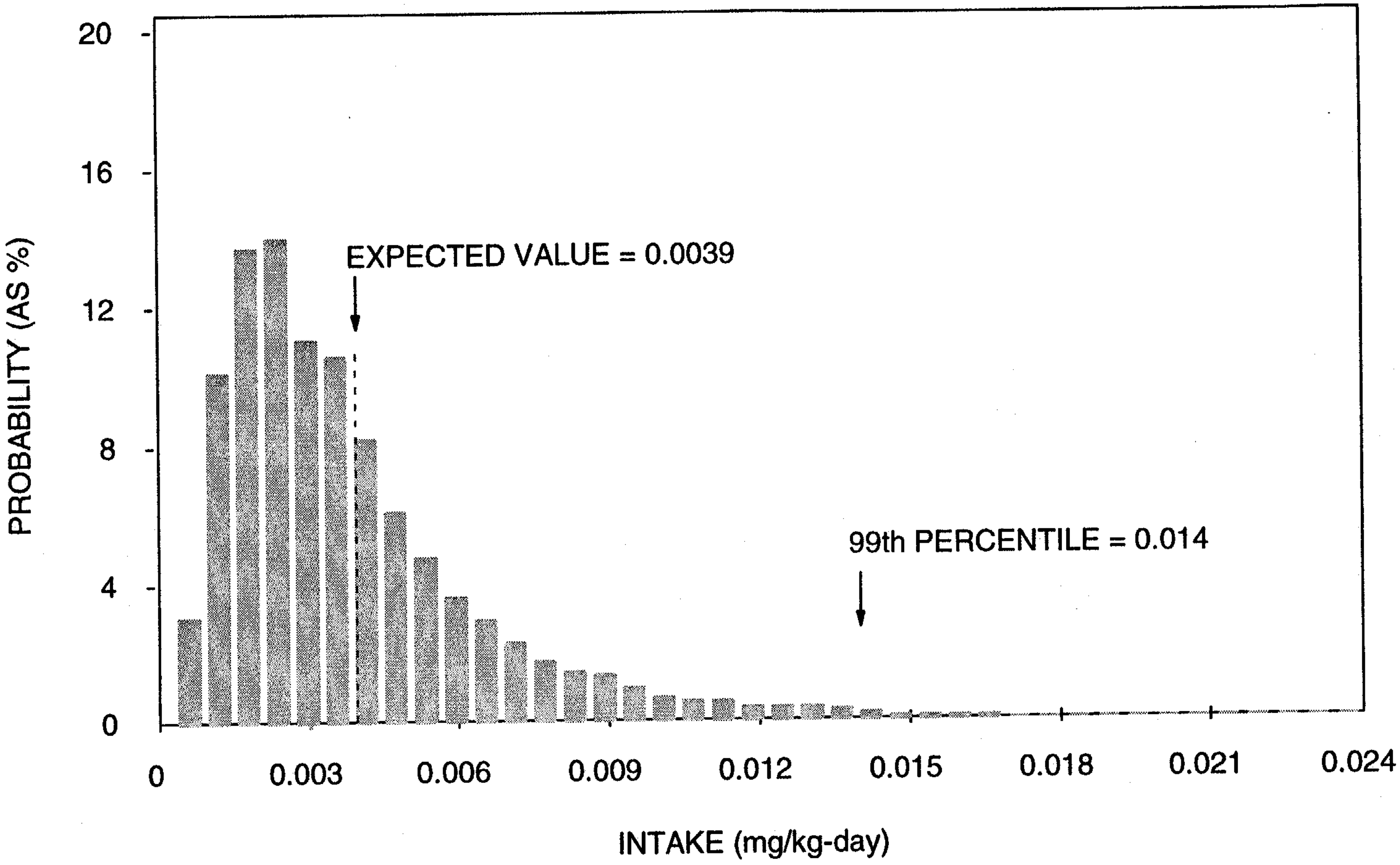
MANGINT.DRW

PATH I:\FL3.01\LKVBRA



NOTE: AVERAGE POTENTIAL INTAKE IS 0.016 mg MOLYBDENUM/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.19
DISTRIBUTION OF POTENTIAL MOLYBDENUM INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION LAKEVIEW, OREGON, SITE

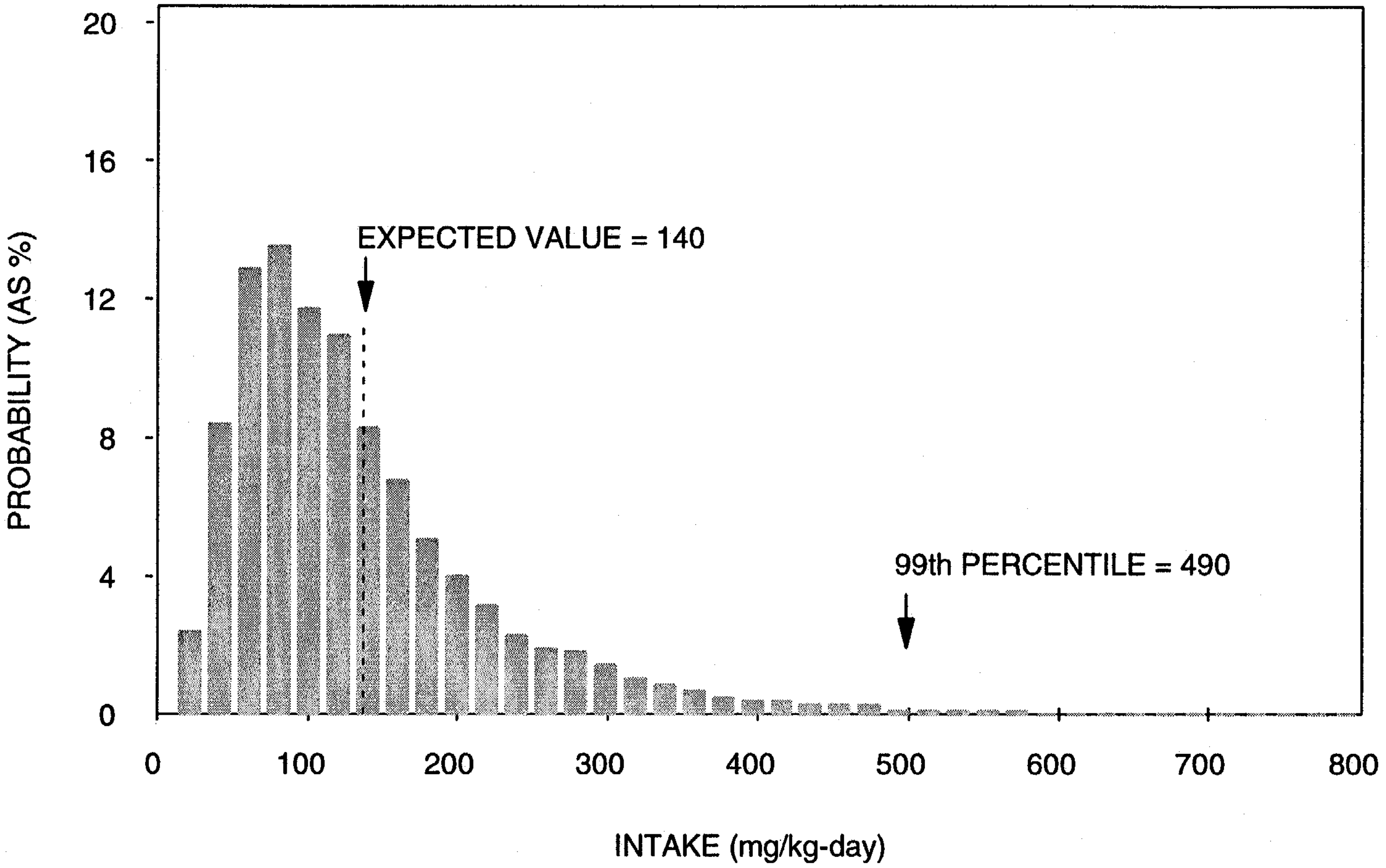


NOTE: AVERAGE POTENTIAL INTAKE IS 0.0039 mg NICKEL/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.20
DISTRIBUTION OF POTENTIAL NICKEL INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

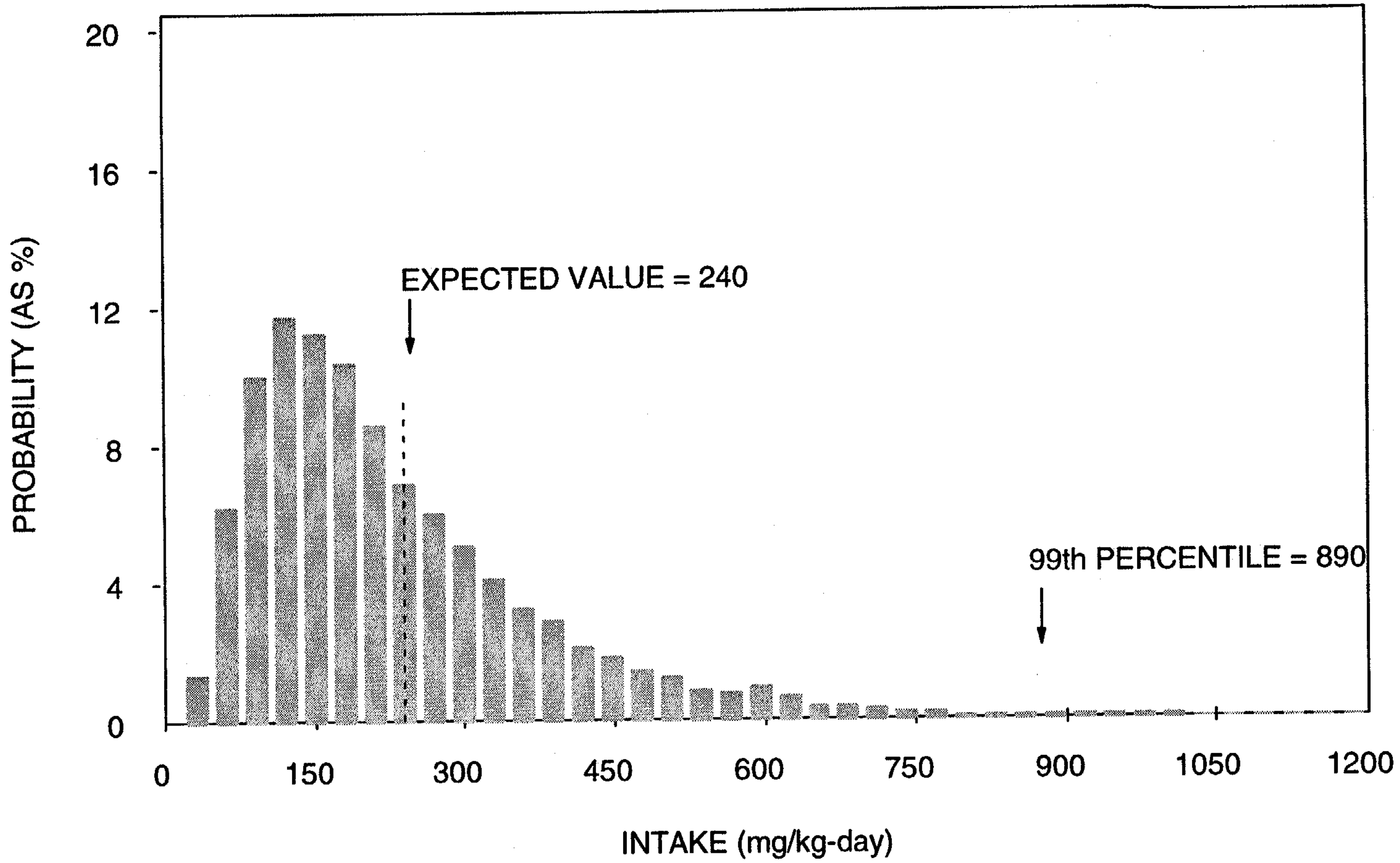
NICKINT.DRW

PATH I:\FL3.01\LKVBRA



NOTE: AVERAGE POTENTIAL INTAKE IS 140 mg SODIUM/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.21
DISTRIBUTION OF POTENTIAL SODIUM INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

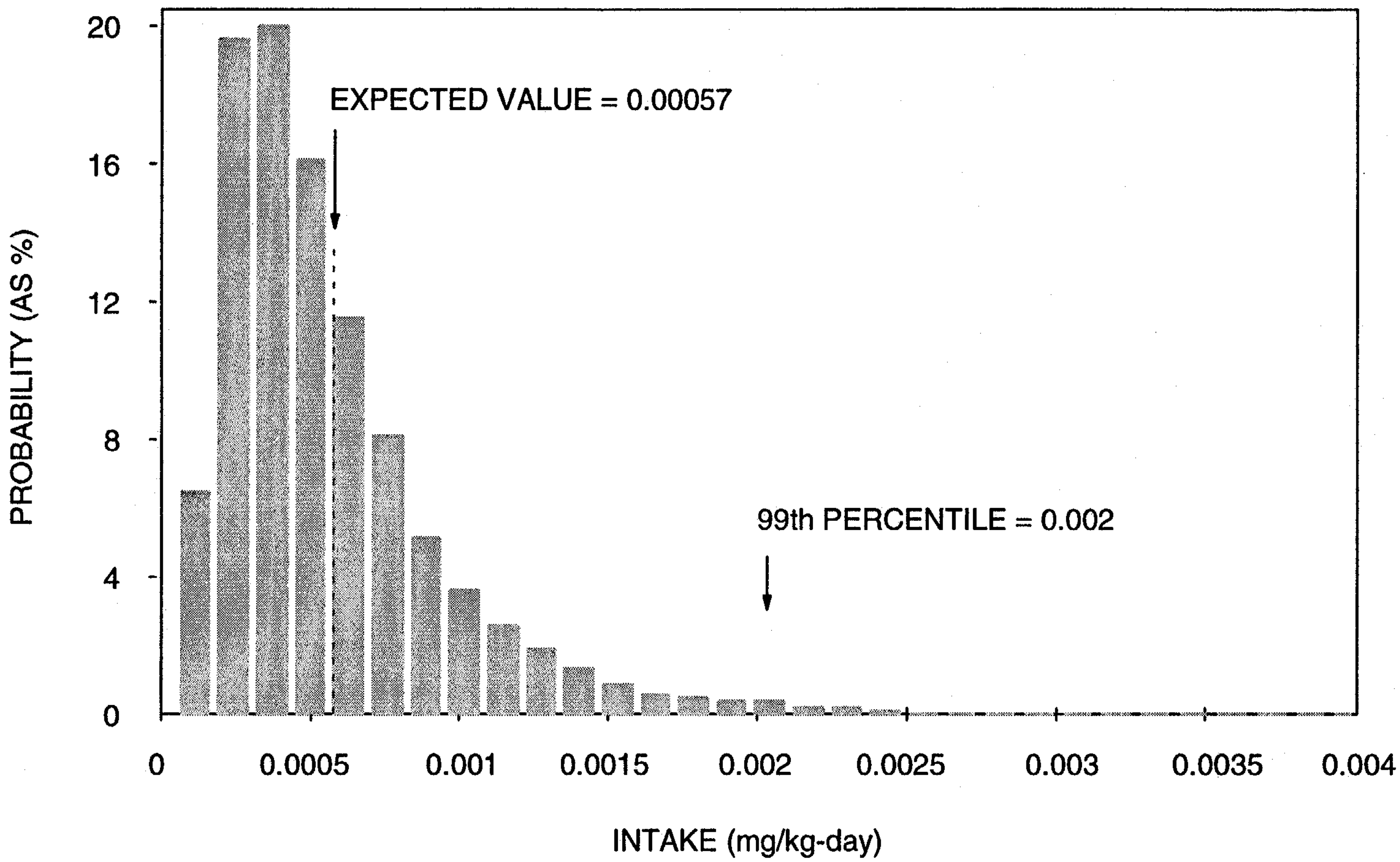


NOTE: AVERAGE POTENTIAL INTAKE IS 240 mg SULFATE/kg BODY WEIGHT/DAY. INTAKES FOR INFANTS HAVE BEEN MODELED BECAUSE THEY ARE THE MOST SUSCEPTIBLE RECEPTORS TO SULFATE TOXICITY.

FIGURE 4.22
DISTRIBUTION OF POTENTIAL SULFATE INTAKES FOR INFANTS
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

PATH I:\FL3.01\LKVBRA

SULINT.DRW



NOTE: AVERAGE POTENTIAL INTAKE IS 0.00057 mg URANIUM/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.23
DISTRIBUTION OF POTENTIAL URANIUM INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE

- Uncertainties arising from the assumption that the ground water contaminant source term at the site has reached a steady state and that contaminant concentrations at the exposure point will remain constant for chronic periods of exposure (generally greater than 7 years). Because the source of contamination at Lakeview has been removed, and although ground water contamination appears to be decreasing slowly, the assumption of a constant source probably will lead to an overestimation of risk.

- Uncertainties associated with the model used to estimate contaminant uptake into plants for the irrigated garden produce pathway. Site-specific plant uptake factors could vary substantially from the default literature estimates. Additionally, bioconcentration factors (BCF) used for livestock are not well defined. Therefore, uncertainty is associated with the calculation of potential exposure levels in livestock. As with environmental sampling, the net effect on risk estimates of this uncertainty cannot be predicted.

Despite these uncertainties, the use of probability distributions that incorporate all definable sources of variability provide a representative picture of the potential range of exposures at the Lakeview site.

5.0 TOXICITY ASSESSMENT

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

The EPA Office of Research and Development has calculated acceptable intake values, or reference doses (RfD), for long-term (chronic) exposure to noncarcinogens. These values are estimates of route-specific exposure levels that would not be expected to cause adverse effects when exposure occurs for a significant portion of the 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 HEAST or are provided through the EPA's IRIS database.

The toxicity profiles presented here focus on drinking water-based toxicity source material for humans when they are available. Animal data are included only if human data are not available. Animal data on the toxicity range graphs are represented by widely spaced dotted lines. Uncertainty about the beginning or ending points of an exposure range that produces specific toxic effects is represented by closely spaced dots at the appropriate end of the line denoting range.

5.1 CONTAMINANT TOXICITY SUMMARIES

The following summaries address the basic toxicokinetics and toxicity of the contaminants of potential concern at the Lakeview site based on the preliminary screening discussed above. These contaminants of potential concern are arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium. Although these contaminants have a wide range of toxic effects, depending on the exposure levels, the following discussions focus on toxic effects observed in the exposure range most relevant to contamination at the Lakeview site.

5.1.1 Arsenic

Absorption

Arsenic effectively is absorbed through the gastrointestinal tract and inhalation. Relative to gastrointestinal absorption, dermal absorption is negligible. In humans, approximately 80 percent of an ingested amount of dissolved inorganic trivalent (arsenite) or pentavalent arsenic (arsenate) is absorbed from the gastrointestinal tract (Pershagen and Vahter, 1979; Marafante and Vahter, 1987).

Tissue accumulation and clearance

After absorption by the gastrointestinal tract, arsenic is transported through the blood to most tissues. In humans as well as in most animal species, exposure to either arsenite or arsenate leads to initial accumulation in the liver, kidneys, and lungs. Clearance from these tissues is very rapid, and long-term retention of arsenic is seen in organs rich in sulfhydryl-containing proteins, such as hair, skin, squamous epithelium of the upper gastrointestinal tract, epididymis, thyroid, lens, and skeleton (Lindgren et al., 1982). Specific target tissue is dependent on the form of arsenic. Higher arsenic retention occurs after exposure to trivalent arsenic than to the pentavalent form and tissue distribution is altered (Webb, 1966; Casarett and Doull, 1991).

In humans and rats, inorganic arsenic passes through the placental barrier. It also has been demonstrated to enter both cow and human milk (Marcus and Rispin, 1988).

In the human body, where methylcobalamine acts as a major methyl group donor in the biotransformation process, inorganic arsenic is converted to methylated compounds. Marcus and Rispin (1988) demonstrated that the major site of arsenic methylation is the liver. Trivalent arsenic is the substrate for methylation, and pentavalent arsenic must be reduced to trivalent arsenic before methylation can occur. Dimethylarsenic acid is a major metabolite found in animals and humans. Methylation results in a detoxification of inorganic arsenic (about 1 order of magnitude per methyl group) and increases the rate of arsenic excretion from the body.

The major excretion route following human exposure to inorganic arsenic is through the kidneys (Ishinishi et al., 1986). Only a low percentage is excreted in feces. The excretion rate in urine varies, depending on the chemical form of arsenic and the species exposed. In humans exposed to a single low dose of arsenite, about 35 percent was excreted in urine over a period of 48 hours (Buchet et al., 1980; 1981). With continuous human intake over a few days, 60 to 70 percent of the daily dose is excreted in urine (Buchet et al., 1981). Following exposure to arsenate, the limited human data available indicate an arsenate excretion rate similar to that of arsenite. Other less important routes of elimination of inorganic arsenic include skin, hair, nails, and sweat.

After oral intake of radiolabeled pentavalent arsenic, 66 percent was excreted with a half time of 2.1 days, 30 percent with a half time of 9.5 days, and 3.7 percent with a half time of 38 days (Marcus and Rispin, 1988).

Environmental sources of arsenic

Arsenic is ubiquitous in nature in both inorganic and organic compounds. Water is the major means of transport of arsenic under natural conditions. In oxygenated water, arsenic occurs in a pentavalent form; under reducing conditions (e.g., in artesian well waters), the trivalent form predominates. Sedimentation of arsenic in association with iron and aluminum represents a considerable factor in environmental transport and deposition of this element (Marcus and Rispin, 1988).

As a result of its widespread occurrence, the general human population is exposed to arsenic primarily from drinking water and foodstuffs. Certain target groups are exposed to arsenic from industrial and agricultural uses. Medicinal use also has been a significant means of human exposure.

Drinking water usually contains a few micrograms of arsenic, predominantly as inorganic salts in the trivalent and pentavalent states (WHO, 1981). However, concentrations of up to 1.1 mg/L in drinking water have been reported in Chile, Argentina, Taiwan, the United States, and the United Kingdom (WHO, 1981).

Certain foodstuffs contain appreciable amounts of arsenic. Arsenic concentrations in fish and seafood, particularly shellfish, is generally 1 or 2 orders of magnitude higher than in other foods. Wine and mineral waters can contain several hundred micrograms of arsenic per liter (Crecelius, 1977; WHO, 1981).

Toxicity of arsenic

Levels of exposure associated with acute arsenic toxicity vary with its valency form. Trivalent arsenicals (arsenites) are generally more toxic than pentavalent (arsenates) (Morrison et al., 1989), and inorganic arsenic compounds are more toxic than organic (Shannon and Strayer, 1989). Based on geochemical models for the Lakeview site, arsenic occurs primarily in the pentavalent form in ground water (Table 3.7). For arsenic trioxide, the reported estimated acute oral lethal dose in humans ranges from 70 to 300 mg (1 to 4 milligrams per kilogram per day [mg/kg-day]) (EPA, 1984). Acute exposure to inorganic arsenic compounds may lead to severe inflammation of the gastrointestinal tract, encephalopathy, and acute renal failure after ingestion.

Teratogenic effects of arsenic compounds administered intravenously or intraperitoneally at high doses have been demonstrated in laboratory animals only (Ferm, 1971; Hood, 1972; EPA, 1984).

Chronic arsenic intoxication results from exposure to small doses of arsenic over a long period of time. These intoxications frequently are caused by arsenic in drinking water and in food. Skin changes leading to skin cancer commonly are seen in populations exposed to high concentrations of arsenic in drinking water. Endemic arsenic poisoning is seen in Cordoba, Argentina, where the concentration of arsenic in drinking water ranges from 0.9 to 3.4 mg/L (equivalent to 0.026 to 0.097 mg/kg-day). Certain areas in Taiwan also have high natural arsenic concentrations in drinking water that cause blackfoot disease (a peripheral extremity vascular disorder resulting in gangrene). A dose-response relationship between the incidence of blackfoot disease and the duration of exposure to arsenic has been documented (Tseng, 1977; EPA, 1994a).

Hyperpigmentation, hyperkeratoses, and skin cancer with prevalence of 7.1 percent, 18.4 percent, and 1.1 percent, respectively, were reported in Taiwanese studies of more than 40,000 people exposed to arsenic in drinking water at daily intakes ranging from 1.4 to 6.3 mg per day. However, these adverse health effects also were observed at lower exposure levels.

Increasing chronic oral ingestion doses of arsenic progressively produces systemic effects, including 1) arterial thickening in children and adults (0.02 mg/kg-day); 2) neurological symptoms, including peripheral neuropathy (0.04 mg/kg-day); 3) fibrosis of the liver (0.05 mg/kg-day); and 4) cirrhosis of the liver (0.8 mg/kg-day) (DHHS, 1993).

Certain characteristics of exposed human populations may influence arsenic toxicity at high exposure levels. Genetic dispositions (rapid versus poor acetylators) and protein-deficient diet may decrease arsenic methylation. This can result in an increased arsenic deposition in the target organs (e.g., lung or skin).

The EPA classifies inorganic arsenic as a Group A (human) carcinogen (EPA, 1994a), based on the occurrence of increased lung cancer mortality in populations exposed primarily through inhalation, and of increased skin cancer in populations exposed through consumption of drinking water containing high arsenic concentrations. The EPA currently is reviewing the cancer slope factor (SF) in light of recent data suggesting arsenic ingestion may result in increased cancers in internal organs as well as skin cancers. Figure 5.1 summarizes the health effects of exposure to arsenic as a function of dose.

5.1.2 Boron

Absorption

Although boron can be measured in blood following ingestion and recovered in urine, no quantitative studies on absorption have been found (DHHS, 1992a).

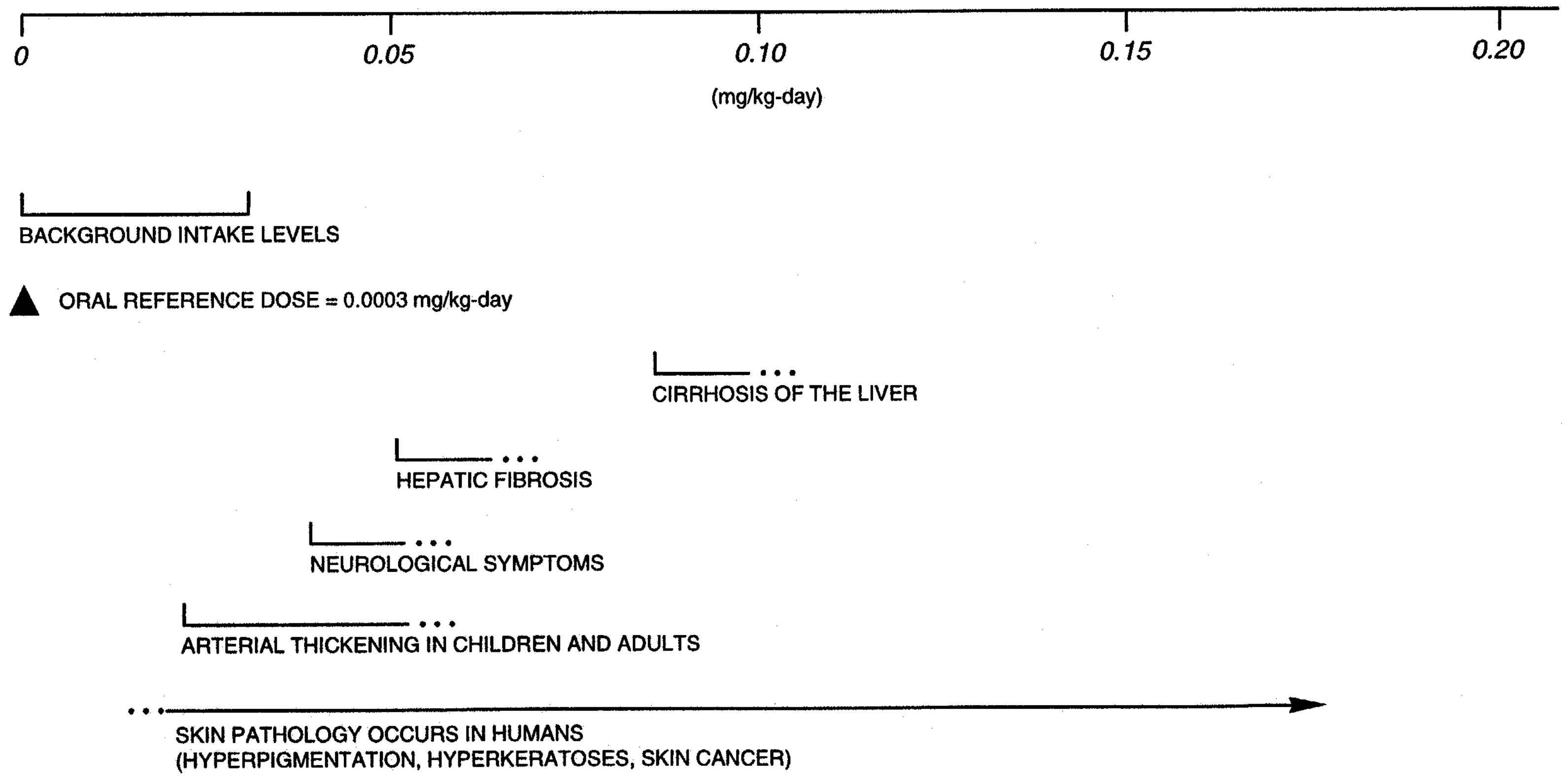


FIGURE 5.1
ARSENIC TOXICITY RANGES

MAC: SITE/LKV/BRA/ARSENTOX

Tissue accumulation and clearance

No studies on tissue distribution and clearance of boron are available. Observations of liver, kidney, and neurological damage following ingestion of very high doses of boron (more than 500 mg/kg-day) suggest these tissues receive some concentration of an ingested dose.

One study of six male volunteers indicated urine was the main route of excretion, with 93 percent of the ingested dose recovered in urine within 96 hours of ingestion. Boric acid could be detected in the patients' urine 23 days after a single ingestion (DHHS, 1992a).

Environmental sources of boron

A major source of boron in water is the natural weathering of boron-containing rocks. Environmental releases of boron from industrial sources include fertilizers, herbicides, glass production, coal-burning power plants, and copper smelters. Although the average surface water concentration in the United States is 0.1 mg/L, concentrations as high as 360 mg/L have been reported in areas of California with boron-rich rock formations (DHHS, 1992a). The concentration of boron in drinking water is not generally reported, but studies of boron in tap water and public water supply systems have shown 99 percent contain less than 1 mg/L. The maximum reported value was 3.28 mg/L (DHHS, 1992a).

Plants take up boron from soil, and boron occurs in many foods (primarily fruits and vegetables). The estimated average daily intake from food and water ranges from 10 to 25 mg per day (0.14 to 0.35 mg/kg-day).

Accidental home exposures to boron occur primarily through ingestion of insecticides used to control cockroaches. Boron also can be absorbed from cosmetics or medical preparations through mucous membranes or damaged skin.

Toxicity of boron

No epidemiological studies are available on human chronic ingestion of boron. Some data are available on acute toxicity in humans. These data are from infants ingesting formula prepared with boric acid (DHHS, 1992a). The dose ranged from 4.5 to 14 grams. Based on the body weights of two infants who died, the lethal acute doses ranged from 505 to 765 mg/kg-day. Doses of 184 mg/kg-day and greater produced nausea, persistent vomiting, diarrhea, and colicky abdominal pain. Doses greater than 500 mg/kg-day could damage the liver and kidneys and produce skin lesions. Neurological effects, including headache, tremors, restlessness, and convulsions followed by weakness and coma, also were reported in some of the infants who ingested high levels of boron (more than 500 mg/kg-day). In adults, ingestion of 241 or 895 mg/kg in suicide attempts produced only vomiting, which resolved in 24 to 96 hours. In one case, extensive redness and irritation of the skin occurred following ingestion of 14 grams of boron.

Chronic toxicity data are available only from animal studies. In both rats and dogs, the primary toxicity was observed in the reproductive organs. Male dogs showed testicular atrophy and no spermatogenesis following 38 weeks of ingesting 29 mg/kg-day of boron. Following 2 years of dietary boron intake of 58.5 mg/kg-day, rats showed a significant decrease in both testicular weight and in the testes-to-body-weight ratio, as well as other histological changes in testicular and seminiferous tubules. Rats in this study also showed decreased brain weight and brain-to-body-weight ratios. The next lower doses tested showed no adverse effects in either species: 8.8 mg/kg-day for dogs and 17.5 mg/kg-day for rats. Intermediate doses were not tested for either species. However, a lifetime study in mice given 8.1 mg/kg-day also showed no effects.

Because no human data are available to evaluate the toxicity of chronic oral boron ingestion, the EPA has applied a standard uncertainty factor of 2 orders of magnitude to allow for extrapolating from the no-observed-adverse-effect level (NOAEL) in animals (8.8 mg/kg-day in dogs) to humans. This correction results in an oral RfD of 0.09 mg/kg-day.

No human cancer has been associated with boron exposure, and only one negative study in mice has evaluated the carcinogenicity of boron. Therefore, no conclusive evaluation of boron can be made with respect to cancer. Boron toxicity is summarized in Figure 5.2 as a function of dose.

5.1.3 Chloride

Absorption

Chloride is rapidly and fully absorbed from the gastrointestinal tract.

Tissue accumulation and clearance

The chloride concentration in the human body is approximately 2000 mg/kg of fat-free body mass in newborns and 1920 mg/kg in adults (National Research Council, 1980). It occurs in plasma at concentrations of 96 to 106 milliequivalents per liter (meq/L) (3408 to 3763 mg/L), and in more concentrated form in cerebrospinal fluid and gastrointestinal secretions (National Research Council, 1989). Its concentration in most cells is low. A daily chloride turnover in adults (intake/output) ranges from 3018 to 8875 mg. Chloride excretion is mainly urinary, with appreciable amounts also excreted in feces, sweat, and tears.

Environmental sources of chloride

Dietary chloride comes almost entirely from sodium chloride (National Research Council, 1989). Much smaller amounts are supplied from potassium chloride.

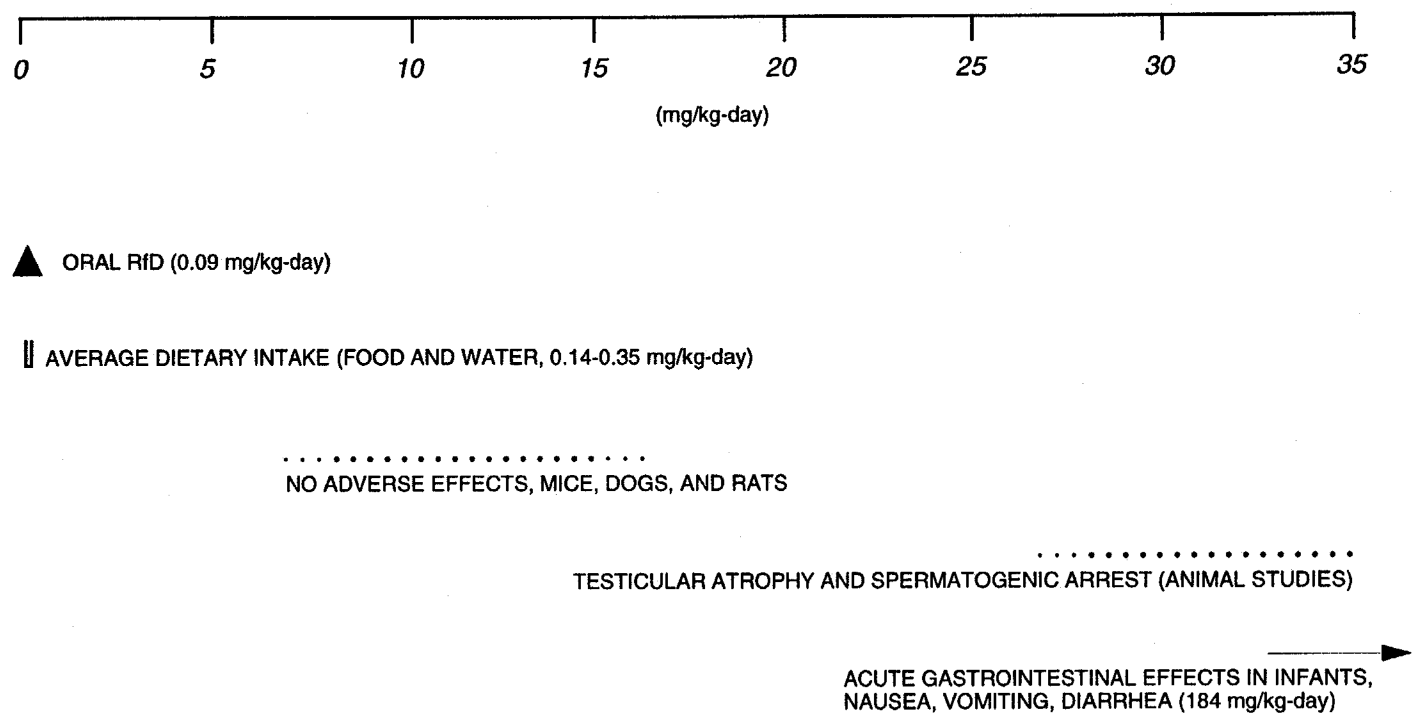


FIGURE 5.2
BORON TOXICITY RANGES

Thus, dietary sources of chloride are essentially the same as those for sodium, and processed foods are the major source of chloride. Rich sources of chloride are salt, cereals, breads, dried skim milk, teas, eggs, margarine, salted butter, bacon, ham, corn beef, canned meats, seafood, vegetables, salted snack food, and olives. Dietary chloride intakes varies largely with salt intake. Estimates range from 2400 to 14,400 mg chloride per day from sodium chloride (equivalent to 34 to 206 mg/kg-day for a 70-kg man).

Human milk contains 11 meq/L of chloride (391 mg/L), and a similar level has been suggested by the American Academy of Pediatrics for infant formulas (National Research Council, 1989).

Chloride is found in all natural waters. Surface waters contain only a few milligrams of chloride per liter, whereas streams in arid or semiarid regions contain several hundred milligrams per liter, especially in drained areas where chlorides occur in natural deposits or are concentrated from soils through evaporation processes. Contamination with sewage increases the chloride content of river water. Ground water usually contains larger quantities of chloride than surface water. Some public supply wells may contain 100 mg chloride per liter (about 3 mg/kg-day, assuming a daily consumption of 2 L and a 70-kg body weight) (National Research Council, 1980).

A typical chloride concentration in drinking water of about 21 mg/L would contribute 0.6 mg chloride per kilogram per day (assuming 2 L per day consumption rate and 70-kg body weight). This would be about 2 percent of the lower estimates of total chloride intake.

The recommended drinking water limit for chloride is set at 250 mg/L (equivalent to 7 mg chloride per kilogram per day, for a 70-kg man and assuming consumption of 2 L of water per day) (National Research Council, 1980). This amount of chloride in drinking water causes a salty taste in water, which may be objectionable to many people. The taste threshold for the chloride anion in water varies from 210 to 310 mg/L.

Toxicity of chloride

Chloride is the most important inorganic anion in the extracellular fluid compartment. It is essential in maintaining fluid and electrolyte balance and is a necessary component of gastric juice.

Chloride loss from the body generally parallels sodium loss. Thus, conditions associated with sodium depletion (e.g., heavy, persistent sweating, chronic diarrhea or vomiting, trauma, and renal disease) will also cause chloride loss, resulting in hypochloremic metabolic alkalosis.

Although the essentiality of the chloride ion is generally recognized, no recommended dietary allowance (RDA) has been determined. The estimated minimum chloride requirement for good health ranges from 180 mg per day for infants to 750 mg per day for adults (National Research Council, 1989).

The toxicity of salts containing the chloride ion depends primarily on the characteristic of the cation (National Research Council, 1980; 1989).

Large amounts of chloride intake may cause an increased chloride plasma concentration and a decreased bicarbonate plasma concentration, with an acidifying effect. This results in hyperkalemic metabolic acidosis (National Research Council, 1980). When metabolic acidosis develops, potassium leaves cells and is excreted by the kidney (Brater, 1992).

The only known dietary hyperchloremia results from water-deficiency dehydration. Sustained ingestion of high chloride levels (as salt) is associated with hypertension in sensitive individuals and in animal models. Although the level of chloride attributable to hypertension has not been determined, it may be estimated based on the level of sodium intake (from sodium chloride) causing hypertension in approximately 15 percent of adults (Freis, 1976). This indirectly estimated amount of chloride presumably associated with hypertension in sensitive individuals would be in the range of 36 to 180 mg/kg-day. Figure 5.3 summarizes chloride toxicity as a function of dose.

5.1.4 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 (Casarett and Doull, 1986; 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 is also 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, it is clear that a reducing agent such as ascorbic acid increases the absorption of iron in food (National Research Council, 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 decreases in the presence of high levels of phosphate, cobalt, copper, and zinc (Elinder, 1986). Excess manganese can significantly decrease

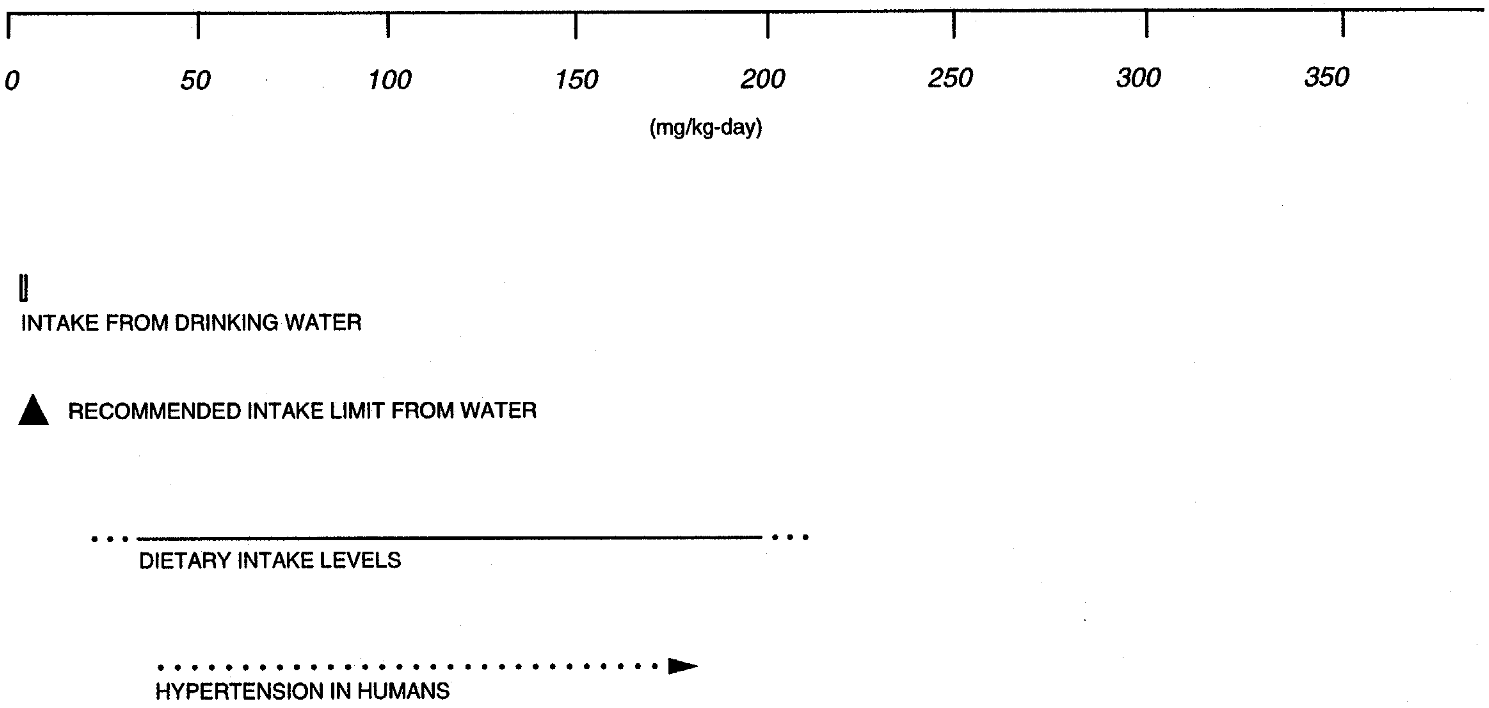


FIGURE 5.3
CHLORIDE TOXICITY RANGES

MAC: SITE/LKV/BRA/CHL-GEN

iron absorption by impairing hemoglobin regeneration in the blood (National Research Council, 1980). In the contaminated ground water at the Lakeview site, iron is present predominantly in the form of ferric ion (Table 3.7).

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 I (Casarett and Doull, 1986).

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 per 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 elemental iron per day is eliminated through the feces, about 0.1 to 0.3 mg per day in urine, and the remainder through normal dermal losses in sweat, hair, and nails. Based on these elimination rates, the biological half-life of iron in the body is 10 to 20 years (Elinder, 1986).

Environmental sources of iron

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

Average daily intakes of iron range from 9 to 35 mg per day (0.1 to 0.5 mg/kg-day) (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 (National Research Council, 1980).

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 daily consumption of 2 L of water by a 70-kg (body weight) adult, this range would result in an intake of 0.0003 to 0.03 mg/kg-day of iron from drinking water.

The RDA for iron is 10 mg (approximately 0.14 mg/kg-day) for adult males and 18 mg (approximately 0.25 mg/kg-day) for females of reproductive age (National Research Council, 1980).

Toxicity of iron

Iron intoxication is most frequent in children aged 1 to 3 years who eat iron supplements formulated for adults in the form of ferrous sulfate tablets with candy-like coatings. Severe poisoning may occur in children following ingestion of more than 0.5 gram (approximately 22 mg/kg) of iron or about 2.5 grams (110 mg/kg) of ferrous sulfate. This acute iron poisoning has occurred in children who ingested as few as six 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 (Casarett and Doull, 1986). 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 be delayed effects (Elinder, 1986).

Long-term iron intake 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 male) 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 pigmentation of the skin 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 (Casarett and Doull, 1986) (Figure 5.4).

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 (Casarett and Doull, 1986).

5.1.5 Manganese

Absorption

Following ingestion, manganese absorption is homeostatically controlled: the rate of absorption depends on both the amount ingested and tissue levels of manganese. For adult humans, approximately 3 to 4 percent of dietary manganese is absorbed (Saric, 1986). Manganese can be absorbed following exposure by inhalation, ingestion, and dermal contact. In humans, available data indicate that only 3 percent of an ingested dose of manganese chloride is absorbed (Mena et al., 1969). The rate of absorption is influenced by iron and other metals. 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

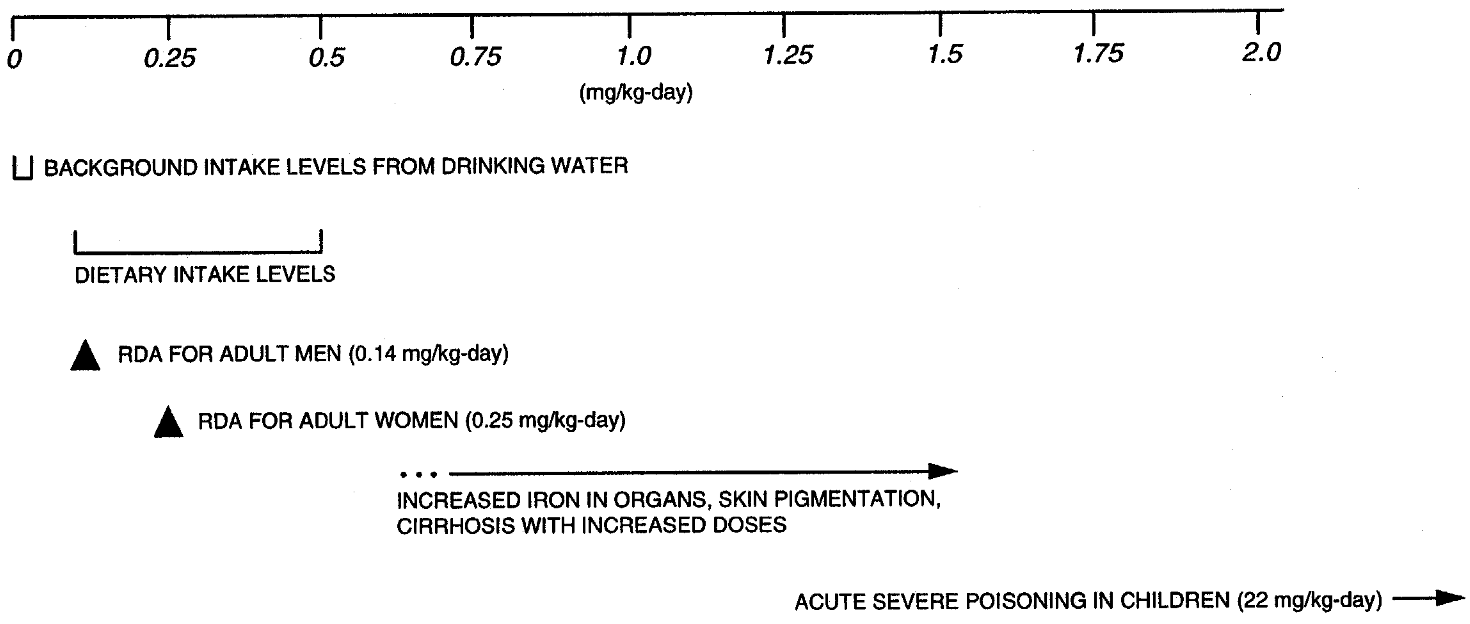


FIGURE 5.4
IRON TOXICITY RANGES

phosphorus have been shown to increase the requirements for manganese in several species (Lönnerdal et al., 1987).

Tissue accumulation and clearance

Manganese is widely distributed throughout the body. The highest concentrations are found in the liver and kidney and, to a lesser extent, the hair. 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 brain than from other tissues (Casarett and Doull, 1986). 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 (National Research Council, 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 (Casarett and Doull, 1986; 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 range from 1 to 7000 mg/kg, with an average of 600 to 900 mg/kg. Mining and natural geological background variation can contribute to this variability. Manganese bioaccumulates in marine mollusks up to 12,000-fold, and there is evidence for 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 with cereals and rice as main food sources, the intake will be higher compared to areas where meat and dairy products make up a larger part of the diet. The average daily intake has been estimated as 2.0 to 8.8 mg per day (0.03 to 0.13 mg/kg-day) (EPA, 1994a), but intakes as high as 12.4 mg (about 0.2 mg/kg-day) have been reported in countries with high cereal intake (Saric, 1986).

Drinking water generally results in an intake of less than 0.2 mg (0.003 mg/kg-day), 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 NOAEL for drinking water is set 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 RfD for drinking water is 0.005 mg/kg-day. 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 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).

Manganese inhalation 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 may occur when exposure ceases (DHHS, 1992b). Metal-chelating agents are ineffective in treatment, but L-dopa has been effective in treatment (Casarett and Doull, 1986).

Limited information is available on the effects of manganese ingestion. Because the effects from drinking water seem to differ from those from food sources, only studies on water consumption are 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) 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 intoxicated people. A Greek study was conducted of more than 4000 individuals who drank water with manganese concentrations varying from 0.081 to 2.3 mg/L (estimated intake at 2 L per day for a 70-kg individual ranges from 0.002 to 0.07 mg/kg-day). Those who drank 0.007 to 0.07 mg/kg-day showed varying degrees of neurological effects but no effects were apparent in individuals drinking less than 0.005 mg/kg-day (Kondakis et al., 1989).

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 clear more rapidly. 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 5.5 summarized the toxicity of manganese.

5.1.6 Molybdenum

Absorption

Molybdenum absorption in the gastrointestinal tract depends on the species. Both inorganic and hexavalent forms such as molybdenum trioxide, sodium molybdate, and ammonium molybdate are readily absorbed from both food and water, whereas molybdenite is not. Human absorption rates of 40 to 70 percent have been observed for soluble forms of molybdenum (Robinson et al., 1973; Alexander et al., 1974).

Tissue accumulation and clearance

In humans, the highest concentrations of molybdenum occur in the liver, kidney, and adrenals (Casarett and Doull, 1991). With normal dietary intake, molybdenum levels in the body slowly increase until approximately age 20, then begin to decline steadily. The principal excretion route in humans is the urine. Human studies indicate the biological half-life in humans is considerably longer than in animals and may be as long as 2 weeks (Rosoff and Spencer, 1964).

Environmental sources of molybdenum

Natural molybdenum occurs in combination with other metals, including uranium, lead, iron, cobalt, and calcium. Native soil concentrations can vary by as much as 2 orders of magnitude (from 0.1 mg/kg to 10 mg/kg), leading to large variations in molybdenum concentrations in plant materials. Natural concentrations in ground water have been reported from 0.00011 mg/L to 0.0062 mg/L. Human dietary intake of molybdenum has been estimated at 0.05 to 0.24 mg per day (0.002 to 0.004 mg/kg-day). The contribution of drinking water is estimated to range from 0 percent to 95 percent. The nutritional range of intake for molybdenum is 0.0015 to 0.0054 mg/kg-day. No symptoms of molybdenum deficiency have been reported in humans. Nonetheless, molybdenum is an essential trace element that functions as a necessary constituent of several enzymes, including xanthine oxidase (which is involved in the metabolism of uric acid) and nitrate reductase (Friberg et al., 1986).

Toxicity of molybdenum

Acute toxic effects of molybdenum have not been reported. No adverse health effects have been reported with chronic intake of less than 0.008 mg/kg-day of molybdenum. The primary toxicity of molybdenum is related to its interactions with copper and sulfur, leading to altered excretion patterns for these elements.

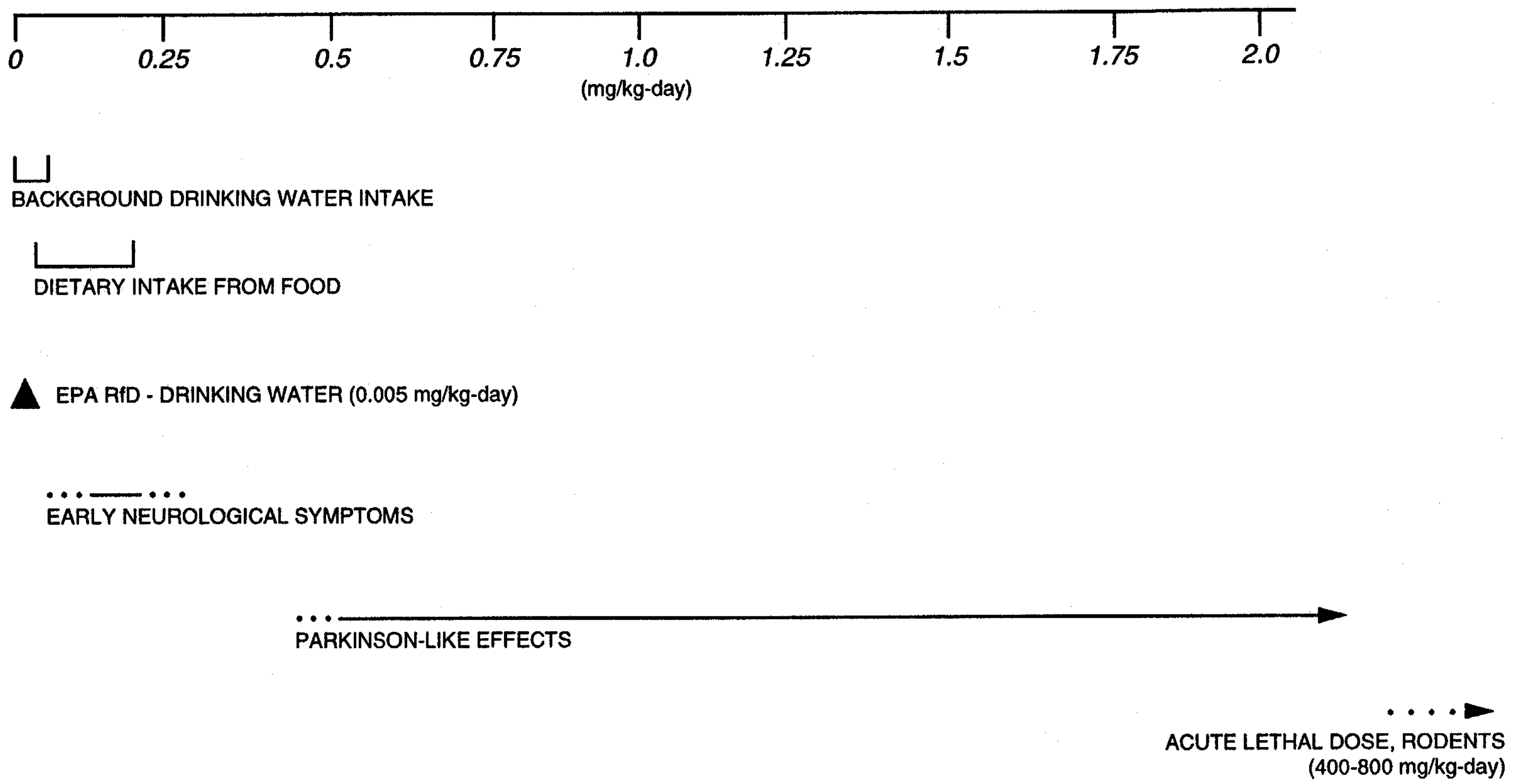


FIGURE 5.5
MANGANESE TOXICITY RANGES

MAC: SITE/LKV/BRA/MAN-GEN

Increased levels of molybdenum also increase the levels of xanthine oxidase, which is responsible for the production of uric acid. High levels of uric acid can accumulate in joints and lead to symptoms of gout and other joint disorders.

Intake of 0.008 to 0.022 mg/kg-day of molybdenum can produce mineral imbalance as a result of increased copper excretion. Copper excretion has been reported to double with molybdenum intakes at the upper end of this range. Copper is an essential nutrient important in many metabolic pathways, including the synthesis and function of hemoglobin. A copper deficiency resulting from excess excretion will impair the oxygen-carrying capacity of the blood, and severe copper deficiencies can lead to hypochromic microcytic anemia. In humans, gout-like symptoms and joint deformities have been reported in regions of Russia where elevated molybdenum concentrations in soil and subsequent increased molybdenum concentrations in food would lead to molybdenum intakes in the range of 0.14 to 0.21 mg/kg-day. Figure 5.6 summarizes these health effects as a function of dose.

5.1.7 Nickel

Absorption

Absorption studies in humans report that 27 percent of inorganic nickel (administered as nickel sulfate) was absorbed when it was administered in drinking water, whereas only 0.7 percent was absorbed when it was given in food. In a separate study, the bioavailability of nickel, as measured by serum nickel levels, increased by 80 micrograms per liter (µg/L) after 3 hours in fasted individuals who ingested nickel sulfate in drinking water; levels were not elevated in individuals who ingested nickel in food (DHHS, 1991). Other human studies show that generally less than 10 percent of ingested nickel is absorbed by the gastrointestinal tract. This finding is consistent with studies reporting from 1 to 10 percent oral absorption in several animal species (Friberg et al., 1986). Absorbed nickel is transported in the plasma bound to serum albumin and various organic ligands, amino acids, or polypeptides (Casarett and Doull, 1986). Nickel has been found to affect gastrointestinal absorption of iron, but only when iron was administered as ferric sulfate (Fe^{+3}) (DHHS, 1991).

Tissue accumulation and clearance

In humans, serum nickel levels reportedly peak 2.5 to 3 hours after ingestion of nickel sulfate. In individuals who accidentally drank water contaminated with nickel sulfate and nickel chloride, nickel mean serum half times were 60 hours. No human data were located regarding nickel levels in specific tissues or organs following ingestion of nickel compounds.

In animals, various nickel compounds administered orally distributed primarily to the kidneys, with significant nickel levels also found in the liver, heart, lung, fat, peripheral nervous tissues, and brain. Increased nickel levels also were found in

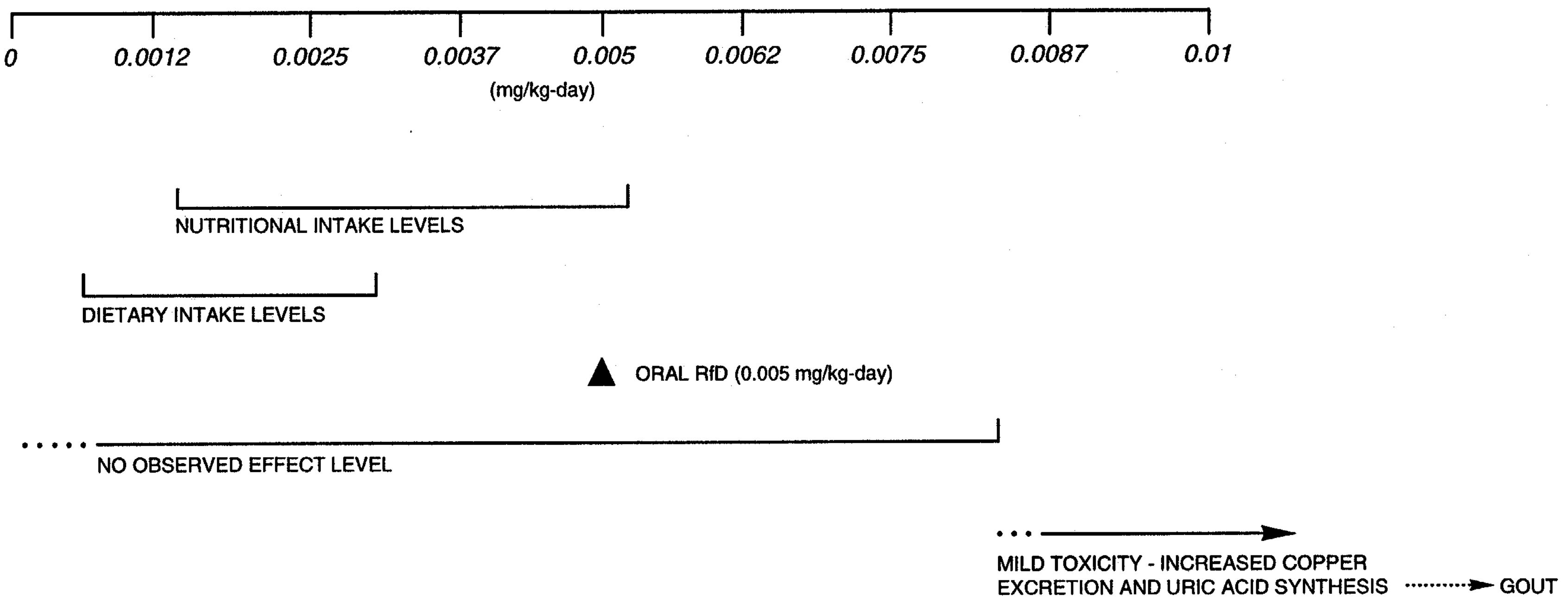


FIGURE 5.6
MOLYBDENUM TOXICITY RANGES

MAC: SITE/LKV/BRA/MOL-GEN

the fetuses of animals exposed orally to nickel compounds, suggesting that nickel crosses the placental barrier (DHHS, 1991).

In humans, most ingested nickel is excreted in the feces, due to limited absorption. Nickel absorbed in the gastrointestinal tract is excreted in the urine. Excretion of a given dose of nickel is nearly complete in 4 or 5 days (Casarett and Doull, 1986), with approximately 26 percent of the dose excreted in the urine and the remainder eliminated in the feces (DHHS, 1991).

Environmental sources of nickel

Exposure to nickel can occur through inhalation of ambient air and tobacco smoke and by ingesting water and food. Most intake occurs through the diet (DHHS, 1991). In grains, fresh weight nickel concentrations reportedly range from 0 to 6.45 µg per gram. In vegetables and fruits, levels range from 0 to 2.59 µg per gram and in seafood from 0.3 to 107 µg per gram. Average daily dietary intake is approximately 165 µg (Friberg et al., 1986).

The drinking water daily intake averages 2 µg (DHHS, 1991). Nickel is not commonly present at harmful levels in ground water. In a nationwide ground water survey, 97 percent of all samples (total of 2053 samples) contained less than 20 µg/L of nickel and 80 percent had less than 10 µg/L, although in areas near nickel mining operations, levels as high as 200 µg/L have been reported (Friberg et al., 1986).

Toxicity of nickel

Acute exposure to high nickel levels in drinking water (1-day duration) reportedly produced symptoms of gastrointestinal distress, including nausea, abdominal cramps, diarrhea, and vomiting. The estimated exposure dose of 7.1 to 35.7 mg/kg also produced transient hematological effects, muscular pain, transient increases in urine albumin, and neurological effects (giddiness and weariness).

The effects of chronic nickel ingestion in humans are not well documented. In laboratory animals (dogs and rats), the primary effects of long-term dietary administration of nickel sulfate were decreases in body weight and changes in organ weights. Low hematocrit and polyuria also were reported for dogs (DHHS, 1991). Rats appear to be the more sensitive of the two species. The lowest nickel dose of 35 mg/kg-day administered to rats in water by gavage resulted in decreased body and internal organ weights, as determined in a subchronic toxicity study (EPA, 1994a).

A susceptible population may exhibit a different or enhanced response to nickel than will most people who may be exposed to the same level of nickel in the environment. Chemical exposure history, genetic makeup, developmental state, health, and nutritional status can affect the detoxification and excretory processes (mainly hepatic and renal). For these reasons, it is expected that the elderly (with declining function) and the youngest of the population (with immature and developing organs) generally will be more vulnerable to toxic substances than healthy adults.

Exposure to nickel may lead to sensitization. Available data indicate oral exposure to relatively low levels of nickel may elicit allergic dermatitis in sensitized (previously exposed to nickel) individuals (DHHS, 1991). Epidemiologic studies indicate that blacks have a higher sensitivity than whites and that women of either racial group have higher reaction rates (DHHS, 1991). The incidence of allergic reactions may be higher in women because they wear more metal jewelry than men. The suggested threshold for elicitation of the response is approximately 0.007 mg/kg-day following oral challenge. Cross-sensitivity of nickel and other metals (e.g., cobalt) has also been reported (DHHS, 1991).

For the rat, a NOAEL of 100 parts per million (ppm) of nickel in diet (5 mg/kg-day) was reported. The EPA chronic RfD was derived based on this NOAEL for human oral exposure to nickel. Considering the uncertainties with interspecies extrapolation and protection of sensitive populations, an oral RfD of 0.02 mg/kg-day has been developed for nickel (EPA, 1994a). This value represents a chronic daily ingestion dose that would not be expected to produce adverse health effects in humans. Figure 5.7 summarizes these health effects as a function of dose.

5.1.8 Sodium

Absorption

Sodium is rapidly and fully absorbed from the intestinal tract. The skin and lungs also absorb sodium rapidly by simple diffusion and ion exchange. Sodium travels in the blood, where it ultimately passes through the kidneys. The kidneys filter out all the sodium; then the adrenal hormone aldosterone maintains sodium concentration in the bloodstream at the amount needed (National Research Council, 1980).

Tissue accumulation and clearance

Sodium is the major extracellular ion. The sodium ion is essential to regulating the acid-base balance and is an important contributor to extracellular osmolarity. It is an essential constituent in the electrophysiological functioning of cells and is required for the propagation of impulses in excitable tissues. Furthermore, sodium is essential for active nutrient transport, including the active transport of glucose across the intestinal mucosa. About 30 to 40 percent of the body's sodium is thought to be stored on the surfaces of the bone crystals, where it is easy to recover if the blood sodium level drops.

Sodium excretion is mainly urinary, with appreciable amounts also excreted in feces, sweat, and tears (Venugopal and Luckey, 1978). Mammalian renal

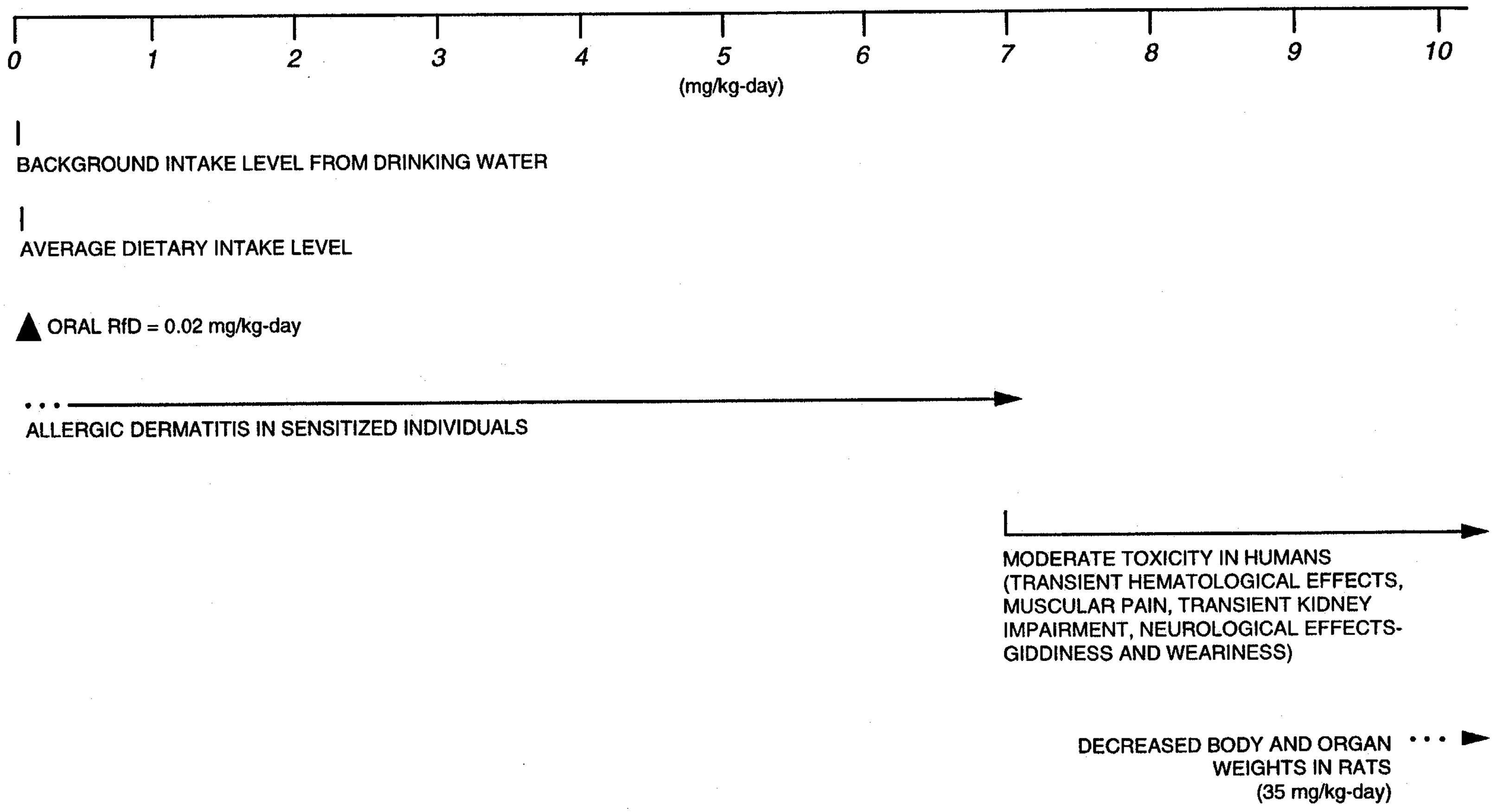


FIGURE 5.7
NICKEL TOXICITY RANGES

excretion of sodium is a two-phase process involving glomerular filtration and reabsorption in proximal tubules; of about 600 grams of sodium involved in 24-hour glomerular filtration, approximately 99.5 percent is reabsorbed in adult humans. A homeostatic mechanism for sodium functions at the renal excretory level.

Environmental sources of sodium

The total intake of sodium is influenced mainly by the extent that salt (sodium chloride) is used as an additive to food, the inherent salt content of the foods consumed, and the intake of other sodium salts in the diet and in medications. Sodium is a natural constituent of both vegetable and animal products in varying concentrations. Other sources of sodium are medications, drinking water, cooking water, soft drinks, and alcoholic beverages.

At 2 months, infants consume approximately 300 mg of sodium a day; at 12 months, approximately 1400 mg a day. Human milk contains 161 mg/L and cow's milk contains approximately 483 mg/L (Carson et al., 1986).

No RDA has been set for sodium. The National Research Council recommendations advise limiting daily sodium intake to less than 2400 mg (34 mg/kg-day). A healthy person requires about 115 mg sodium daily (1.6 mg/kg-day), yet sodium dietary intake is estimated at 57 to 85 mg/kg-day. However, dietary sodium intake levels as high as 134 mg/kg-day have been reported (National Research Council, 1980). The American Heart Association recommends limiting sodium intake to 3000 mg daily.

The sodium content of drinking water is extremely variable. The Chemical Analysis of Interstate Carrier Water Supply Systems reports sodium concentrations in 630 systems ranged from less than 1 to 402 mg/L (resulting in drinking water ingestion rates from less than 0.03 to 11 mg/kg-day), with 42 percent greater than 20 mg/L and 3 percent over 200 mg/L (Carson et al., 1986).

Toxicity of sodium

In healthy adult males, acute toxicity symptoms of sodium chloride accompanied by visible edema may occur with an intake as low as 35 to 40 grams of salt per day (20 to 23 mg sodium per kilogram per day, because sodium is 39 percent of the weight of sodium chloride) (Meneely and Battarbee, 1976). Venugopal and Luckey (1978) report the mean lethal dose of sodium for humans is 3230 mg/kg.

Epidemiological studies indicate long-term excessive sodium intake is one of many factors associated with hypertension in humans. A high sodium/potassium ratio in the diet may be detrimental to persons susceptible to high blood pressure. Some adults, however, tolerate chronic intake above 40 grams of sodium chloride per day (equivalent to 23 mg/kg-day) (Carson et al., 1986).

Research shows apparently critical levels of sodium ingestion that cause elevated blood pressure with age and hypertension in some people. Freis (1976) reported that with sodium intake below 227 mg per day (3 mg/kg-day for a 70-kg adult), hypertension was absent. In the range of 227 to 1591 mg per day (3 to 23 mg/kg-day for a 70-kg adult), a few cases of hypertension may appear, while in the range of 1590 to 8000 mg per day (23 to 114 mg/kg-day for a 70-kg adult), approximately 15 percent of adults would exhibit hypertension. When sodium intake rises above 8000 mg per day, hypertension may be found in about 30 percent of the population.

Because sodium chloride is present in nearly all processed and packaged foods, it is difficult to limit dietary sodium intake. In the United States, average daily intakes from dietary sources are often within a range where hypertensive effects occur. Drinking water generally contains relatively low levels of sodium; therefore, it does not contribute significantly to the total intake unless sodium is at higher than average levels in the water supply. However, people on sodium-restricted diets can obtain a significant portion of daily sodium from drinking water. Because the kidneys are the major organs involved in regulating sodium balance, individuals with compromised kidney function may be placed on a low sodium diet. In addition, other individuals may be on low sodium diets to control hypertension. Because of the high prevalence of such individuals in our society, the American Heart Association has proposed that public drinking water supplies in the United States adopt a standard of 20 mg/L sodium to protect individuals on low sodium diets (Calabrese and Tuthill, 1977). This would limit the additional intake of sodium from drinking water to approximately 0.6 mg/kg-day for a 70-kg adult. Figure 5.8 summarizes the potential sodium health effects as a function of dose.

5.1.9 Sulfate

Absorption

Sulfate absorption from the gastrointestinal tract is similar in humans and other animals. Generally, greater than 90 percent absorption has been reported for sulfate doses below 150 mg/kg, decreasing to 50 to 75 percent as the dose increases into the grams-per-kilogram range.

Tissue accumulation and retention

Ingesting high levels of sulfate results in transient increases in both blood and urine concentrations. Approximately 50 percent of a sulfate dose of 75 mg/kg is excreted over 72 hours. The urinary excretion mechanism is transport-limited and therefore can become saturated at high doses. Excess sulfate is also excreted in feces in its inorganic form. To date, no available data indicate sulfate is accumulated, even with chronic ingestion of above-normal levels. However, extremely high chronic doses apparently have not been examined in humans.

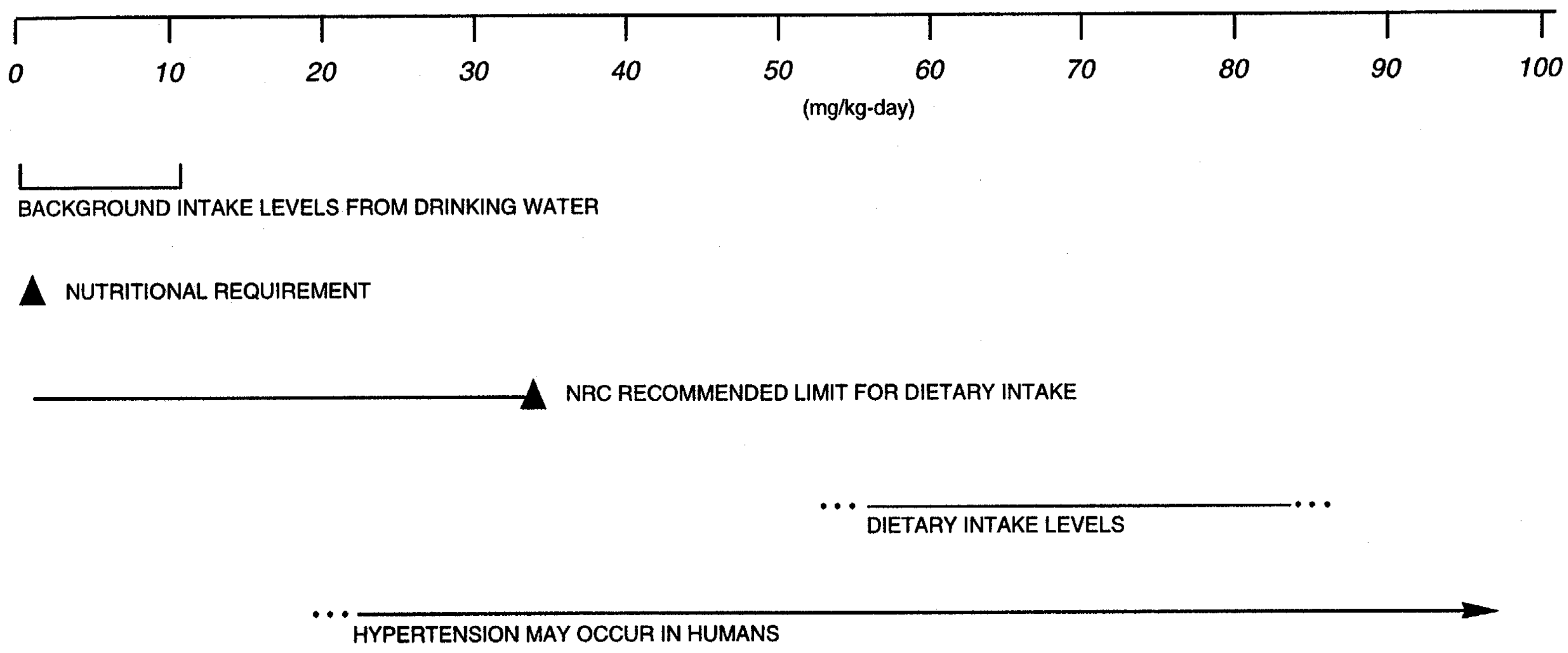


FIGURE 5.8
SODIUM TOXICITY RANGES

MAC: SITE/LKV/BRA/SODITOX

Sulfate is used in the biosynthesis of collagen, cartilage, and dentin and in the formation of 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, which enhances their excretion in the urine. Exposure to high concentrations of compounds that are conjugated with sulfate and excreted can produce a transient decrease in sulfate concentrations in plasma.

Environmental sources of sulfate

Drinking water in the western United States in 1978 showed a range of sulfate concentrations from 0 to 820 mg/L, with a mean concentration of 99 mg/L. The EPA estimates a normal sulfate intake range of 0.00023 to 0.0064 mg/kg-day from air and 0.000 to 2.9 mg/kg-day from drinking water in the concentration range found in the western United States. No estimates are available on sulfate intake from food sources.

Toxicity of sulfate

The acute and chronic effects of sulfate toxicity differ more in severity than in symptoms or mechanisms. Therefore, this discussion combines acute and chronic toxicities. There are no data to indicate a sulfate bioaccumulates with chronic exposure. Sulfate salts of magnesium and sodium are used medicinally as cathartics. The presence of 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 also is the basis of sulfate toxic effects.

Toxicity in humans primarily manifests as diarrhea; the severity of the diarrhea is dose-dependent. Chronic sulfate ingestion can result in persistent diarrhea, leading to ionic imbalances and dehydration similar to that seen with extremely high acute doses. When drinking water is contaminated with sulfate, the taste of the water may make it unpalatable and reduce consumption. However, this is not always the case. In regions such as Saskatchewan with high sulfate concentrations in the drinking water, residents adapt to the taste and find the water palatable (EPA, 1992a). A lower water intake could compound the dehydration effects of the diarrhea. Extreme dehydration can lead to death.

Infants seem to be the population most susceptible to sulfate-induced diarrhea. Also, some data indicate diabetic and elderly populations with compromised kidney function may be more sensitive than healthy adults to the effects of sulfates (EPA, 1992a). Figure 5.9 summarizes these health effects as a function of dose.

Data on sulfate toxicity are based primarily on epidemiologic studies of human adults and infants who report to hospitals with symptoms of sulfate exposure. In most cases, exposure doses have been back-calculated by sampling their drinking water. Therefore, these data do not represent well-controlled studies where dosage ranges can be readily defined.

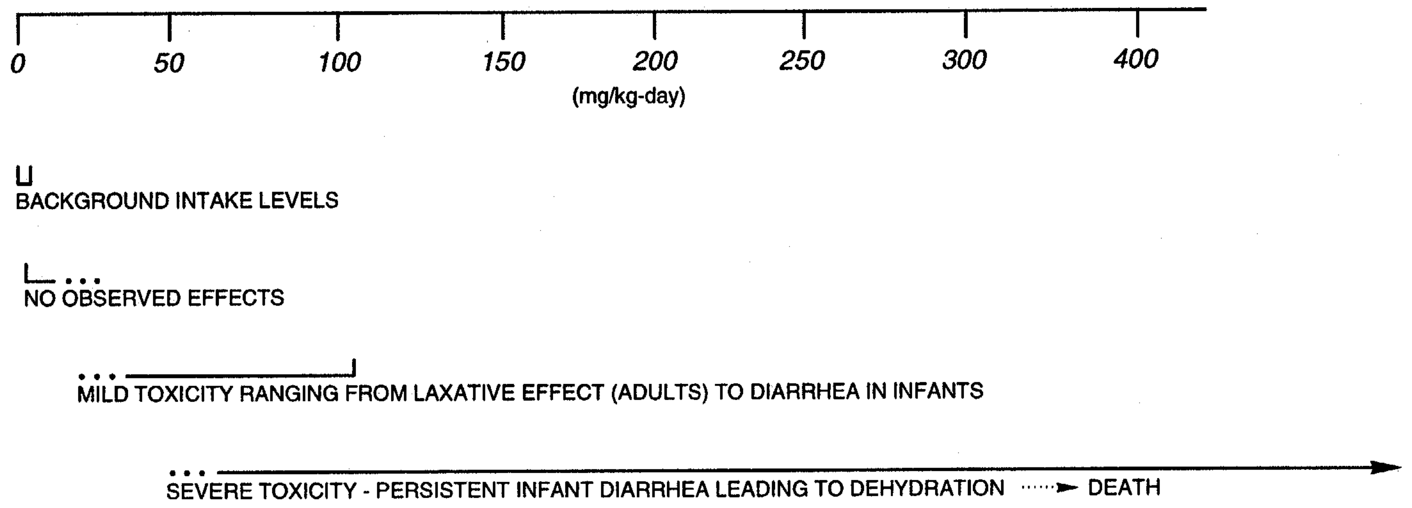


FIGURE 5.9
SULFATE TOXICITY RANGES

MAC: SITE/LKV/BRA/SULFTOX

5.1.10 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 as 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 5.10 summarizes the radioactive decay chain of uranium-238 and uranium-234. 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

Uranium absorption 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). Even with soluble compounds, only a small fraction is absorbed. Wrenn et al. (1985) have determined human gastrointestinal absorption rates of 0.76 to 7.8 percent.

Tissue accumulation and clearance

In humans exposed to background uranium, the highest concentrations 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 convert 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 in humans, more than 90 percent of uranium is excreted in the feces and not absorbed in the gastrointestinal tract. Animal studies indicate that approximately 60 percent of the small amount that is absorbed (typically less than 5 percent) is excreted in the urine within 24 hours and 98 percent 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 ppm. Uranium concentrations in ground water and surface water averaged 1 pCi/L and 3 pCi/L, respectively (NCRP, 1984). The extent of absorption from

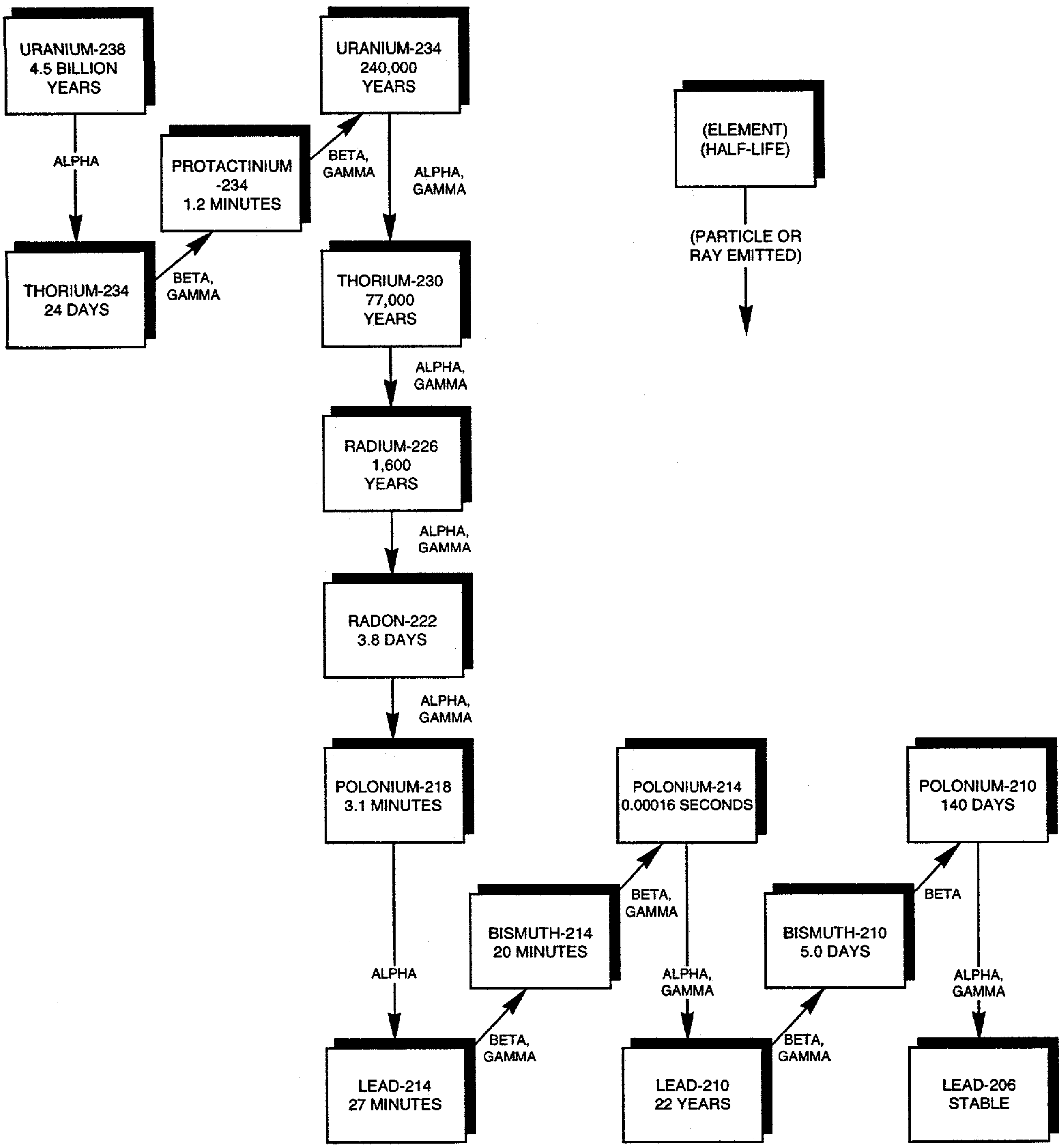


FIGURE 5.10
HALF-LIVES AND EMISSIONS FROM DECAY CHAIN OF URANIUM-238

MAC: SITE/LKV/BRA/HALFLIVES

soil into plant tissues depends on the plant species and the depth of its root system (Berlin and Rudell, 1986). Plant concentrations of uranium averaged 0.075 mg/kg of fresh plant material (Tracy et al., 1983).

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

Toxicity of uranium

Exposing the general public to natural uranium is unlikely to pose an immediate lethal threat. No human deaths have been reported that are definitely attributable to uranium ingestion; therefore, no lethal dose has been determined for humans. In animals, lethal doses of uranium ($LD_{50,23}$) have been reported 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. High doses of uranium cause complete kidney and respiratory failure.

Also, chronic toxic effects have not been reported in humans following oral exposure to uranium. Data available from populations occupationally exposed to high concentrations of uranium compounds through inhalation and data from studies on experimental animals indicate 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 are limited for the toxic effects of uranium on the human kidney.

The lowest dose of uranyl nitrate that caused moderate renal damage was given to rabbits in diet at 2.8 mg/kg-day (Maynard and Hodge, 1949). Figure 5.11 summarizes the health effects for uranium as a function of dose.

5.2 CONTAMINANT INTERACTIONS

Some information is available on potential interactions between contaminants found at UMTRA Project sites. However, discussions of potential interactions generally can be presented only qualitatively. In addition to individual physiological variables that can affect toxicity, uncertainties in interactions also result from 1) differences between the relative exposure concentrations of the different contaminants and the concentrations tested experimentally; and 2) the presence of additional ground water constituents in sufficient quantities to modify predicted toxicities, even though these constituents are not considered contaminants of potential concern for human health. Therefore, the interactions described below should be recognized as factors that can influence the

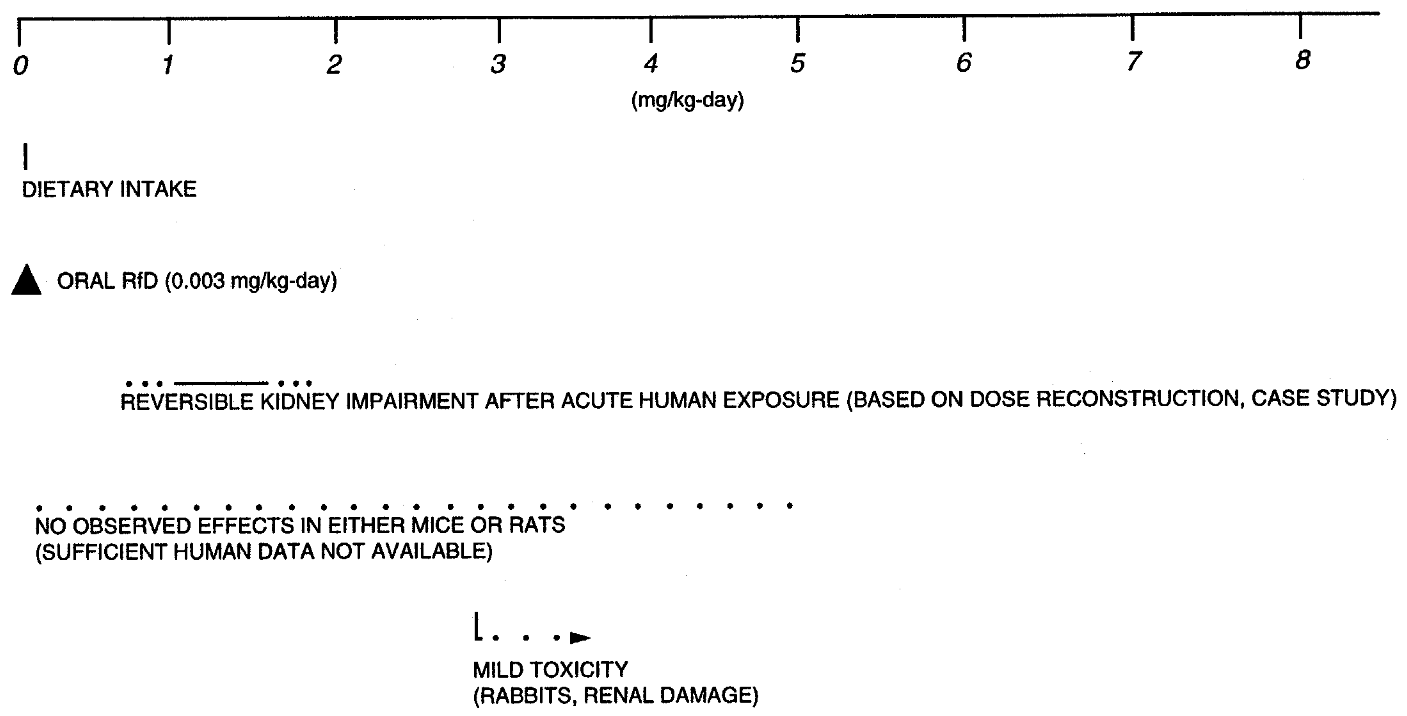


FIGURE 5.11
URANIUM TOXICITY RANGES

predicted toxicity, although the precise nature and magnitude of that influence cannot be determined.

Contaminant interactions particularly between manganese and iron, are of the primary concern at the former Lakeview processing site. Excess manganese significantly decreases iron absorption and thereby impairs hemoglobin regeneration in the blood (National Research Council, 1980). However, excess iron inhibits the manganese absorption (Saric, 1986). Even though an excess of either contaminant can inhibit the absorption of the other, the net result of a significant excess of the two in combination is not clear.

Nickel has been found to interact with other heavy metals such as iron, chromium, magnesium, manganese, zinc, and cadmium. A relationship was reported between nickel and cobalt sensitization (DHHS, 1991). It has been suggested that nickel acts as a bioligant cofactor, facilitating the gastrointestinal absorption of iron. This interaction occurs only when the iron is present in the form of a ferric ion (DHHS, 1991). In the Lakeview ground water, most of the detected iron exists in the form of ferric ions (Table 3.7). Therefore, nickel in the soil at this site may affect iron absorption.

In animal studies, iron status affected uranium absorption (EPA, 1989b). No other information on uranium interactions with other metals has been found. However, the common target organ suggests interaction with nickel in the production of kidney toxicity. The carcinogenic effects of the radionuclides and arsenic probably combine at least additively.

5.3 CONTAMINANT RISK FACTORS

Table 5.1 summarizes the most recent oral RfDs for the noncarcinogenic contaminants of concern. RfDs for chloride, iron, polonium-210, sodium, 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 evidence provided by epidemiological studies of radiation-induced cancer in humans. At sufficiently high doses, ironizing radiation acts as a complete carcinogen (both initiator and promoter), capable of increasing the probability of cancer development. However, the actual risk is difficult to estimate, particularly for the low dose and dose rates encountered in the environment. Most of the reliable data were obtained under conditions of high doses delivered acutely. It is not clear if cancer risks at low doses are dose proportional (i.e., the linear dose-response hypothesis) or if risks are greatly reduced at low doses and rates (the threshold hypothesis). A conservative assumption rejects the threshold hypothesis and assumes that any dose and dose rate add to the risk of cancer. Risk factors are published in HEAST and IRIS for correlating lifetime carcinogen intake with the increased excess cancer risk from that exposure. Table 5.2 gives the most recent cancer SFs for arsenic, polonium-210 and the uranium-234/-238 radioactive decay series.

Table 5.1 Toxicity values: potential noncarcinogenic effects

Chemical	Chronic oral RfD (mg/kg-day)	Confidence level	Critical effect/organ	RfD basis/RfD source	Uncertainty factor
Arsenic (inorganic)	3.0E-04	Medium	Keratosis, hyperpigmentation, vascular complications	Water/IRIS	3
Boron	9.0E-02	Medium	Testis	EPA, 1994a	100
Chloride	NA	NA	NA	NA	NA
Iron	NA	NA	NA	NA	NA
Manganese	5.0E-03	Not applicable	Central nervous system	Water/IRIS	30
Molybdenum	5.0E-03	Medium	Increased uric acid levels	Water/IRIS	30
Nickel (soluble salts)	2.0E-02	Medium	Decreased body weight (whole body and major organs)	Diet/IRIS	300
Polonium-210	NA	NA	NA	NA	NA
Sodium	NA	NA	NA	NA	NA
Sulfate	NA	High	Diarrhea	Water/NA	NA
Uranium (soluble salts)	3.0E-03	Medium	Kidney, decreased body weight	Diet/IRIS	1000

NA – not available.

Table 5.2 Toxicity values: potential carcinogenic effects

Parameter	Oral SF $(pCi)^{-1}$	Weight of evidence classification	Type of cancer	SF basis/ SF source
Arsenic[a] (inorganic)	1.8E+0[b]	A	Skin	Water/IRIS
Polonium-210	1.5E-10	A	Liver, kidneys, spleen	Water/HEAST
Uranium-234	4.44E-11	A	Note[c]	Water/HEAST
Uranium-238	6.2E-11	A	Note[c]	Water/HEAST

[a] SF based on oral unit risk of 5×10^{-5} (EPA, 1994a).

[b] In $(mg/kg\text{-}day)^{-1}$.

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

Two potential sources of uncertainty are the following:

- Uncertainties associated with the relationship of an applied dose (used in this assessment) and an absorbed or effective toxic dose.
- Uncertainties associated with differing sensitivities of subpopulations such as individuals with chronic illnesses, which could alter predicted responses to contaminants.

6.0 HUMAN 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 contaminant an individual takes into his or her body. Indeed, at lower levels some of the contaminants associated with the mill tailings are beneficial to health, because they are essential nutrients (e.g., calcium and magnesium). At higher levels, these same elements can cause adverse health effects. Exposure doses from using site-related contaminated ground water within the plume as drinking water are evaluated by correlating potential exposure doses to expected adverse health effects (Sections 6.1 and 6.2).

6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

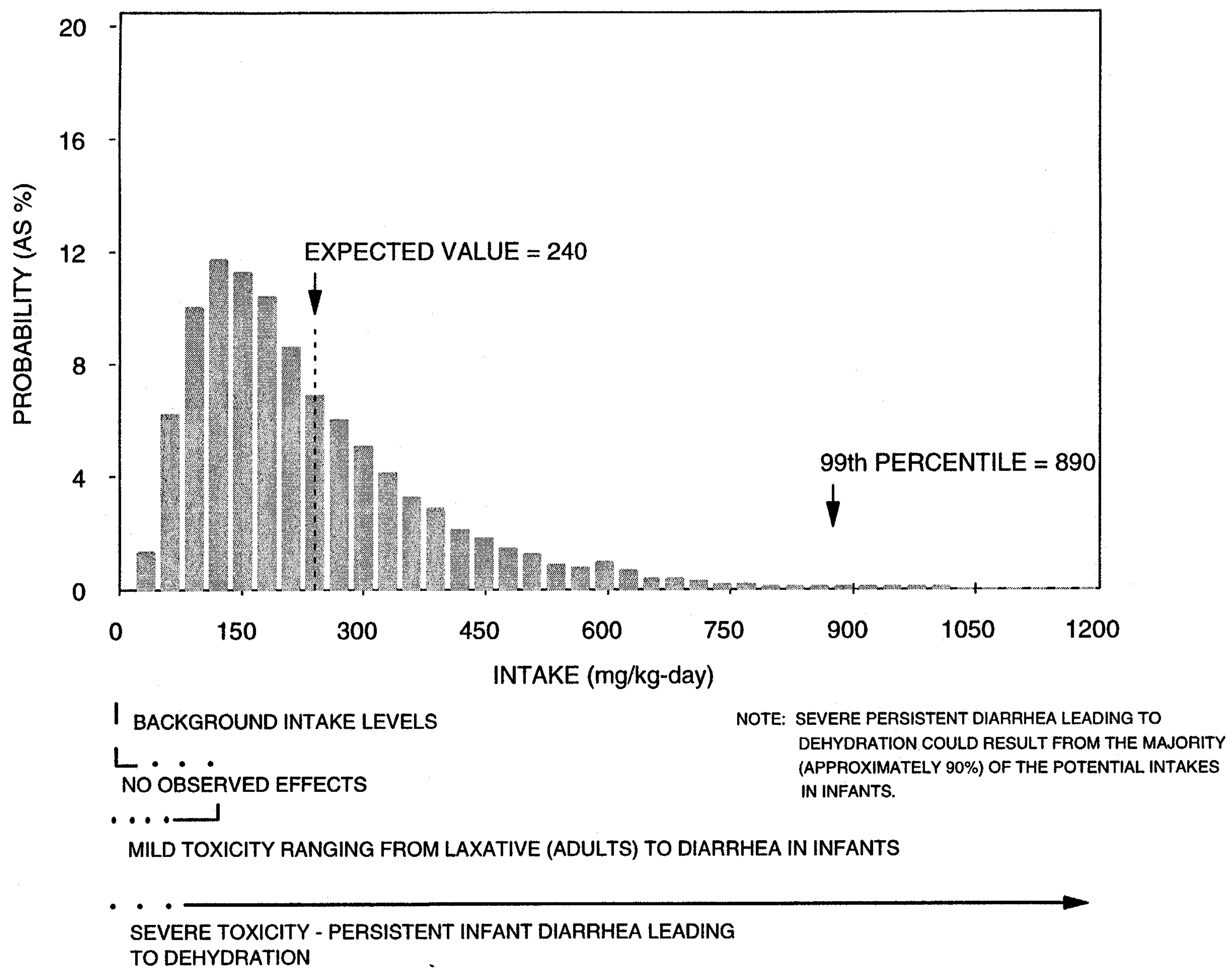
The results from the exposure assessment showing either the highest intake-to-body-weight ratios (or highest doses) or the toxicologically most sensitive group are used to evaluate potential health effects for noncarcinogens. For the contaminants of potential concern at the Lakeview site, the highest intake-per-body-weight group is children 1 to 10 years old. Because infants are the age group most susceptible to sulfate toxicity, infants are used to model potential health risks of sulfate ingestion.

The primary hazard to human health from ingesting contaminated ground water at the Lakeview site is exposure to sulfate, manganese, and to a lesser extent, chloride, iron, and molybdenum.

The potential range of sulfate exposure from drinking contaminated ground water lie in the range expected to produce mild toxicity exhibited by laxative effects in adults and severe persistent diarrhea in infants) (Figure 6.1). Diarrhea resulting from sulfate exposure could alter manganese excretion or reabsorption, although no data are available to evaluate this possibility. The toxic effects of sulfate for adults drinking this water would be less, because adults would be exposed to lower doses (per kilogram of body weight) and would be less sensitive to the effects.

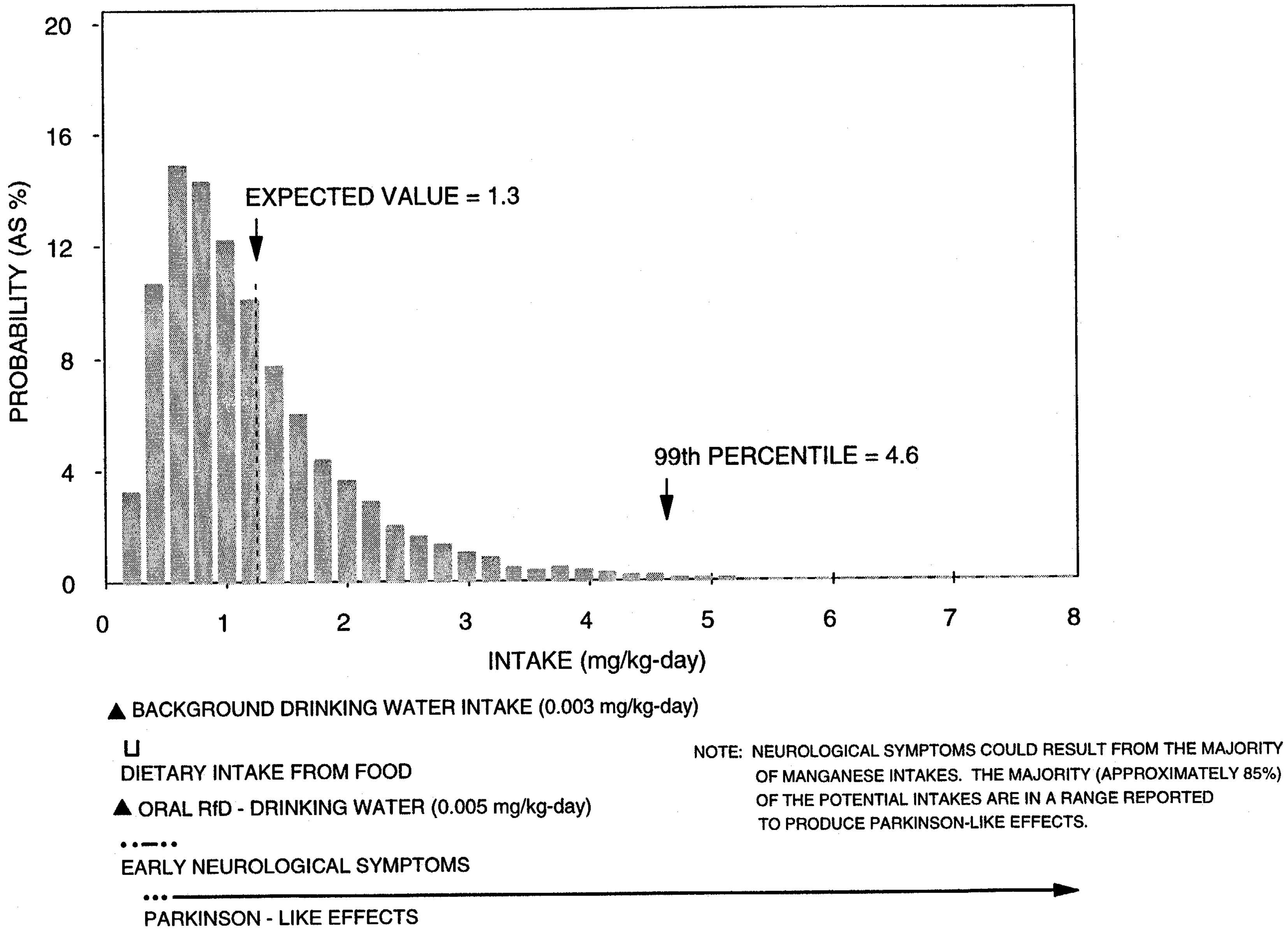
Nearly all simulated exposures for manganese are greater than the EPA oral RfD and are greater than the levels that produce mild neurological symptoms; these symptoms often are similar to early Parkinson's disease. Approximately 85 percent of the distribution falls above levels that could result in more developed Parkinson-like effects with chronic exposure through drinking water (Figure 6.2).

A large portion (approximately 45 percent) of the simulated exposure range for iron is within normal dietary intake levels (Figure 6.3). The upper 40 percent of the simulated exposure range is above the dose that could produce chronic iron toxicity, including skin pigmentation, potential disruption of liver and endocrine



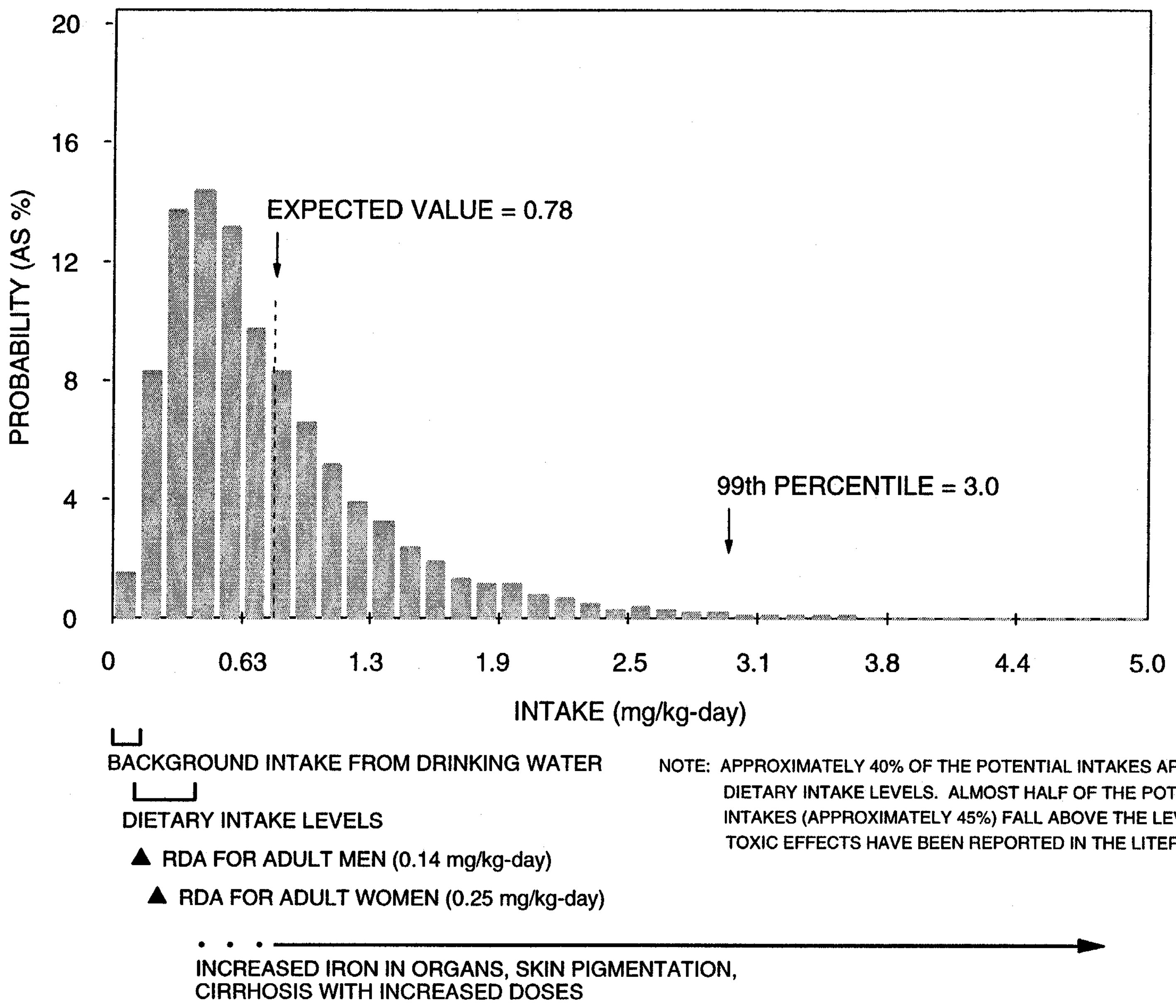
NOTE: SEVERE PERSISTENT DIARRHEA LEADING TO DEHYDRATION COULD RESULT FROM THE MAJORITY (APPROXIMATELY 90%) OF THE POTENTIAL INTAKES IN INFANTS.

FIGURE 6.1
HEALTH EFFECTS OF POTENTIAL SULFATE EXPOSURE RANGES FOR INFANTS
LAKEVIEW, OREGON, SITE



NOTE: NEUROLOGICAL SYMPTOMS COULD RESULT FROM THE MAJORITY OF MANGANESE INTAKES. THE MAJORITY (APPROXIMATELY 85%) OF THE POTENTIAL INTAKES ARE IN A RANGE REPORTED TO PRODUCE PARKINSON-LIKE EFFECTS.

FIGURE 6.2
HEALTH EFFECTS OF POTENTIAL MANGANESE EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE



NOTE: APPROXIMATELY 40% OF THE POTENTIAL INTAKES ARE WITHIN DIETARY INTAKE LEVELS. ALMOST HALF OF THE POTENTIAL INTAKES (APPROXIMATELY 45%) FALL ABOVE THE LEVEL WHERE TOXIC EFFECTS HAVE BEEN REPORTED IN THE LITERATURE.

FIGURE 6.3
HEALTH EFFECTS OF POTENTIAL IRON EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

function, and, after long-term exposure, cirrhosis of the liver and/or development of diabetes. Based on geochemical modeling for the Lakeview site, almost all of the iron in this ground water is expected to be in the trivalent form, which is less readily absorbed (and therefore less toxic) than the divalent form.

Nearly the entire range (99 percent) of the simulated sodium intake falls above the level reported to cause hypertension and, in some people, kidney impairment (Figure 6.4).

Approximately 75 percent of the simulated molybdenum exposure range falls above the daily intake range that could be associated with mild toxicity manifested primarily by mineral imbalances (increased copper and uric acid excretion) (Figure 6.5). If these exposures increased by a 20 percent contribution from milk ingestion and by a 20 percent contribution from meat ingestion, the doses still would be within the dose range expected to result in mild toxicity. Even with these increased dose contributions, however, the predicted molybdenum intake would be well below the range reported to result in gout-like symptoms.

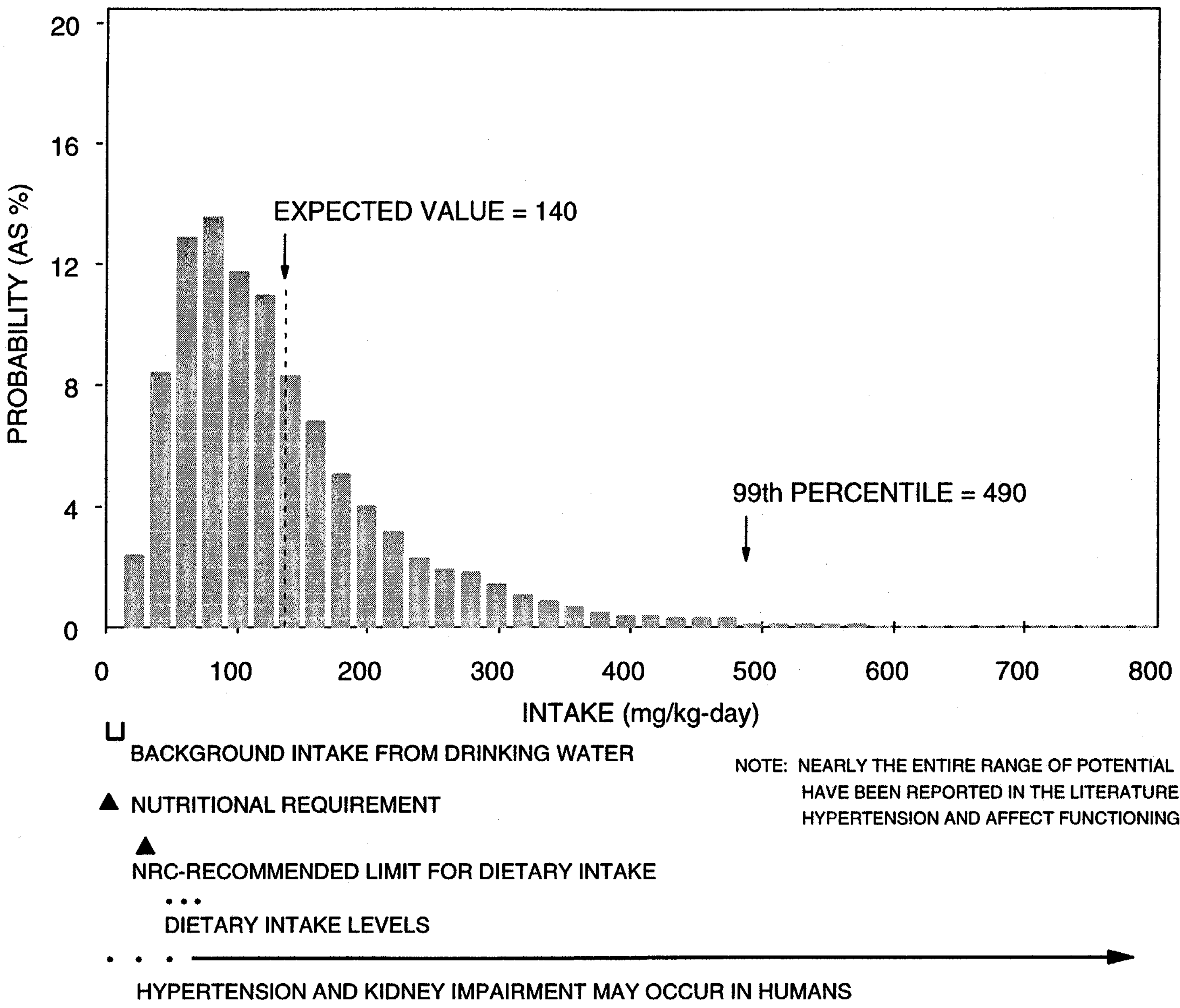
Most (approximately 90 percent) of the simulated chloride exposure range falls above the level associated with hypertension. However, 80 percent of the exposure range is within normal dietary intake levels (Figure 6.6).

All the simulated exposure range for nickel is below the oral RfD and would not be expected to adversely affect most people. For individuals sensitized to nickel, roughly 85 percent of the potential exposure range has been reported to result in allergic dermatitis, even with oral exposure (Figure 6.7).

For the noncarcinogenic effects of uranium, the entire exposure distribution falls below the oral RfD and below ranges resulting in adverse effects in animal studies (Figure 6.8). Although animal data are not always predictive of human toxicity, the lack of human data to evaluate oral toxicity of uranium at these concentrations should not suggest that toxic effects will not occur. RfDs incorporate safety and uncertainty factors and therefore are generally conservative values designed to protect human health. However, they are based on careful evaluation of existing databases; therefore, exposures that significantly exceed an RfD should be considered potential problems.

For the noncarcinogenic effects of arsenic, nearly all the simulated exposure range (from ingesting contaminated ground water) is within the range associated with normal background intake levels (Figure 6.9). A very small portion (less than 0.5 percent) of the exposure range simulated here has been associated with skin pathology and arterial thickening.

No adverse health effects have been reported for any of the simulated boron intakes, even though a large portion of the intake range exceeds the oral RfD (Figure 6.10).



NOTE: NEARLY THE ENTIRE RANGE OF POTENTIAL INTAKES HAVE BEEN REPORTED IN THE LITERATURE TO RESULT IN HYPERTENSION AND AFFECT FUNCTIONING OF THE KIDNEY.

FIGURE 6.4
HEALTH EFFECTS OF POTENTIAL SODIUM EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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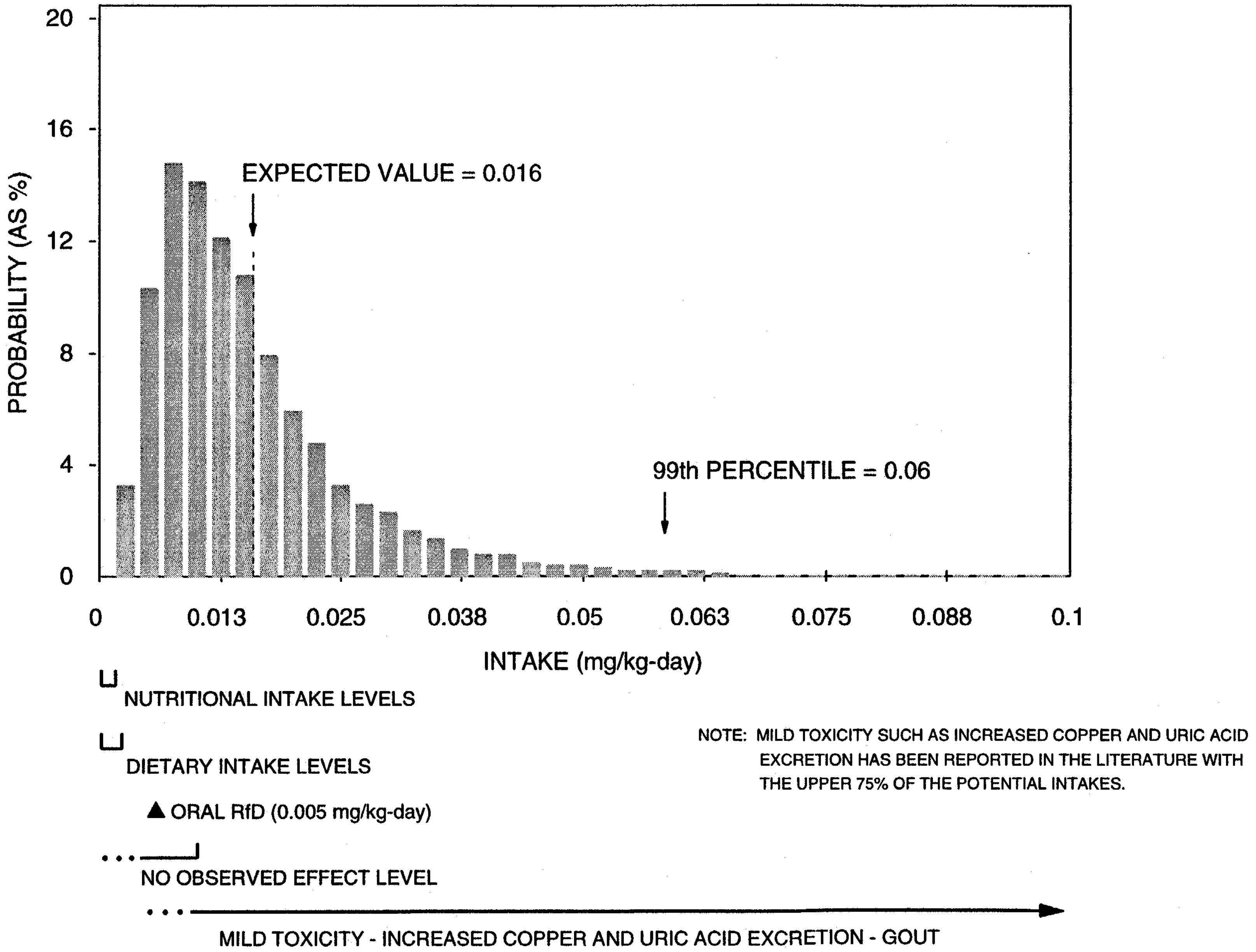

NOTE: MILD TOXICITY SUCH AS INCREASED COPPER AND URIC ACID EXCRETION HAS BEEN REPORTED IN THE LITERATURE WITH THE UPPER 75% OF THE POTENTIAL INTAKES.

FIGURE 6.5
HEALTH EFFECTS OF POTENTIAL MOLYBDENUM EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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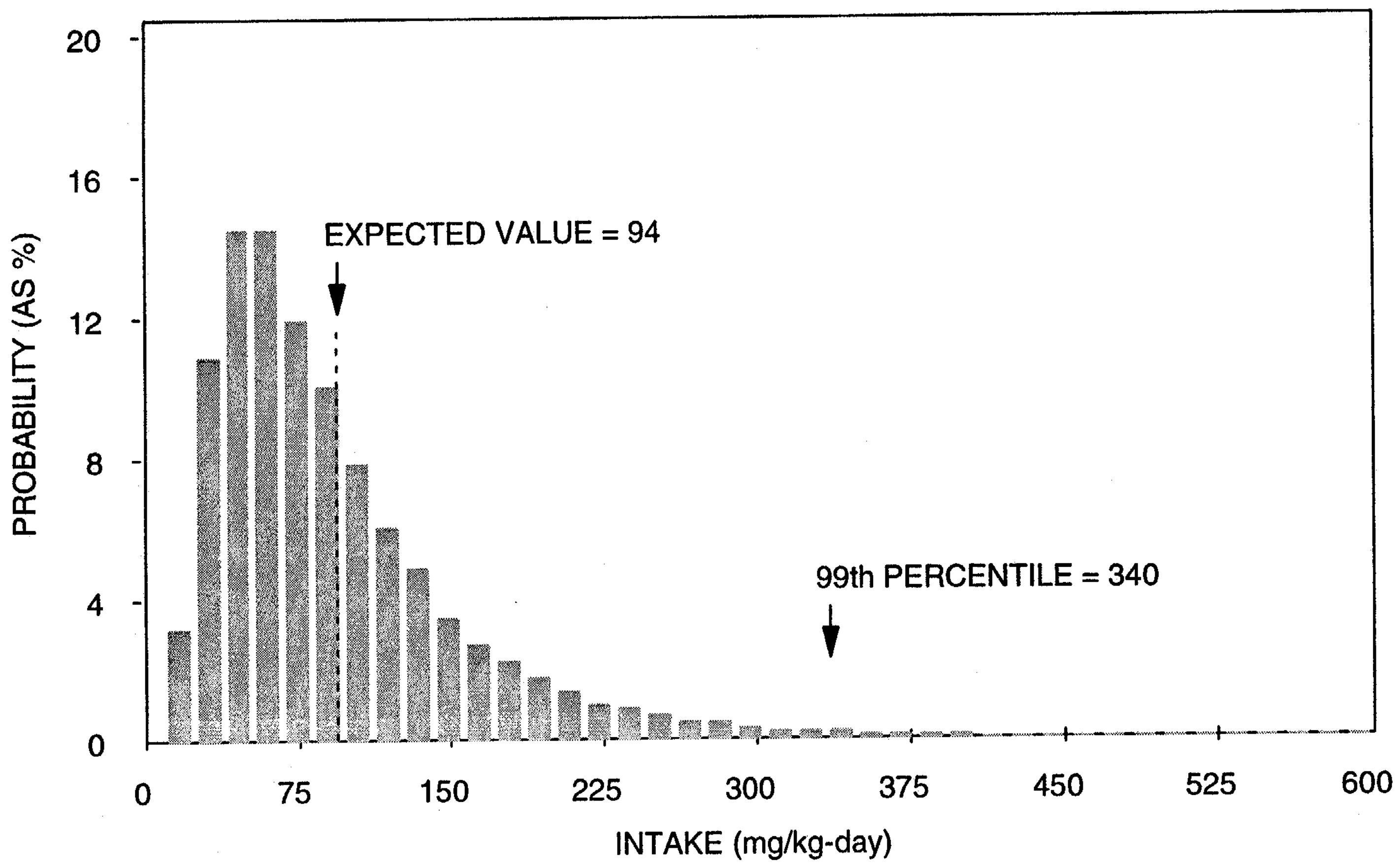


U
INTAKE FROM DRINKING WATER

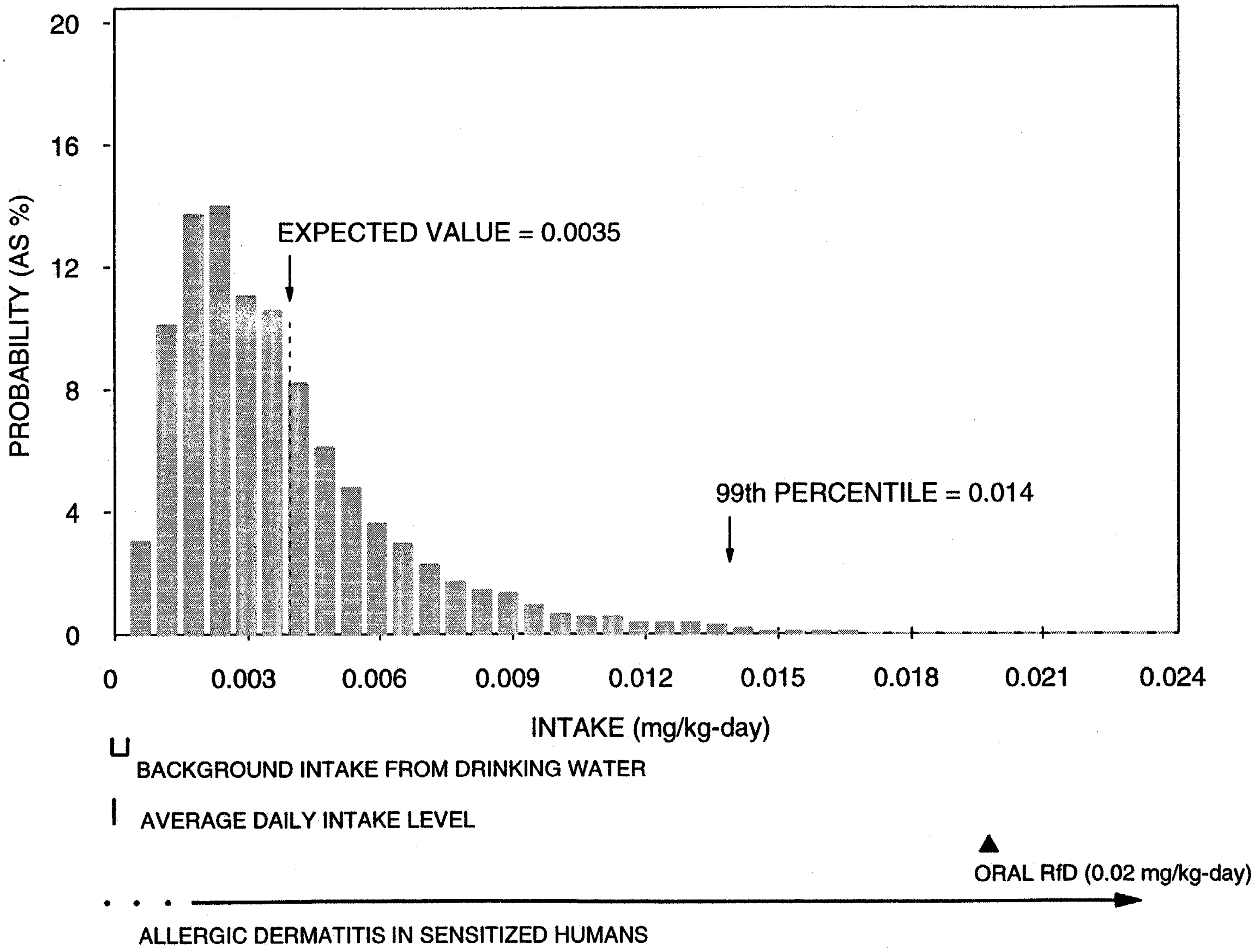
▲ RECOMMENDED INTAKE FROM WATER

DIETARY INTAKE LEVELS

HYPERTENSION IN HUMANS

NOTE: ALTHOUGH A LARGE PORTION (APPROXIMATELY 90%) OF THE POTENTIAL INTAKES FALL ABOVE THE LEVEL WHERE HYPERTENSION HAS BEEN REPORTED IN THE LITERATURE, APPROXIMATELY 80% OF THE POTENTIAL INTAKES FALL WITHIN REPORTED DIETARY INTAKE LEVELS.

FIGURE 6.6
HEALTH EFFECTS OF POTENTIAL CHLORIDE EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE



NOTE: THE MAJORITY OF THE POTENTIAL INTAKES IN CHILDREN WOULD NOT BE ASSOCIATED WITH ADVERSE HEALTH EFFECTS IN NONSENSITIZED INDIVIDUALS. HOWEVER, A LARGE PORTION (APPROXIMATELY 85%) OF THE POTENTIAL INTAKES COULD RESULT IN ALLERGIC DERMATITIS IN SENSITIZED INDIVIDUALS.

FIGURE 6.7
HEALTH EFFECTS OF POTENTIAL NICKEL EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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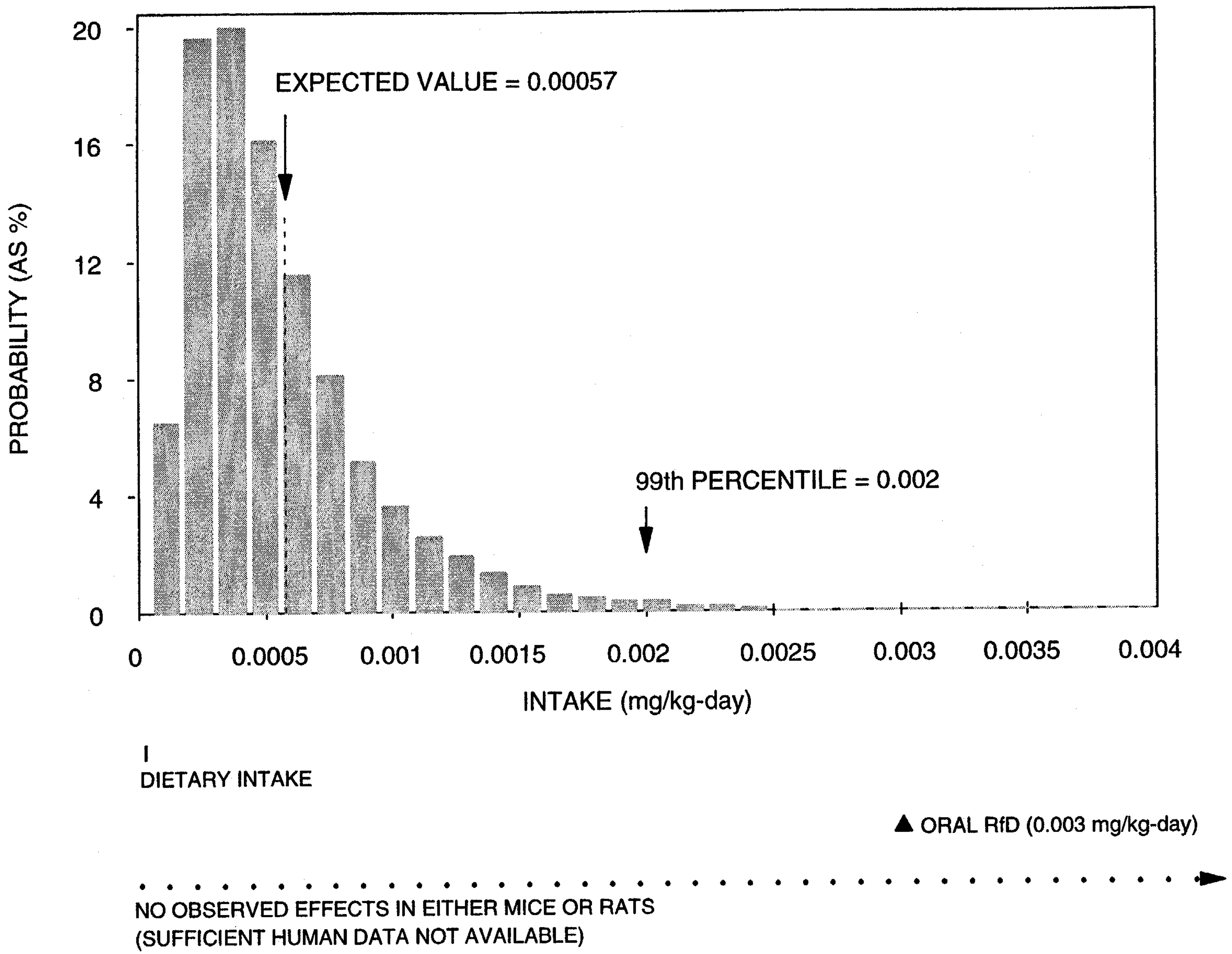

NOTE: NO HEALTH EFFECTS HAVE BEEN OBSERVED FOR ANY OF THE INTAKES AT THIS SITE.

FIGURE 6.8
HEALTH EFFECTS OF POTENTIAL URANIUM EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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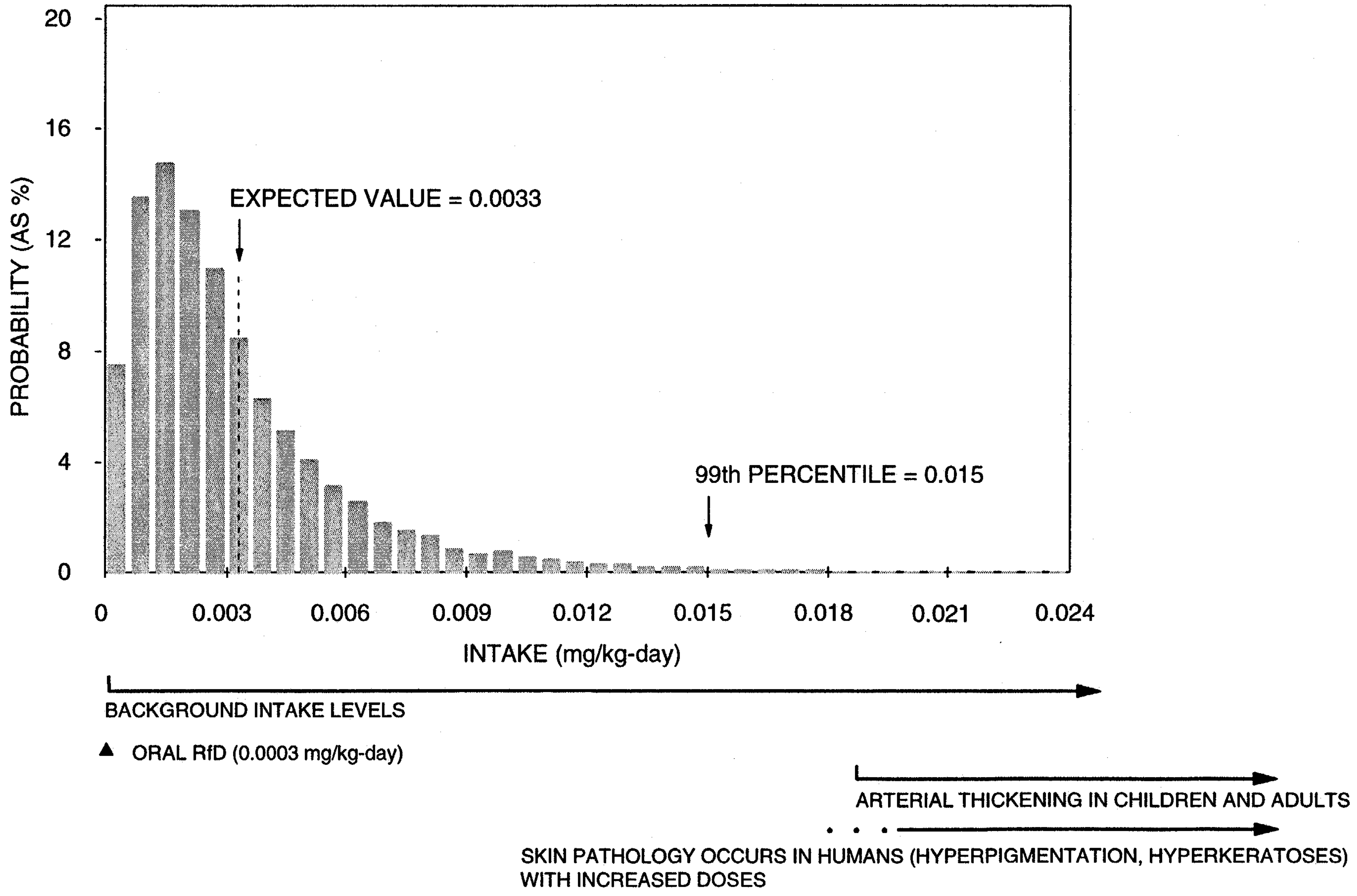

NOTE: ONLY A SMALL AMOUNT (LESS THAN 0.5%) OF THE POTENTIAL INTAKES IN CHILDREN COULD BE ASSOCIATED WITH SKIN PATHOLOGY AND CARDIOVASCULAR SYMPTOMS.

FIGURE 6.9
HEALTH EFFECTS OF POTENTIAL ARSENIC EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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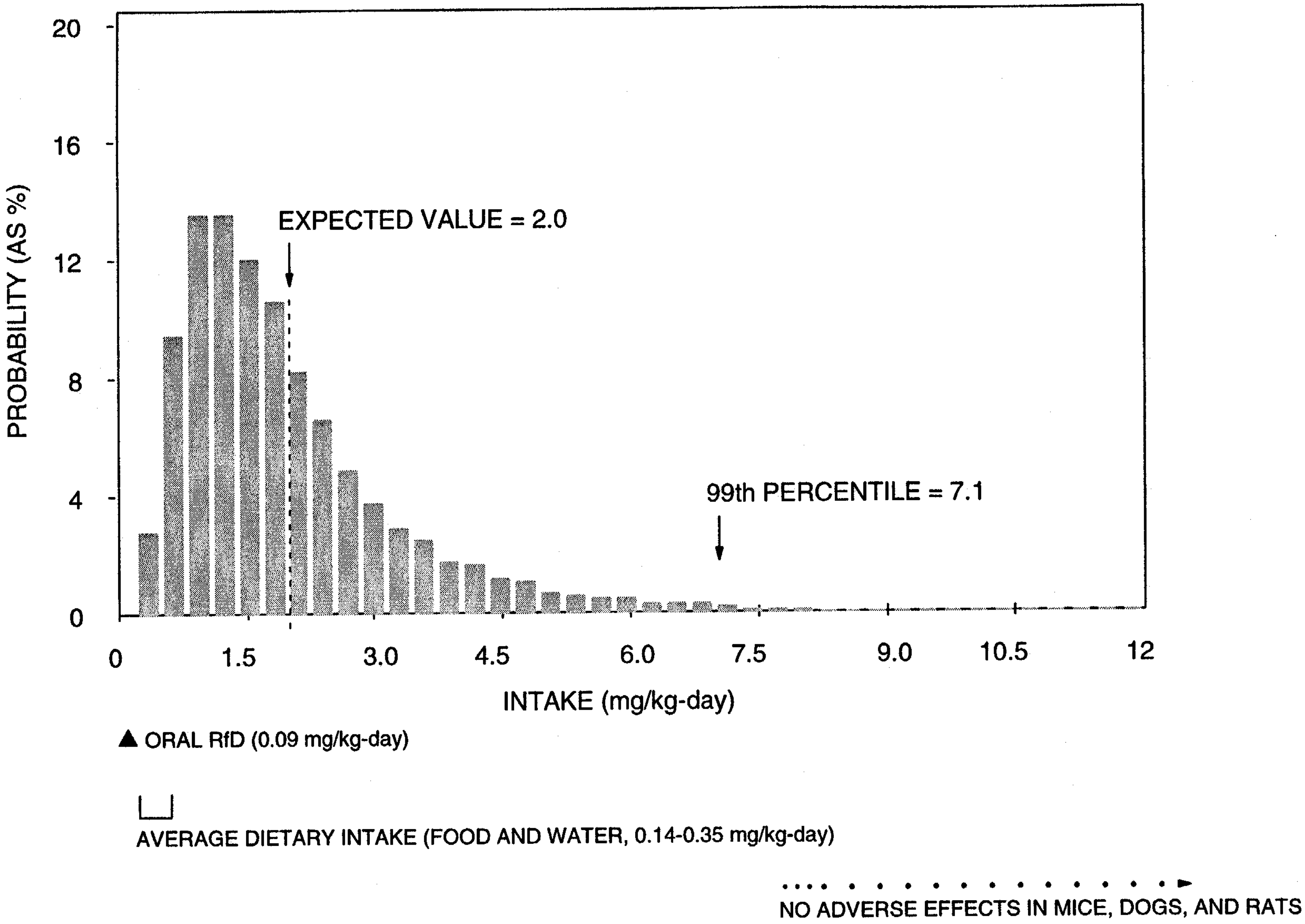

NOTE: ALTHOUGH THE MAJORITY OF THE SIMULATED INTAKES FALL ABOVE THE ORAL RfD, NO ADVERSE HEALTH EFFECTS HAVE BEEN ASSOCIATED WITH THE SIMULATED INTAKES FOR BORON.

FIGURE 6.10
HEALTH EFFECTS OF POTENTIAL BORON EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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BORHLT.DRW

For each contaminant of potential concern except molybdenum, the exposure contribution from the other pathways discussed in Section 4.2 would be approximately 7 percent or less. For molybdenum, the exposure contribution from other pathways would be approximately 20 percent. These increases in the potential exposure for each contaminant of potential concern would not significantly increase the risk over that predicted from the drinking water pathway alone.

As discussed in Section 5.2, physiologic interactions are likely among the numerous metal contaminants at this site. These interactions are likely to alter 1) the absorption of some metals in the presence of others, resulting in competition for binding sites once absorbed; 2) the concentrations of binding proteins that often transport metals in the bloodstream and through cell membranes; and 3) the tissue distribution of individual metals. The complexity of these interactions and the number of metal contaminants at Lakeview make it impossible to predict an increase or decrease in the toxicity of a given metal with the concurrent exposure to many other metals.

6.2 POTENTIAL CARCINOGENIC HEALTH EFFECTS

All uranium isotopes are radioactive and, as such, are considered potential carcinogens. Arsenic is not radioactive but is carcinogenic due to its chemical properties. Table 6.1 predicts the individual excess lifetime cancer risk from exposure to arsenic and the two radionuclide contaminants of potential concern (polonium-210 and uranium) through ingesting contaminated ground water at the Lakeview site. These estimates are based on the cancer SFs developed by the EPA; however, ingesting natural uranium has not been demonstrated to cause cancer in humans or animals. The individual excess lifetime cancer risk estimates for polonium-210 and uranium are within the range recommended by the National Contingency Plan (NCP) of 1E-04 and 1E-06. The individual excess lifetime cancer risk for arsenic is 2E-03 which exceeds the range recommended by the NCP. Summing the risks for the radionuclides calculated in Table 6.1 to one significant figure, the excess cancer risk estimate from ground water ingestion is 1E-04 , which is within the recommended range.

6.3 EVALUATION OF GROUND WATER FROM PRIVATE WELLS

The potential for arsenic, chloride, iron, manganese, molybdenum, sodium, and sulfate to cause adverse health effects is evaluated (except for manganese) by comparing the total exposure doses (Table 4.5) to the toxicity ranges that are presented in Section 5.0.

The results of the evaluation show that manganese would have the greatest potential to cause adverse health effects. The total manganese dose of 0.2 mg/kg-day, from dermal absorption and ingestion of water through cooking, falls in the range where early neurological symptoms such as irritability, speech disturbance, and compulsive behaviors could be expected to occur. The estimated amount of chloride that could be associated with hypertension in

Table 6.1 Excess lifetime cancer risk calculations for ground water ingestion by a hypothetical future adult resident, Lakeview, Oregon, site

Contaminant of potential concern	Concentration in ground water[a]	Ground water exposure dose[b]	Oral SF	Excess lifetime cancer risk[c]
	UCL	UCL	(mg/kg-day; pCi)$^{-1}$	UCL
Arsenic	0.11	1.0E-03	1.8E+0	2E-03
Polonium-210	8.5	2.9E+05	3.26E-10	9E-05
Uranium	14	4.9E+05	5.32E-11[d]	3E-05
			Total radionuclide risk:	1E-04

[a] Concentrations in milligrams per liter for arsenic and in picocuries per liter for polonium-210 and uranium.

[b] Exposure doses in milligrams per kilogram per day for arsenic and in picocuries per lifetime for polonium-210 and uranium.

[c] Calculated by multiplying the exposure dose by the slope factor (SF). The SF is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen.

[d] Uranium isotopes -234 and -238 are averaged for this calculation.

Notes:
1. UCL is the upper 95 percent confidence limit on the mean.
2. Ingestion rate: 2 L per day.
3. Exposure frequency: 350 days per year.
4. Exposure duration: 30 years.

sensitive individuals is in the range of 36 to 180 mg/kg-day. From ingesting chloride in water used to cook with (10 mg/kg-day) would not appreciably increase the potential for hypertension to occur in sensitive individuals. However, the effect of chloride doses from sources other than water; e.g., meat and milk, 47 and 36 mg/kg-day respectively, where the exposure doses are higher than in water, cannot be predicted at this time because of little research data in this area.

Noncarcinogenic adverse health effects would not be expected if all of the exposure doses are combined for arsenic, molybdenum, sodium, and sulfate. Also, the total carcinogenic risk estimate (8E-05) falls within the acceptable risk range of 1E-04 to 1E-06.

As stated in Section 4.2, the evaluation of the private wells is conservative because of the conservative assumptions. It was assumed that the people would get all of their garden produce, meat, and milk from their own garden or livestock and ingest the constituents for 350 days per year for 30 years, concurrently. Using these assumptions likely overestimates the true risks associated with using the ground water from these wells.

6.4 LIMITATIONS OF RISK EVALUATION

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

- This risk assessment evaluates only risks related to inorganic ground water contamination. Potential contamination with any of the organic constituents used in uranium processing is not addressed here.

- Except for individuals sensitized to nickel by previous nickel exposure and infants exposed to sulfate, subpopulations that might have increased sensitivity to specific contaminants are not specifically addressed on the graphs.

- Uncertainties are associated with the derivation of the toxicity values (RfDs and SFs). Derivations for these values intentionally overestimate rather than underestimate risk.

- 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 well locations sampled may not be in the most contaminated portion of the plume.

- Only the drinking water exposure pathway has been considered in depth, although other pathways were screened to determine their contribution.

This evaluation incorporates these limitations and compensates wherever possible by presenting toxicity ranges rather than point estimates, incorporating as much variability as could be reasonably defined. The impacts of these potential limitations is discussed more fully in Section 8.2.

7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION

The objective of the environmental portion of this risk assessment is to determine if contaminants detected at the Lakeview site have the potential to adversely affect biological communities at or in the area surrounding the site. This section describes the qualitative methodology used to evaluate the ecological risk at the site (EPA, 1989b). The EPA recommends conducting ecological assessments in a phased approach to ensure the most effective use of resources while all necessary work is conducted (EPA, 1992b). This approach consists of four increasingly complex phases: identifying potentially exposed habitats (phase 1); collecting analytical data from potentially affected media such as surface water and sediment (phase 2); collecting biological samples such as plant and animal tissue (phase 3); and toxicity testing (phase 4). If the early phases of the assessment indicate contaminants may be adversely affecting the ecological receptors, a higher level of analysis may be warranted. However, if the early phases of the evaluation indicate little or no potential for ecological risk, the assessment likely will be complete.

This ecological evaluation is a screening level assessment of the risks associated with the Lakeview site, using only phases 1 and 2. This approach compares contaminant concentrations in environmental media to aquatic life criteria and guidelines for sediment, vegetation, wildlife, livestock watering, and crop irrigation. The comparisons determine if the contaminants of potential concern potentially threaten ecological, livestock, and agricultural receptors.

It should be noted that sources of uncertainty in the ecological assessments can arise from limited analyses of media, limited toxicological information, and the inherent complexities of the ecosystem. In addition, methods of predicting nonchemical stresses such as drought, biotic interactions, behavior patterns, biological variability (differences in physical conditions, nutrient availability), and resiliency and recovery capacities are often unavailable. Therefore, it often is difficult to determine if contaminants can affect the biological component of an ecosystem and to predict adverse effects to the ecosystem.

7.1 EXPOSURE CHARACTERIZATION

The potential exposure pathways associated with the Lakeview site are discussed below. For risk to exist, a receptor must be exposed to contaminants. The evaluation of limited data for surface water and sediment in Section 3.6 suggest surface water has not been discernibly impacted and the levels of contaminants detected above background in sediment could reflect the natural variation associated with drainage areas other than the site. However, the limited database creates uncertainly about the impact of site-related contamination on surface water and sediment. Therefore, contaminant concentrations detected in these two media are evaluated for potential ecological impacts.

Currently, ground water is the only impacted medium at the site. Because the tailings pile and contaminated soil have been removed from the processing site, some direct exposure pathways (such as incidental soil ingestion, dermal contact with soil, and inhalation of air containing particulates) do not represent an ecological concern and are not evaluated here. Other direct and indirect exposure pathways are possible at the site. Direct exposure pathways include ingesting surface water and sediments potentially affected by contaminated ground water or by particulate transport and contaminant bioconcentration in surface water by aquatic organisms. Indirect exposure pathways include consumption of previously exposed organisms.

Bioconcentration is the net accumulation of a constituent by an organism directly from the surrounding environment. Net accumulation as a result of all routes of exposure, including diet, is known as bioaccumulation. Generally, bioconcentration is measured for chemical uptake 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.

The main surface water bodies in the site vicinity are Hunters Hot Springs, Warner Creek, Thomas Creek, and Hammersley Creek. These surface water bodies were evaluated in this screening assessment because they are potential exposure points for resident aquatic life and for terrestrial wildlife (including domestic animals) to come in contact with surface water and/or sediments.

Another potential current pathway is plant uptake of contaminants in ground water. Due to the shallow depth to ground water in the site area (approximately 10 to 20 ft [3 to 6 m] or less below land surface), plant roots can reach contaminated shallow ground water. To evaluate plant uptake, the roots were assumed to reach soil saturated with ground water containing the mean concentrations of the contaminants of potential concern in the most contaminated wells. Plant BCFs from the literature were used in the evaluation.

This screening assessment conservatively assumed a future private well could be placed at a location that intercepts the most contaminated ground water in the plume. The water from this hypothetical well could be used in a livestock watering pond (which could be stocked with fish) or to irrigate agricultural crops.

7.2 ECOLOGICAL RECEPTORS

Ecological resources present at the site and its vicinity that are potentially exposed to site-related contaminants are identified below.

The following information on ecological receptors is based primarily on surveys conducted before tailings removal began; it provides an historical perspective (DOE, 1985b). Limited observations of aquatic organisms were conducted at the surface water and sediment sampling locations during the October 1993 qualitative field survey.

7.2.1 Terrestrial plant and animal communities

The Lakeview site is in the ecoregion known as the Intermountain Sagebrush Province. This semiarid region is in the rain shadow of the Cascade Mountains to the west. The natural vegetation type for the region is the ponderosa shrub forest (DOE, 1985b). The site is characterized primarily by introduced grassland vegetation. The site also has been colonized by desert saltgrass (*Distichlis spicata* var. *stricta*). Hammersley Creek is distinguished by a narrow riparian zone that traverses the former processing site.

Wildlife observed or determined to be present (based on habitat availability) are the coyote, red fox, mountain cottontail, and jackrabbit. Mule deer may occur irregularly as vagrants.

Bird species that may be in the site vicinity include the gray partridge, ring-necked pheasant, Canada goose, mallard, great blue heron, and a variety of small waterfowl such as cinnamon teal, blue-winged teal, and green-winged teal along the drainages. Although some nesting by these ducks (as well as mallards and perhaps other species) is possible, most waterfowl use probably occurs during migration. Raptors hunting across the open terrain of the site probably include red-tailed hawks, northern harriers, and short-eared owls as year-round residents, with rough-legged hawks and ferruginous hawks present in winter and several other species present as migrants or vagrants.

Reptiles at the mill tailings site probably are limited due to the small number of habitat types represented. A few amphibian species would be expected to breed along Hammersley Creek.

A complete list of the plant and animal species that have been observed or are characteristic of the Lakeview area is presented in Appendix F of the environmental assessment (DOE, 1985b).

7.2.2 Threatened and endangered species

Two species of plants in the state of Oregon are on the U.S. Fish and Wildlife Service (FWS) list of endangered and threatened plants. One plant is proposed for protected status, and 113 plants are under review for protected status (50 CFR Part 17; 45 FR 82480; 48 FR 53640). The Oregon Natural Heritage database, maintained for the state by the Oregon Chapter of the Nature Conservancy, indicates neither the two plant species listed as endangered nor the one plant species proposed for protected status occurs in or near the Lakeview site (Wolfin, 1985).

The only wildlife species listed by the FWS that are potentially present at the site are the American peregrine falcon (*Falco peregrinus anatum*) (endangered) and the bald eagle (*Haliaeetus leucocephalus*) (currently proposed as a threatened species) (50 CFR Part 17). Both species could occur as vagrants or migrants because of the proximity to suitable habitat along nearby lakes and rimrocks.

7.2.3 Aquatic organisms

Quantitative surveys of aquatic organisms in the water bodies near the site were not conducted for this baseline risk assessment. A brief qualitative survey of the aquatic organisms was conducted near historic surface water sampling locations 605 through 609 and a new location, 613, which is in a pond near the former mill buildings. The following organisms were observed using a fine-mesh dip net at all water sampling locations: dragonfly and damselfly nymphs (*Odonata*), water striders (*Gerridae*), backswimmers (*Notonectidae*), midge larvae (*Chironomidae*), and unidentified water fleas (*Cladocera*). Fish were not observed at any location, due to the limited stream habitat. All the streams except the open water marshy areas in Hunters Hot Springs were shallow (less than 4 inches [10 cm] deep) and were channelized with no riffle/run areas or undercut banks.

7.3 CONTAMINANTS OF ECOLOGICAL CONCERN

The complete list of ground water contaminant levels that exceed background levels was considered in assessing contaminants of potential concern for ecological receptors potentially exposed to ground water (Table 3.6). Potential receptors and pathways include plants by uptake, livestock by watering, and fish stocked in a ground water-fed pond.

As discussed in Section 3.6, available data suggest that the surface water is not discernibly impacted by site-related contaminated ground water (Table 3.5). Therefore, no contaminants were identified as ecological contaminants of potential concern for surface water. Also as discussed in Section 3.6, data indicate that sediments adjacent to and downstream of the site have higher levels of iron, manganese, and uranium than samples collected from Hunters Creek upstream of the site (Table 3.9). Therefore, these three constituents are identified as ecological contaminants of potential concern for sediment.

7.4 POTENTIAL IMPACTS TO WILDLIFE AND PLANTS

7.4.1 Ground water

Concentrations of the contaminants of potential concern in plant tissue were estimated using soil-to-plant BCFs. Soil concentrations in the saturated zone were estimated by multiplying the ground water concentration by the soil-water distribution coefficient (Kd). Table 7.1 presents the methodology and parameters used to estimate root uptake and plant tissue concentrations at harvestable maturity for the contaminants of potential concern. This methodology is described in detail elsewhere in the literature (Baes et al., 1984) and therefore will not be presented here.

Table 7.1 Comparison of estimated plant concentrations to phytotoxic concentrations, Lakeview, Oregon, site

Contaminant of potential concern	UCL concentration in ground water (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factor		Estimated concentration in vegetative growth[a] (mg/kg DW)	Estimated concentration in fruits/tubers[b] (mg/kg DW)	Approximate concentration in mature leaf tissue that is toxic[c] (mg/kg DW)
				Bv	Br			
Arsenic	0.11	19	2.0	0.04	0.006	0.084	0.013	5-20
Boron	58	1.3	75	4	2	300	150	50-200
Chloride	3,000	0.25	750	70	70	52,500	52,500	NA
Iron	27	15	400	0.004	0.001	1.6	0.41	NA
Manganese	40	25	1,000	0.25	0.05	250	50	200[d]-1000
Molybdenum	0.46	120	55	0.25	0.06	14	3.3	10-50
Nickel	0.13	59	7.7	0.06	0.06	0.46	0.46	10-100
Sodium	3970	0.2	790	0.075	0.055	60	60	NA
Sulfate[e]	6860	0	NC	0.5	0.5	NC	NC	NA
Uranium	0.02	50	1.0	0.008	0.004	0.0085	0.004	NA
Radionuclide								
Polonium-210[f]	1.9E-12	15	2.9E-11	0.015	0.0015	7.1E-14	1.1E-14	NA

[a] Estimated concentration in vegetative portions, calculated as estimated soil concentration multiplied by Bv.

[b] Estimated concentration in nonvegetative portions, calculated as estimated soil concentration multiplied by Br.

[c] Concentrations are not presented for very sensitive or for highly tolerant plant species due to a general lack of data for these species (Kabata-Pendias and Pendias, 1992).

[d] The value of 200 mg/kg is a recommended criterion of the Illinois Institute of Environmental Quality (Saric, 1986).

[e] Bv and Br factors available for elemental sulfur only; thus, these factors were reduced by a factor of 3 for sulfate.

[f] The ground water concentration in picocuries per liter was converted to milligrams per liter.

Kd – soil-water distribution coefficient; from PNL (1989).
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.
NC – value cannot be calculated because Kd is zero.
L/kg – liters per kilogram.
A – not available.

No Kd, Bv, or Br values available for ammonium.

The estimated tissue concentrations for the contaminants of potential concern in the vegetative portions (e.g., stems and leaves) and in the nonvegetative portions (e.g., fruits and tubers) of plants at harvestable maturity were compared to approximate concentrations in mature leaf tissue that were toxic to plants (phytotoxic) (Table 7.1). Data on phytotoxic soil concentrations of metals are available in the literature. However, for the metals of concern at this site, more data are available for phytotoxic tissue concentrations than for phytotoxic soil concentrations. As illustrated in Table 7.1, few available data relate tissue concentrations to phytotoxicity. The reported phytotoxic concentrations are not representative of very sensitive or highly tolerant plant species because most of the data are for agronomic crops (Kabata-Pendias and Pendias, 1992). The estimated tissue concentrations for boron in plants that may reach soil saturated with contaminated ground water exceed the available phytotoxicity data, while the estimated concentrations of manganese and molybdenum fall within the range reported to be phytotoxic. The estimated plant tissue concentrations for arsenic and nickel are below phytotoxic levels. For contaminants without comparison data (chloride, iron, polonium-210, sodium, sulfate, and uranium) it is not possible to determine if the estimated tissue concentrations could adversely affect plants.

Bioaccumulation in terrestrial organisms as a function of contaminants of potential concern in ingested plants or animals (for example, birds eating fish) is a potential exposure pathway at the site. Birds and other vertebrates consuming these plants and animals can bioaccumulate some contaminants of potential concern if the amount of contamination ingested exceeds the amount eliminated. Bioaccumulation often is a function of the areal extent of contamination compared to the areal extent of an animal's feeding range. When the contaminated area is small, the amount of food in the diet usually exceeds the impacted food, and bioaccumulation is not a concern. Therefore, while exposure through diet is possible for all food chain species in certain areas (e.g., wetlands), the potential for bioaccumulation is not always a concern.

Biomagnification is more severe: the constituent concentration increases in higher levels of the food chain because the contaminant concentrations accumulate through each successive trophic level. Biomagnification effects are of particular concern for the top predators, especially carnivorous birds and mammals. Only a limited number of metals (e.g., mercury and selenium) have the potential to magnify in the food chain (EPA, 1989b; EPA, 1991). None of the metals detected in media at this site represent a significant biomagnification concern. Available data suggest low potential for the contaminants of potential concern to represent a concern via food chain transfer.

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 UCL ground water concentrations for the contaminants of potential concern were compared to available comparison water quality criteria (Table 7.2). Neither federal nor state criteria or standards have been established to protect terrestrial wildlife from exposure to contaminated water. Therefore, it is

Table 7.2 Comparison of contaminants of potential concern in ground water with available water quality values, Lakeview, Oregon, site

Contaminant of potential concern	UCL concentration in ground water	Aquatic life water quality value[a]	Concentration in irrigation water protective of plants[b]	Water concentration protective of livestock[b]
Arsenic	0.11	0.048	0.10	0.20
Boron	58	1.0[c]	0.75	5.0
Chloride	3000	230[d]	NA	NA
Iron	27	1.0	5.0	NA
Manganese	40	NA	0.20	NA
Molybdenum	0.46	50 (0.79)[e]	0.010	NA
Nickel	0.13	1.6[f]	0.20	NA
Polonium-210	8.5 pCi/L	NA	NA	NA
Sodium	3960	NA	NA	NA
Sulfate	6860	NA	NA	1000[g]
Uranium	0.02	31[h]	NA	NA

[a] From ODEQ (1992), unless specified otherwise. These values are standards protective of aquatic life via chronic exposure.

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

[c] No state or federal water quality value available. Value presented is the current boron criterion recommended by the FWS for the protection of aquatic life (Eisler, 1990).

[d] No state water quality value available. Value presented is the Federal Water Quality Criterion for the protection of freshwater aquatic life via chronic exposure (EPA, 1992b).

[e] No state or federal water quality value available. Value presented is the current molybdenum criterion recommended by the FWS for the protection of aquatic organisms, with the exception of newly fertilized eggs of rainbow trout, which are sensitive to molybdenum concentrations above 0.79 mg/L (Eisler, 1989).

[f] Water-hardness-related state standard (ODEQ, 1992). Criterion presented was calculated using the median hardness (milligrams per liter) determined from concentrations of calcium and magnesium in the plume wells (Table 3.2).

[g] From National Research Council (1971).

[h] No state or federal water quality value available. Value presented is the state of Colorado's chronic water quality standard for uranium (CDH, 1991). This is a hardness-dependent standard.

Concentrations reported in milligrams per liter unless otherwise noted.
NA – not available.

difficult to evaluate the potential hazards to terrestrial receptors. However, surface water quality values for the protection of freshwater aquatic life are available and include state of Oregon standards (ODEQ, 1992).

UCL ground water concentrations for arsenic, boron, chloride, and iron exceeded the comparison aquatic life water quality values (Table 7.2), indicating this water would be unacceptable for aquatic organisms. Ground water concentrations for molybdenum, nickel, and uranium were below the comparison values. No comparison water quality values are available for manganese, polonium-210, sodium, and sulfate.

Another future hypothetical use of contaminated ground water in the area is crop irrigation. Table 7.2 compares the approximate concentrations in irrigation water that should be protective of plants (EPA, 1972). The EPA developed these approximate irrigation water concentrations to protect agricultural crops from toxicity associated with constituent buildup in the soil. Six contaminants of potential concern (arsenic, boron, iron, manganese, molybdenum, and nickel) have comparison values. The UCL ground water concentrations for all these inorganics exceed the comparison values, with the exception of nickel. No comparison values are available for the remaining contaminants of potential concern. Thus, it is not possible to evaluate the potential for these compounds to adversely affect plants through irrigation water.

Based on the available information, using contaminated alluvial ground water near the site (containing the UCL concentrations) as a continuous source of irrigation water could adversely affect crops, primarily because of the elevated concentrations of arsenic, boron, iron, manganese, and molybdenum.

The sodium hazard is a basic criterion in evaluating irrigation water quality. Sodium concentrations can contribute to the total salinity of an irrigation water and may be directly toxic to sensitive crops. However, the primary concern with elevated sodium concentrations is the adverse effect on soil characteristics (e.g., increase in colloidally absorbed sodium, resulting in hard compact soil). The sodium hazard of water is expressed as the sodium adsorption ratio (SAR), which is calculated as the proportion of sodium to the total of calcium and magnesium in the water. Using the 95 percent UCL ground water concentrations for sodium, calcium, and magnesium, an SAR of 50 was calculated. This SAR is well above the upper SAR limit of 10 (Follett and Soltanpour, 1985). Water with a SAR value greater than 10 should not be used as the sole source of irrigation water for long periods of time.

7.4.2 Sediments

There are no established state or federal sediment quality criteria (SQC) for the protection of aquatic life for the contaminants of potential concern (iron, manganese, and uranium) at this site (EPA, 1988; NOAA, 1990). Because there are no guidelines for iron, manganese, and uranium, it is not possible to evaluate their potential threat to ecological receptors without further study.

Many of the contaminants that exceed background ground water quality have not been monitored in sediments from water bodies in the site vicinity. Thus, their potential threat to the ecological environment cannot be evaluated at this time.

7.5 POTENTIAL IMPACTS TO LIVESTOCK

Ingestion of vegetation that may have bioconcentrated contaminants from alluvial ground water is a potential exposure route for livestock. However, without actual plant tissue concentrations or exposure and toxicity information for livestock, this potential exposure route cannot be evaluated.

It is possible that contaminated ground water could be used in the future to provide water for livestock. To evaluate the potential impact to livestock in this future scenario, the UCL ground water concentrations for the contaminants of potential concern were compared to drinking water concentrations considered protective of livestock (Table 7.2). The UCL ground water concentrations exceed the comparison water quality guidelines for boron and sulfate, while the arsenic UCL concentration is below the guideline. The boron ground water concentration (58 mg/L) is more than 10 times the livestock drinking water value (5 mg/L). Sulfate was detected in ground water at the UCL concentration of 6860 mg/L. This level is above the value (1000 mg/L) that can cause diarrhea in cattle (National Research Council, 1971; Church, 1984).

No livestock drinking water guidelines have been reported for the remaining contaminants of potential concern. However, the available information suggests that the use of ground water as a source of drinking water for livestock would be hazardous due to elevated levels of boron and sulfate.

7.6 LIMITATIONS OF THE ECOLOGICAL RISK ASSESSMENT

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

- Only a small amount of ecological data were collected during this screening.
- Many of the contaminants that exceed background ground water quality levels have not been measured in surface water or sediments from the water bodies in the site vicinity.

- Land adjacent to the site is used for pasture. Due to the shallow depth to ground water at and downgradient of the site, plant roots might access contaminated ground water. However, because little is known about uptake rates of plant site-specific contaminants, this potential exposure pathway was not evaluated. Additionally, because of the lack of plant tissue data, the potential effect on livestock could not be evaluated.

- Only limited ecotoxicological reference data are available.

- Site-specific biota tissue analysis has not been conducted.

7.7 SUMMARY

Recent surface water data from the water bodies near the site indicate the constituent concentrations detected at adjacent locations from the Lakeview site were not elevated relative to the background location. This suggests that site-related contamination has not adversely affected the quality of the surface water bodies in the site vicinity.

Three contaminants (iron, manganese, and uranium) detected in the sediments exceed background concentrations. However, no state or federal sediment quality guidelines for the protection of aquatic life are available to evaluate these contaminants.

Available data and criteria indicate potential hazards to plants at harvestable maturity that may have roots in contact with soil saturated with the most contaminated ground water in the alluvial/lacustrine aquifer. The concentrations of boron, and possibly of manganese and molybdenum, could cause phytotoxicity following accumulation in plant tissue. Additionally, this ground water would not be suitable for continuous long-term use as irrigation water for crops due to the elevated levels of arsenic, boron, iron, manganese, molybdenum, and sodium.

This ground water would also not be suitable as a long-term source of drinking water for livestock due to the adverse effects associated with elevated levels of boron and sulfate. Water from the most contaminated wells in this aquifer would not be suitable as a source of water (e.g., a pond) for fish to live in.

Available surface water, sediment, and ground water data indicate that the contaminants of concern detected in media at the site have a low potential to threaten the food chain of terrestrial and aquatic wildlife (via bioaccumulation and biomagnification). However, tissue from biota (e.g., invertebrates and plants) has not been analyzed.

Insufficient water quality and sediment quality values were available to allow a comprehensive and statistical evaluation of the impact of surface water, sediment, and contaminated ground water on ecological receptors. However, available data show no compelling evidence that the surface water and sediment of the drainage features near the site are affected by former milling activities at the Lakeview site.

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 for the Lakeview site to evaluate the potential occurrence of these hazards.

Currently, no uses of site-related contaminated ground water by human receptors have been identified at the Lakeview site. However, shallow ground water is used in the Lakeview area for domestic purposes such as drinking, bathing, and cooking. Additionally, ground water in this area is used for agricultural purposes such as crop irrigation and livestock watering, although the current use of contaminated ground water for these purposes has not been identified. Surface water is used primarily for irrigation in the former site area. Access to site-related contaminated ground water at and downgradient of the site is possible, thereby providing potential future exposure routes to site-related contaminated ground water.

This risk assessment is a conservative estimate of risk to human health at the Lakeview site because ground water data from the most contaminated part of the shallow on-site zone were used. Additionally, with the uranium mill tailings (the source of the contamination) removed, contaminated ground water at this site will tend to disperse and dilute (although slowly), thereby reducing exposure concentrations over time.

8.1.1 Human health

If the site-related contaminated ground water in the shallow zone is used for drinking water in the future, potential adverse human health effects could occur.

Serious adverse health effects could result from nearly the entire range of potential exposures to sulfate, manganese, and sodium, as well as from the upper range of potential exposures to iron. The individual excess lifetime cancer risk of arsenic (2 in 1000) is at a level that exceeds the NCP's upper-bound criterion of 1 increased chance in 10,000 of developing cancer. However, the total carcinogenic risk from the radionuclides uranium and polonium-210 fall at the NCP criterion of 1 increased chance in 10,000 of developing cancer.

8.1.2 Ecological

The ground water would not be suitable as a long-term source of drinking water for livestock, due to the adverse effects of sulfate and boron. In addition, the site-related contaminated ground water would not be acceptable as a water source for fish habitat (e.g., a ground water-fed pond) or as a source of continuous irrigation water for agricultural crops and other plants.

Available surface water data from water bodies in the site vicinity suggest contaminated ground water from the site has not adversely affected surface water. Three contaminants (iron, manganese, and uranium) detected in the sediments exceed background concentrations. However, no state or federal sediment quality guidelines for the protection of aquatic life are available to evaluate these contaminants.

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 contaminants of ground water at the UMTRA Project site near Lakeview. Potential organic contaminants (those few related to uranium processing) have not been considered.

- In general, the results presented in this document are based on filtered (0.45-micron) water samples. Unfiltered data were used for arsenic. Filtration effects vary with different elements; 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.

- Contaminant toxicity varies from person to person. Normal variability in biochemical factors among individuals, differences in medical history, previous exposure to toxicants, and dietary and exercise habits all affect susceptibility to chemical toxicity. Chemicals also exert different toxic effects at different exposure levels. Because 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.

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 levels 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 responses as animals.

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

- Considerable effort has been directed at determining plume movement and placing monitor wells in locations that capture maximal contamination. Nevertheless, physical systems and models used to determine contaminant plume migration vary widely and may result in well placements that do not measure the highest contaminant concentrations or determine the fullest extent of plume impact.

- Variability can be 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. The use of multiple samples should introduce 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. Although other pathways were screened and determined not to contribute significantly to the total exposure, the additivity of exposure from these pathways should be kept in mind.

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 EPA final rule ground water standards for the UMTRA Project were published on 11 January 1995 (60 FR 2854). These standards consist of ground water protection standards to evaluate disposal cell performance and ground water cleanup standards for existing contamination at processing sites. The contaminants that have MCLs are summarized in Table 8.1. While the standards apply only to the UMTRA Project, the EPA also has published drinking water health advisory levels for both long- and short-term exposures. These advisories are also shown in Table 8.1.

Table 8.1 Constituent maximum concentration limits and health advisories of constituents

Constituent	UMTRCA MLA (mg/L)	Health advisories 10-kg child, 10-day (mg/L)	Health advisories 70-kg adult lifetime (mg/L)
Chemicals (inorganic)			
Antimony	-	0.01	0.003
Arsenic	0.05^{b}	-	-
Barium	1.0	-	2
Boron	-	0.9^{b}	0.6^{b}
Cadmium	0.01^{b}	0.04	0.005
Chromium	0.05	1	0.1
Cobalt	-	-	-
Copper	-	-	-
Fluoride	-	-	-
Iron	-	-	-
Lead	0.05	-	-
Manganese	-	-	-
Mercury	0.002	-	0.002
Molybdenum	0.1^{b}	0.04	0.04
Nickel	-	1	0.1
Nitrate	44^{b,c}	10^{d}	-
Selenium	0.01	-	-
Silver	0.05	0.2	0.1
Strontium	-	25	17
Sulfate	-	-	-
Thallium	-	0.007	0.0004
Vanadium	-	-	-
Zinc	-	6	2
Radionuclides			
Lead-210	-	-	-
Polonium-210	-	-	-
Radium-226/-228	5 pCi/L	-	-
Thorium-230	-	-	-
Uranium (U-234/-238)	30 pCi/L (0.044 mg/L)	-	-

[a] From 40 CFR §192.02.
[b] Exceeded in plume wells.
[c] Equal 10 mg/L nitrate as nitrogen.
[d] Under review, 10 mg/L as nitrogen.

Dash indicates not available.

8.4 RISK MITIGATION MEASURES/INSTITUTIONAL CONTROLS

In the preamble to the final ground water standards for the UMTRA Project, institutional controls are defined as mechanisms that can be effectively used to protect human health and the environment by controlling access to contaminated ground water (60 FR 2854). Although the preamble refers to institutional controls for long periods of time (e.g., up to 100 years during natural flushing), this concept also can be applied to short-term restrictions of access to ground water. Because all 24 UMTRA Project sites cannot be evaluated simultaneously, institutional controls may be needed pending Ground Water Project compliance decisions or during the implementation of the selected ground water compliance strategy at individual sites.

The Oregon Water Resources Department (formerly the State Engineer's Office) administers certain Oregon water laws that allow the Oregon Water Resources Department to curtail use of any ground water wells, including privately owned wells, that are known to be contaminated (Oregon Water Resources Department, 1994). In accordance with this law, the Water Resources Department could potentially prohibit further use of a well determined to be contaminated (Carter, 1994).

The Oregon Department of Environmental Quality (ODEQ) policy is to notify owners of private ground water wells that they are required to restrict the use of a well that presents a major contamination problem to other wells in the vicinity. ODEQ rules can be implemented if ground water well contamination is proved a major source of contamination. The ODEQ also sends written notification to any private well owner of a known contamination problem (Bailey, 1994).

8.5 RECOMMENDATIONS

Currently, human health is not at risk from the use of site-related contaminated shallow zone ground water. However, using ground water from the shallow zone as drinking water in the future could result in potential adverse effects to human health. Therefore, it is recommended that no one drink the site-related contaminated ground water from the shallow zone.

This risk assessment was performed without complete site characterization of the contaminated ground water at the Lakeview site. To fully characterize potential risks at the site the recommendations presented below should be followed:

- Consideration should be given to placement of monitor wells between the site and the private wells south of the site to characterize the farthest southern extent of the contaminant plume. This would need to be done to verify that the constituents detected in the private wells are not site-related.

- Substantiation of the conceptual model of saline ground water should be considered by performing direct tests for soluble salts in soil from the area near the private wells south of the site and in the area of the former tailings pile.

- Dilution and dispersion in conjunction with ground water flow is identified as a major control on the fate and transport of contaminants at the Lakeview site. The present rates of ground water transport of site-related contaminants, indicated by little or no change in contaminant concentrations over the last 12 years, appear to be too slow to explain the known extent of contamination. The apparent contradiction between the extent of contamination and the lack of change in contaminant concentrations in individual wells needs to be further studied.

- Three contaminants (iron, manganese, and uranium) are identified as above background in the sediments adjacent to the site. These observed contaminants could reflect natural variation associated with a different drainage area, such as Warner Creek, rather than site-related contamination. However, the limited database for this medium precludes definitive conclusions about the origins of these three contaminants. Additionally, surface water data suggest the surface water is not impacted by site-related contaminated ground water. However, the database for this medium is limited. Therefore, additional data would be needed to confirm any observations concerning surface water and sediment contamination.

- Although the use of organic chemicals has not been identified at the Lakeview site, organics (e.g., kerosene used as a carrier) are known to have been used at other UMTRA Project sites. Consequently, an investigation of the presence of organics in the tailings pile and mill areas should be considered.

- Depth to ground water in the area of the Lakeview site is 10 to 20 ft (3 to 6 m) or less in some areas. In addition to areas containing indigenous plantlife, much of the land downgradient from the site is agricultural or pastureland. Some plants such as alfalfa have deep root systems that could reach contaminated ground water in the downgradient plume where access to ground water is not restricted. Therefore, an investigation of plant access to ground water should be considered to further determine if potential impacts could occur.

9.0 LIST OF CONTRIBUTORS

The following individuals contributed to the preparation of this report.

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