

RECEIVED

MAR 25 1996

OSTI

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

March 1996

DISCLAIMER

This report was prepared as an account of work sponsored by an agency of the United States Government. Neither the United States Government nor any agency thereof, nor any of their employees, makes any warranty, express or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed, or represents that its use would not infringe privately owned rights. Reference herein to any specific commercial product, process, or service by trade name, trademark, manufacturer, or otherwise does not necessarily constitute or imply its endorsement, recommendation, or favoring by the United States Government or any agency thereof. The views and opinions of authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof.

MASTER

INTENDED FOR PUBLIC RELEASE

This report has been reproduced from the best available copy. Available in paper copy and microfiche.

Number of pages in this report: 196

DOE and DOE contractors can obtain copies of this report from:

**Office of Scientific and Technical Information
P.O. Box 62
Oak Ridge, TN 37831
(615) 576-8401**

This report is publicly available from:

**National Technical Information Service
Department of Commerce
5285 Port Royal Road
Springfield, VA 22161
(703) 487-4650**

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

March 1996

Work performed under DOE Contract No. DE-AC04-91AL62350.

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

**Prepared by
Jacobs Engineering Group Inc.
Albuquerque, New Mexico**



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, 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 mill operated from February 1958 to November 1960. 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, no current users of the site-related contaminated ground water (for domestic or agricultural purposes) have 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 (ft) (900 meters [m]) south of the site, several private wells contain some of the same constituents as the site-related contaminated ground water. Although it is unlikely 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 Lakeview site area. 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, however, 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. Additionally, arsenic exceeds the U.S. Environmental Protection Agency (EPA) maximum concentration limit for ground water.

Site-related ground water quality

There are two types of contaminated ground water at the Lakeview site. One type is associated with constituents 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 constituents that seeped from the former evaporation ponds; it 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 sulfate distribution has been used to determine how far the site impact extends from the site. Based on the sulfate distribution, it appears that the shallow zone contains two impacted areas. 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 ft (900 m) 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 due to site-related contaminated ground water, primarily because their pattern of occurrence in the private wells is inconsistent with site-related constituents. 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 constituents 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 the 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 constituents, 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 constituents 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. Constituents detected in the ground water that could cause adverse human health effects if taken into the body are called constituents of potential concern. Arsenic, boron, chloride, iron, manganese,

molybdenum, nickel, polonium-210, sodium, sulfate, and uranium are the constituents of potential concern detected in the shallow zone at the Lakeview site.

This risk assessment also 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 constituents 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 site-related contaminated 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.

Table CS.1 Constituents of potential concern to human health, Lakeview, Oregon, site

| Constituent of potential concern | Results |
|----------------------------------|---|
| 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).

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 were the only source of exposure.

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 constituents 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 constituents 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 constituents 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 constituents 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 constituents of potential concern were identified. Three constituents (iron, manganese, and uranium) were detected above background concentrations in sediments and were retained as constituents 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 constituents.

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 exceed guidelines that are toxic to plants. Additionally, the concentrations of arsenic, boron, iron, manganese, molybdenum, and nickel exceed levels considered protective of plants in water used for irrigation. However, preliminary results of a DOE-sponsored irrigation and plant growth study at the University of Arizona suggests that for manganese and molybdenum the guidance values may be considerably overprotective.

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 site-related constituents of potential concern and human health is not at risk. This situation will continue if land and water use at the site does not change.

The source of the constituents detected in the private wells south of the site is not understood. It is unlikely that the constituents in these wells are 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 site were used in this evaluation. Furthermore, the uranium mill tailings (the source of the constituents) 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), 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.

The environmental evaluation of the UMTRA Lakeview site is ongoing. This risk assessment and future investigations will be used to determine how to comply with the UMTRA ground water standards.



TABLE OF CONTENTS

| <u>Section</u> | <u>Page</u> |
|--|-------------|
| CITIZENS' SUMMARY | CS-1 |
| 1.0 INTRODUCTION | 1-1 |
| 2.0 SITE DESCRIPTION..... | 2-1 |
| 2.1 Site background | 2-1 |
| 2.2 Climate..... | 2-2 |
| 2.3 Hydrogeology..... | 2-4 |
| 2.4 Surface water | 2-8 |
| 2.5 Land use..... | 2-10 |
| 2.6 Water use | 2-11 |
| 3.0 EXTENT OF CONTAMINATION..... | 3-1 |
| 3.1 Background ground water quality summary | 3-10 |
| 3.2 Magnitude and extent of site-related constituents | 3-19 |
| 3.2.1 Tailings and raffinate chemistry..... | 3-19 |
| 3.2.2 Contaminated ground water chemistry..... | 3-20 |
| 3.2.3 Statistical comparison of background and plume water quality | 3-23 |
| 3.3 Private well water quality | 3-25 |
| 3.4 Constituents of potential concern..... | 3-30 |
| 3.5 Constituent fate and transport | 3-30 |
| 3.6 Surface water and sediment | 3-36 |
| 4.0 EXPOSURE ASSESSMENT | 4-1 |
| 4.1 Potentially exposed populations and exposure pathways | 4-1 |
| 4.1.1 Drinking water ingestion..... | 4-3 |
| 4.1.2 Dermal absorption..... | 4-3 |
| 4.1.3 Ingestion of ground water-irrigated produce..... | 4-3 |
| 4.1.4 Ingestion of milk or meat from ground water-fed livestock | 4-6 |
| 4.1.5 Ingestion of fish..... | 4-6 |
| 4.2 Private wells | 4-6 |
| 4.3 Exposure concentrations | 4-10 |
| 4.4 Estimation of intake | 4-11 |
| 4.5 Exposure assessment uncertainties | 4-23 |
| 5.0 TOXICITY ASSESSMENT | 5-1 |
| 5.1 Constituent toxicity summaries | 5-1 |
| 5.1.1 Arsenic | 5-2 |
| 5.1.2 Boron | 5-4 |
| 5.1.3 Chloride | 5-7 |
| 5.1.4 Iron | 5-10 |
| 5.1.5 Manganese | 5-13 |
| 5.1.6 Molybdenum | 5-17 |
| 5.1.7 Nickel | 5-19 |

TABLE OF CONTENTS (Concluded)

| <u>Section</u> | <u>Page</u> |
|---|-------------|
| 5.1.8 Sodium..... | 5-22 |
| 5.1.9 Sulfate | 5-25 |
| 5.1.10 Uranium..... | 5-29 |
| 5.2 Constituent interactions | 5-31 |
| 5.3 Constituent risk factors..... | 5-33 |
| 6.0 HUMAN RISK EVALUATION..... | 6-1 |
| 6.1 Potential noncarcinogenic health effects | 6-1 |
| 6.2 Potential carcinogenic health effects..... | 6-13 |
| 6.3 Evaluation of ground water from private wells | 6-13 |
| 6.4 Limitations of risk evaluation..... | 6-15 |
| 7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION | 7-1 |
| 7.1 Exposure characterization..... | 7-1 |
| 7.2 Ecological receptors | 7-2 |
| 7.2.1 Terrestrial plant and animal communities..... | 7-3 |
| 7.2.2 Threatened and endangered species | 7-3 |
| 7.2.3 Aquatic organisms | 7-4 |
| 7.3 Contaminants of ecological concern | 7-4 |
| 7.4 Potential impacts to wildlife and plants | 7-4 |
| 7.4.1 Ground water..... | 7-4 |
| 7.4.2 Sediments..... | 7-9 |
| 7.5 Potential impacts to livestock | 7-9 |
| 7.6 Limitations of the ecological risk assessment..... | 7-10 |
| 7.7 Summary | 7-10 |
| 8.0 INTERPRETATION AND RECOMMENDATIONS..... | 8-1 |
| 8.1 Risk summary | 8-1 |
| 8.1.1 Human health..... | 8-1 |
| 8.1.2 Ecological | 8-1 |
| 8.2 Limitations of this risk assessment | 8-2 |
| 8.3 Ground water criteria | 8-3 |
| 8.4 Risk mitigation measures/institutional controls | 8-4 |
| 8.5 Recommendations | 8-4 |
| 9.0 LIST OF CONTRIBUTORS..... | 9-1 |
| 10.0 REFERENCES | 10-1 |

LIST OF FIGURES

| Figure | | Page |
|---------------|---|-------------|
| 2.1 | Location map, Lakeview, Oregon, site | 2-2 |
| 2.2 | Land use, Lakeview, Oregon, site vicinity | 2-3 |
| 2.3 | Geologic cross section, former Lakeview, Oregon, processing site | 2-5 |
| 2.4 | DOE monitor well locations, Lakeview, Oregon, site | 2-6 |
| 2.5 | Water table elevation map for shallow wells, Lakeview, Oregon, site | 2-7 |
| 2.6 | Locations of selected private wells, Lakeview, Oregon, site..... | 2-14 |
| 3.1 | Sulfate concentrations versus time in selected wells, Lakeview, Oregon, processing site vicinity | 3-3 |
| 3.2 | Trilinear diagram of major chemistries for ground water, Lakeview, Oregon, site | 3-12 |
| 3.3 | Sulfate isopleths, Lakeview, Oregon, site | 3-22 |
| 3.4 | Distribution of sulfate in shallow DOE monitor wells and private wells, Lakeview, Oregon, site | 3-29 |
| 3.5 | Surface water and sediment sampling locations, Lakeview, Oregon, site..... | 3-37 |
| 4.1 | Conceptual model, Lakeview, Oregon, site | 4-2 |
| 4.2 | Simulated distribution of arsenic concentrations, Lakeview, Oregon, site | 4-12 |
| 4.3 | Simulated distribution of boron concentrations, Lakeview, Oregon, site | 4-13 |
| 4.4 | Simulated distribution of chloride concentrations, Lakeview, Oregon, site | 4-14 |
| 4.5 | Simulated distribution of iron concentrations, Lakeview, Oregon, site..... | 4-15 |
| 4.6 | Simulated distribution of manganese concentrations, Lakeview, Oregon, site | 4-16 |
| 4.7 | Simulated distribution of molybdenum concentrations, Lakeview, Oregon, site | 4-17 |
| 4.8 | Simulated distribution of nickel concentrations, Lakeview, Oregon, site | 4-18 |
| 4.9 | Simulated distribution of sodium concentrations, Lakeview, Oregon, site | 4-19 |
| 4.10 | Simulated distribution of sulfate concentrations, Lakeview, Oregon, site..... | 4-20 |
| 4.11 | Simulated distribution of uranium concentrations, Lakeview, Oregon, site..... | 4-21 |
| 4.12 | Probability distributions for tap water ingestion rates | 4-24 |
| 4.13 | Probability distributions for body weight | 4-25 |
| 4.14 | Distribution of potential arsenic intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-26 |
| 4.15 | Distribution of potential boron intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-27 |
| 4.16 | Distribution of potential chloride intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-28 |
| 4.17 | Distribution of potential iron intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-29 |
| 4.18 | Distribution of potential manganese intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-30 |
| 4.19 | Distribution of potential molybdenum intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-31 |
| 4.20 | Distribution of potential nickel intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-32 |

LIST OF FIGURES (Concluded)

| <u>Figure</u> | | <u>Page</u> |
|---------------|--|-------------|
| 4.21 | Distribution of potential sodium intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-33 |
| 4.22 | Distribution of potential sulfate intakes for infants as a result of ground water ingestion, Lakeview, Oregon, site | 4-34 |
| 4.23 | Distribution of potential uranium intakes for children as a result of ground water ingestion, Lakeview, Oregon, site | 4-35 |
| 5.1 | Arsenic toxicity ranges | 5-5 |
| 5.2 | Boron toxicity ranges..... | 5-8 |
| 5.3 | Chloride toxicity ranges | 5-11 |
| 5.4 | Iron toxicity ranges | 5-14 |
| 5.5 | Manganese toxicity ranges..... | 5-18 |
| 5.6 | Molybdenum toxicity ranges | 5-20 |
| 5.7 | Nickel toxicity ranges | 5-23 |
| 5.8 | Sodium toxicity ranges | 5-26 |
| 5.9 | Sulfate toxicity ranges..... | 5-28 |
| 5.10 | Half-lives and emissions from decay chain of uranium-238..... | 5-30 |
| 5.11 | Uranium toxicity ranges | 5-32 |
| 6.1 | Health effects of potential sulfate exposure ranges for infants, Lakeview, Oregon, site..... | 6-2 |
| 6.2 | Health effects of potential manganese exposure ranges for children, Lakeview, Oregon, site | 6-3 |
| 6.3 | Health effects of potential iron exposure ranges for children, Lakeview, Oregon, site..... | 6-4 |
| 6.4 | Health effects of potential sodium exposure ranges for children, Lakeview, Oregon, site..... | 6-6 |
| 6.5 | Health effects of potential molybdenum exposure ranges for children, Lakeview, Oregon, site | 6-7 |
| 6.6 | Health effects of potential chloride exposure ranges for children, Lakeview, Oregon, site..... | 6-8 |
| 6.7 | Health effects of potential nickel exposure ranges for children, Lakeview, Oregon, site..... | 6-9 |
| 6.8 | Health effects of potential uranium exposure ranges for children, Lakeview, Oregon, site..... | 6-10 |
| 6.9 | Health effects of potential arsenic exposure ranges for children, Lakeview, Oregon, site..... | 6-11 |
| 6.10 | Health effects of potential boron exposure ranges for children, Lakeview, Oregon, site..... | 6-12 |

| Table | | Page |
|--------------|---|-------------|
| CS.1 | Constituents of potential concern to human health, Lakeview, Oregon, site..... | CS-5 |
| 2.1 | Static ground water levels, Lakeview, Oregon, site | 2-9 |
| 2.2 | Private and public wells, Lakeview, Oregon, site vicinity..... | 2-12 |
| 3.1 | Summary of DOE monitor wells, Lakeview, Oregon, site | 3-2 |
| 3.2 | Summary of filtered ground water quality in the shallow zone, Lakeview, Oregon, site, 1984-1993 | 3-4 |
| 3.3 | Summary of filtered ground water quality in the deeper zone, Lakeview, Oregon, site, 1984-1992 | 3-13 |
| 3.4 | Summary of private wells sampled, 1984 through 1994, Lakeview, Oregon, site vicinity..... | 3-26 |
| 3.5 | Summary of unfiltered water quality for private wells, Lakeview, Oregon, site vicinity | 3-27 |
| 3.6 | Contaminants of potential concern for human health evaluation, Lakeview, Oregon, site..... | 3-31 |
| 3.7 | Predominant ionic species of contaminants of potential concern, Lakeview, Oregon, site..... | 3-32 |
| 3.8 | Constituents monitored in surface water collected from water bodies in the Lakeview, Oregon, site vicinity | 3-38 |
| 3.9 | Occurrence of constituents monitored in sediment collected from surface water bodies in the Lakeview, Oregon, site vicinity | 3-40 |
| 4.1 | Exposure dose calculations for ground water ingestion and dermal contact, Lakeview, Oregon, site | 4-4 |
| 4.2 | Exposure dose calculations for ground water-irrigated garden produce ingestion, Lakeview, Oregon, site..... | 4-5 |
| 4.3 | Exposure dose calculations for ingestion of milk from ground water-fed livestock, Lakeview, Oregon, site | 4-7 |
| 4.4 | Exposure dose calculations for ingestion of meat from ground water-fed livestock, Lakeview, Oregon, site..... | 4-8 |
| 4.5 | Exposure dose results for the private wells south of the Lakeview, Oregon, site | 4-9 |
| 5.1 | Toxicity values: potential noncarcinogenic effects | 5-34 |
| 5.2 | Toxicity values: potential carcinogenic effects..... | 5-35 |
| 6.1 | Excess lifetime cancer risk calculations for ground water ingestion by a hypothetical future adult resident, Lakeview, Oregon, site | 6-14 |
| 7.1 | Comparison of estimated plant concentrations to phytotoxic concentrations, Lakeview, Oregon, site | 7-5 |

LIST OF TABLES (Concluded)

| <u>Table</u> | | <u>Page</u> |
|--------------|---|-------------|
| 7.2 | Comparison of contaminants of potential concern in ground water with available water quality values, Lakeview, Oregon, site | 7-7 |
| 8.1 | Constituent maximum concentration limits and health advisories of constituents..... | 8-5 |

LIST OF ACRONYMS

| <u>Acronym</u> | <u>Definition</u> |
|----------------|--|
| BCF | bioconcentration factor |
| DHHS | U.S. Department of Health and Human Services |
| DOE | U.S. Department of Energy |
| EPA | U.S. Environmental Protection Agency |
| FWS | 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 processing site was determined to be "surface clean" by U.S. Nuclear Regulatory Commission (NRC) approval of the completion report.

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 a ground water compliance strategy is selected, 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, 1994a). 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 constituents 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 constituents 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

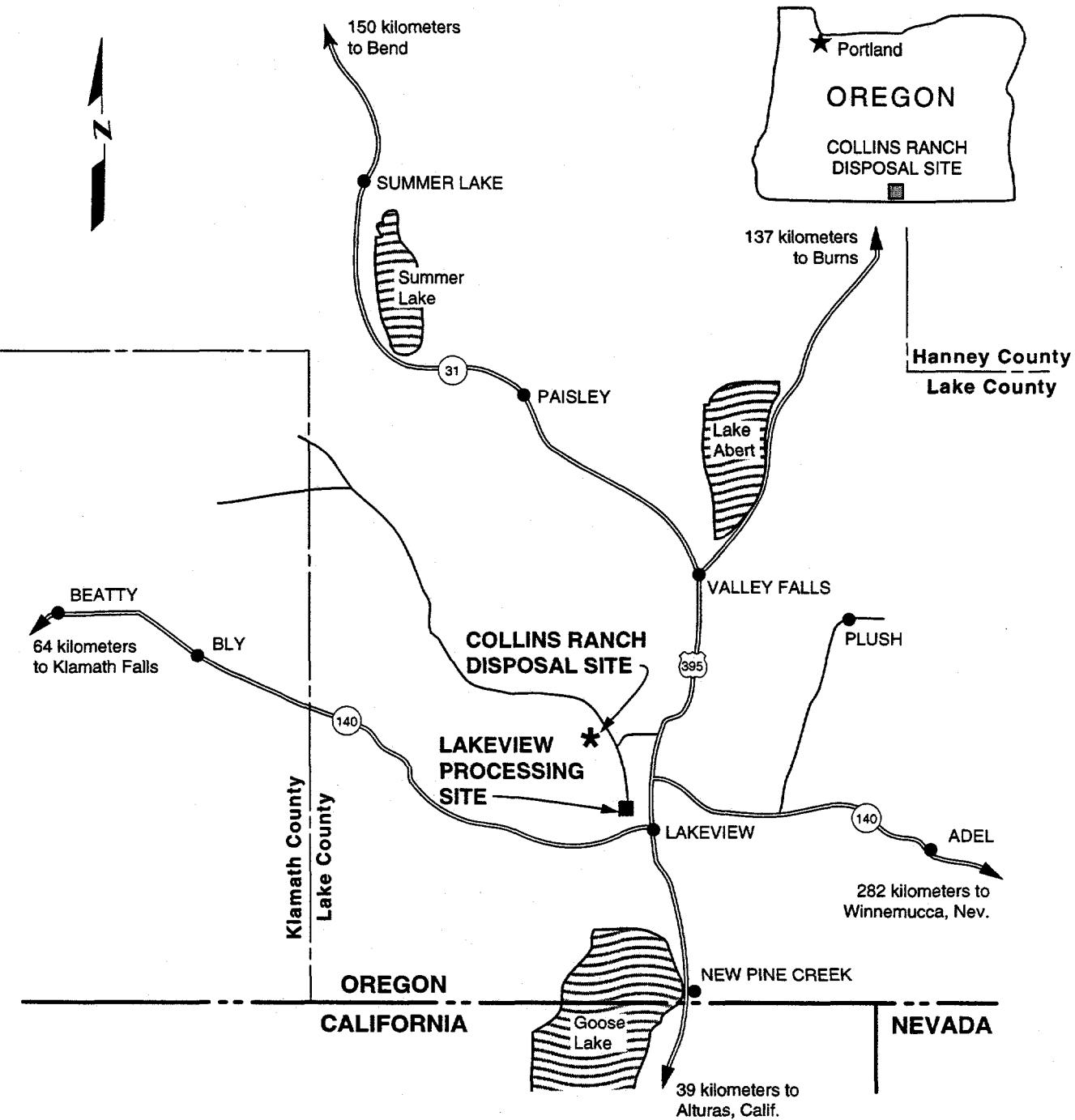
The Lakeview site is approximately 1.5 mile (mi) (2.4 kilometer [km]) north-northwest of the town of Lakeview in Lake County, Oregon (Figure 2.1). Klamath Falls is about 96 mi (150 km) west of the site and the California-Oregon state line is approximately 16 mi (26 km) south of the site. The Lakeview 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 and approximately 64 ac (26 ha) by the evaporation ponds; 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, 1994b). 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 NRC.

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 ($^{\circ}F$) (7.8 degrees Celsius [$^{\circ}C$]) and ranges from an



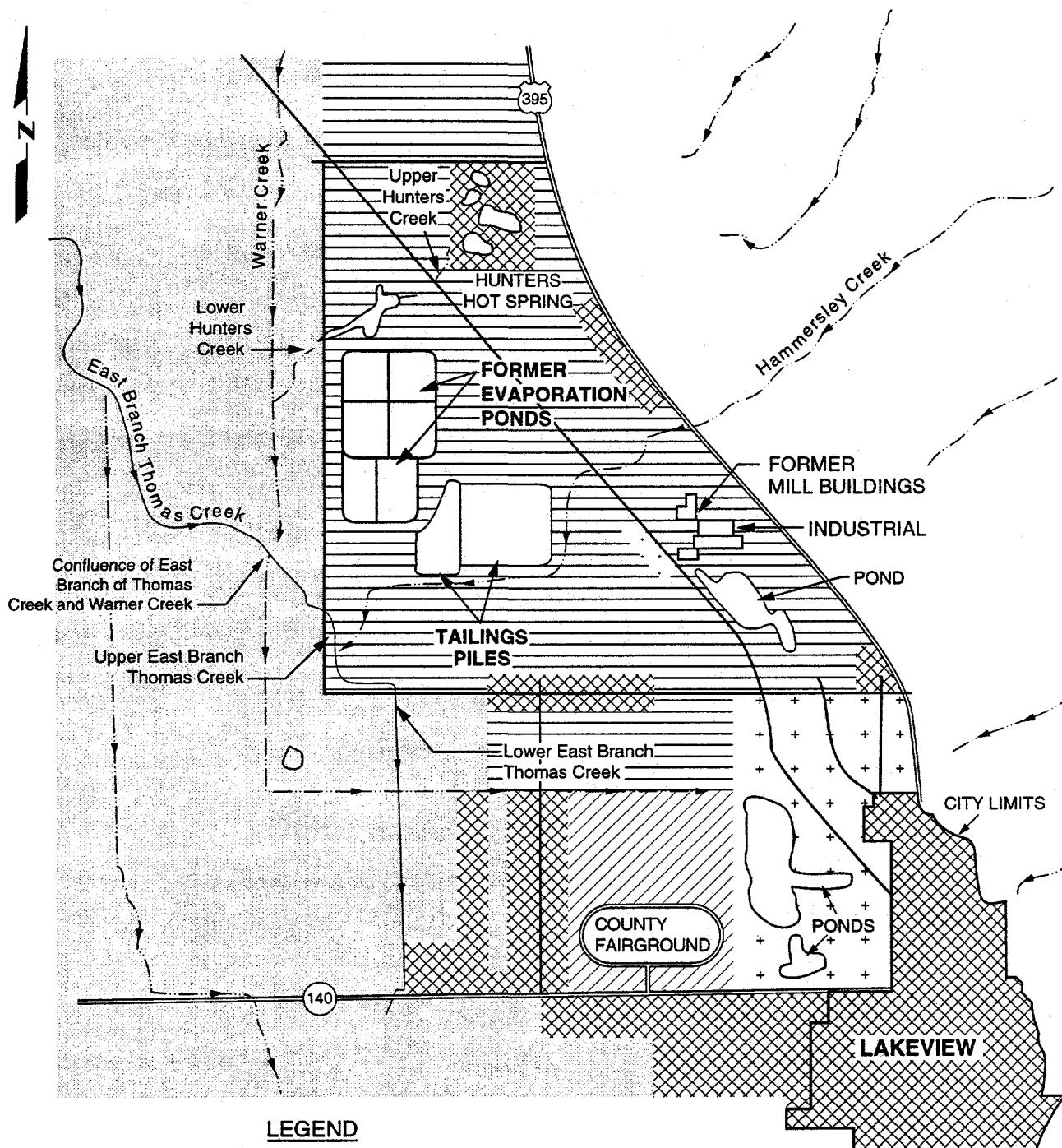
LEGEND

- U.S HIGHWAY
- STATE HIGHWAY
- STATE LINE
- COUNTY LINE

5 0 5 10 15 MILES

8 0 8 16 24 KILOMETERS

FIGURE 2.1
LOCATION MAP
LAKEVIEW, OREGON, SITE



LEGEND

| | |
|--|-----------------------------|
| | PASTURE/RANCHLAND |
| | RESIDENTIAL/COMMERCIAL |
| | LUMBER MILLS |
| | RANCHLAND/IRRIGATION FIELDS |
| | PUBLIC PROPERTY |
| | U.S. HIGHWAY |
| | STATE HIGHWAY |
| | EPHEMERAL STREAM OR DITCH |

1000 0 1000 2000 3000 FEET

500 0 500 1000 METERS

FIGURE 2.2
LAND USE
LAKEVIEW, OREGON, SITE VICINITY

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 (Figure 2.3). 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 2.4.

Regional ground water recharge occurs from mountains on the west (Fremont Mountains) and east (Warner Mountains) sides of Goose Lake valley, precipitation, snowmelt recharge from the unlined irrigation ditches, 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 2.5. 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).

Downward vertical gradients have been noted between the shallow and deep monitor well clusters (DOE, 1993). The background well cluster (501 and 502) shows head 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 similar, indicating no clear vertical gradient. Static water level elevations in

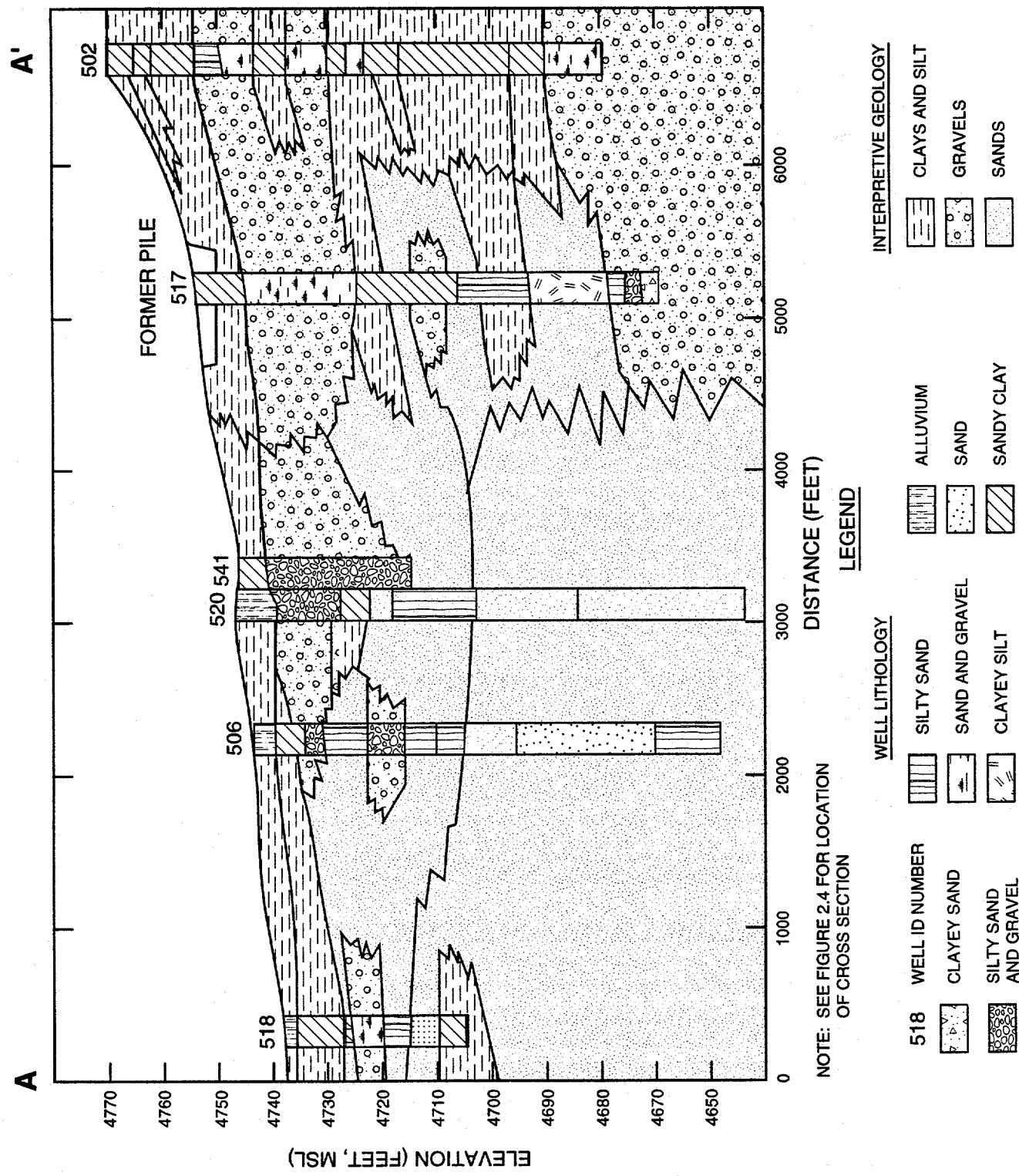


FIGURE 2.3
GEOLOGIC CROSS SECTION, FORMER LAKEVIEW, OREGON, PROCESSING SITE

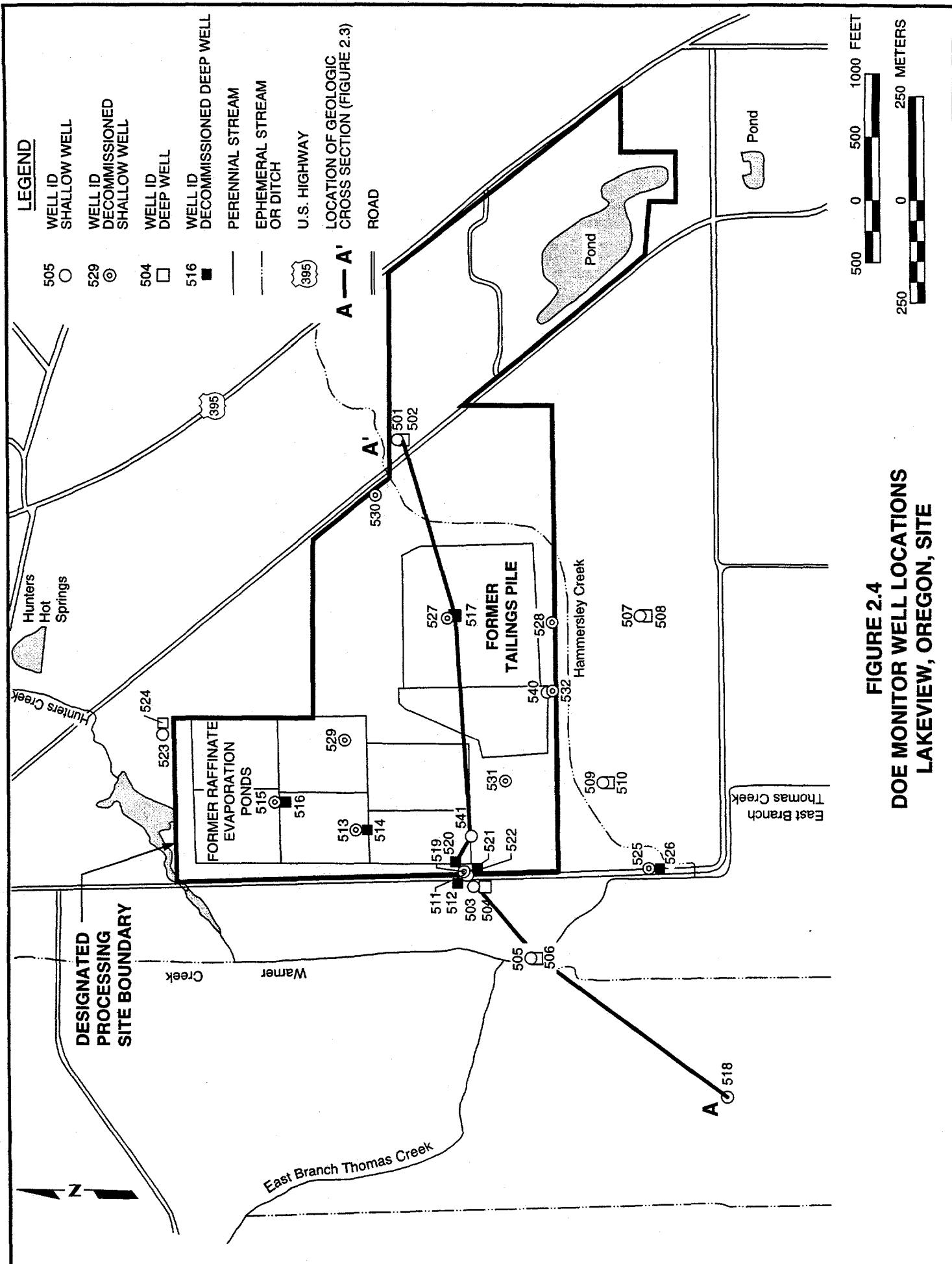


FIGURE 2.4
DOE MONITOR WELL LOCATIONS
LAKEVIEW, OREGON, SITE

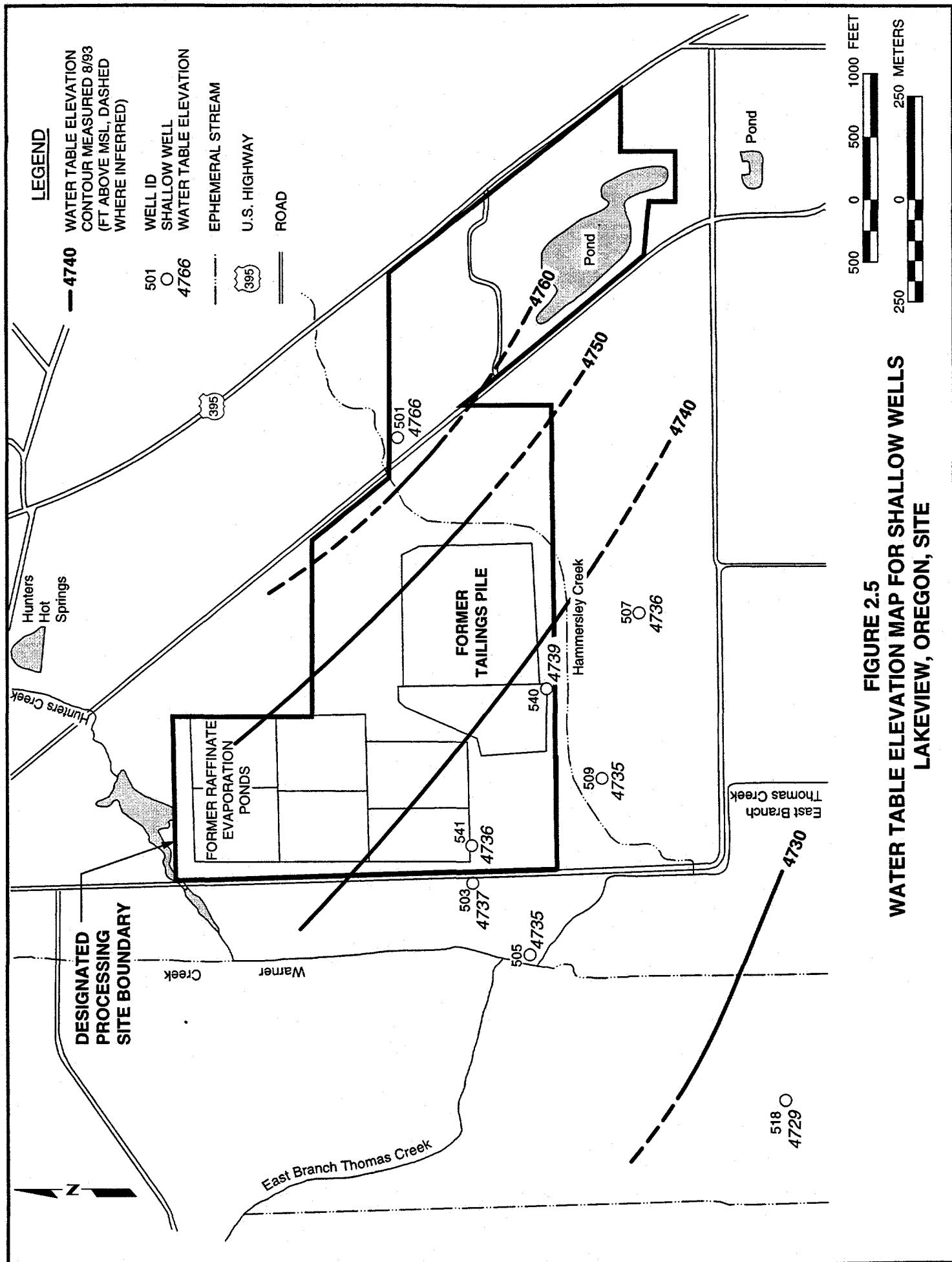


FIGURE 2.5
WATER TABLE ELEVATION MAP FOR SHALLOW WELLS
LAKEVIEW, OREGON, SITE

other downgradient monitor well clusters range from similar 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; hydraulic parameter 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 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 [cm³] 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 cm³ 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 average hydraulic conductivities for the shallow and deep zones, 1.4 and 4.4 ft (0.4 and 1.3 m) per day, respectively; an average hydraulic gradient of 0.01; 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 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

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 | 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 |
| LKV-01-0503 | 7931.9 | 3921.1 | SP | Downgradient | 4747.73 | 4745.89 | 08/25/93 | 15.54 | 10.99 | 9.15 |
| LKV-01-0505 | 7476.2 | 3342.9 | SP | Downgradient | 4744.64 | 4742.74 | 08/25/93 | 15.59 | 9.88 | 7.98 |
| LKV-01-0507 | 6599.3 | 6102.0 | SP | Crossgradient | 4742.77 | 4741.00 | 08/25/93 | 16.35 | 6.76 | 4.99 |
| LKV-01-0509 | 6897.1 | 4762.6 | SP | Downgradient | 4742.14 | 4740.06 | 08/25/93 | 16.47 | 7.36 | 5.28 |
| LKV-01-0518 | 5914.7 | 2231.0 | SP | Downgradient | 4739.79 | 4737.55 | 08/25/93 | 16.18 | 10.31 | 8.07 |
| LKV-01-0540 | 7340.20 | 5486.57 | AL | Downgradient | 4747.89 | 4745.93 | 08/25/93 | 15.20 | 9.09 | 7.13 |
| LKV-01-0541 | 7938.71 | 4223.85 | AL | Downgradient | 4747.56 | 4745.55 | 08/25/93 | 15.10 | 11.34 | 9.33 |

Report date: 1 July 1994.

SP – sand or gravelly sand, poorly graded.

AL – alluvium.

MSL – mean sea level.

(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 produce irrigated forage 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 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. These private wells are south of the site.

Figure 2.6 shows the locations of 15 private wells sampled by the UMTRA Project. 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 ditches 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 15 and 16. 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 constituents. 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. The Lakeview municipal wells comply with Oregon State Health Department wellhead protection regulations. These regulations include requirements that 1) wells must be installed with adequate surface seals to prevent contamination from surface waters, 2) well casings must extend at least 1 ft (0.3 m) above the floodplain, and 3) potential subsurface contamination sources are excluded within 100 ft (30 m) of the well.

Approximately 10 residences are along Missouri Avenue, about 1100 ft (340 m) south of the site. Another group of up to 10 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

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

| Location | Number of wells | Depth range | | Screen range | | |
|---------------------------------------|-----------------|-------------------------------------|---------|--------------|--------|--|
| | | (ft) | (m) | (ft) | (m) | |
| East of processing site | | | | | | |
| <i>Upgradient Section 3</i> | | | | | | |
| | 1 | 685 | 209 | None | None | |
| | 3 | 105-317 | 32-97 | 50-280 | 15-85 | |
| <i>Crossgradient Section 10</i> | | | | | | |
| | 4 | 155-310 | 47-94 | 70-310 | 21-94 | |
| | 5 | 58-603 | 18-184 | 38-603 | 12-184 | |
| <i>Downgradient Section 15</i> | | | | | | |
| 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 | | | | | | |
| <i>Upgradient Section 4</i> | | | | | | |
| 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 | |
| <i>Downgradient Section 9</i> | | | | | | |
| | 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 | |
| <i>Far downgradient Section 16</i> | | | | | | |
| 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) |
|------------------------------------|-----------------|------------------|-----------------|-------------------|------------------|
| <i>Far downgradient Section 16</i> | | | | | |
| (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 |
| <i>West of processing site</i> | | | | | |
| <i>Crossgradient Section 5</i> | | | | | |
| | 1 | 165 | 50 | 40-95 | 12-29 |
| | 1 | 130 | 40 | NA | NA |
| | 3 | 48-223 | 15-68 | 48-210 | 15-64 |
| <i>Downgradient Section 8</i> | | | | | |
| 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.

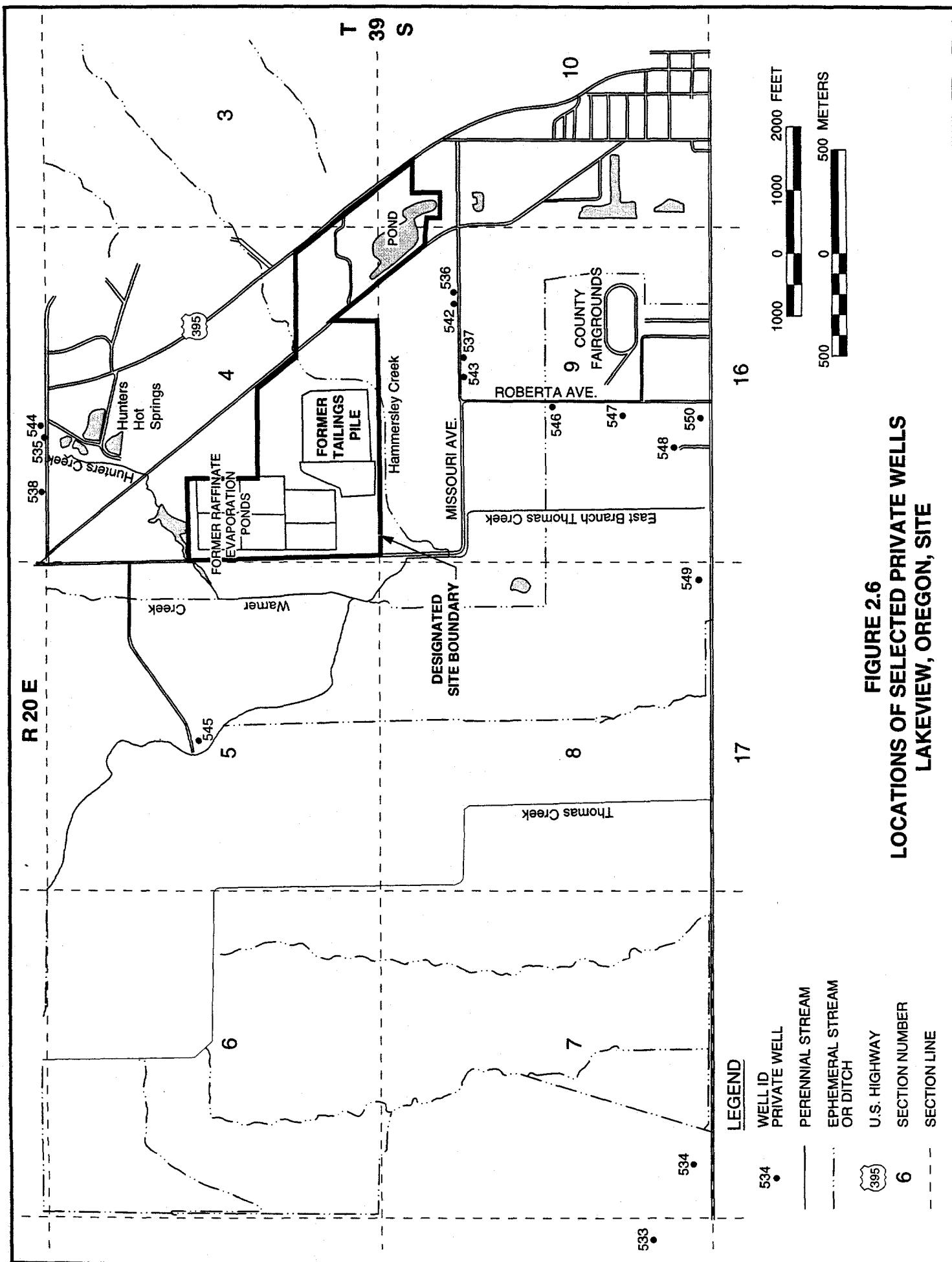


FIGURE 2.6
LOCATIONS OF SELECTED PRIVATE WELLS
LAKEVIEW, OREGON, SITE

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.) (Figure 2.6). Bottled water is used for drinking.



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 2.4 and 2.6, 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 the 1989 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.

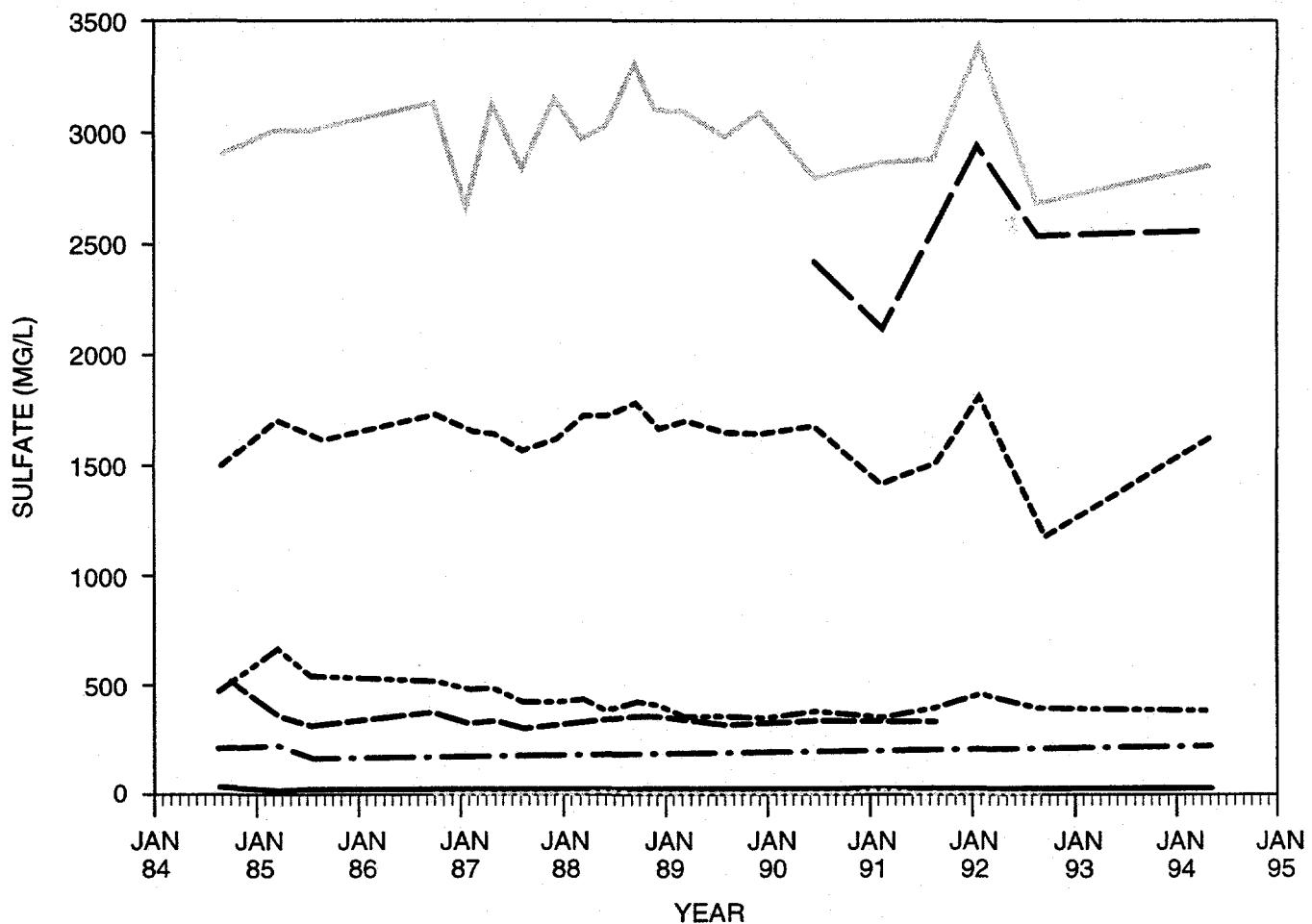
With regard to background, the use of data collected both before and after 1988 allows a more rigorous statistical evaluation of background water quality. With regard to contaminated ground water, prior to surface remediation (1988 to 1989) the greatest concentrations of some site-related constituents were located beneath the evaporation pond and tailings pile. Monitor wells in these areas were removed during surface remediation of the site and water quality data are not available for these wells after 1988.

In the period following 1989, concentrations observed in areas beneath the tailings piles and evaporation ponds may have decreased. This is because constituents in ground water may have been diluted and dispersed after surface remediation. However, a review of data immediately downgradient of the evaporation pond and tailings pile areas has not supported this supposition. Constituent concentrations have remained relatively constant downgradient of the site (for example, sulfate) (Figure 3.1). Without information to support a decrease of concentrations beneath the site, water quality data collected prior to 1988 were used to characterize the possible range of site-related contamination. These data may overestimate current concentrations of ammonium, boron, chloride, iron, molybdenum, potassium, sodium, strontium, and sulfate. Therefore, while the highest historical levels of site-related constituents have been used to determine potential risk, information on the maximum observed concentrations in downgradient wells is also provided (Table 3.2).

An additional possible limitation to the use of older (pre-1988) data is that laboratory analytical methods and quality control have improved over time. However, a review of all data from the site indicates the analytical data collected since 1984 have been consistent through time, and there is no reason to expect analytical errors in the older data. Note that this is not true of data collected prior to 1984, and these data have not been incorporated into this risk assessment.

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

| Location ID | Depth | | Screen | | Sampling date | Number of rounds |
|---|-------|------|--------|-----------|------------------|------------------|
| | (ft) | (m) | (ft) | (m) | | |
| 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 |



LEGEND

| | |
|----------------|------------|
| WELL 503 | — WELL 518 |
| ----- WELL 505 | — WELL 523 |
| — WELL 507 | — WELL 540 |
| — WELL 509 | |

FIGURE 3.1
SULFATE CONCENTRATIONS
VERSUS TIME IN SELECTED WELLS
LAKEVIEW, OREGON, PROCESSING SITE VICINITY

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

| Parameter ^c | Frequency of detection ^d | Observed concentration ^a | | |
|-------------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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)

| Parameter ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|---------------------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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) ^e | 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 | | | | |
| | | | | |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|--------------------------------|---------|
| | | Minimum | Median ^b (pCi/L) | Maximum |
| 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 |

^aTrend analyses with time indicate virtually no change in the chemistry of ground water in 12 wells (both upgradient and downgradient of the site) sampled from 1984 through 1993. Therefore, all data available for that period are used for statistical evaluation of ground water quality. However, no recent data (post-1988) are available for wells decommissioned during surface remediation. Thus, present concentrations in the area of wells 521, 513, and 529 may differ from those listed.

^bThe 50th percentile of the data. When only two data are available, the reported median is the arithmetic average of the two.

^cNongeothermal 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.

^dNumber of samples with concentrations reported above detection limit/total number of samples.

^eUranium 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.

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-210, radium-226, and thorium-230.

Organic solvents were used in the milling process (Merritt, 1971). In November 1988, ground water from the Lakeview processing site was screened for organic constituents (Hill, 1989; DOE, 1992) listed in Appendix IX of 40 CFR Part 264 of the Resource Conservation and Recovery Act. This screening included ground water samples from an upgradient background monitoring well (501) and a shallow well (503) immediately downgradient of the evaporation pond area. None of the organic constituents listed in Appendix IX were detected in samples from these wells.

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.

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).

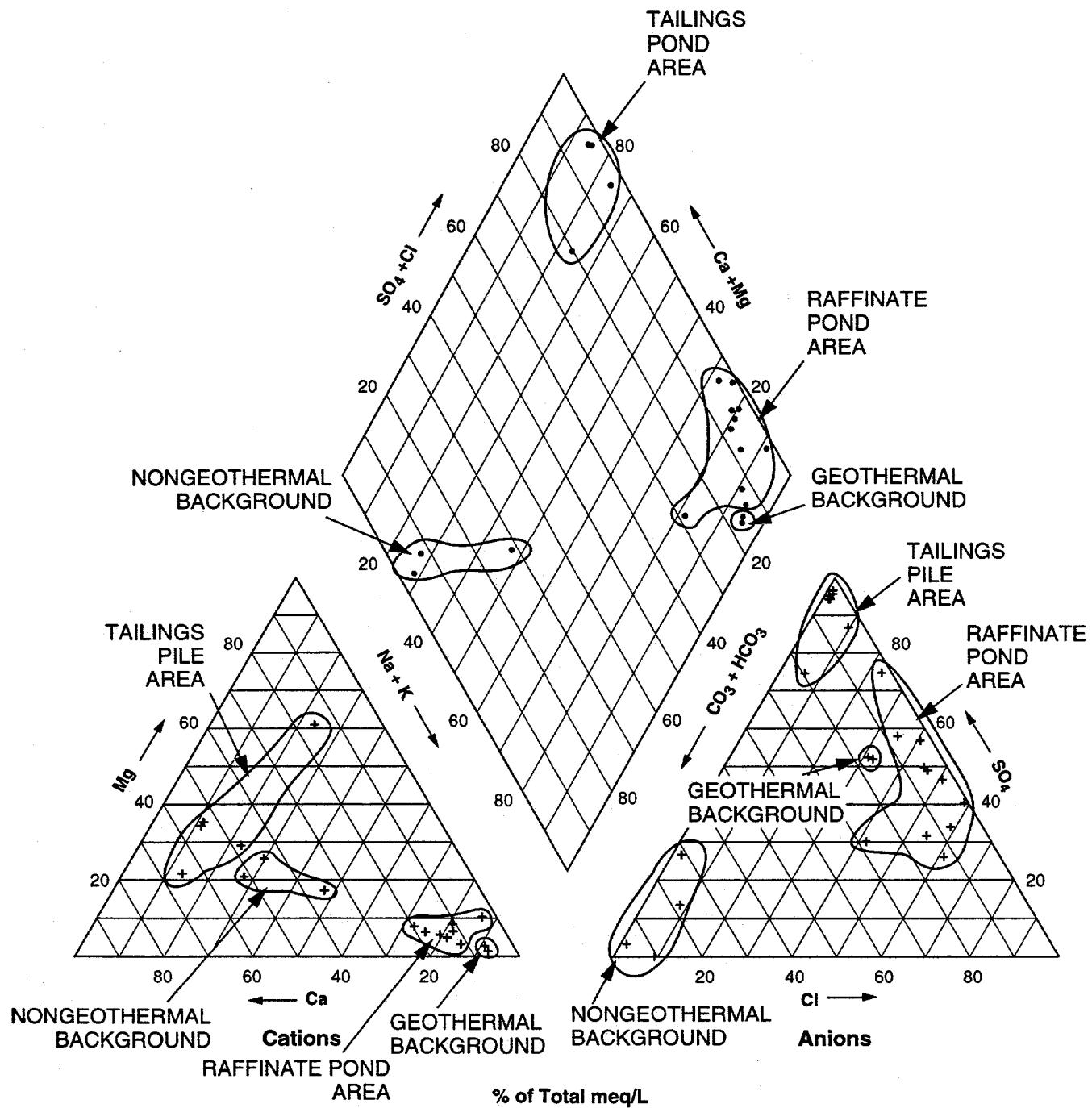
The water is sodium-calcium-magnesium-bicarbonate water, with notably low concentrations of chloride (1 to 5 mg/L) and sulfate (<1 to 14 mg/L). In contrast, the geothermal ground water has relatively high TDS, ranging from 657 to 714 mg/L, and is sodium-sulfate-chloride water, notably depleted in calcium 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 3.2).

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 3.2). 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 effects of geothermal activity on regional ground water quality in the Lakeview area has been studied (Brown et al., 1980). There are over 20 thermal springs and wells in the Lakeview area (Goose Lake basin). Seven of the thermal springs and wells studied and sampled were between 0.2 to 2 miles (0.3 to 3 km) upgradient of the site. Arsenic concentrations in ground water from these seven locations were notable, ranging from <0.005 to 0.162 mg/L, with the highest values occurring in Hunters Hot Springs and nearby thermal wells, just north of the site. Ground water in six of these seven thermal springs and wells upgradient of the site exceed the UMTRCA maximum concentration limit (MCL) for arsenic (0.05 mg/L) and the National Primary Drinking Water Regulation for arsenic (also 0.05 mg/L).

Arsenic also occurs in geothermal background ground water sampled from DOE well 524. Arsenic concentrations in this well, completed in the deeper zone, have ranged from 0.04 to 0.11 mg/L (Table 3.3). These values are similar to arsenic values reported in the thermal wells and springs upgradient of the site. By contrast, arsenic levels in geothermal background well 523 have been low (<0.01 mg/L) (Table 3.2) and are anomalous when compared to nearby and upgradient wells and springs. The reason for this anomaly is not known. There are no other notable differences between water quality in geothermal background wells 524 and 523.

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



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 3.2
TRILINEAR DIAGRAM OF MAJOR GROUND WATER CHEMISTRIES
LAKEVIEW, OREGON, SITE

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

| Parameter ^c | Frequency of detection ^d | Observed concentration ^a | | |
|-------------------------------|-------------------------------------|-------------------------------------|---------------------|---------|
| | | Minimum | Median ^b | Maximum |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|-------------------------------|---------|
| | | Minimum | Median ^b (mg/L) | Maximum |
| 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 ^c | Frequency of detection ^d | Observed concentration ^a | | |
|--------------------------|-------------------------------------|-------------------------------------|--------------------------------|---------|
| | | Minimum | Median ^b (pCi/L) | Maximum |
| 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 |

^aTrend analyses with time indicate virtually no change in the ground water chemistry in 12 wells (both upgradient and downgradient of the site) sampled from 1984 through 1993. Therefore, all data available for that period are used for statistical evaluation of ground water quality. However, no recent data (post-1988) are available for wells decommissioned during surface remediation. Thus, present concentrations in the area of wells 521, 513, and 529 may differ from those listed.

^bThe 50th percentile of the data. When only two data are available, the reported median is the arithmetic average of the two.

^cNongeothermal 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.

^dNumber of samples with reported concentrations above detection limit/total number of samples.

Dash indicates only one measurement.

pCi/L – picocuries per liter.

rigorous statistical comparison to contaminated ground water. Therefore, all downgradient ground water data initially were compared to the nongeothermal ground water to determine constituents 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.

3.2 MAGNITUDE AND EXTENT OF SITE-RELATED CONSTITUENTS

3.2.1 Tailings and raffinate chemistry

Uranium ores were obtained from the White King and Lucky Lass mines, located about 15 mi (24 km) northwest of the mill, and hauled to the mill by truck for processing. The uranium ore deposits near Lakeview were formed by hydrothermal alteration of volcanic rhyolites and tuffs. Ores at the White King mine contained uraninite (UO_2), coffinite ($\text{U}(\text{Si},\text{H}_4)\text{O}_4$) and other secondary uranium minerals in a silicified volcanic breccia. These ores were associated with other hydrothermal minerals including galena (lead sulfide), realgar and orpiment (arsenic sulfides), stibnite (antimony sulfide) and cinnabar (mercury sulfide) (Walker, 1980). The Lucky Lass Mine primarily contained secondary uranium minerals with some uraninite or coffinite. However, none of the other metallic minerals found at the White King mine were found at the Lucky Lass Mine (Walker, 1980).

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 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 ground water show relative ion proportions similar to their respective background water types (Figure 3.2). 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 ground water are 10 to 20 times greater than those in the background ground 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 ground 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 hypothesis 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 ground 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 the constituents 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 the constituents, 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 (Figure 3.1).

The sulfate distribution indicates two constituent 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 3.3). This finding is consistent with the chemical identification of two types of contaminated ground water. Sulfate 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, constituents in the shallow

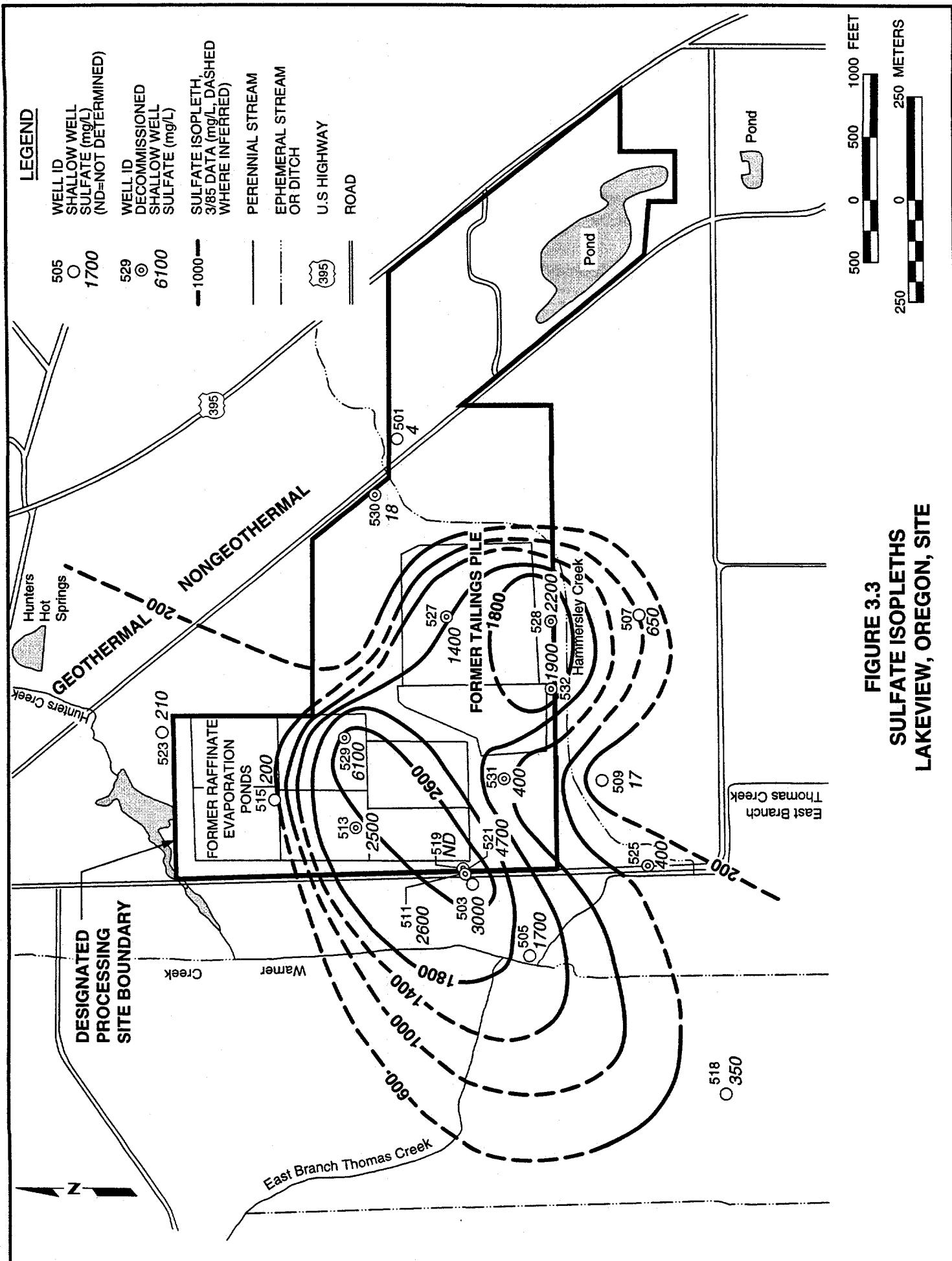


FIGURE 3.3
SULFATE ISOPLETHS
LAKEVIEW, OREGON, SITE

portion of the aquifer (0 to 40 ft [0 to 10 m]) extends beyond existing monitor wells and off the site (Figure 3.3). However, sulfate and other major constituent 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 the constituents appears to be limited to depths of less than 80 ft (20 m). An intermediate depth 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 geothermal background (200 mg/L) may indicate that a small amount of constituents has reached a depth of 80 ft (24 m) in monitor wells 504 and 506.

3.2.3 Statistical comparison of background and plume ground water quality

After determining the extent of the constituents (using sulfate as an indicator), water quality data collected between 1984 and 1993 were used to statistically identify the full suite of constituents elevated above background levels in 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 the constituents. 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 constituents 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 constituent 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). Constituents 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 a site-related constituent.

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 3.3). 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 site-related constituent.

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 ground water is the nongeothermal ground water, which has relatively low silica concentrations. Therefore, silica is considered a site-related constituent 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 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 1983-1985 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 or may have decreased, but are not accessible to current wells. Site-related constituents 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 15 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 2.6. 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 ground water quality (discussed in Section 3.1).

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

| Location ID | Depth | | Sampling dates | Number of sampling rounds |
|-------------|---------|-----|------------------------|---------------------------|
| | (ft) | (m) | | |
| 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 | <0.04 | 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 | 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 | 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 pH (standard units) and temperature.

| | | | |
|-----|------------|------------------|------------------------|
| 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 |

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

Four of five private wells due south of the site have elevated sulfate concentrations (locations 547, 548, 549, and 550), which could indicate constituents from the site have reached these wells. However, the areal pattern of sulfate distribution in these wells is inconsistent with site-related constituents. 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 3.4). Sulfate 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 constituents. Therefore, the private wells with high sulfate concentrations apparently are not affected by site-related constituents. 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 constituents 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 constituents, 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 constituents) 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

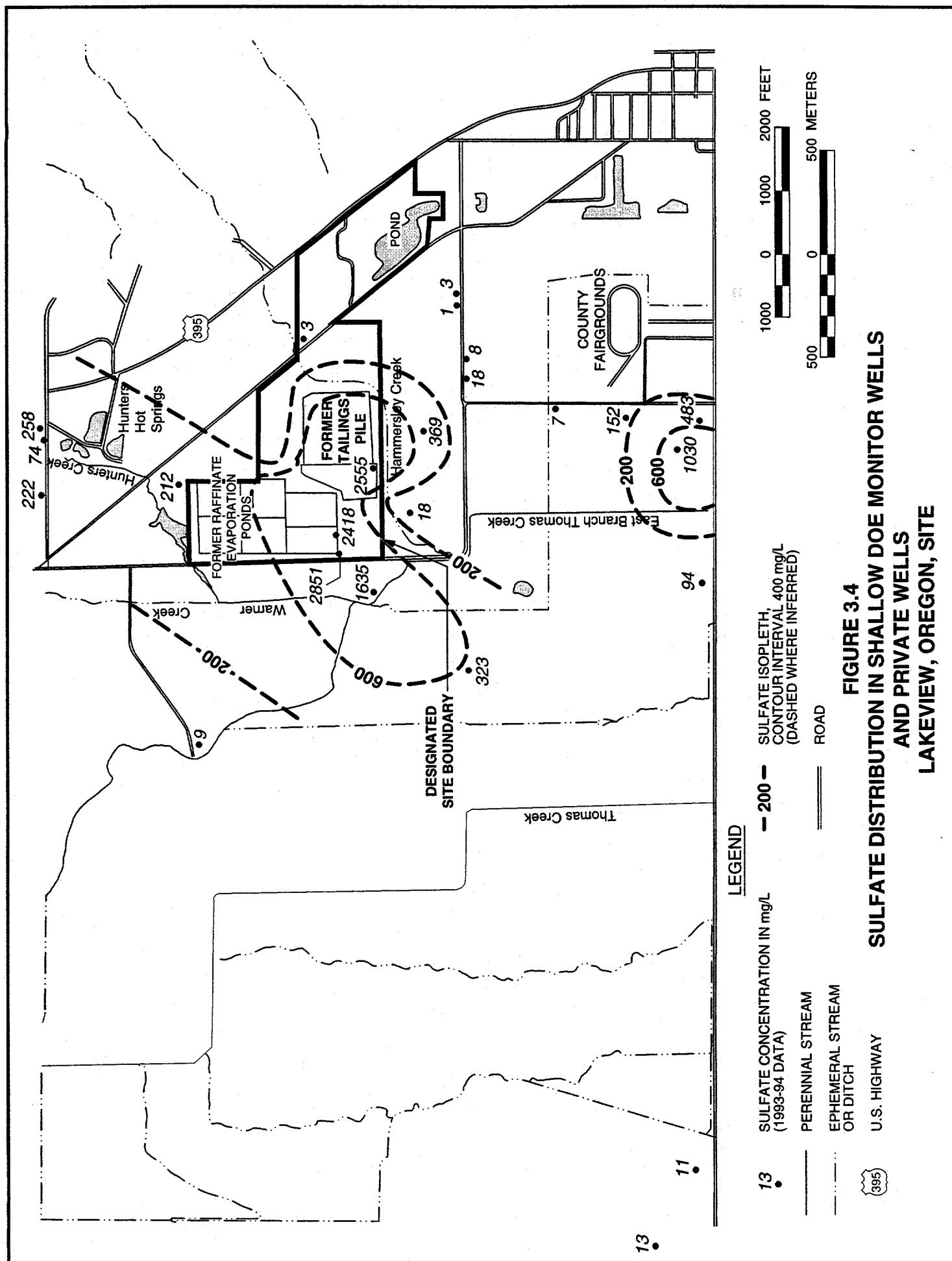


FIGURE 3.4
SULFATE DISTRIBUTION IN SHALLOW DOE MONITOR WELLS
AND PRIVATE WELLS
LAKEVIEW, OREGON, SITE

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.

3.4 CONSTITUENTS OF POTENTIAL CONCERN

The data presented in Sections 3.1 and 3.2 were used to compile a list of constituents of potential concern for this assessment of human health risks at the Lakeview site. A constituent was placed on the list of constituents 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 constituents in column 1 of Table 3.6 (constituents 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 constituents 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 constituents 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 constituents 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 constituents.

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

3.5 CONSTITUENT FATE AND TRANSPORT

Constituent 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 constituents 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, constituent dilution and dispersion in conjunction with ground water flow are cited as a major control on the fate of constituents in ground water at the site. The current rate of ground water transport of constituents appears to be slow, as indicated by little or no change in constituent concentrations downgradient of the site over the last 12 years.

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

| Constituents that exceed background levels | Constituents in nutritional range | Constituents of low toxicity and/or high dietary range | Constituents 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 | | |

^aAlthough 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 constituents 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 | $\text{H}_3\text{BO}_3 \text{ aq}$ | 100 | B(III) |
| Chloride | Chloride | Cl^- | 100 | Cl(I) |
| Iron | Ferric hydroxide | FeOH^{2+} | 96 | Fe(III) |
| | Ferric hydroxide | $\text{Fe(OH)}_3 \text{ aq}$ | 2 | Fe(III) |
| Manganese | Manganese | Mn^{2+} | 68 | Mn(II) |
| | Manganese sulfate | $\text{MnSO}_4 \text{ aq}$ | 26 | Mn(II) |
| Molybdenum | Molybdate | MoO_4^{2-} | 100 | Mo(VI) |
| Nickel | Nickel carbonate | $\text{NiCO}_3 \text{ aq}$ | 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 | $\text{CaSO}_4 \text{ aq}$ | 8 | S(VI) |
| Uranium | Uranyl hypophosphate | $\text{UO}_2(\text{HPO}_4)_2^{2-}$ | 99 | U(VI) |

During that period, for example, sulfate concentrations remained relatively constant in downgradient well 518 near the edge of the contaminated ground water. Nonetheless, the presence of site-related constituents in this downgradient well indicates ground water flow has occurred at the site. The apparent contradiction between the extent of constituents and the lack of time-dependent trends in constituent concentrations in individual wells represents a data gap in both the understanding of the hydrologic regime at the site and the prediction of constituent fate and transport. Without constituent 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.

This is supported by sulfur isotopic studies that indicate that the geothermal system is the primary source of sulfate at distances greater than 600 ft (180 m) from the site or depths greater than 25 ft (7.6 m) (DOE, 1992).

Uranium

Uranium in the contaminated ground 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 ($\text{UO}_2(\text{HPO}_4)_2^{2-}$). 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 five locations (Figure 3.5). 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). Sediment 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 constituents analyzed in the surface water samples that were above background levels in ground water.

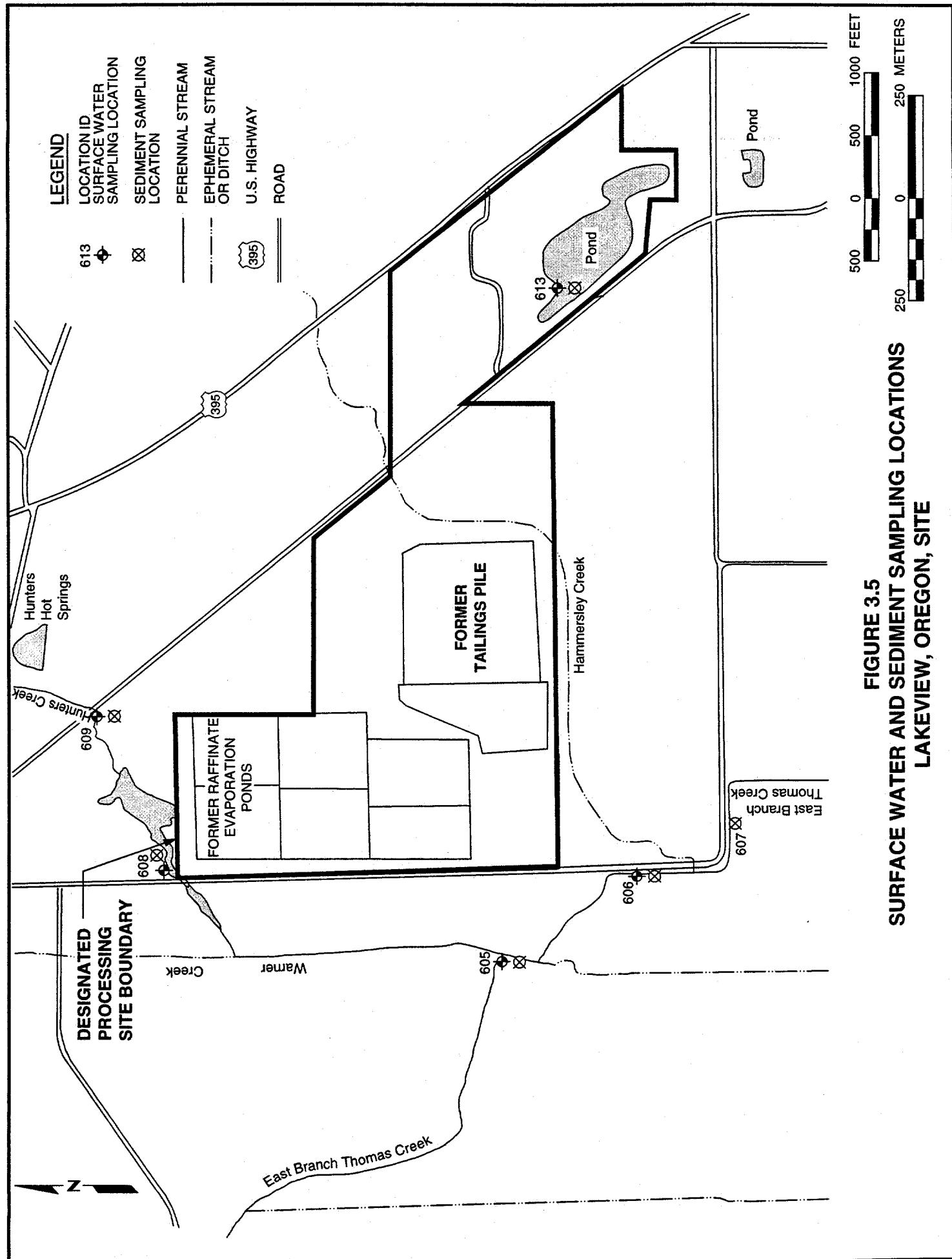


FIGURE 3.5
SURFACE WATER AND SEDIMENT SAMPLING LOCATIONS
LAKEVIEW, OREGON, SITE

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

| Constituent | Upstream of site | | Adjacent to site | | Pond location |
|-------------|---------------------|-------------|------------------|-------------|---------------|
| | 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.

Surface water data from these locations show that most of the constituent levels found at the adjacent locations were no greater than their respective 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 3.4). 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 constituents analyzed in the sediment samples (tin, vanadium, and zinc are not considered site-related constituents). 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 constituents.

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

| Constituent | Upstream of site | | Adjacent to site | | Downstream of site | | Pond |
|-------------|------------------|-------------|------------------|-------------|--------------------|-------------|------|
| | 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.



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 constituent takes from the source to the exposed individual. To complete a pathway there must be a source of contamination, a release mechanism for the constituent (e.g., constituents 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 constituents of potential concern at UMTRA Project sites, direct exposure through drinking water ingestion is expected to result in the most significant constituent 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 constituent 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 forage 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 constituents. Figure 4.1 presents the conceptual site model of potential exposure pathways at the Lakeview site.

Several of the constituents 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

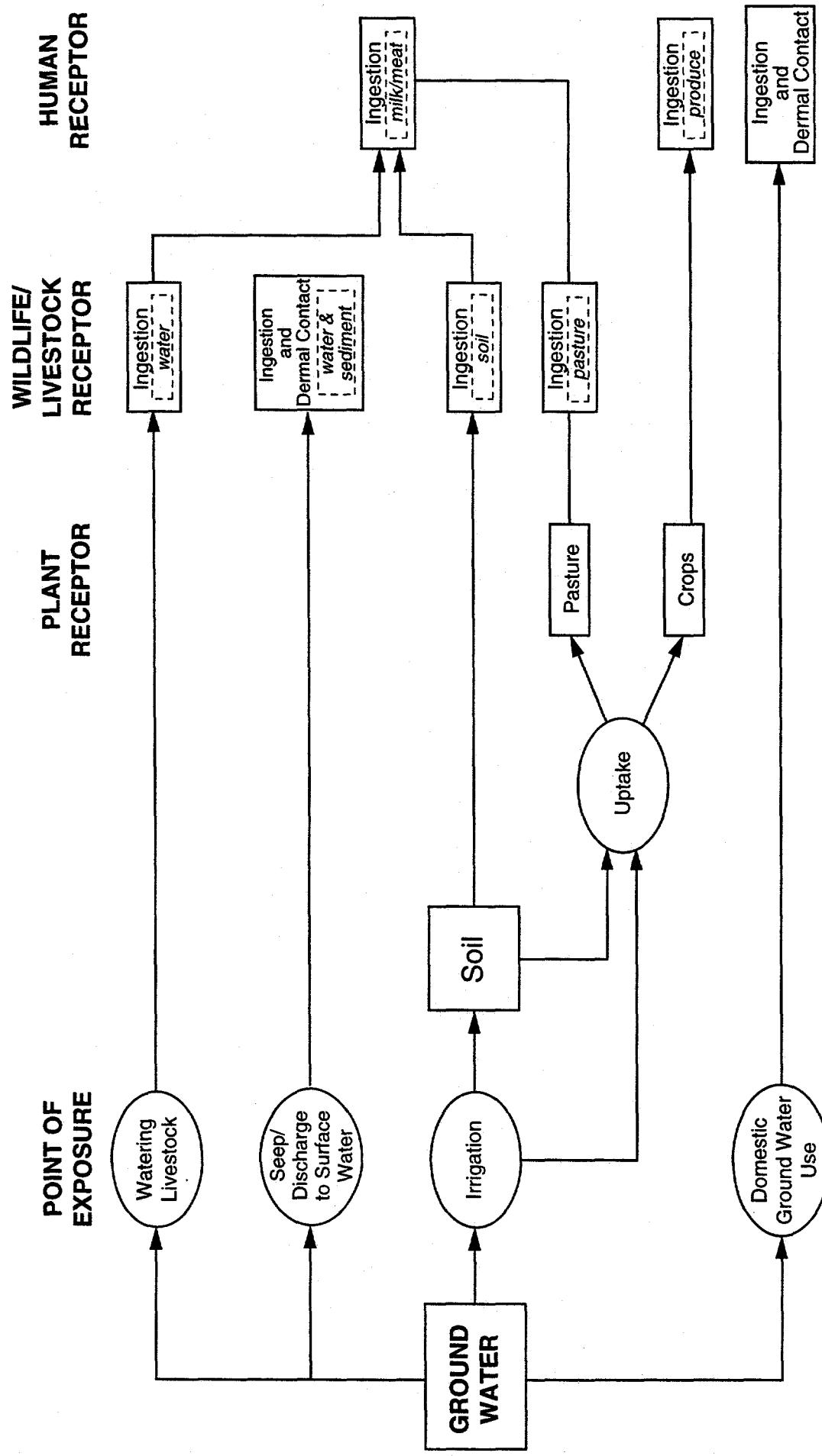


FIGURE 4.1
CONCEPTUAL MODEL
LAKEVIEW, OREGON, SITE

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 constituent 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 constituents of potential concern. The constituents 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 constituents of potential concern at this site, ingesting garden

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

| Constituents 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 = Constituent 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/cm³).
- ET = Exposure time (0.2 hour per day).

^aCalculated by dividing the dermal contact exposure dose by the ground water ingestion exposure dose.

^bMeasured in picocuries per lifetime.

^cUranium-234 and uranium-238 combined.

UCL - 95th upper confidence limit on the mean constituent 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

| Constituent of potential concern | C _w (mg/L) | K _d (L/kg) | B _v | B _r | 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) = $\frac{C_w \times K_d \times B_v \text{ or } B_r^e \times DF \times IR_p \times FI \times EF \times ED}{BW \times AT}$

Radionuclides: Lifetime intake (pCi/lifetime) = $C_w \times K_d \times B_v \text{ or } B_r^e \times DF \times IR_p \times FI \times EF \times ED$

Where:

C_w = Constituent concentration in ground water (95th UCL).

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

B_v = Soil-to-plant concentration ratio for vegetative portions of plants (unitless).

B_r = Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless).

DF = Dry weight fraction of plant (0.066 unitless).

IR_p = 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).

^aExposure doses shown are the sum of the vegetative parts plus the reproductive parts.

^bCalculated by dividing the garden produce ingestion exposure dose by the ground water ingestion exposure dose.

^cMeasured in picocuries per lifetime.

^dValue cannot be calculated because K_d is equal to zero.

^eExposure 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 constituent 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 these constituents 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 constituent 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

| Constituent 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**

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

$$\text{Radionuclides: Lifetime intake (pCi/lifetime)} = Cm \times IRm \times FI \times EF \times ED$$

Where:

Cm = Constituent concentration in milk (mg/L), estimated using the following equation:

$$Cm = Fm \times [(Qp \times Cp) + (Qs \times Cs) + (Qw \times Cw)]$$

where

Fm = Feed-to-milk transfer coefficient (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 = Constituent concentration in pasture (mg/kg); Cp = Kd x Cw x Bv.

Cs = Constituent concentration in soil (mg/kg); Cs = Kd x Cw.

Cw = Constituent 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).

^aKd value was not available for chloride; therefore, it has been omitted.^bCalculated by dividing the milk ingestion exposure dose by the ground water ingestion exposure dose.^cMeasured in picocuries per lifetime.^dValue cannot be calculated because Kd is equal to zero.

UCL - 95th upper confidence limit on the mean constituent 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

| Constituent of potential concern | C _w (mg/L) | K _d (L/kg) | B _v | F _f | 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) = $\frac{C_b \times IR_b \times FI \times EF \times ED}{BW \times AT}$

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

Where:

C_b = Constituent concentration in beef (mg/kg), estimated using the following equation:

C_b = F_f x ([Q_p x C_p] + [Q_s x C_s] + [Q_w x C_w])

where

F_f = Feed-to-flesh transfer coefficient (kilograms per day).

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

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

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

C_p = Constituent concentration in pasture (mg/kg). C_p = K_d x C_w x B_v.

C_s = Constituent concentration in soil (mg/kg). C_s = K_d x C_w.

C_w = Constituent concentration in ground water (95th UCL).

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

B_v = 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).

IR_b = Ingestion rate of meat (0.075 kg per day).

FI = Fraction of meat ingested from constituent source (1.0 unitless).

^aCalculated by dividing the meat ingestion exposure dose by the ground water ingestion exposure dose.

^bMeasured in picocuries per lifetime.

^cValue cannot be calculated because K_d is equal to zero.

UCL - 95th upper confidence limit on the mean constituent 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 |

^aCalculated 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.

- Notes: 1. Values are reported in milligrams per kilogram per day.
2. 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 constituent in ground water is defined as the concentration an individual is assumed to take in over a specific period. In this evaluation, the constituent concentrations are assumed to be in a steady state, although actual constituent concentrations (and therefore exposures) are expected to decrease with time 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 constituent. 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 constituent 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 constituent 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 constituents 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 constituent, 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 constituents 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 constituents 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

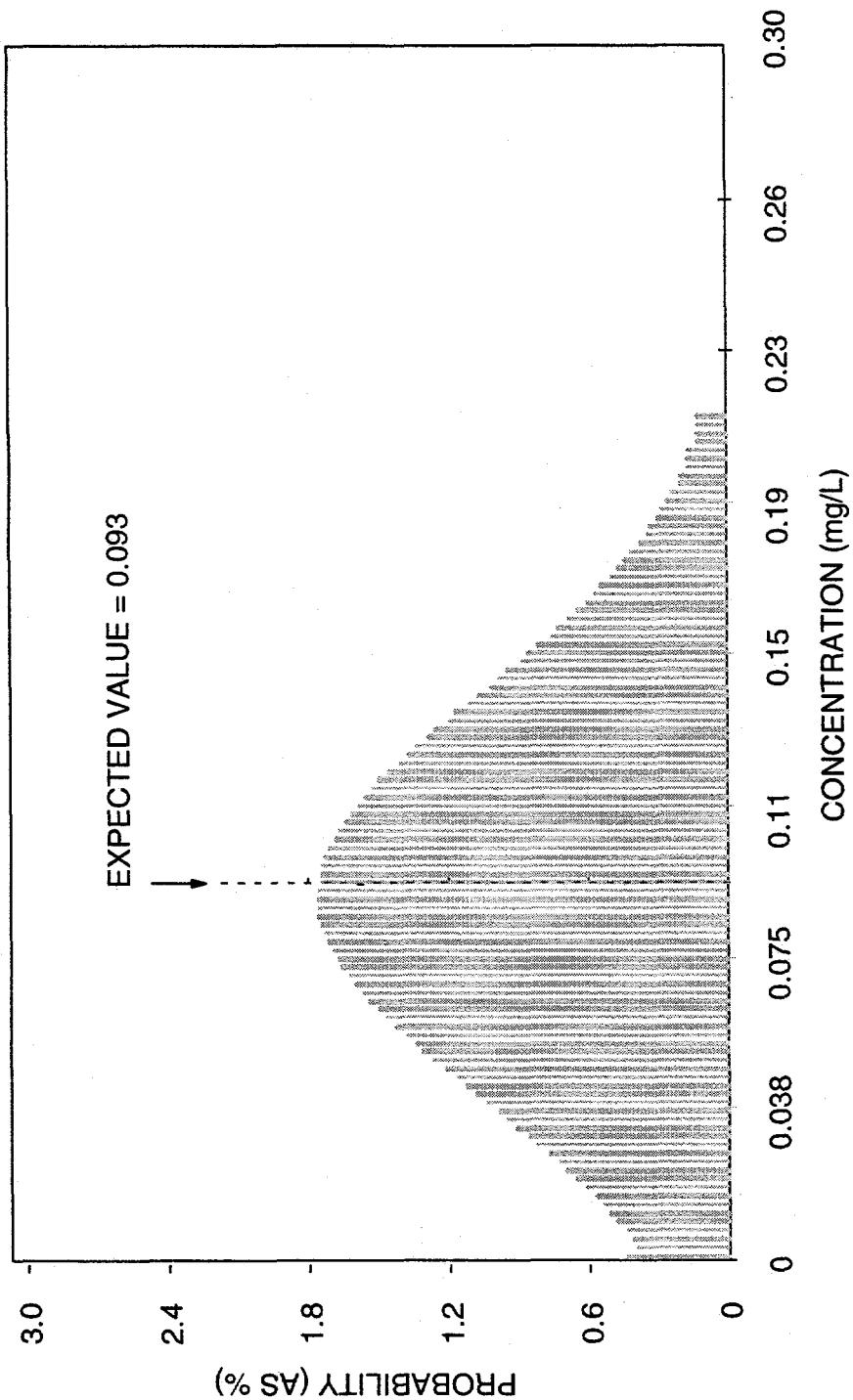


FIGURE 4.2
SIMULATED DISTRIBUTION OF ARSENIC CONCENTRATIONS
LAKEVIEW, OREGON, SITE

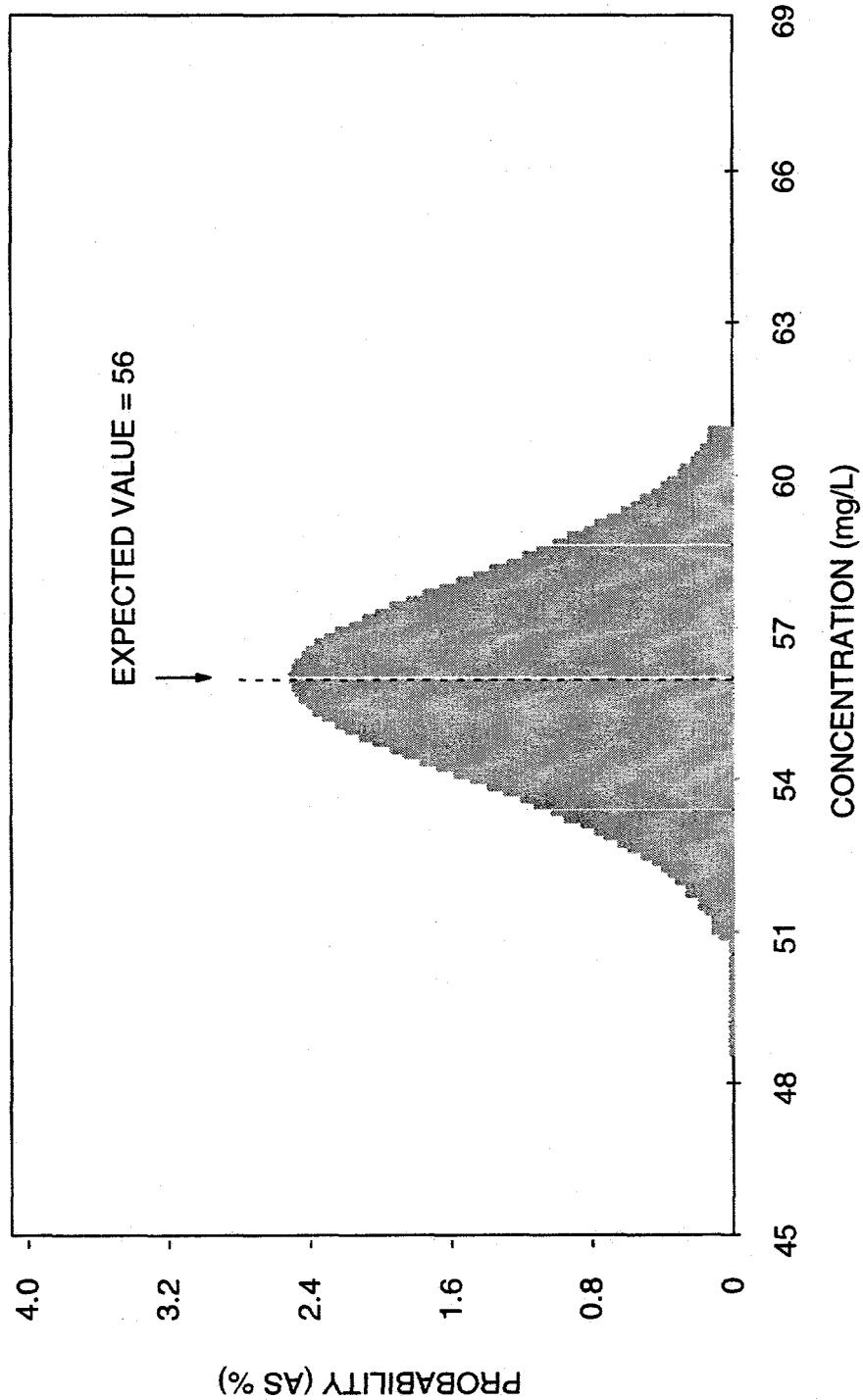


FIGURE 4.3
SIMULATED DISTRIBUTION OF BORON CONCENTRATIONS
LAKEVIEW, OREGON, SITE

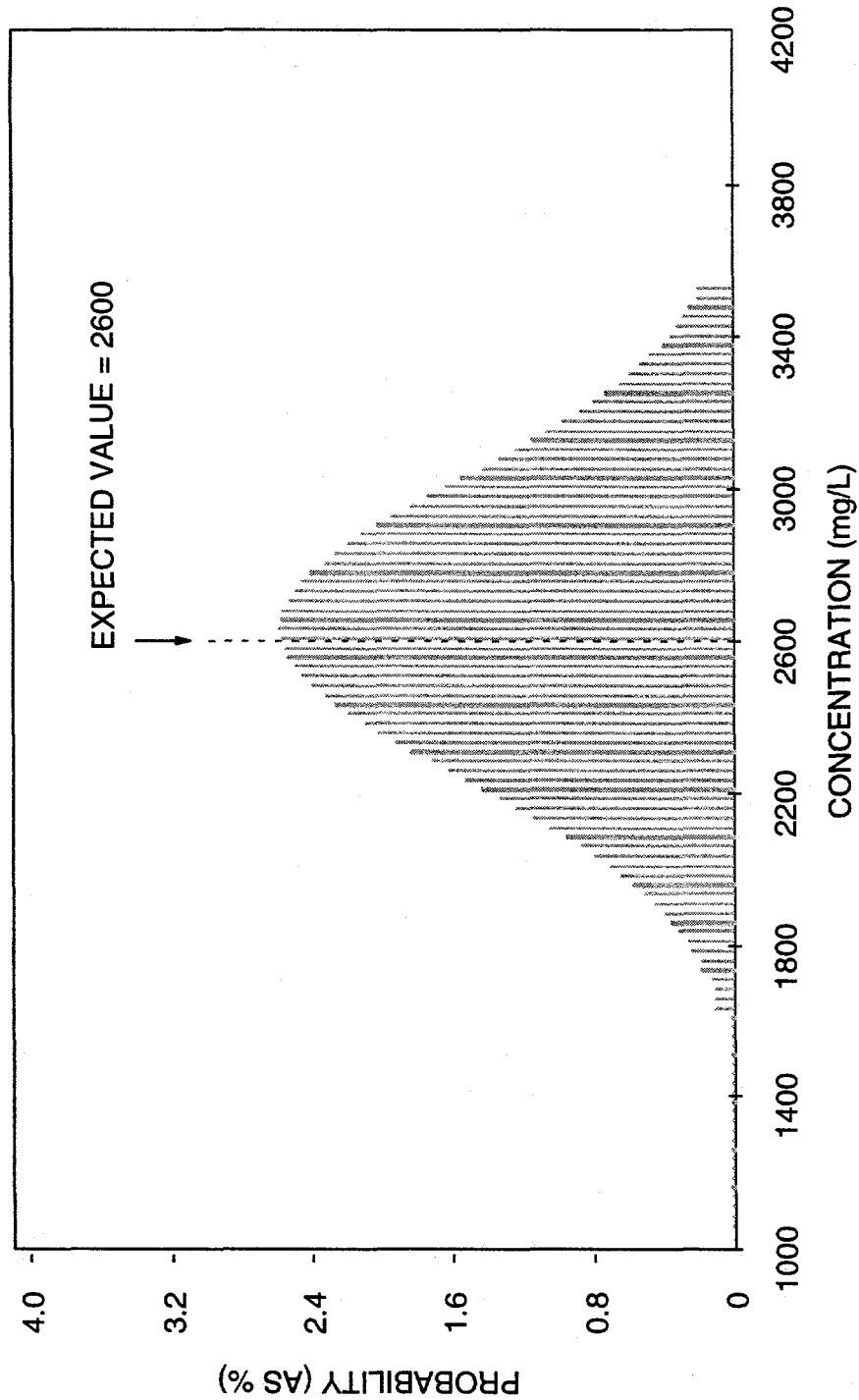


FIGURE 4.4
SIMULATED DISTRIBUTION OF CHLORIDE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

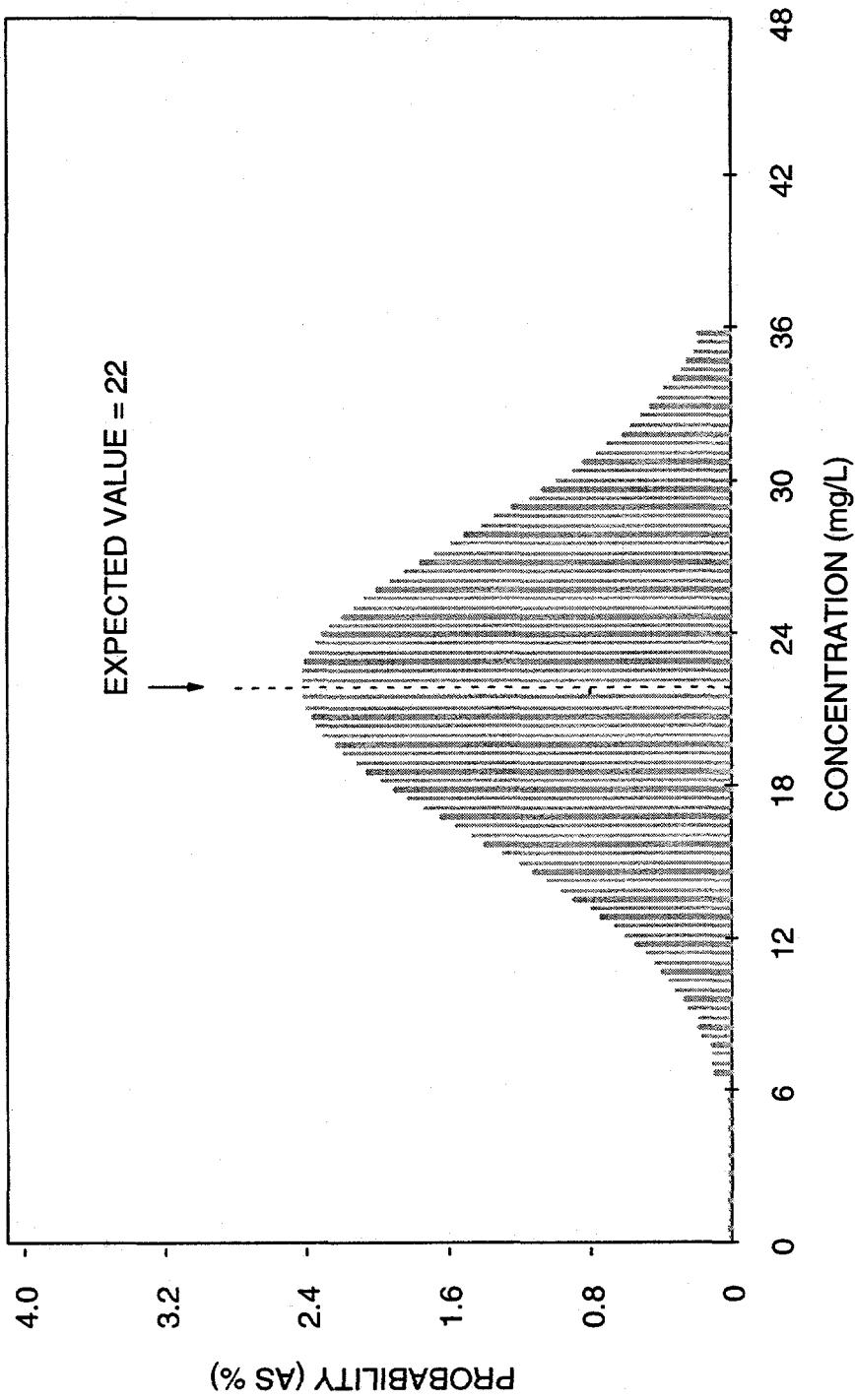


FIGURE 4.5
SIMULATED DISTRIBUTION OF IRON CONCENTRATIONS
LAKEVIEW, OREGON, SITE

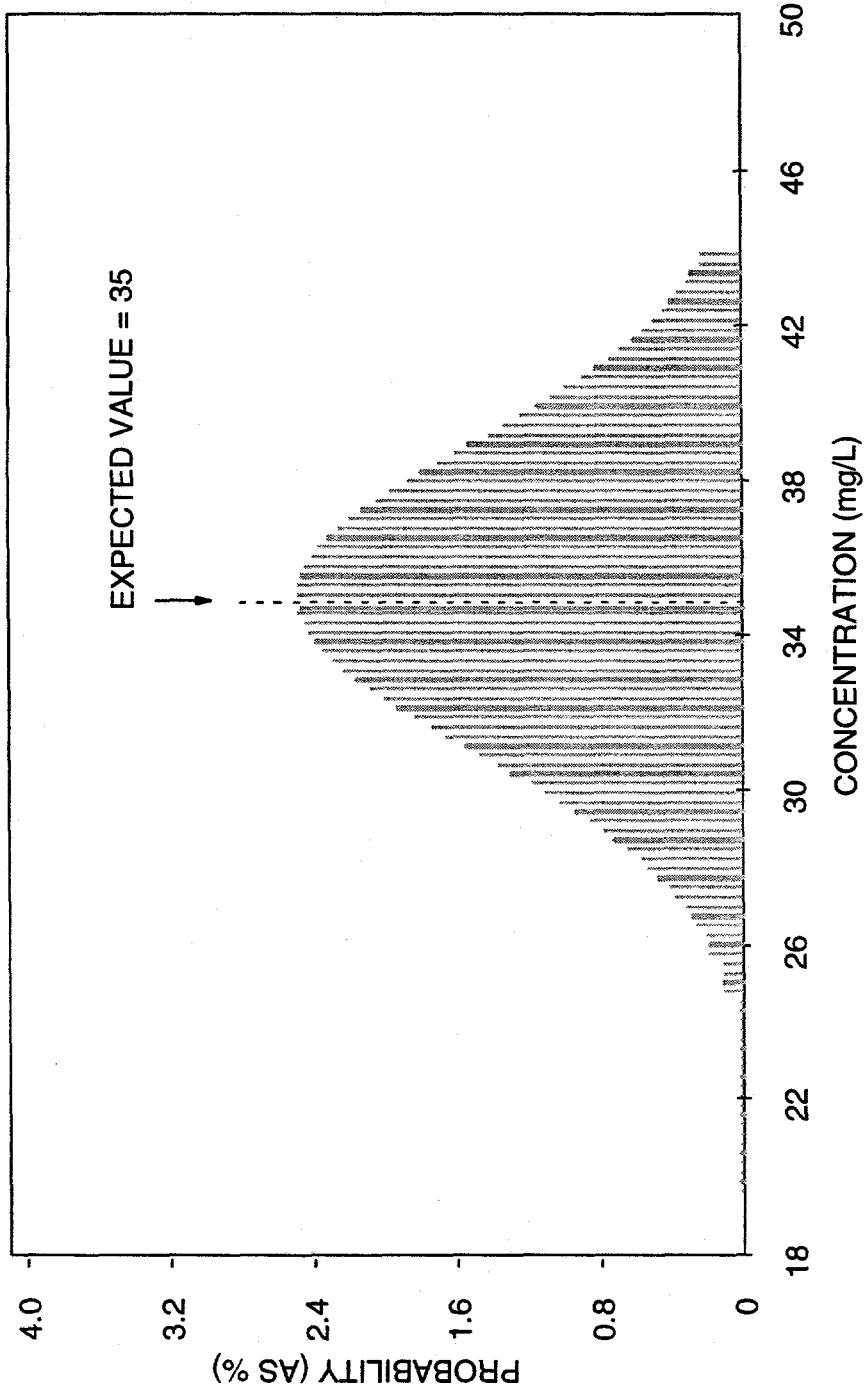


FIGURE 4.6
SIMULATED DISTRIBUTION OF MANGANESE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

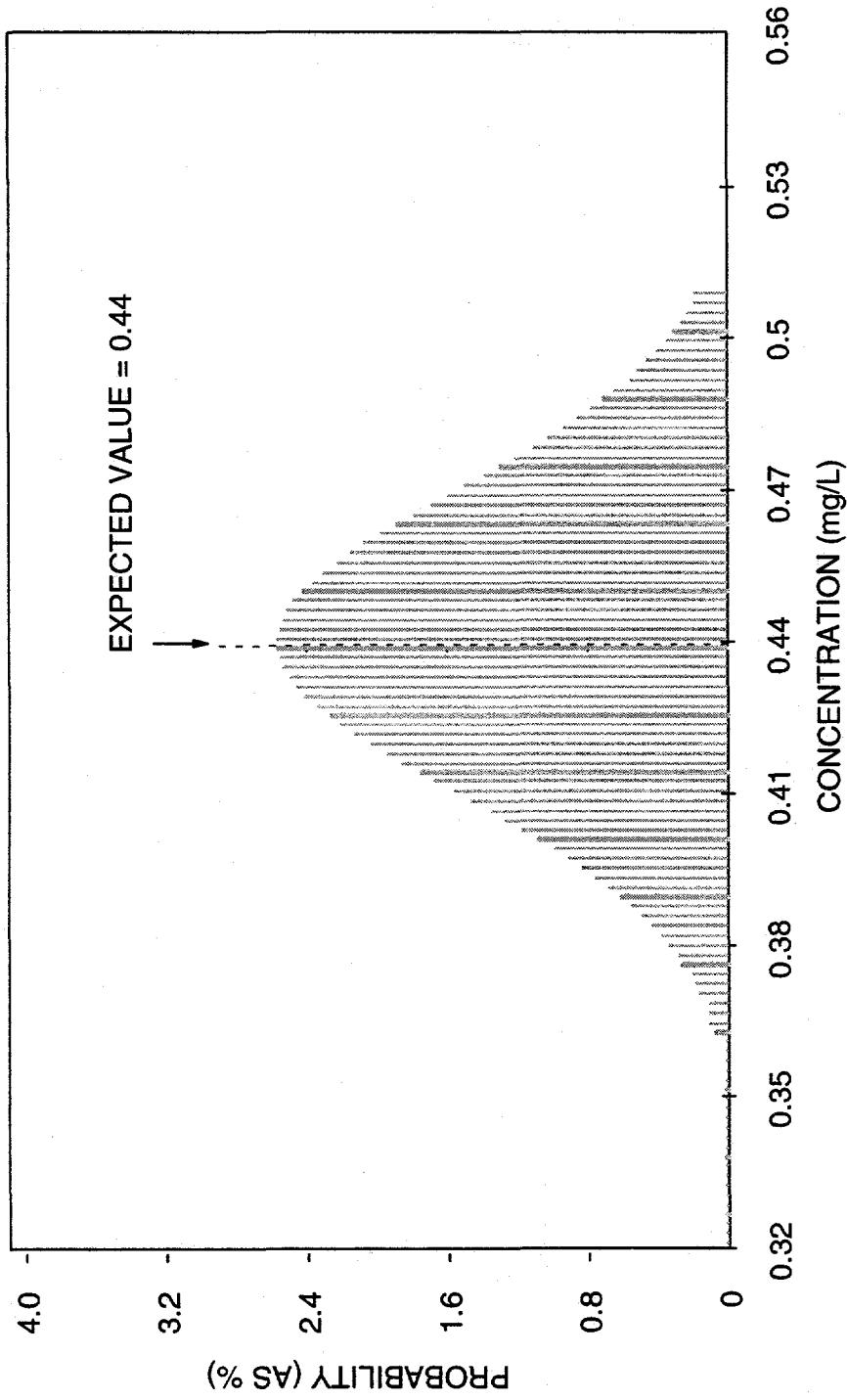


FIGURE 4.7
SIMULATED DISTRIBUTION OF MOLYBDENUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

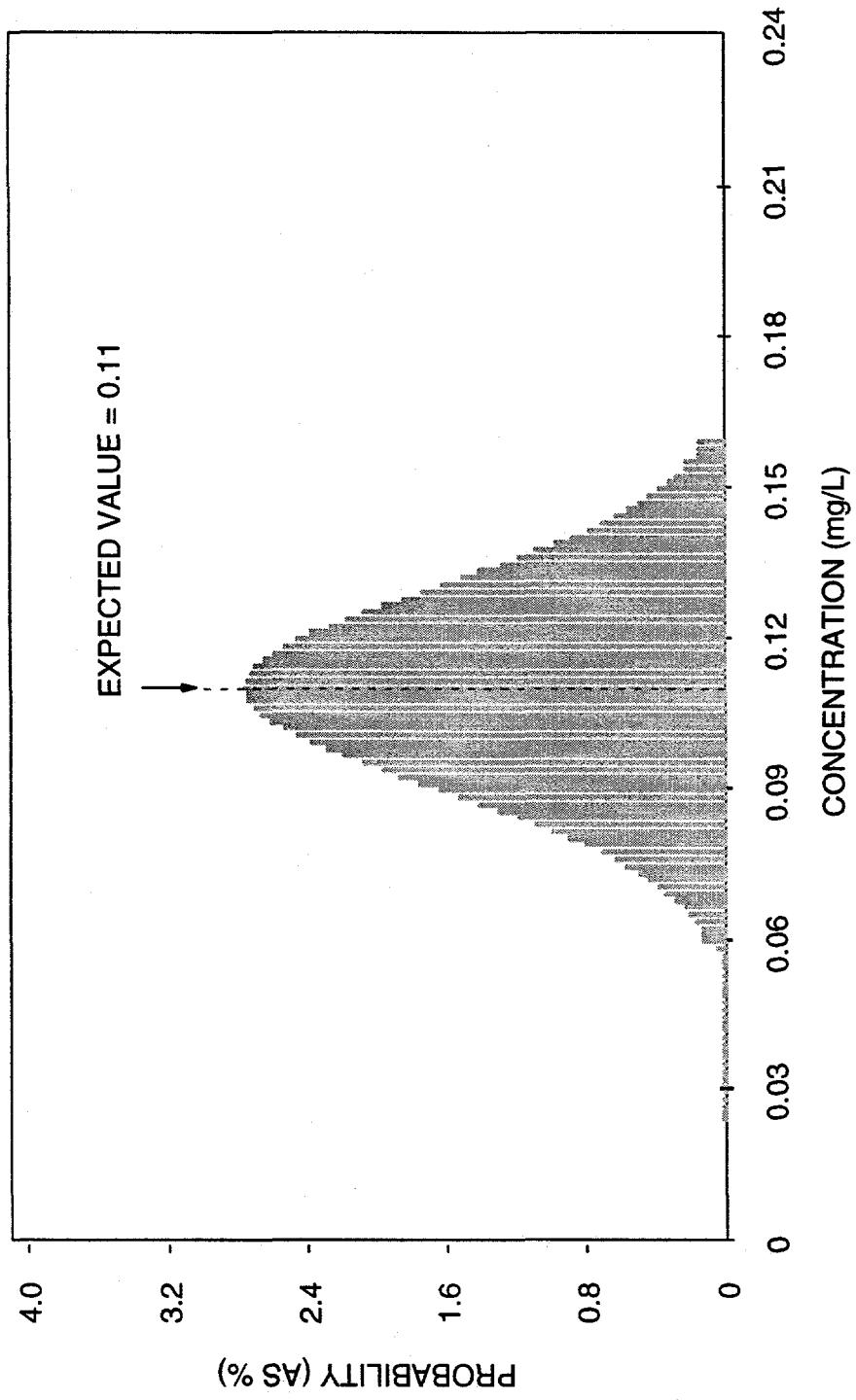


FIGURE 4.8
SIMULATED DISTRIBUTION OF NICKEL CONCENTRATIONS
LAKEVIEW, OREGON, SITE

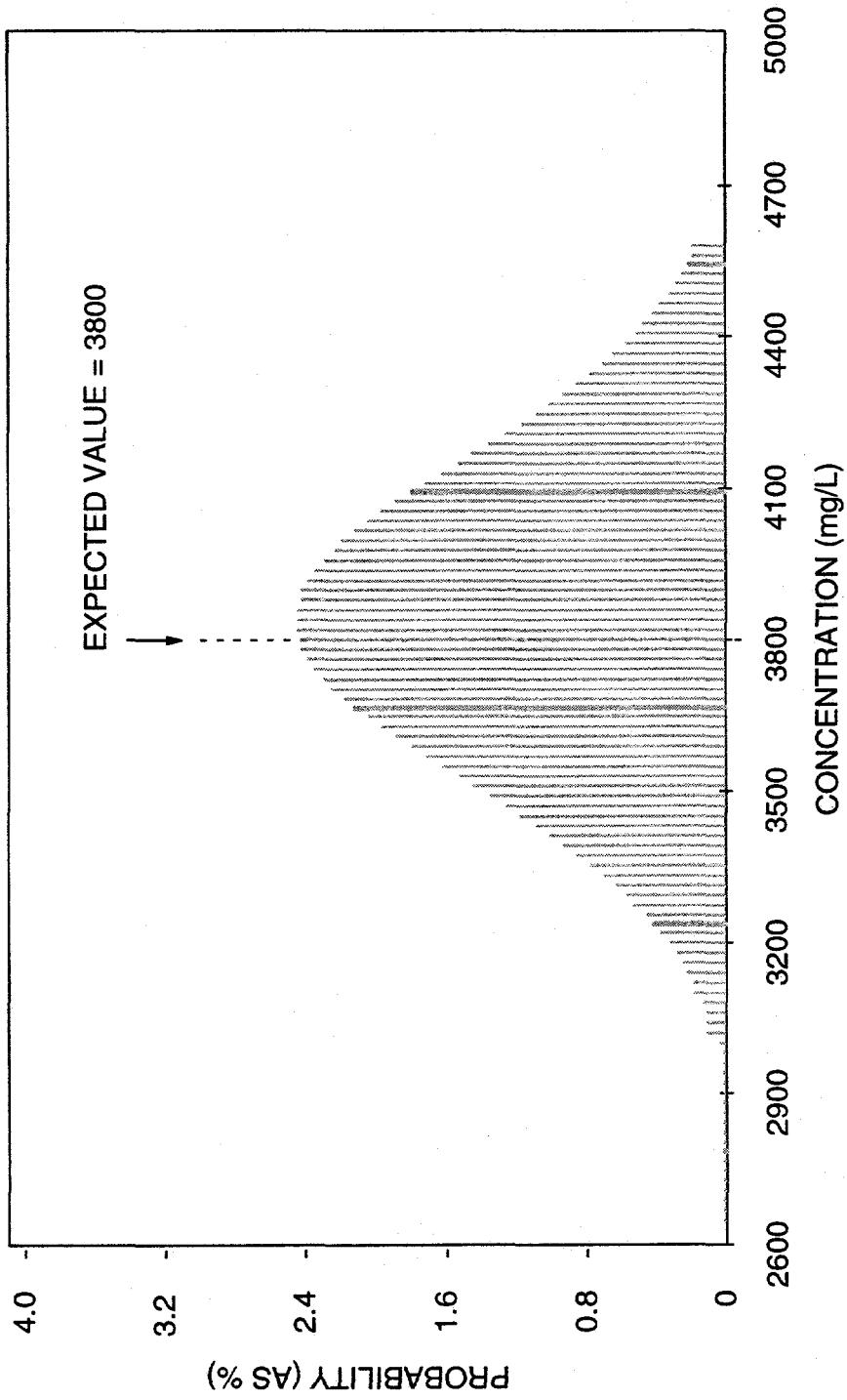


FIGURE 4.9
SIMULATED DISTRIBUTION OF SODIUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

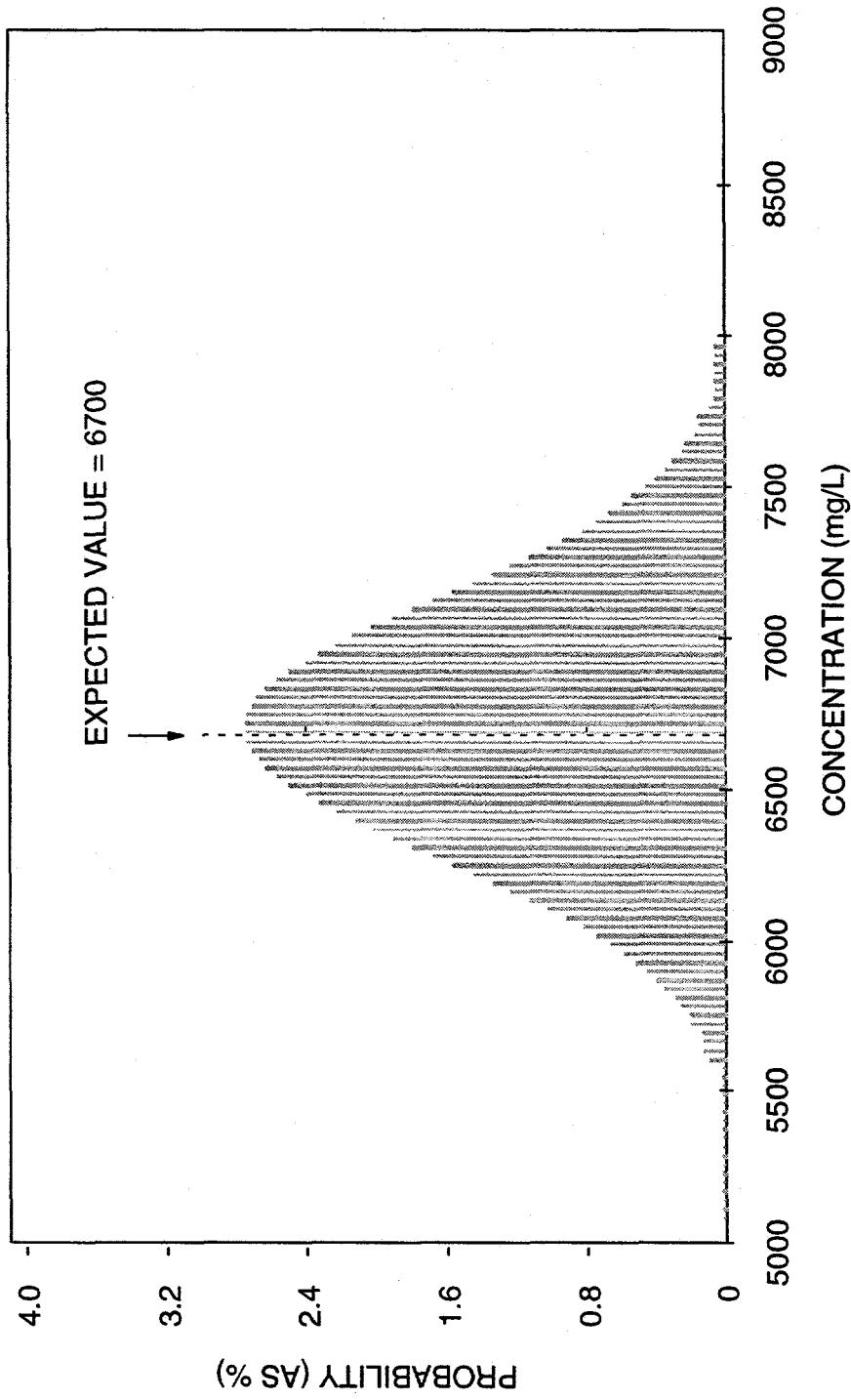


FIGURE 4.10
SIMULATED DISTRIBUTION OF SULFATE CONCENTRATIONS
LAKEVIEW, OREGON, SITE

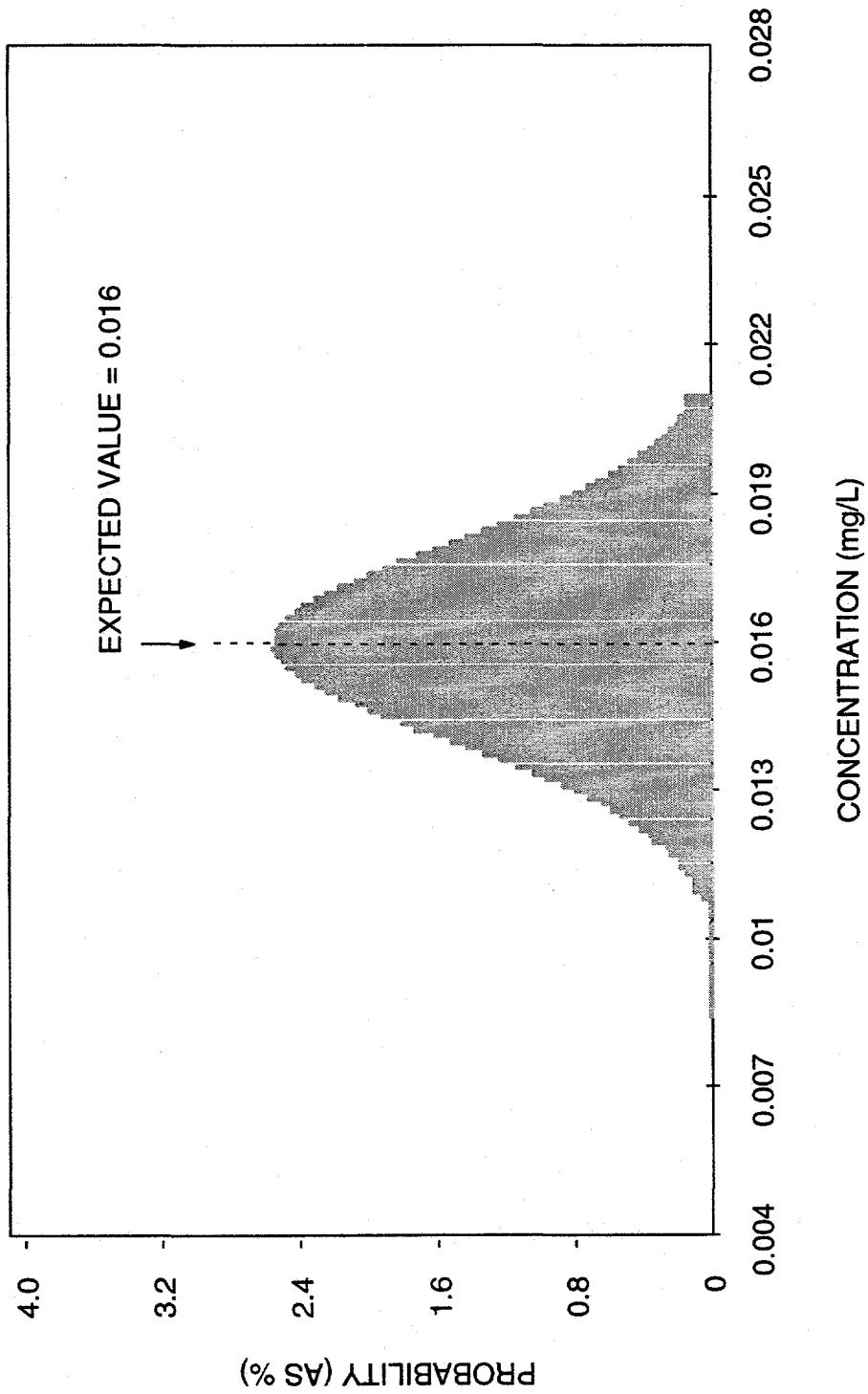


FIGURE 4.11
SIMULATED DISTRIBUTION OF URANIUM CONCENTRATIONS
LAKEVIEW, OREGON, SITE

upper and lower 0.01 percentile. Within the hypothetical population, values disallowed through this truncation may occur with a probability of less than 1 in 10,000.

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

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

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:

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

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

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

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 constituents 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.

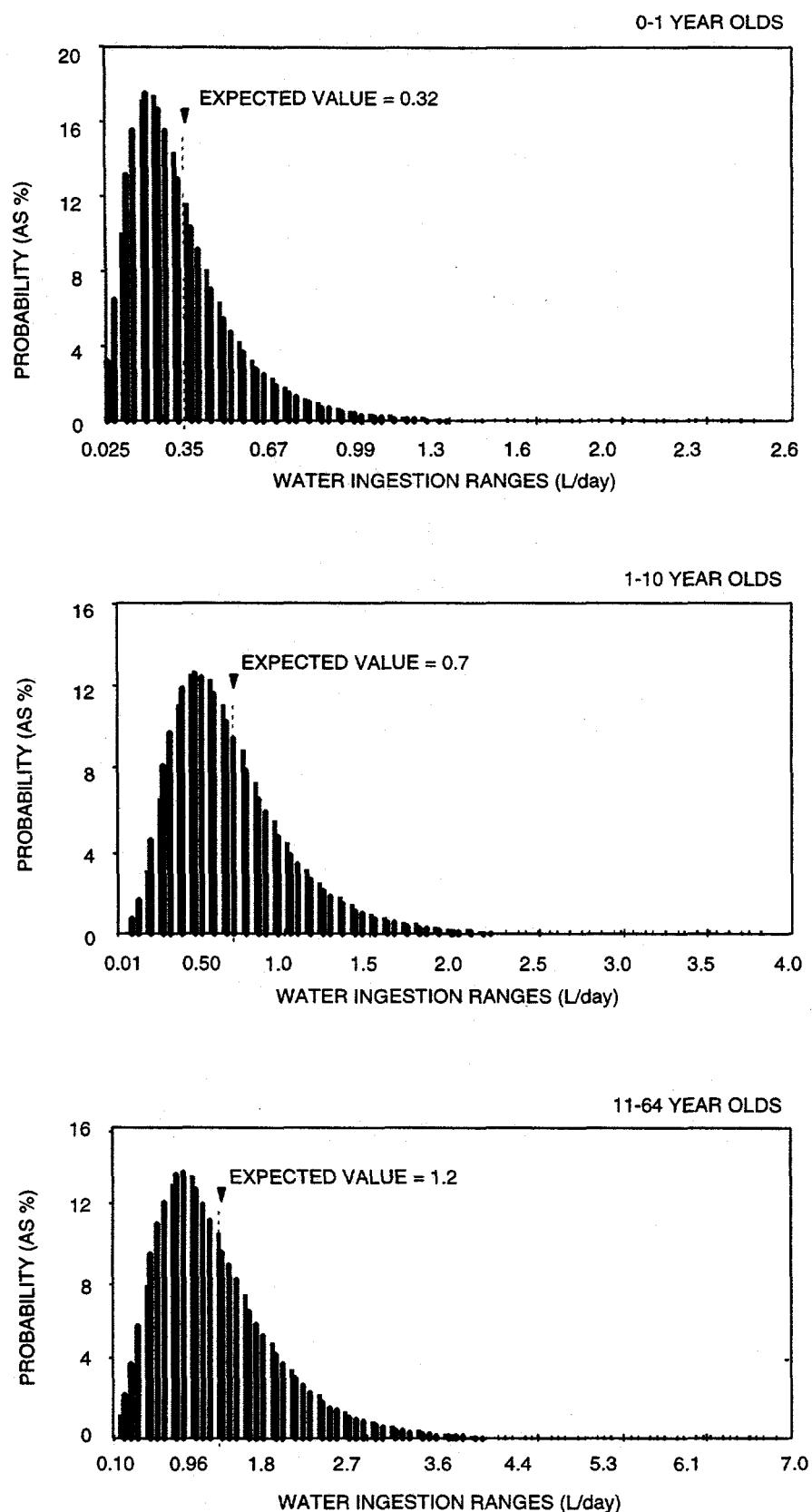
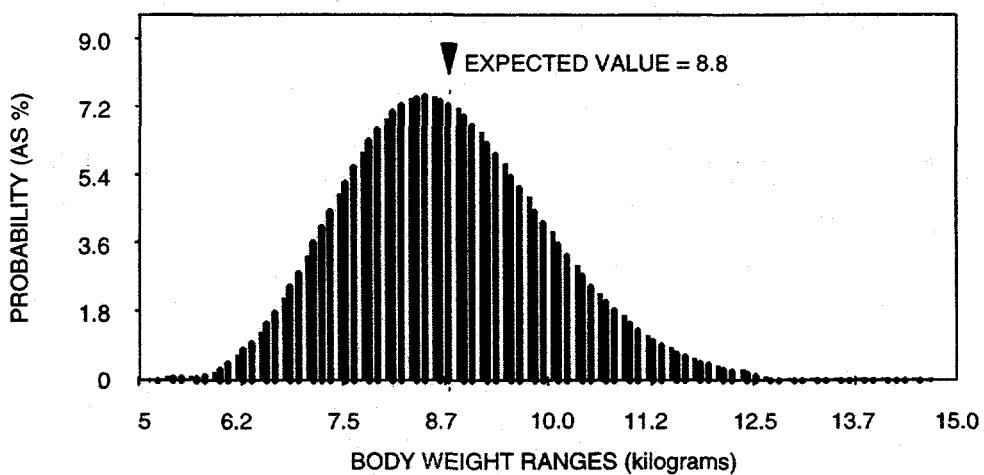
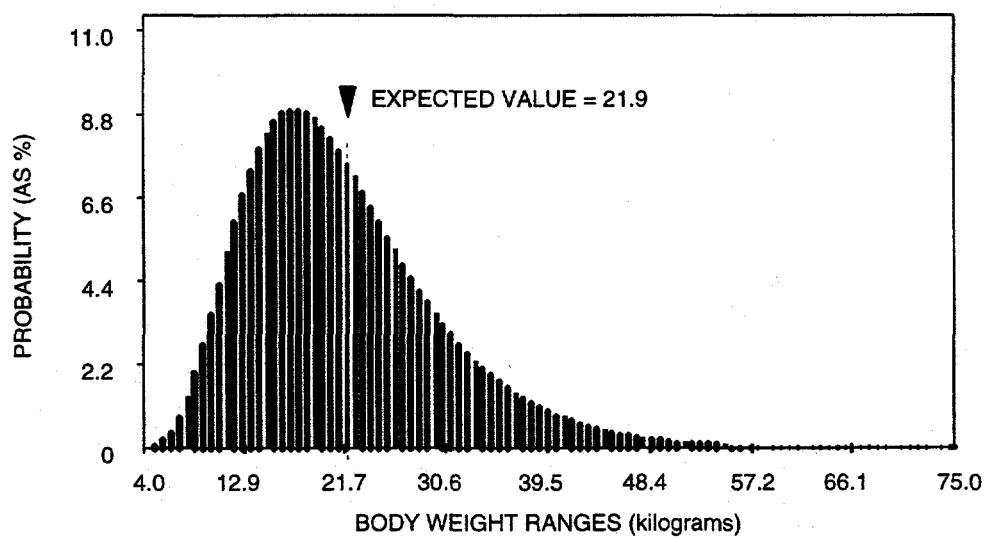


FIGURE 4.12
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES

0-1 YEAR OLDS



1-10 YEAR OLDS



11-64 YEAR OLDS

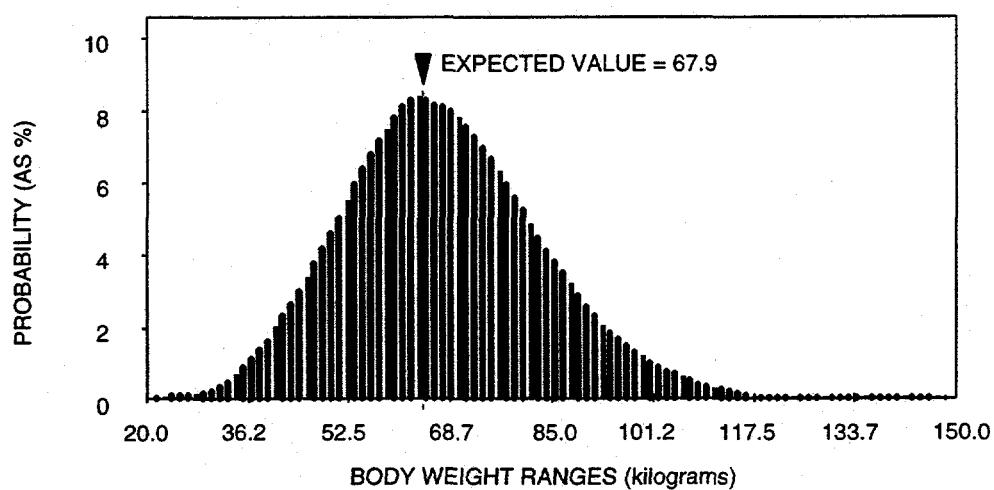
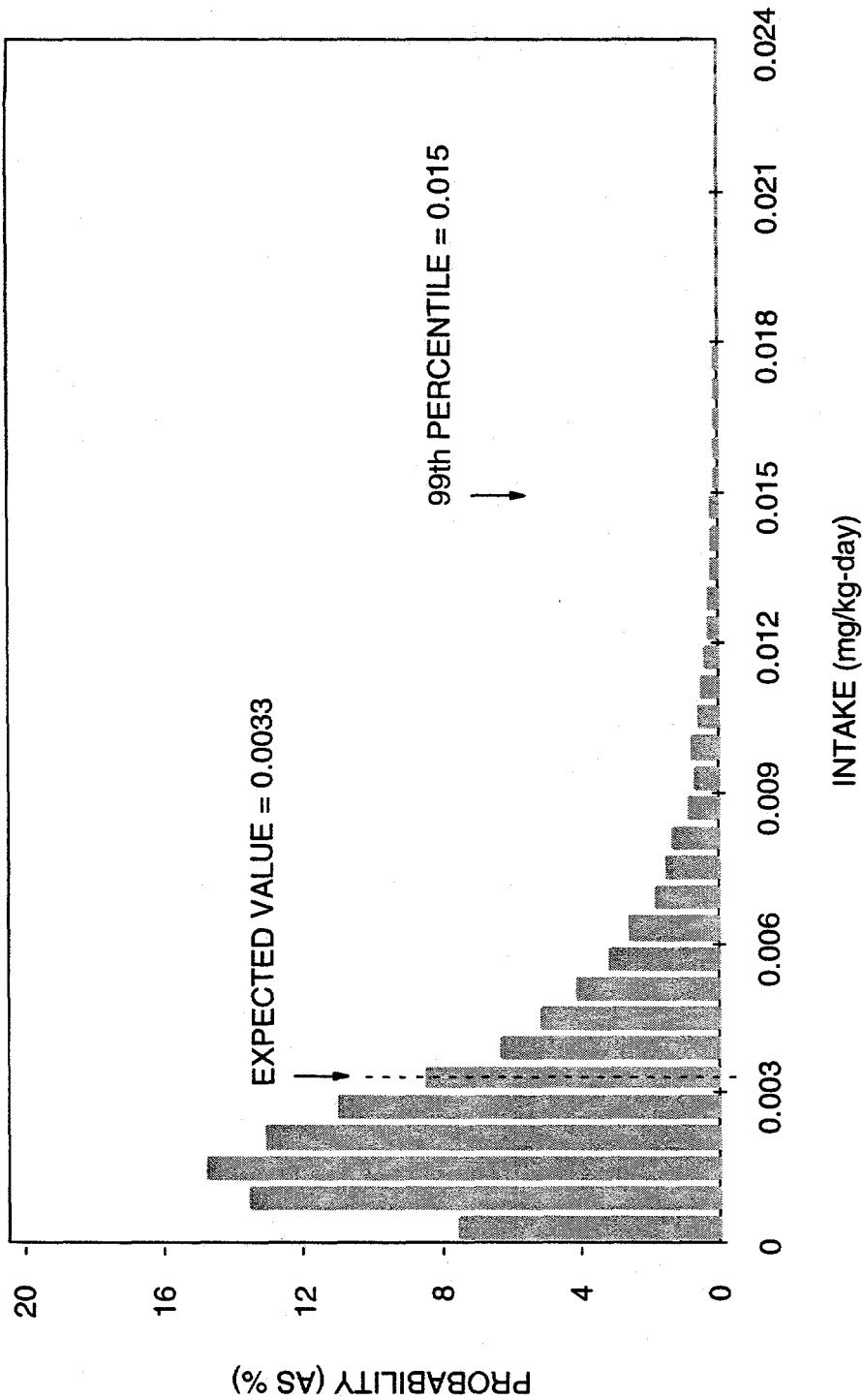
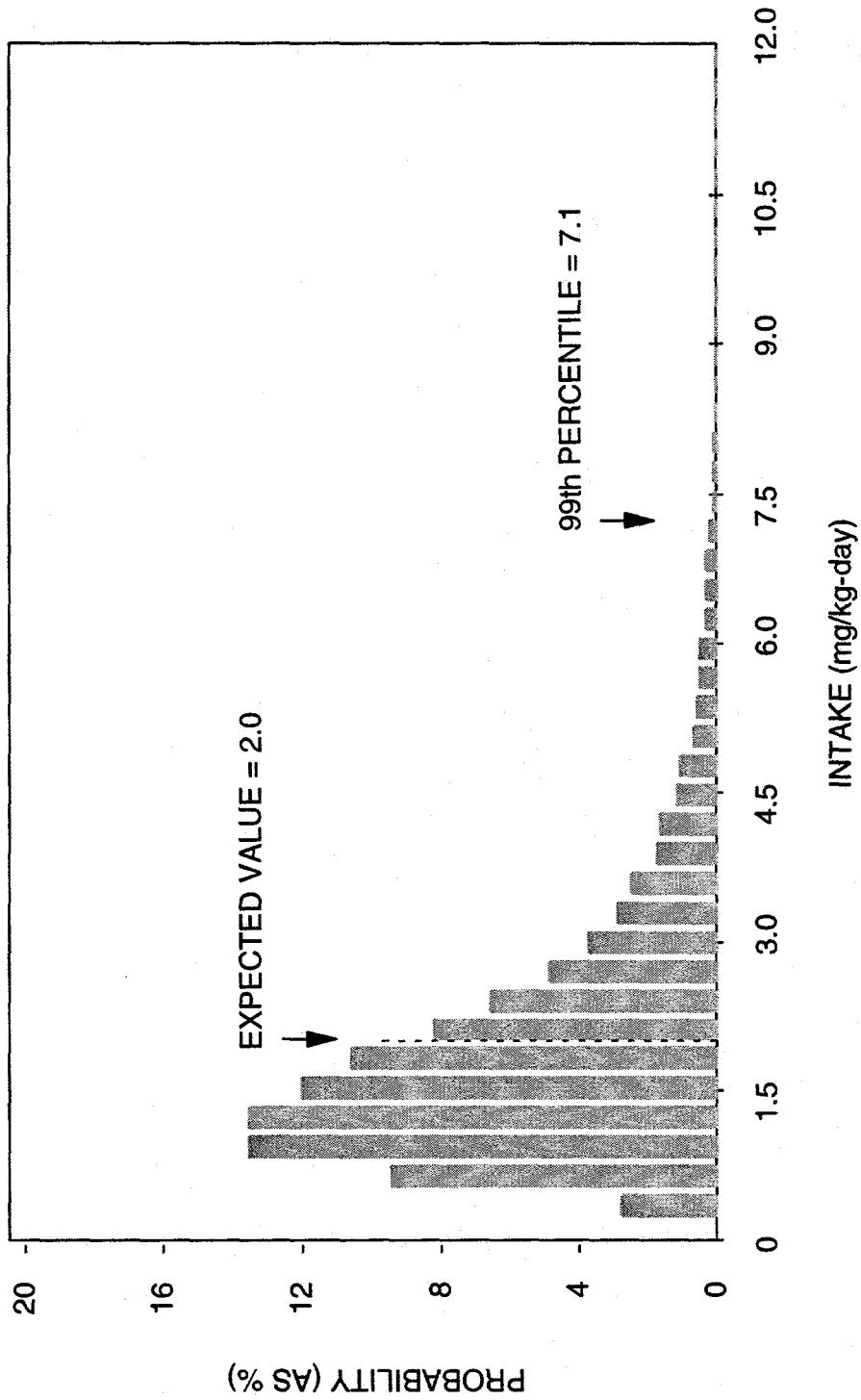


FIGURE 4.13
PROBABILITY DISTRIBUTIONS FOR BODY WEIGHT



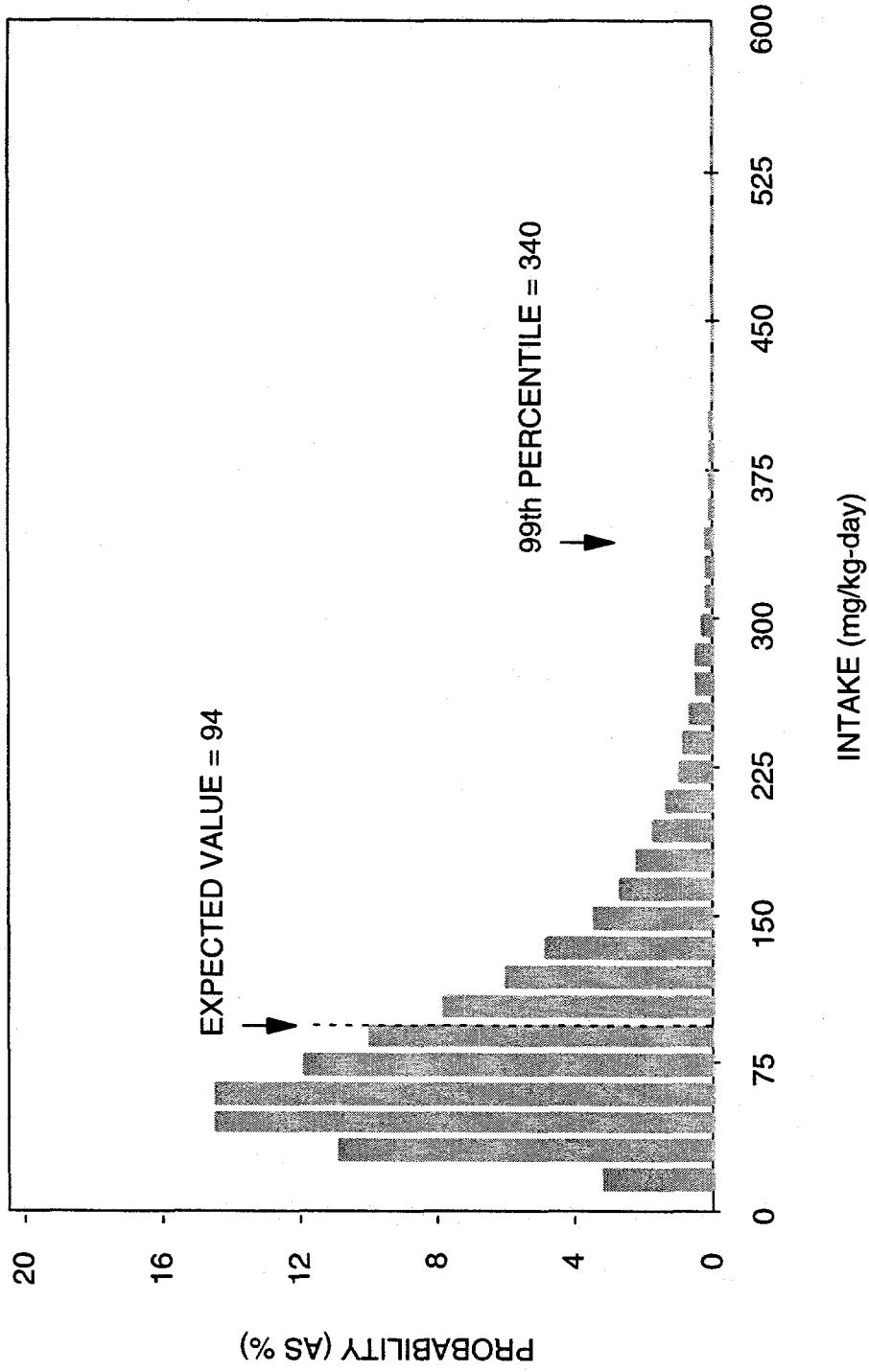
NOTE: AVERAGE POTENTIAL INTAKE IS 0.0033 ARSENIC/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



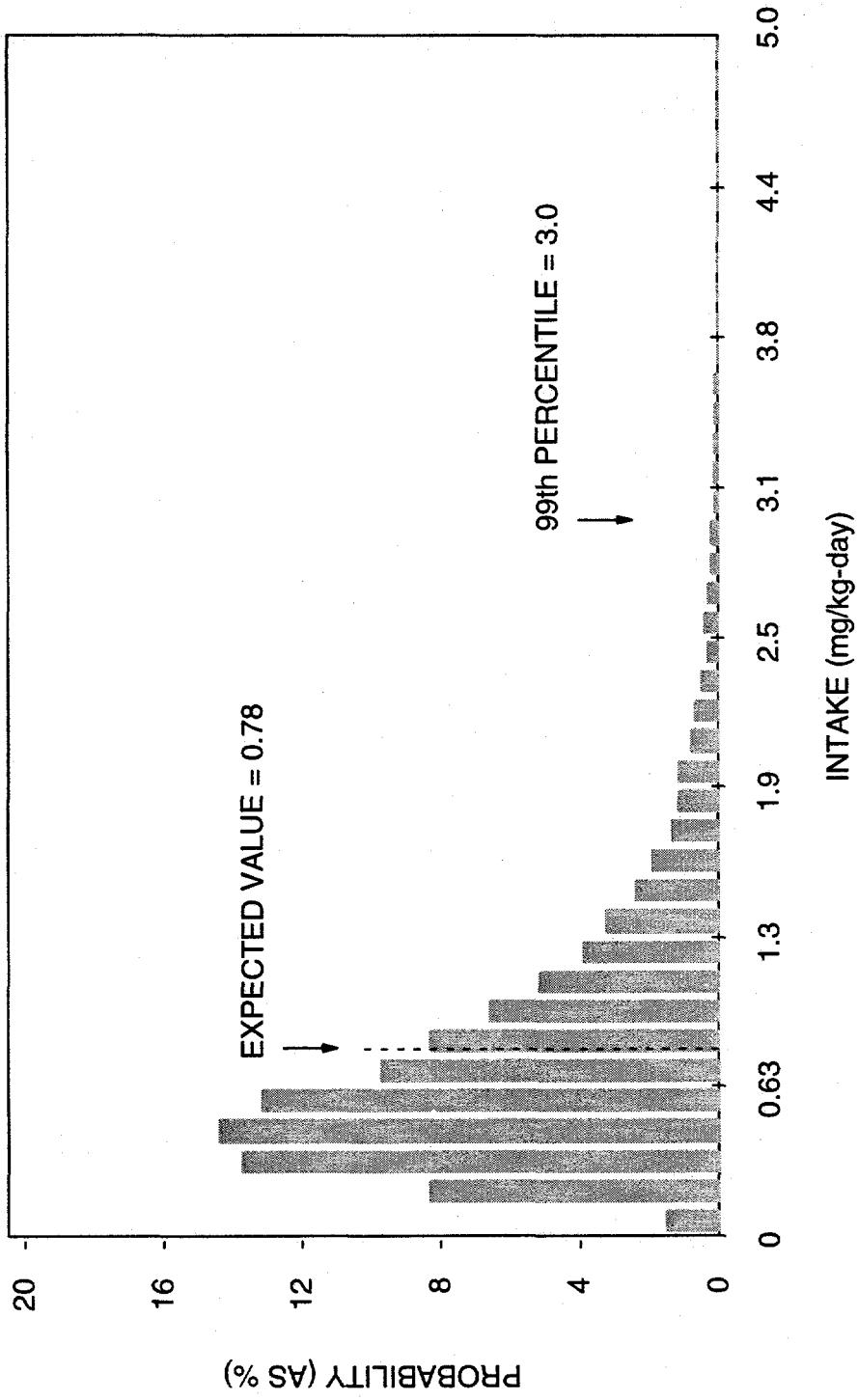
NOTE: AVERAGE POTENTIAL INTAKE IS 2.0 mg BORON/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.15
DISTRIBUTION OF POTENTIAL BORON INTAKES FOR CHILDREN
AS A RESULT OF GROUND WATER INGESTION
LAKEVIEW, OREGON, SITE



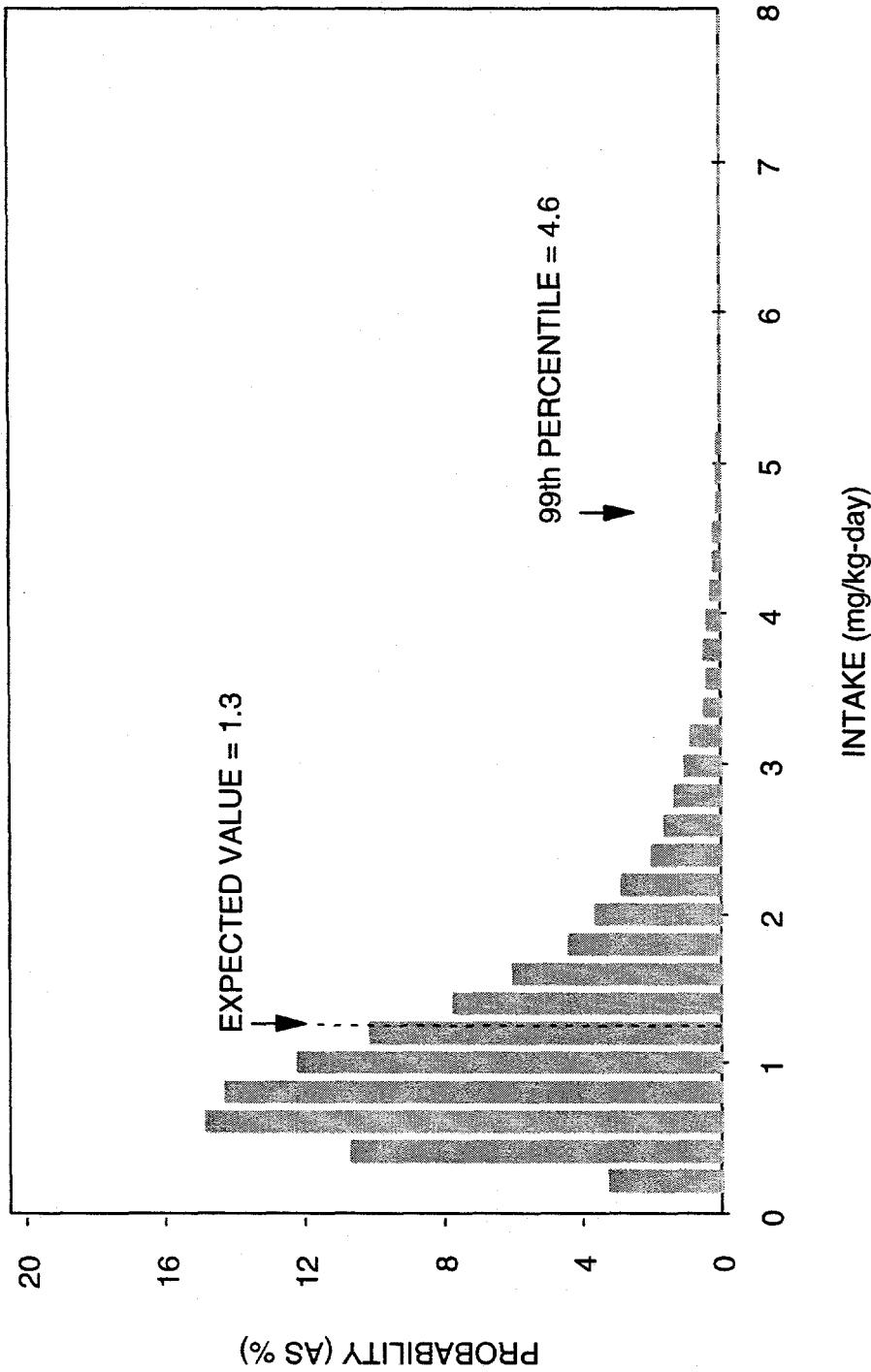
NOTE: AVERAGE POTENTIAL INTAKE IS 94 mg CHLORIDE/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



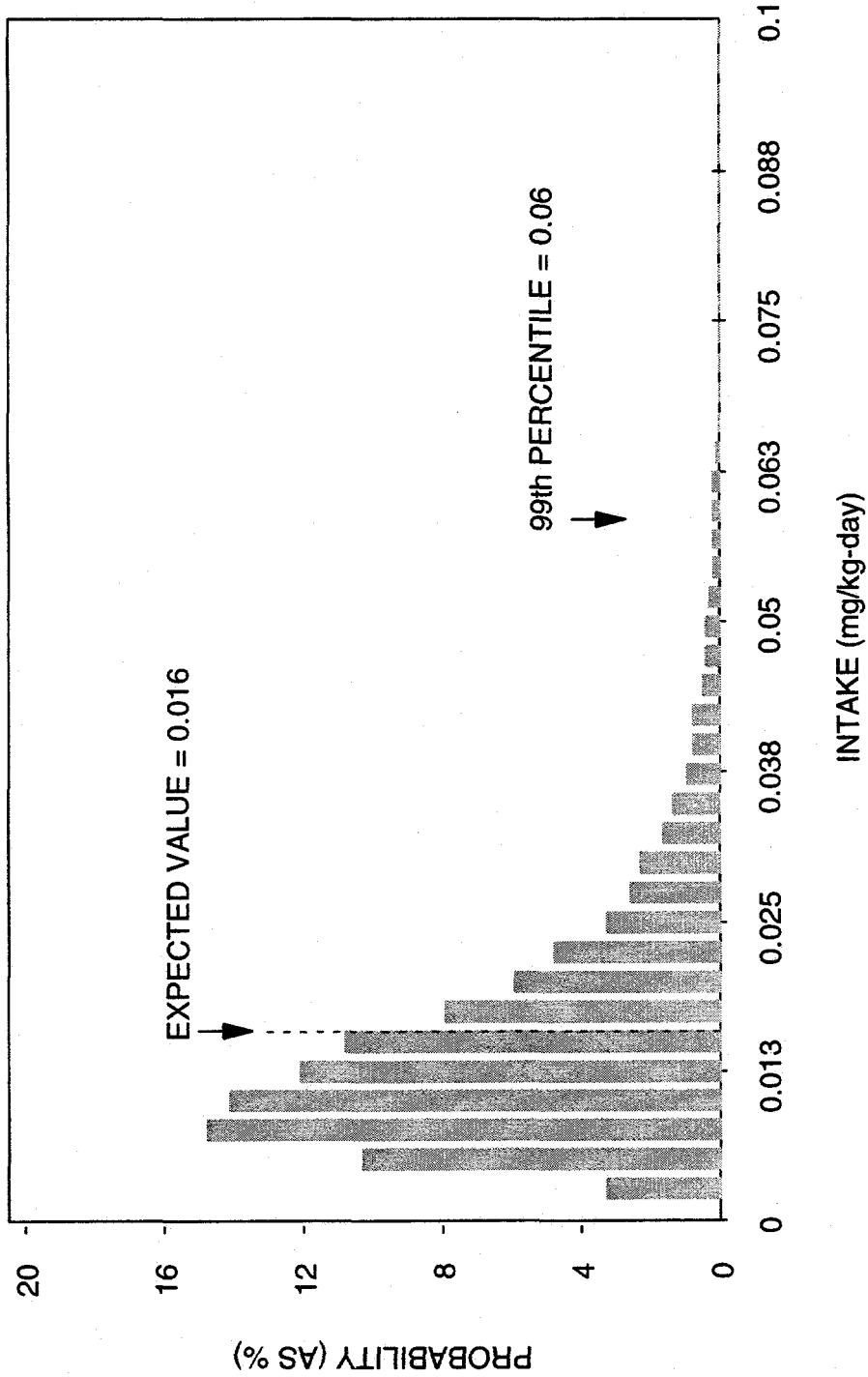
NOTE: AVERAGE POTENTIAL INTAKE IS 0.78 mg IRON/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



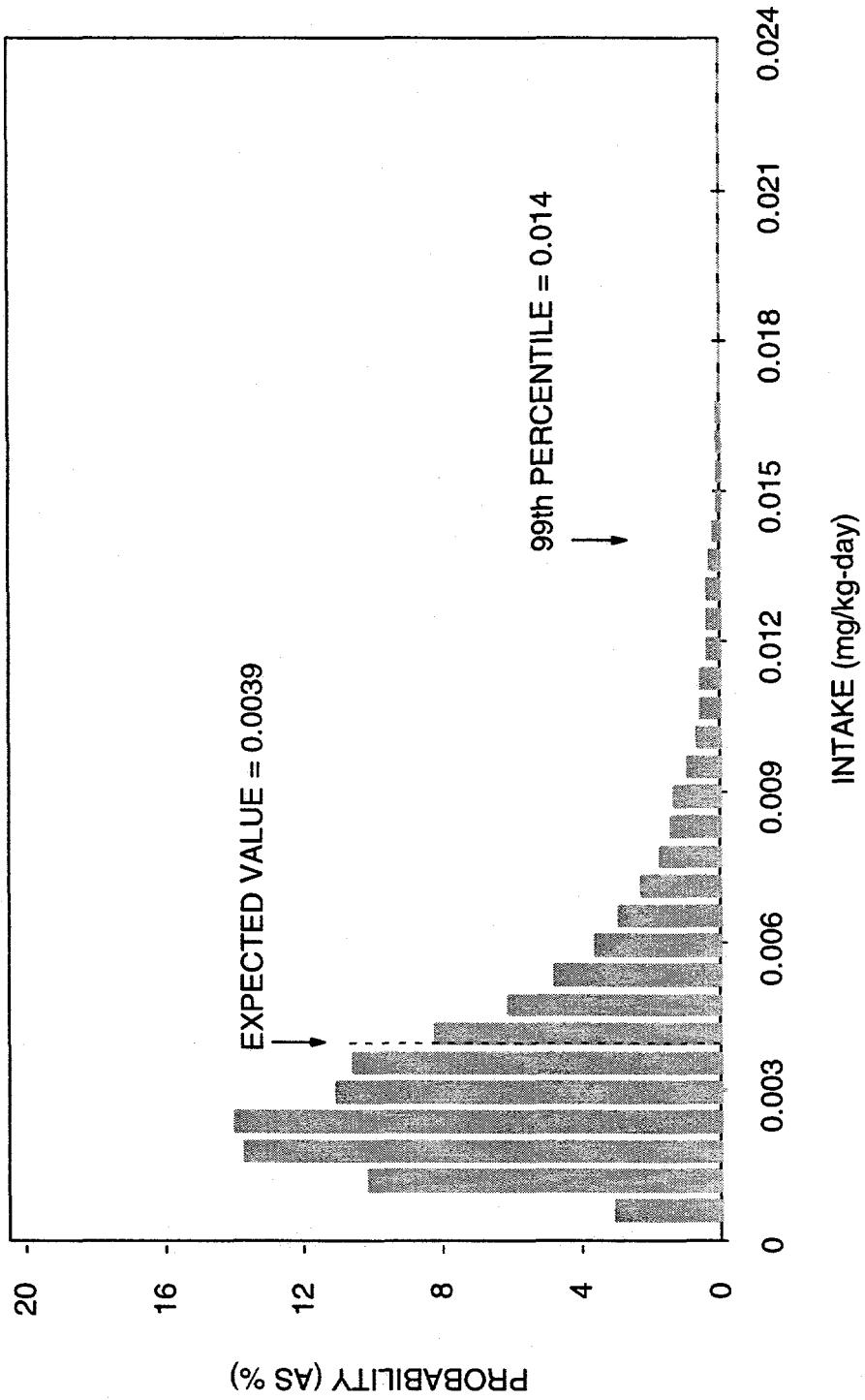
NOTE: AVERAGE POTENTIAL INTAKE IS 1.3 mg MANGANESE/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



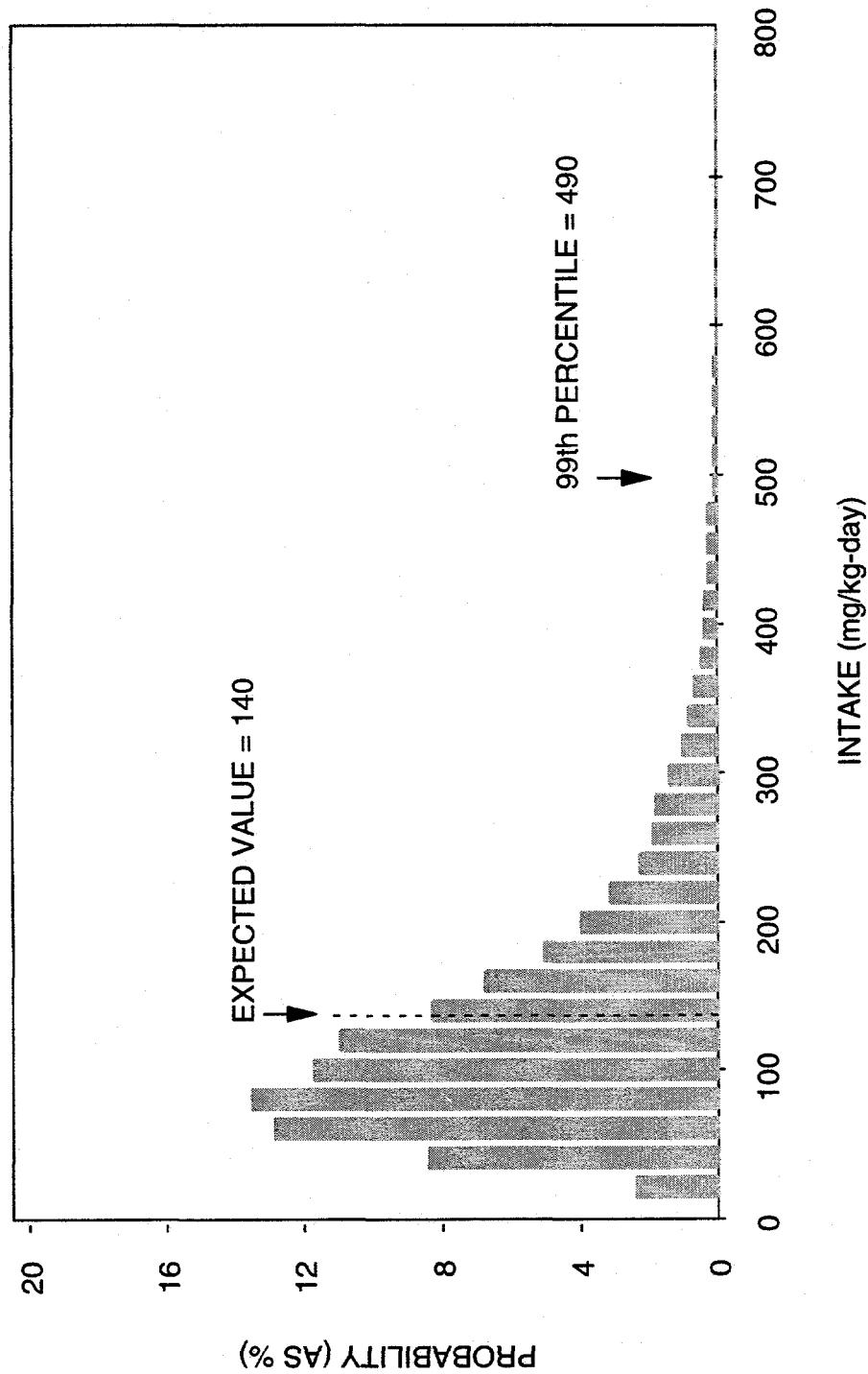
NOTE: AVERAGE POTENTIAL INTAKE IS 0.016 mg MOLYBDENUM/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



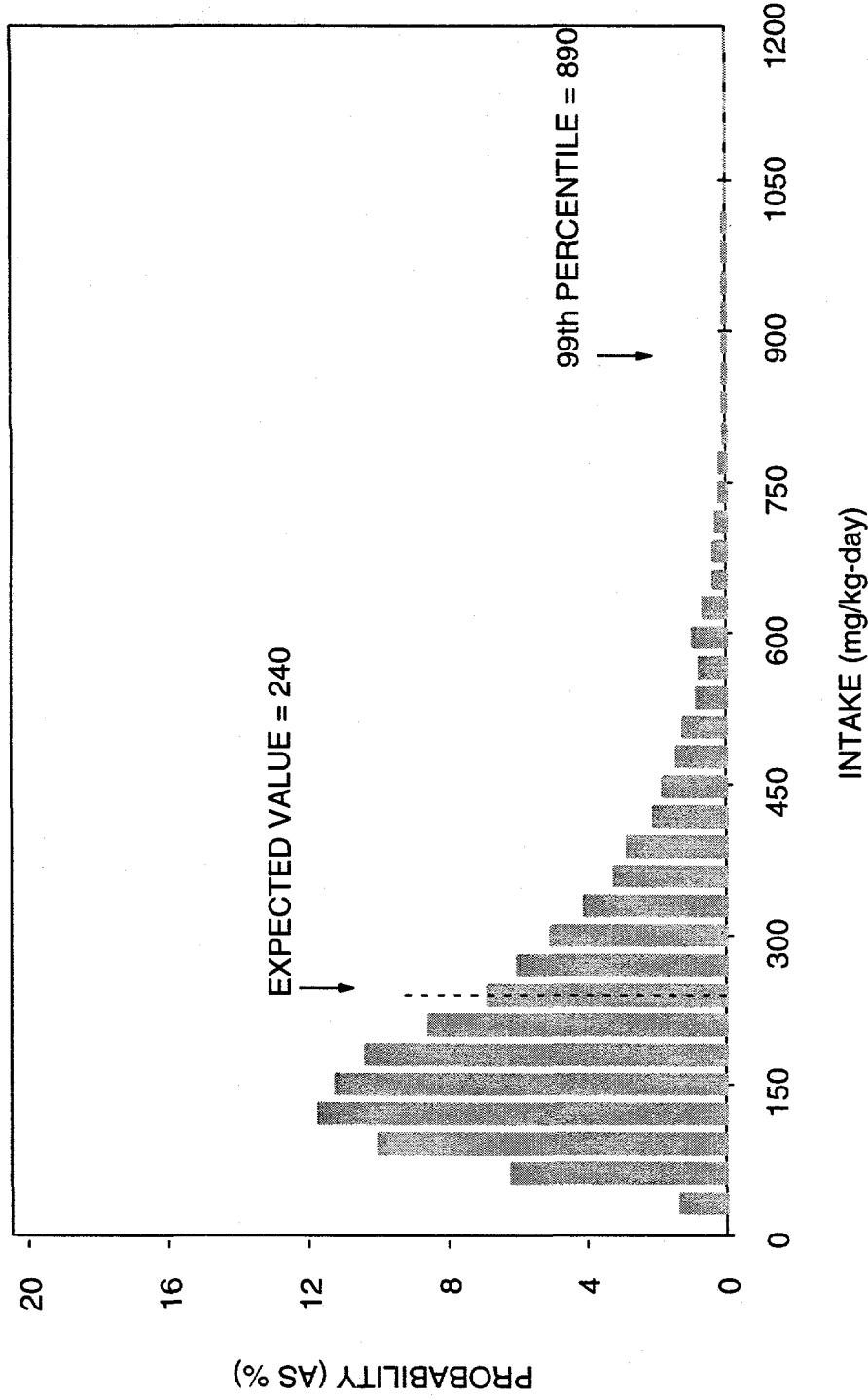
NOTE: AVERAGE POTENTIAL INTAKE IS 0.0039 mg NICKEL/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



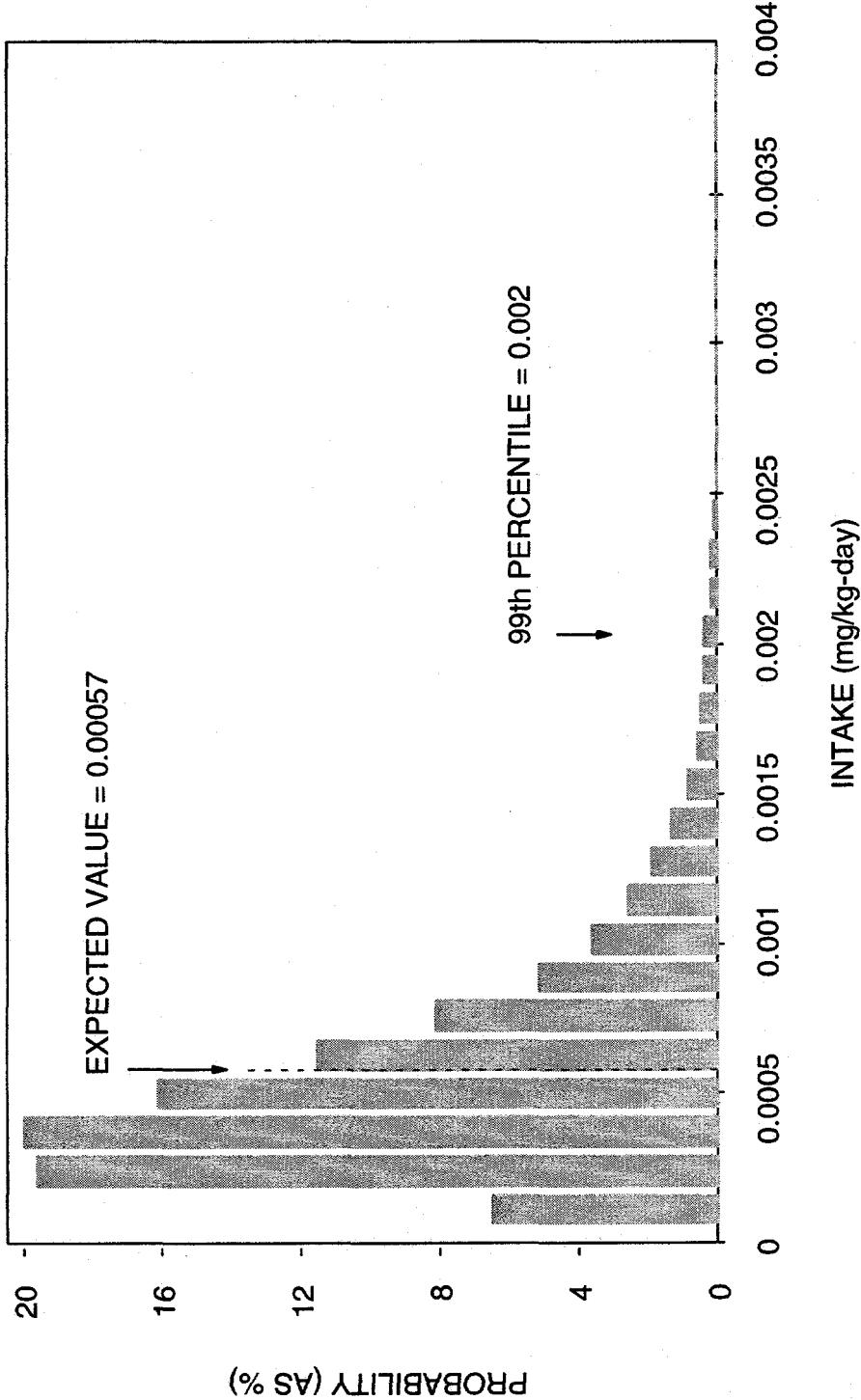
NOTE: AVERAGE POTENTIAL INTAKE IS 140 mg SODIUM/kg BODY WEIGHT/DAY. INTAKES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

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



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 constituent source term at the site has reached a steady state and that constituent concentrations at the exposure point will remain constant for chronic periods of exposure (generally greater than 7 years). Because the source of constituents 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 constituent 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 constituents 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 constituents 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 U.S. Department of Health and Human Services (DHHS); the *Handbook on the Toxicology of Metals* (Friborg 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 CONSTITUENT 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 constituents of potential concern are arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium. Although these constituents 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 sulphydryl-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 methylcobalamin 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).

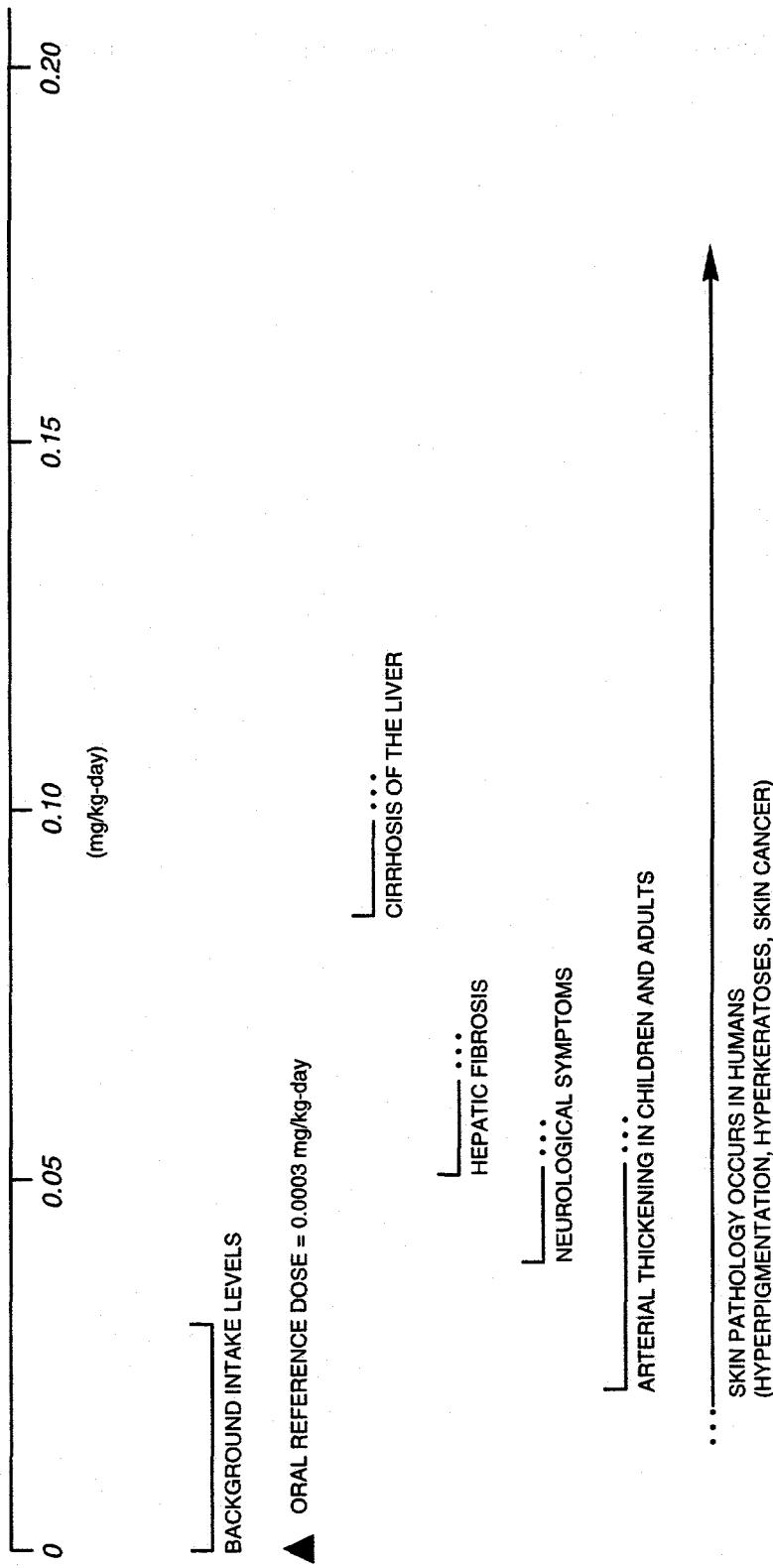


FIGURE 5.1
ARSENIC TOXICITY RANGES

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.

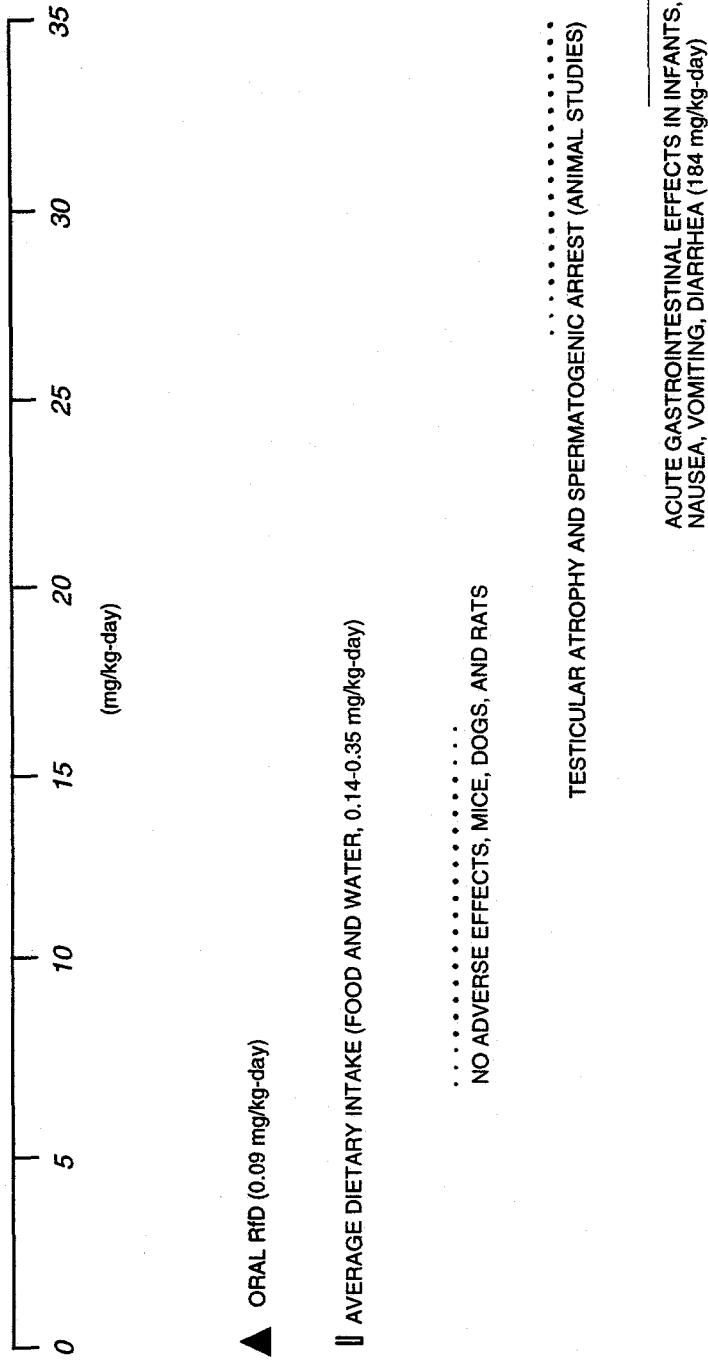


FIGURE 5.2
BORON TOXICITY RANGES

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

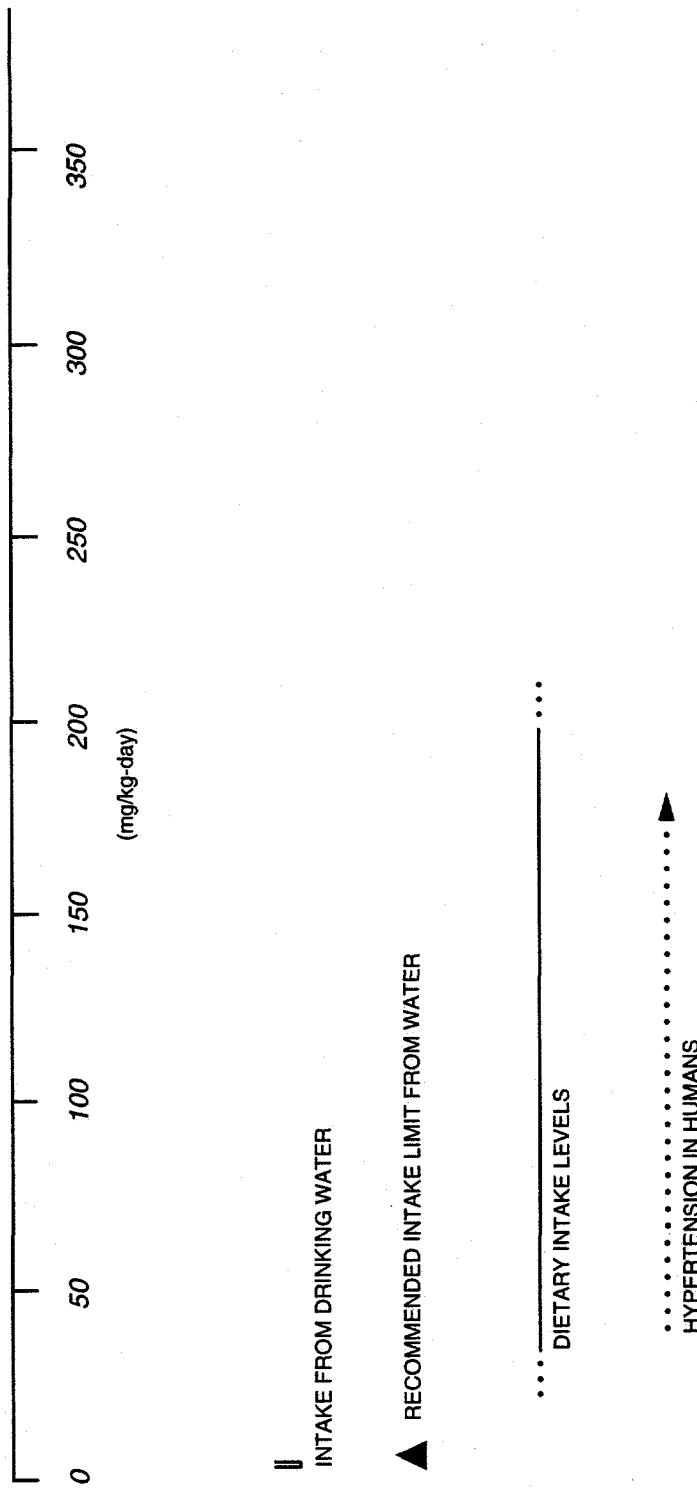


FIGURE 5.3
CHLORIDE TOXICITY RANGES

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

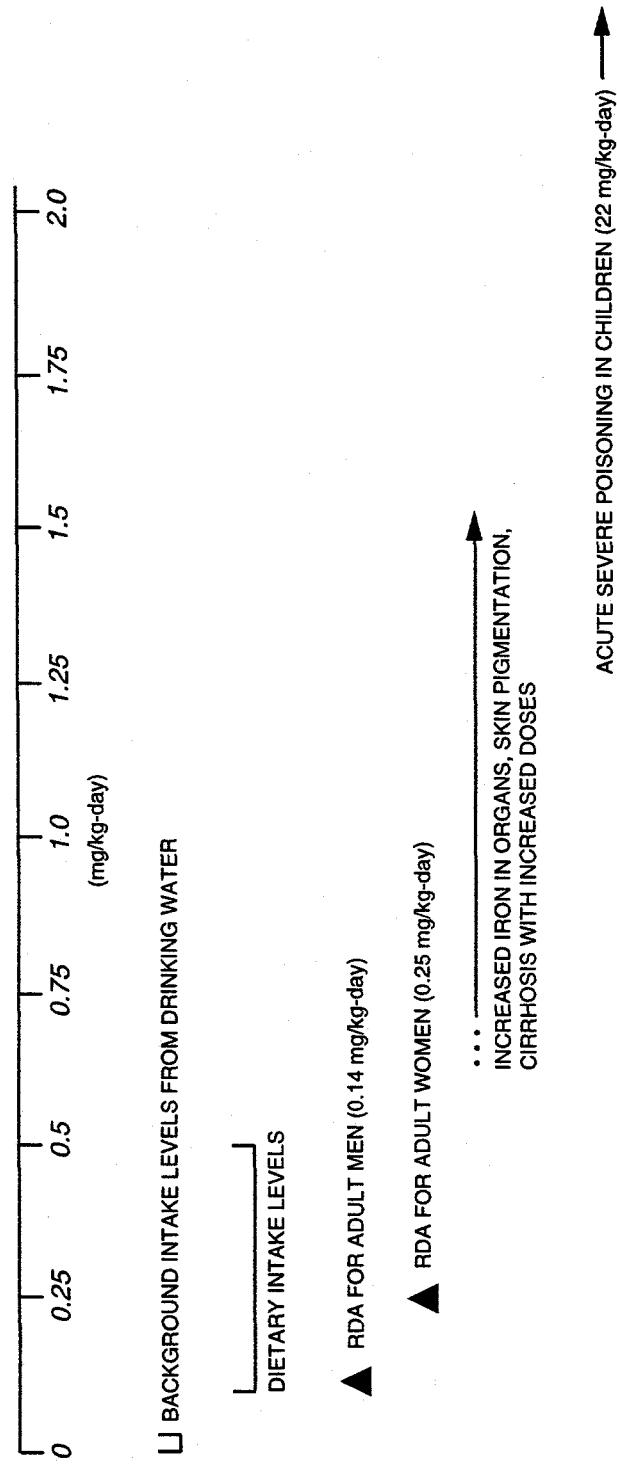


FIGURE 5.4
IRON TOXICITY RANGES

phosphorus have been shown to increase the requirements for manganese in several species (Lönnardal 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.

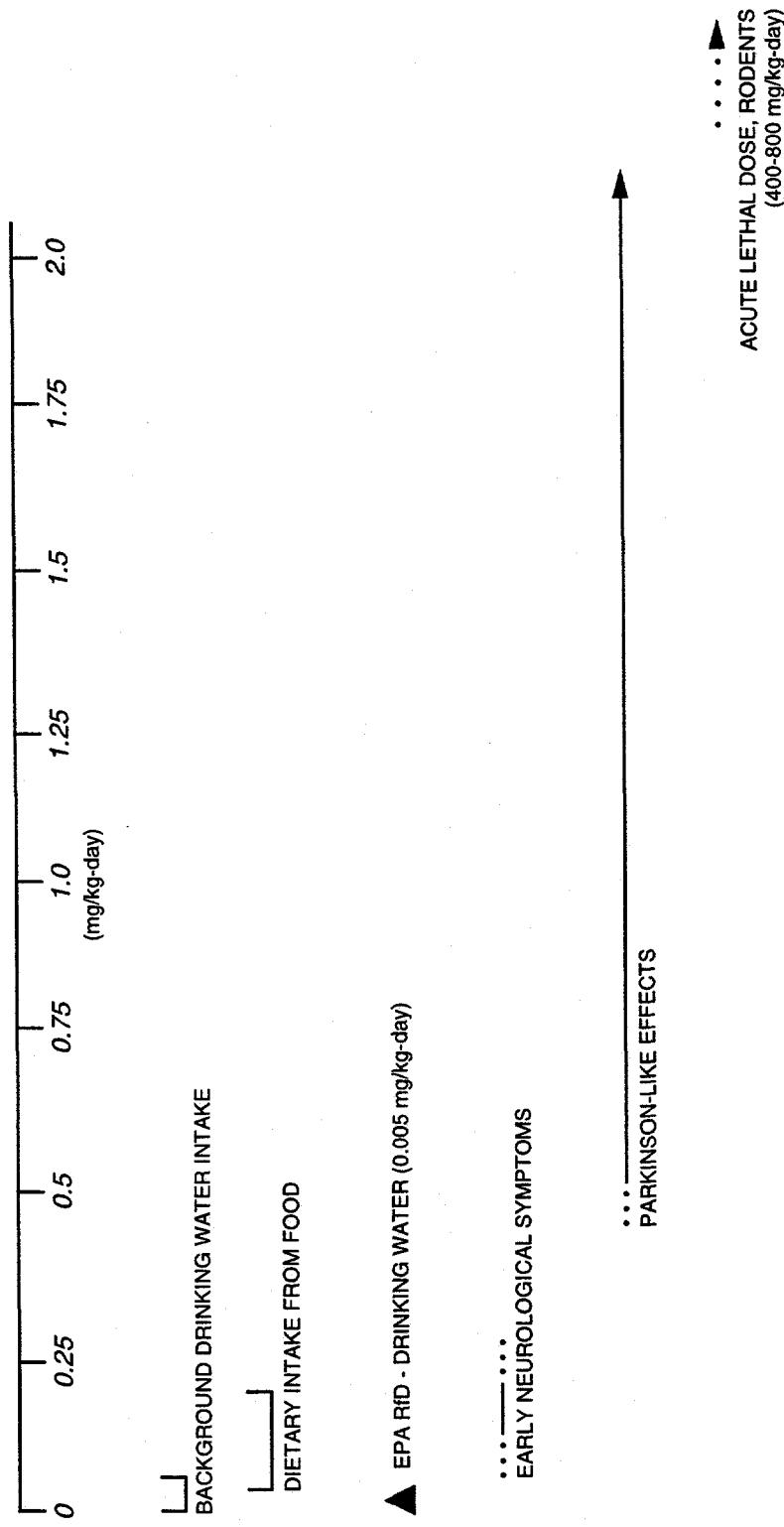


FIGURE 5.5
MANGANESE TOXICITY RANGES

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 ($\mu\text{g/L}$) 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

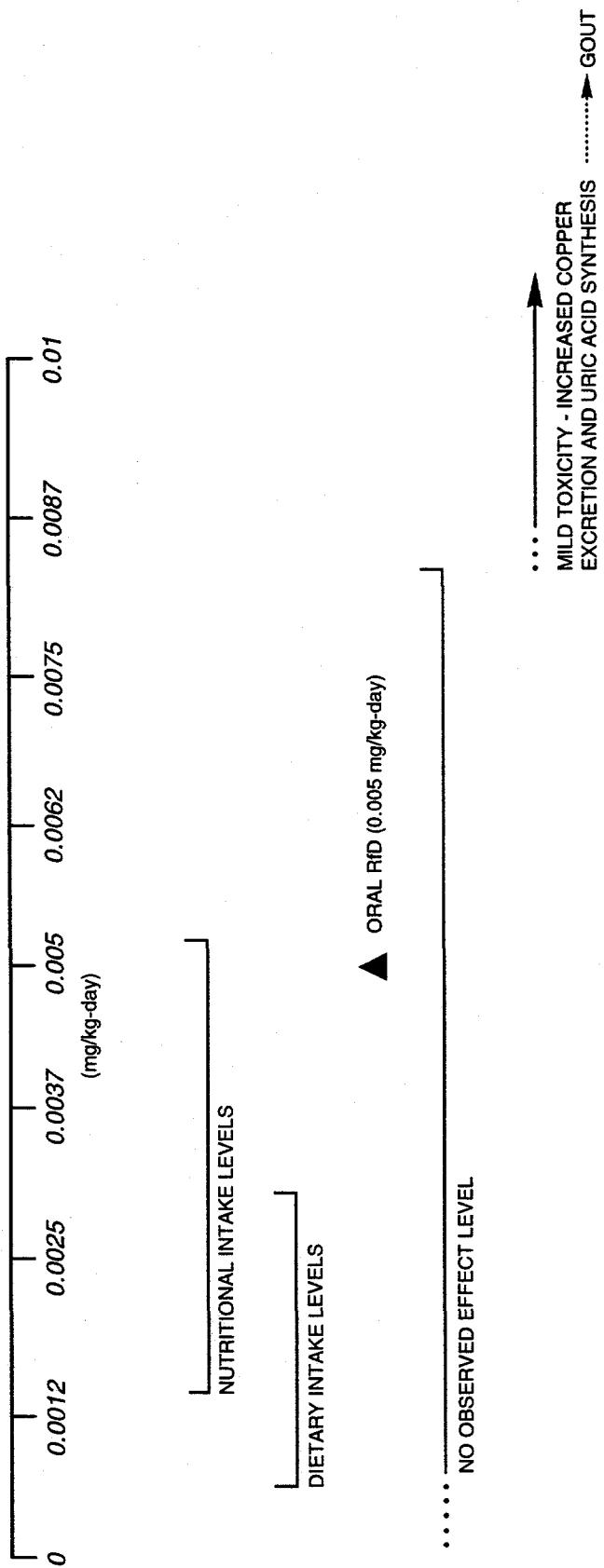


FIGURE 5.6
MOLYBDENUM TOXICITY RANGES

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 $\mu\text{g/L}$ of nickel and 80 percent had less than 10 $\mu\text{g/L}$, although in areas near nickel mining operations, levels as high as 200 $\mu\text{g/L}$ 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

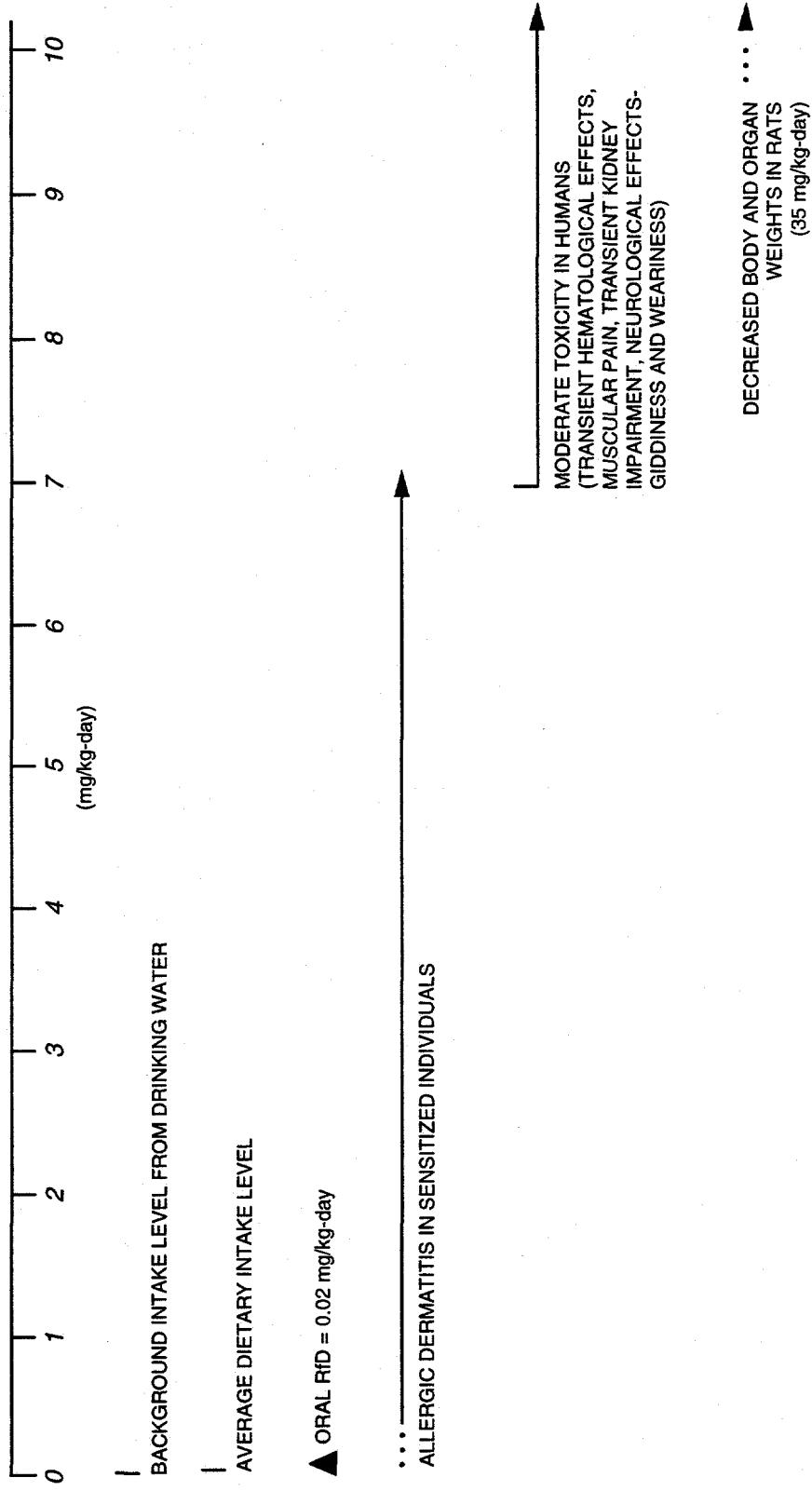


FIGURE 5.7
NICKEL TOXICITY RANGES

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.

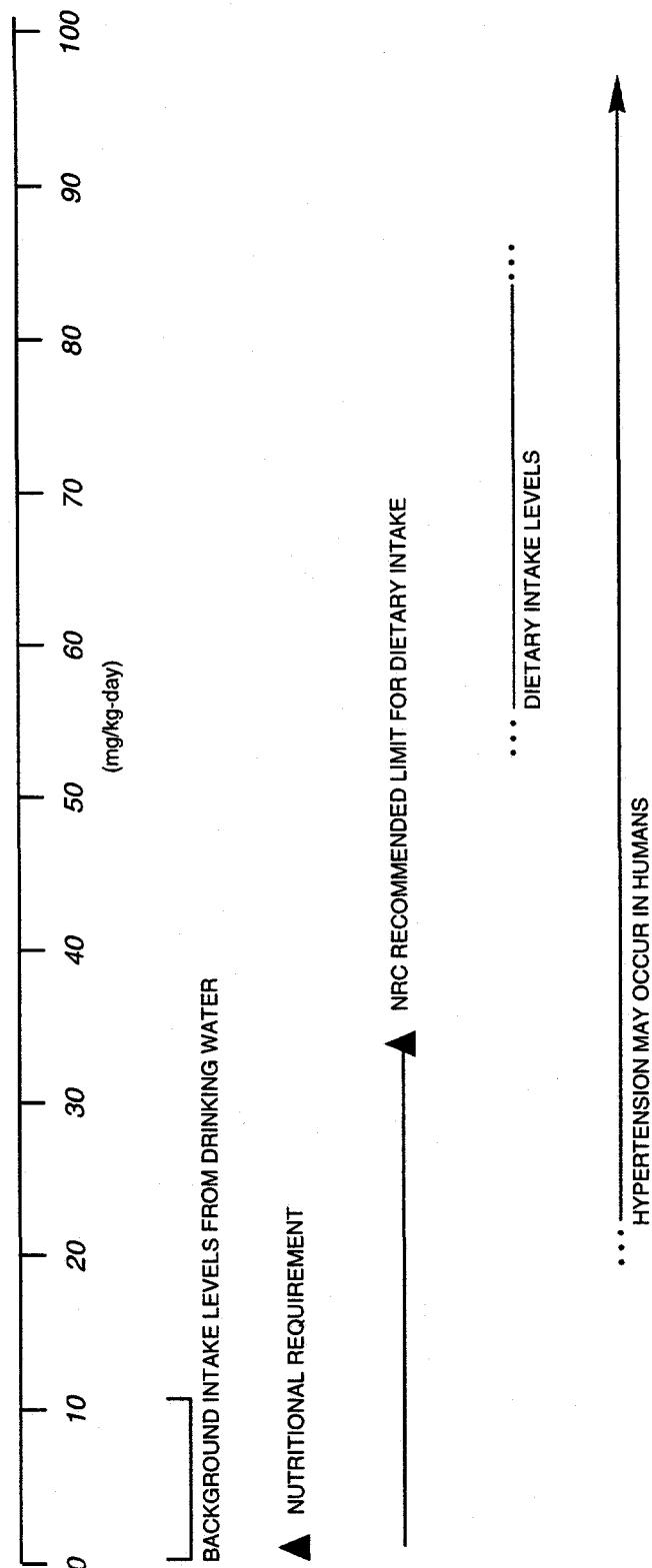


FIGURE 5.8
SODIUM TOXICITY RANGES

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.

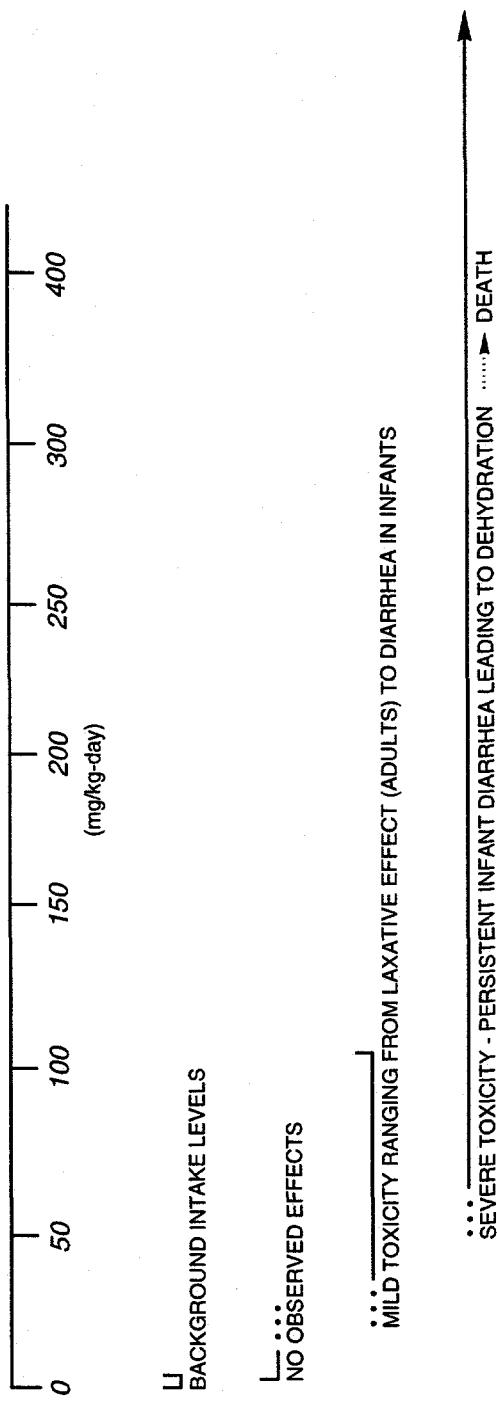


FIGURE 5.9
SULFATE TOXICITY RANGES

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

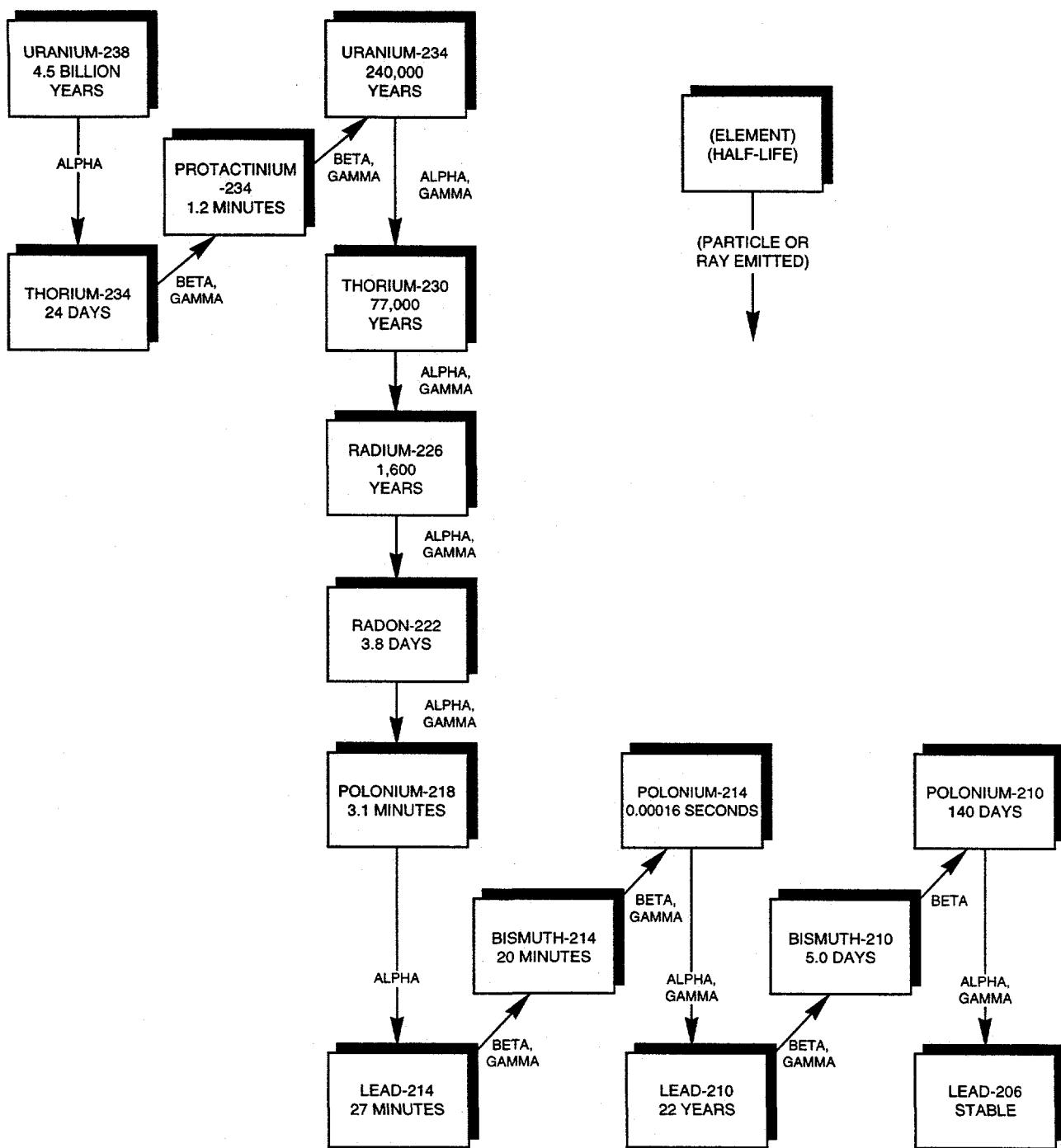


FIGURE 5.10
HALF-LIVES AND EMISSIONS FROM DECAY CHAIN OF URANIUM-238

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 CONSTITUENT INTERACTIONS

Some information is available on potential interactions between constituents 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 constituents 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 constituents of potential concern for human health. Therefore, the interactions described below should be recognized as factors that can influence the

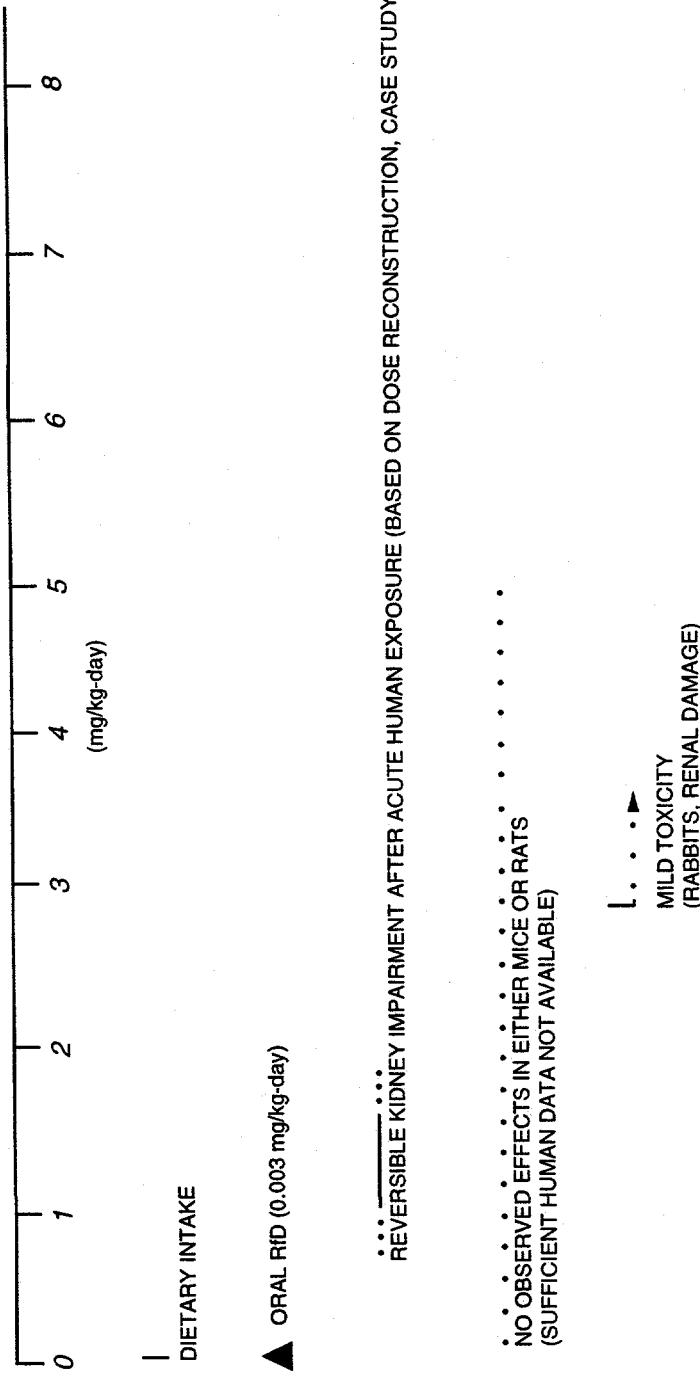


FIGURE 5.11
URANIUM TOXICITY RANGES

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

Constituent 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 CONSTITUENT 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, ionizing 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 |

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

^bIn (mg/kg-day)⁻¹.

^cNo 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 constituents.

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 constituent an individual takes into his or her body. Indeed, at lower levels some of the constituents associated with the mill tailings are beneficial to health, 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

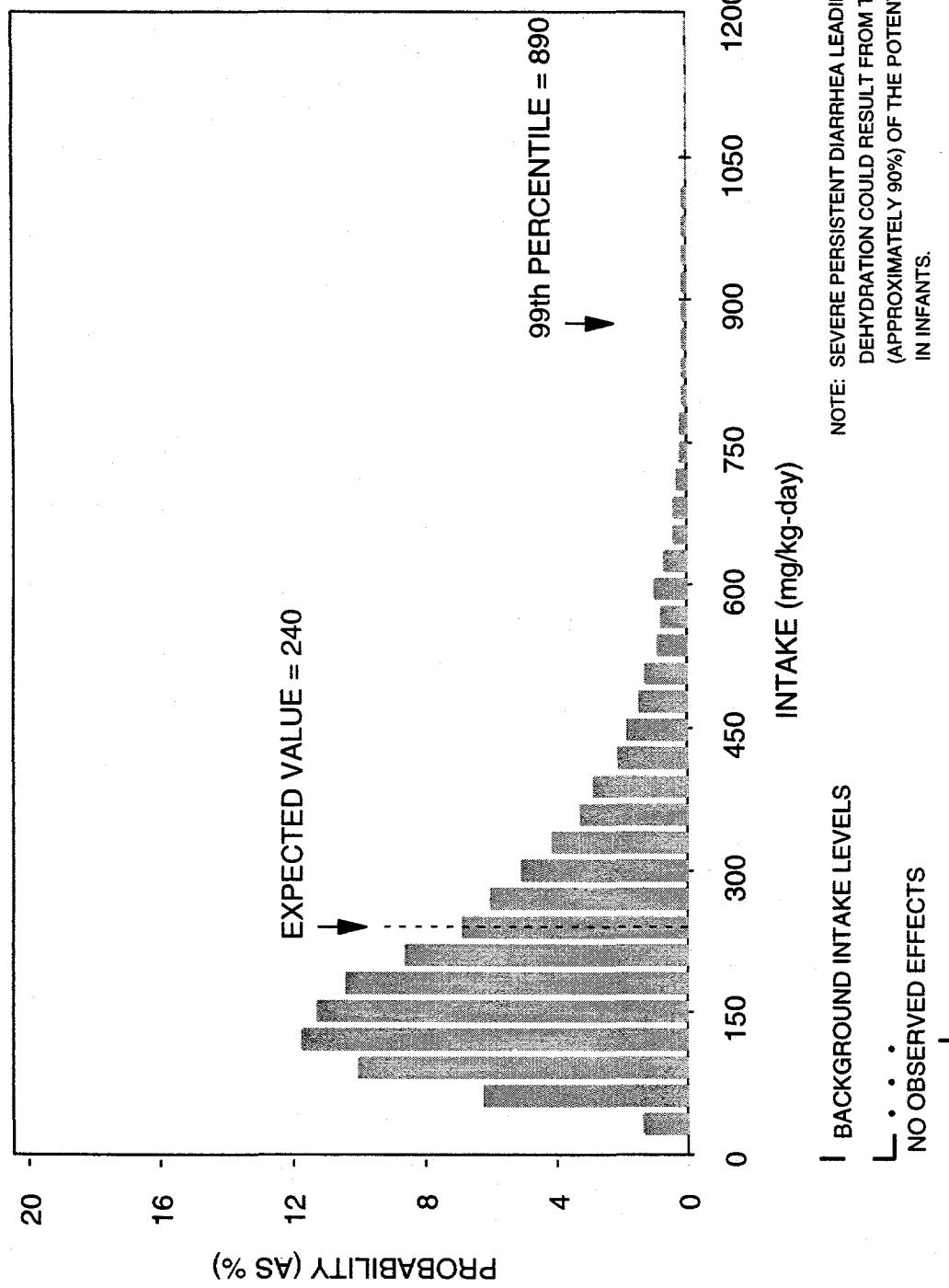
The results from the exposure assessment showing either the highest intake-to-body-weight ratios (or highest doses) or the toxicologically most sensitive group are used to evaluate potential health effects for noncarcinogens. For the constituents of potential concern at the Lakeview site, the highest 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



- SEVERE TOXICITY - PERSISTENT INFANT DIARRHEA LEADING TO DEHYDRATION
- MILD TOXICITY RANGING FROM LAXATIVE (ADULTS) TO DIARRHEA IN INFANTS

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

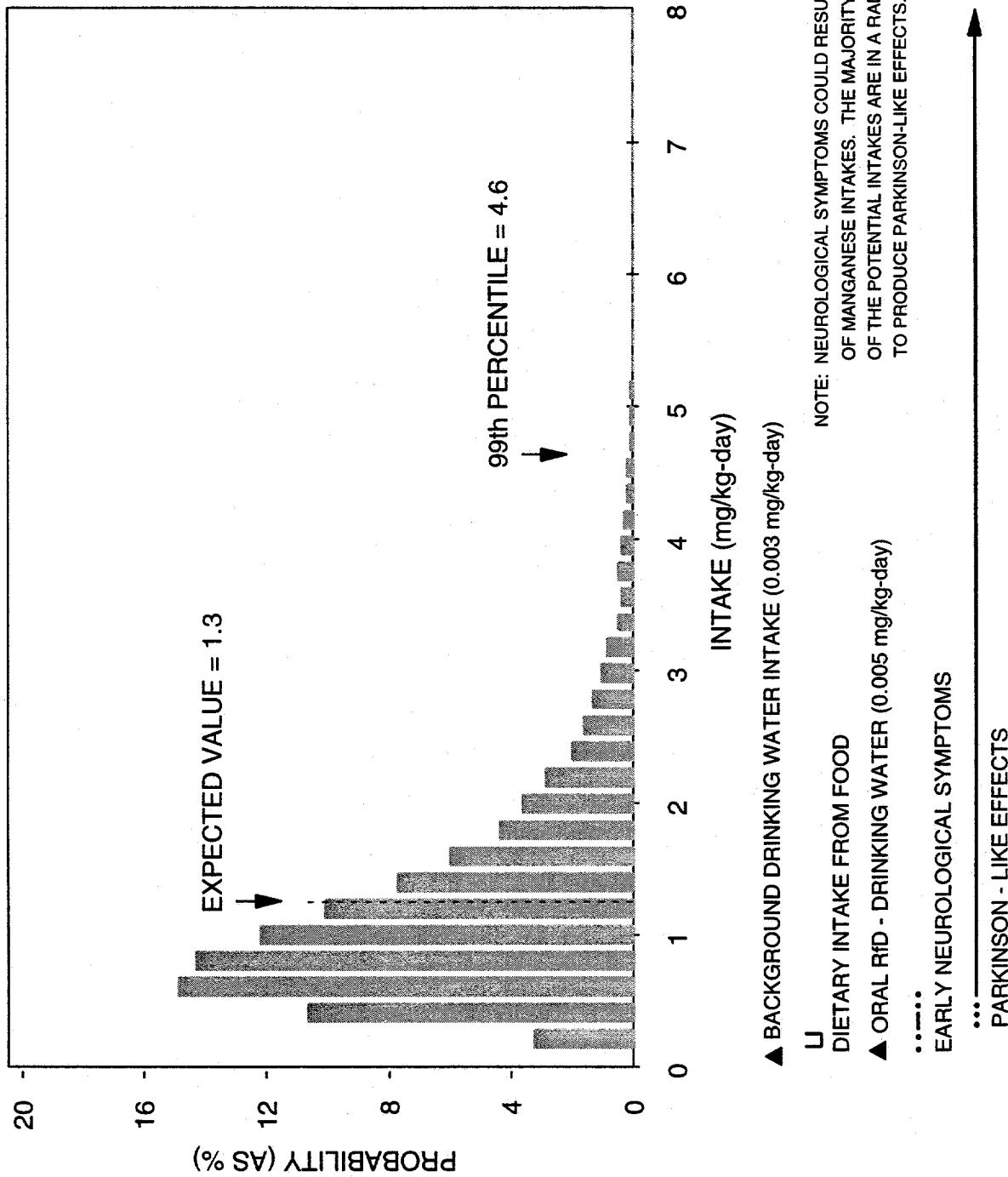


FIGURE 6.2
HEALTH EFFECTS OF POTENTIAL MANGANESE EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

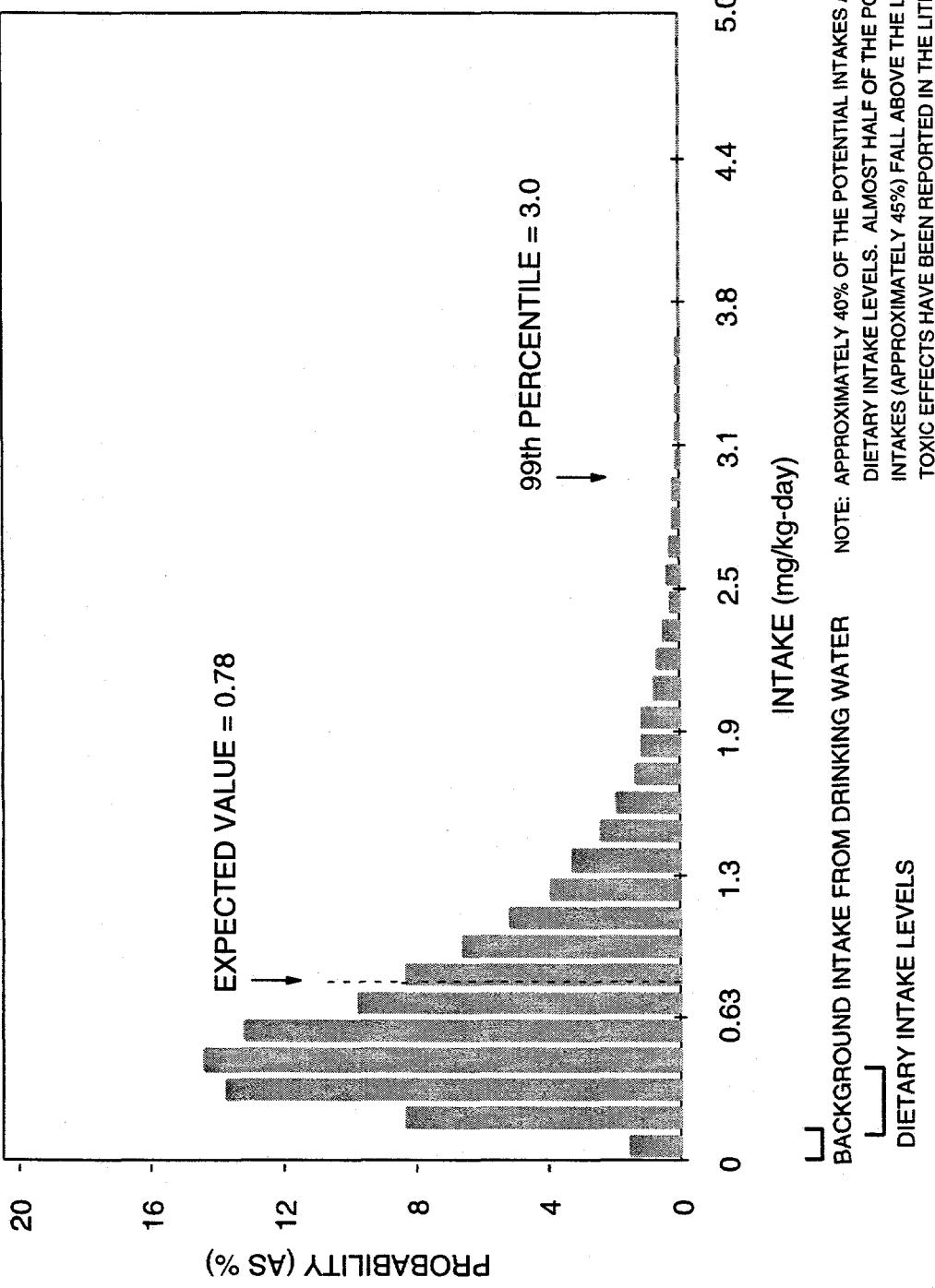


FIGURE 6.3
HEALTH EFFECTS OF POTENTIAL IRON EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

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).

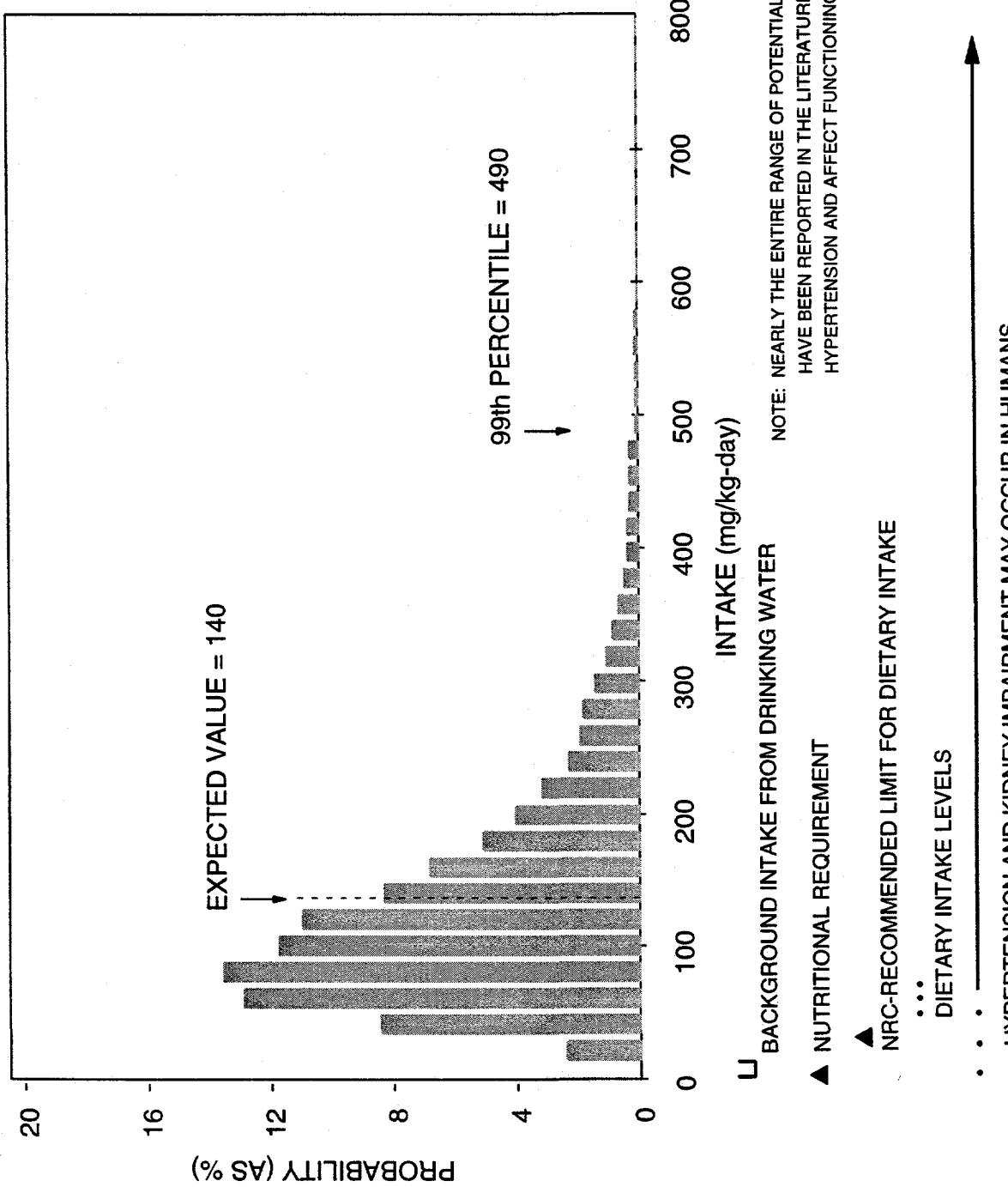


FIGURE 6.4
HEALTH EFFECTS OF POTENTIAL SODIUM EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE

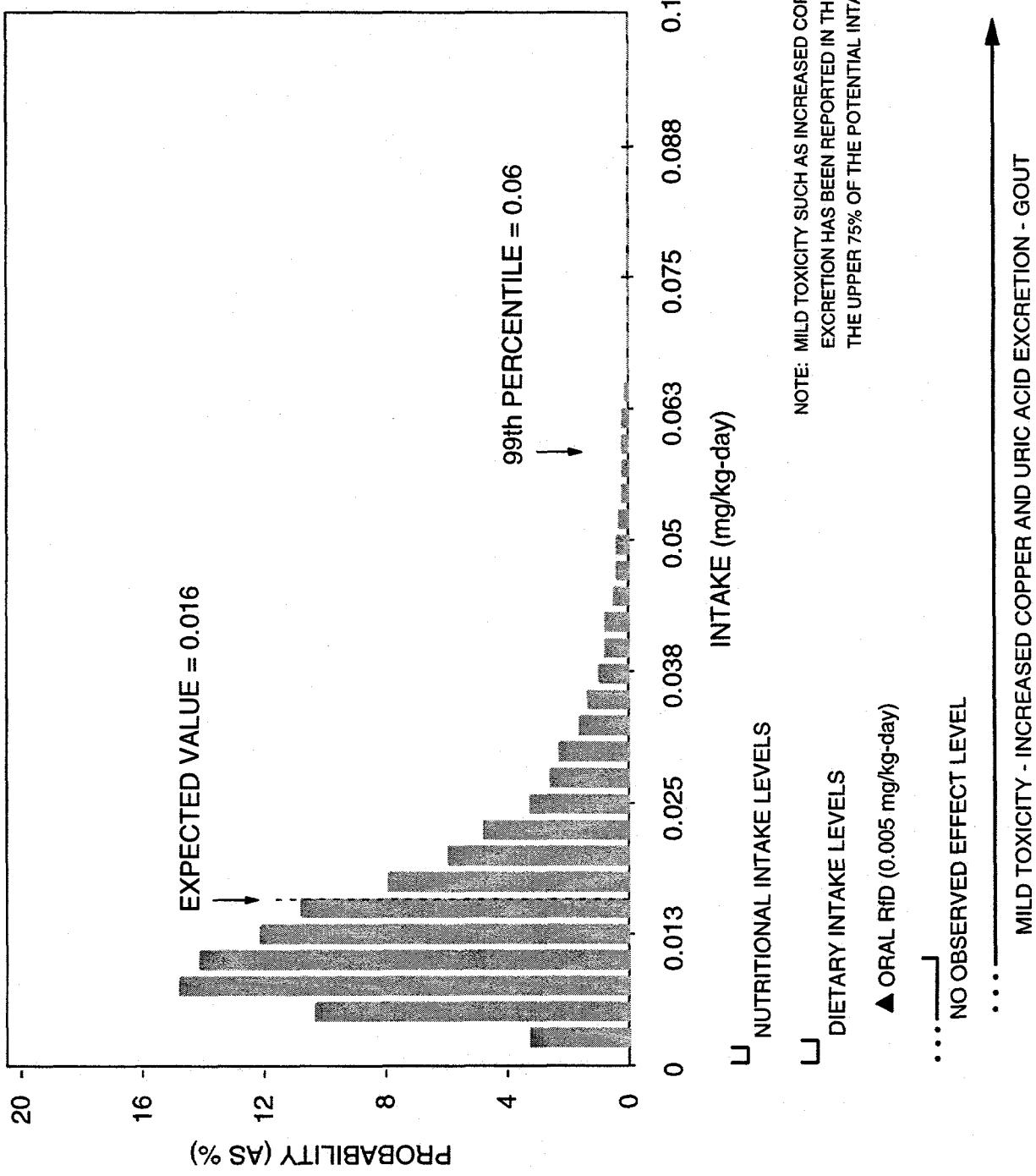
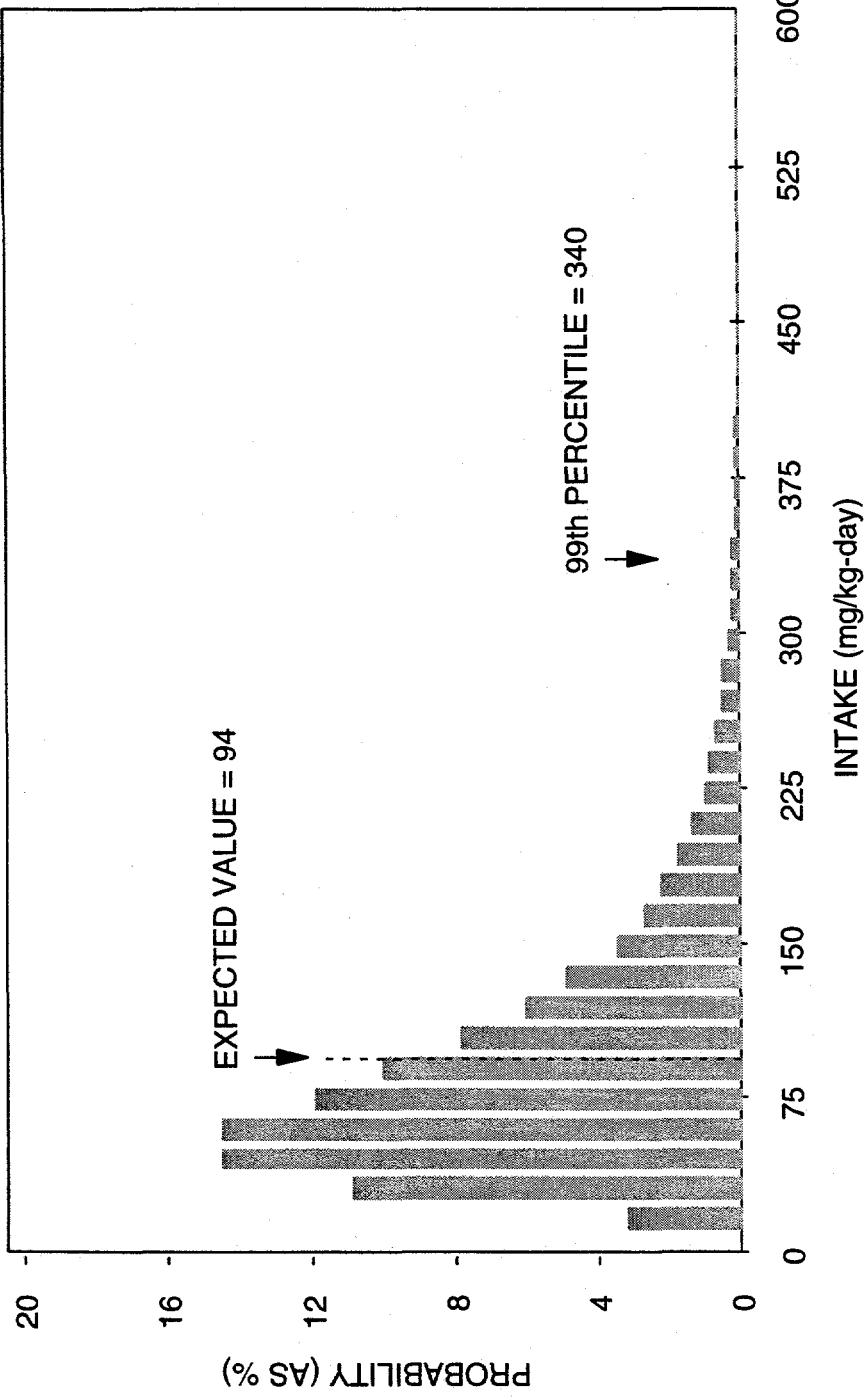
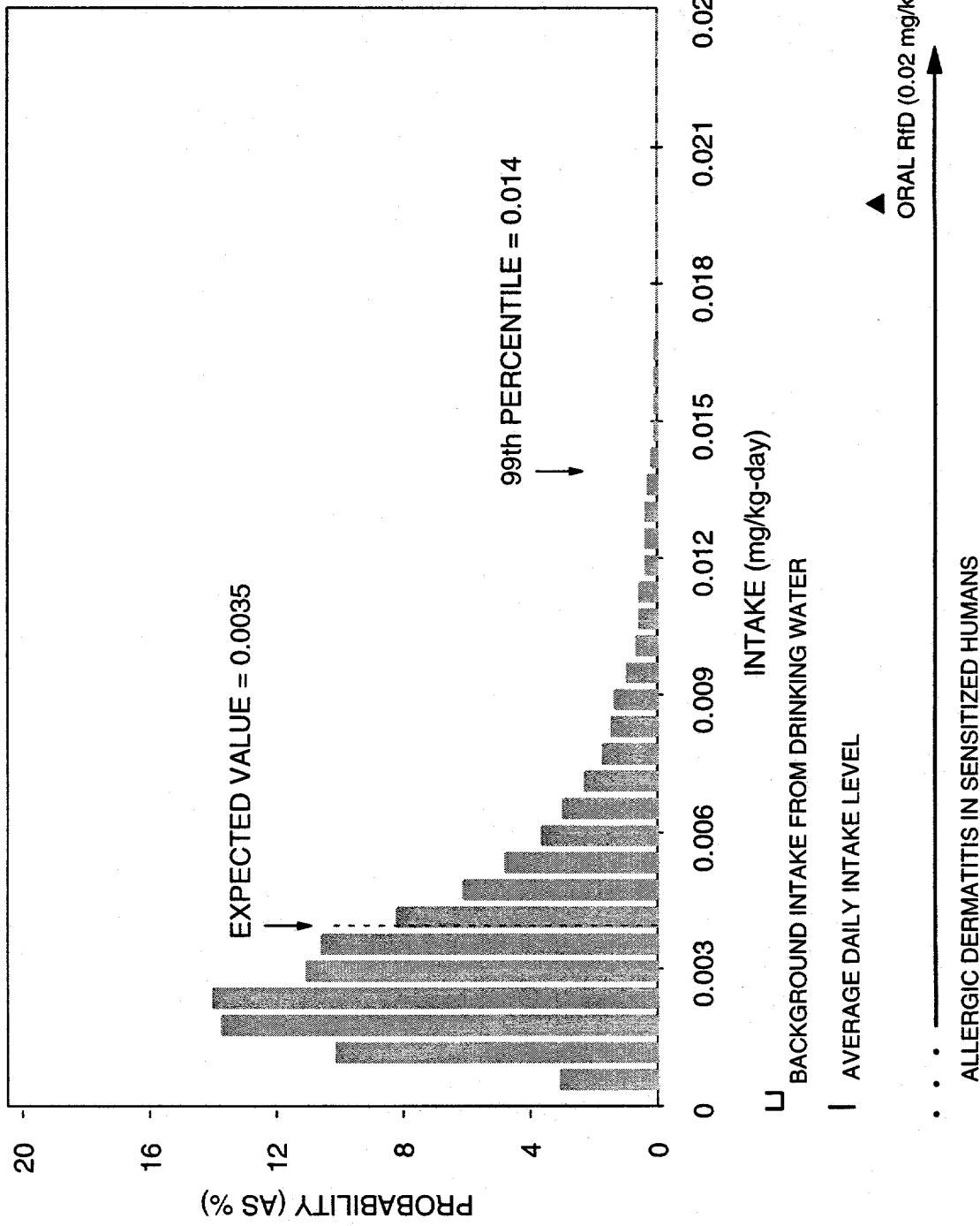


FIGURE 6.5
HEALTH EFFECTS OF POTENTIAL MOLYBDENUM EXPOSURE RANGES FOR CHILDREN
LAKEVIEW, OREGON, SITE



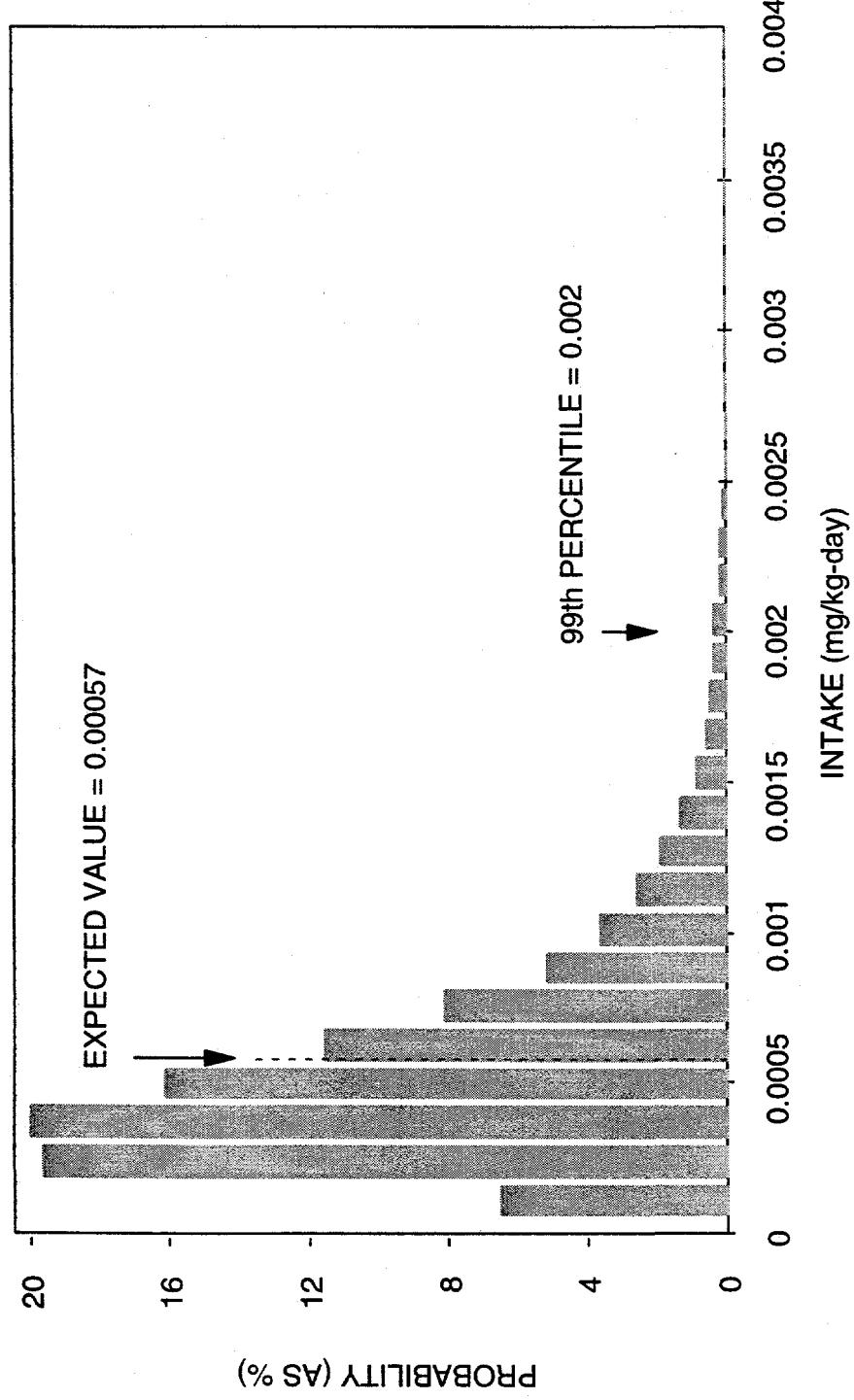
NOTE: ALTHOUGH A LARGE PORTION (APPROXIMATELY 90%) OF THE POTENTIAL INTAKES FALL ABOVE THE LEVEL WHERE HYPERTENSION HAS BEEN REPORTED IN THE LITERATURE, APPROXIMATELY 80% OF THE POTENTIAL INTAKES FALL WITHIN REPORTED DIETARY INTAKE LEVELS.

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

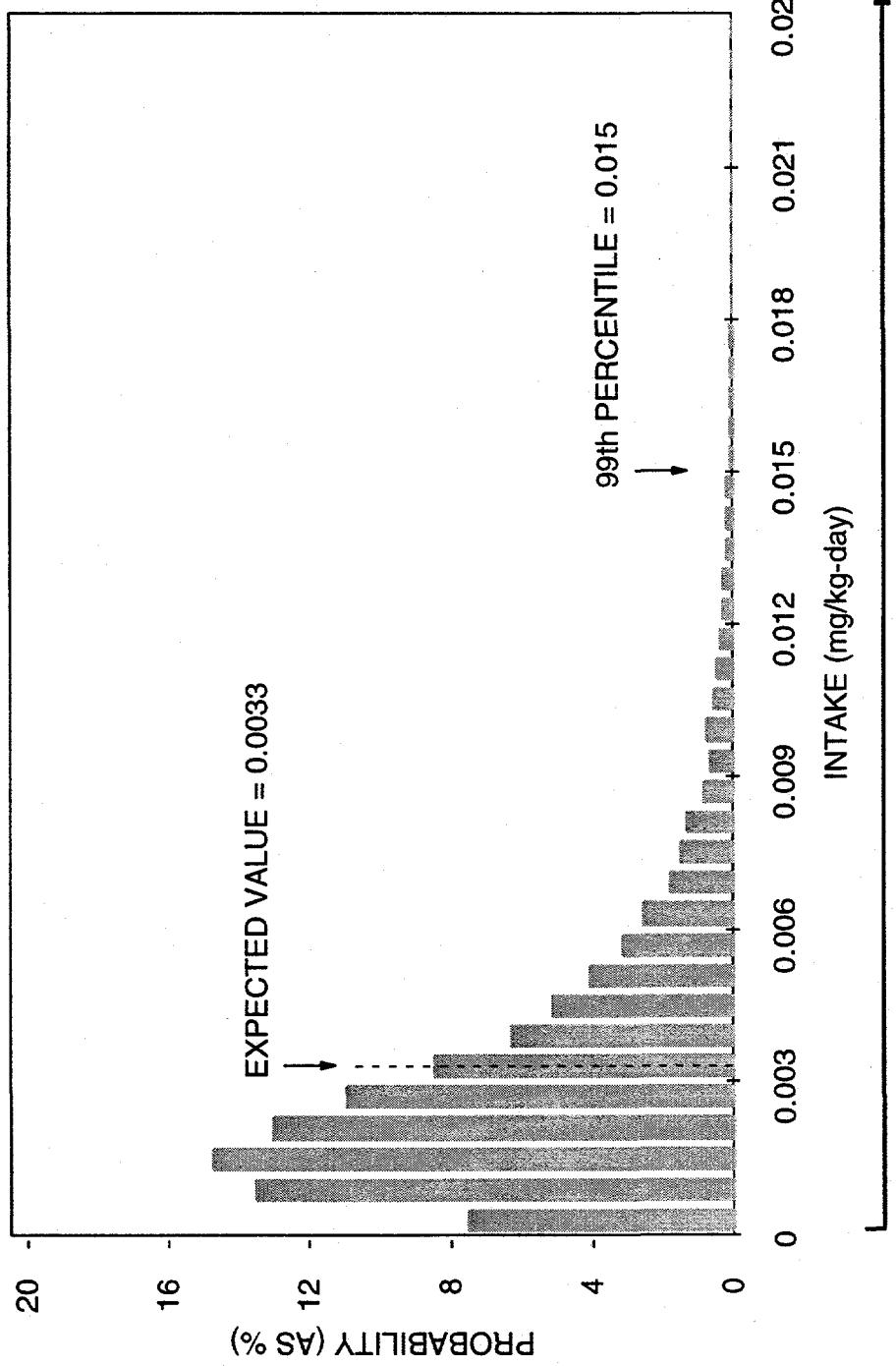


▲ ORAL RfD (0.003 mg/kg-day)

• NO OBSERVED EFFECTS IN EITHER MICE OR RATS
(SUFFICIENT HUMAN DATA NOT AVAILABLE)

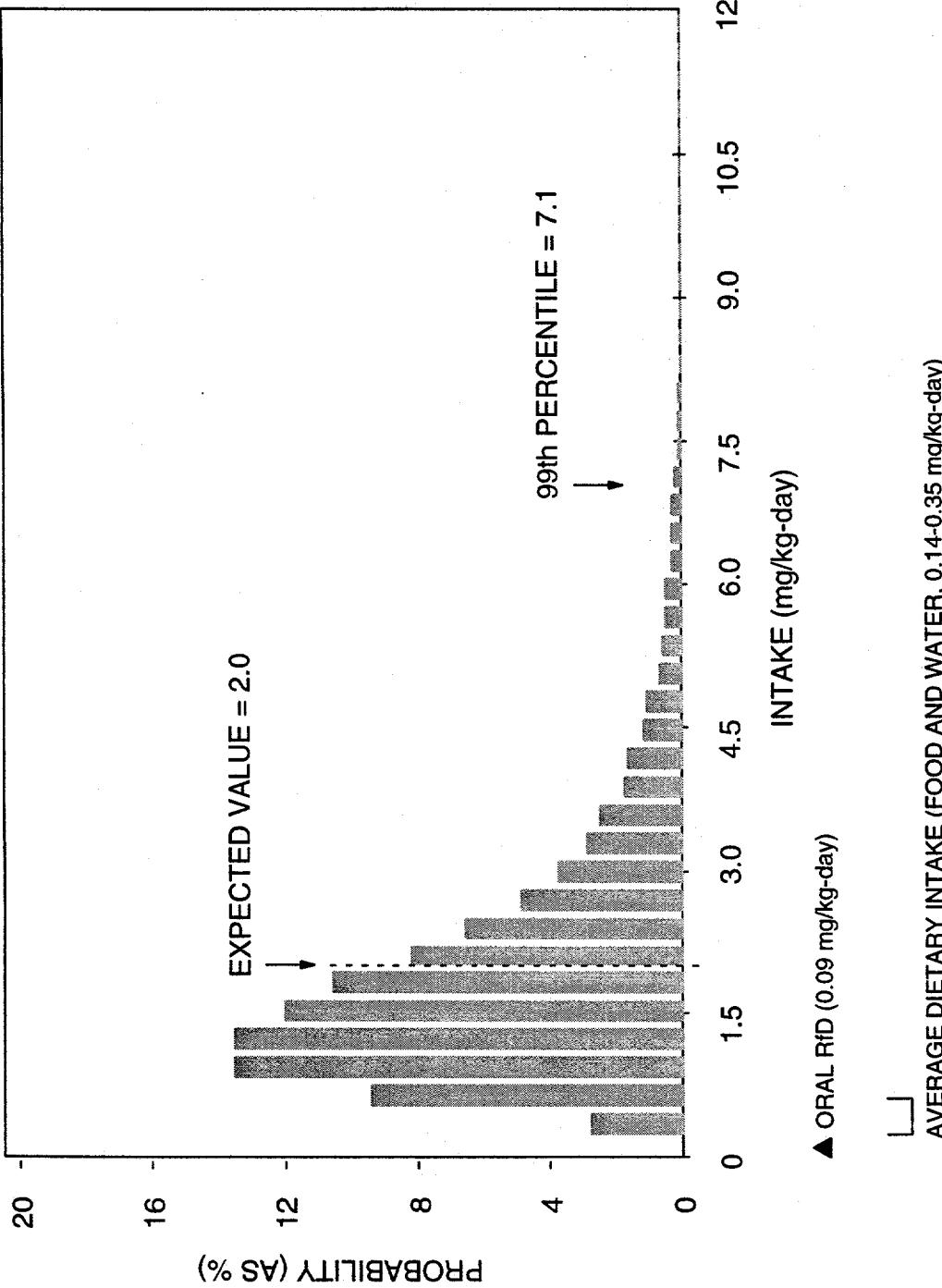
NOTE: NO HEALTH EFFECTS HAVE BEEN OBSERVED
FOR ANY OF THE INTAKES AT THIS SITE.

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



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



NOTE: ALTHOUGH THE MAJORITY OF THE SIMULATED INTAKES FALL ABOVE THE ORAL RID, 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

For each constituent of potential concern, the exposure contribution from the other pathways (dermal contact with ground water, ingestion of garden produce, and the ingestion of meat and milk from livestock) would be approximately 7 percent or less. The exception is molybdenum, where the exposure doses from the milk and meat pathways would be approximately 20 percent per pathway. These increases in the potential exposure for each constituent 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 constituents 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 constituents 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 constituents 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

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

| Constituent of potential concern | Concentration in ground water ^a | Ground water exposure dose ^b | | Excess lifetime cancer risk ^c |
|----------------------------------|--|---|---|--|
| | UCL | UCL | Oral SF (mg/kg-day; pCi) ⁻¹ | |
| 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 |

^aConcentrations in milligrams per liter for arsenic and in picocuries per liter for polonium-210 and uranium.

^bExposure doses in milligrams per kilogram per day for arsenic and in picocuries per lifetime for polonium-210 and uranium.

^cCalculated 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.

^dUranium 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.

disturbance, and compulsive behaviors could be expected to occur. The estimated amount of chloride that could be associated with hypertension in sensitive individuals is in the range of 36 to 180 mg/kg-day. Ingesting chloride in water used to cook with (10 mg/kg-day) would not appreciably increase the potential for hypertension 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:

- One of the primary limitations associated with this risk assessment is the uncertainty of the actual constituent concentrations underneath the site. Several factors add to this uncertainty. There are no data regarding constituent concentrations underneath the site since 1988. However, downgradient concentrations indicate constituent concentrations have not changed. Therefore, the highest historical levels of site-related constituents statistically above background were used in this risk assessment. If the concentrations beneath the site actually have decreased, this risk assessment potentially overestimates risk of ingesting ground water as drinking water at this site.
- 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 constituents 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).
- 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 constituents 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 constituents 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 constituent concentrations in environmental media to aquatic life criteria and guidelines for sediment, vegetation, wildlife, livestock watering, and crop irrigation. The comparisons determine if the constituents 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 constituents 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 constituents. The evaluation of limited data for surface water and sediment in Section 3.6 suggests surface water has not been discernibly impacted and the levels of constituents detected above background in sediment could reflect the natural variation associated with drainage areas other than the site. However, the limited database creates uncertainty about the impact of site-related contamination on surface water and sediment. Therefore, constituent 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 constituent 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 constituents 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 constituents 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 constituents 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 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 CONSTITUENTS OF ECOLOGICAL CONCERN

The complete list of ground water constituent levels that exceed background levels was considered in assessing constituents 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 constituents 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 constituents of potential concern for sediment.

7.4 POTENTIAL IMPACTS TO WILDLIFE AND PLANTS

7.4.1 **Ground water**

Concentrations of the constituents 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 constituents 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

| Constituent 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 |

^aEstimated concentration in vegetative portions, calculated as estimated soil concentration multiplied by Bv.

^bEstimated concentration in nonvegetative portions, calculated as estimated soil concentration multiplied by Br.

^cConcentrations 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).

^dThe value of 200 mg/kg is a recommended criterion of the Illinois Institute of Environmental Quality (Saric, 1986).

^eBv and Br factors available for elemental sulfur only; thus, these factors were reduced by a factor of 3 for sulfate.

^fThe 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 constituents 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 constituents 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 constituents 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 constituents 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 constituent 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 constituents 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 constituents 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 constituents of potential concern in ground water with available water quality values, Lakeview, Oregon, site

| Constituent 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 |

^aFrom ODEQ (1992), unless specified otherwise. These values are standards protective of aquatic life via chronic exposure.

^bFrom EPA (1972), unless specified otherwise. Values shown are for water used continuously on all soils.

^cNo 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).

^dNo 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).

^eNo 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).

^fWater-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).

^gFrom National Research Council (1971).

^hNo 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 constituents 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 constituents 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.

However, the DOE is sponsoring an irrigation and plant growth study at the University of Arizona to determine the uptake of manganese, molybdenum, uranium, and selenium by three plants (sudan grass, carrots, and squash). The measurement of uptake can be used to assess potential exposure doses of inorganics from animal or human consumption of plants. In addition, to study the toxicity of these inorganics to the plants themselves (phytotoxicity), changes in dry-weight biomass is measured.

Preliminary results are available (Baumgartner, 1995). No effect was observed in sudan grass and carrots up to a manganese concentration in irrigation water of 31 mg/L (the highest treatment level). The squash biomass is reduced slightly at the highest level of exposure of 31 mg/L, indicating a small degree of phytotoxicity. Phytotoxicity was not observed for molybdenum in any of the plant species up to the highest treatment level of 2.56 mg/L. Additionally, phytotoxicity was not observed for uranium in any of the three plant species up to the highest treatment level of 5 mg/L. These results suggest that the EPA guidance values used for the preceding evaluation of manganese and molybdenum may be overprotective. However, the UCL concentration for manganese at this site is 40 mg/L, where some phytotoxic response could be observed.

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 constituents of potential concern (iron, manganese, and uranium) at this site (EPA, 1988; NOAA, 1990). There are no guidelines for iron, manganese, and uranium.

Many of the constituents 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 constituents 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 constituents 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 constituents 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 constituents 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 constituents, 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 constituents have not adversely affected the quality of the surface water bodies in the site vicinity.

Three constituents (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 constituents.

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 constituents 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 forage 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 forage 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 constituents (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 constituents.

8.2 LIMITATIONS OF THIS RISK ASSESSMENT

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

- One of the primary limitations associated with this risk assessment is the uncertainty of the actual constituent concentrations underneath the site. Several factors add to this uncertainty. There are no data regarding constituent concentrations underneath the site since 1988. However, downgradient concentrations indicate constituent concentrations have not changed. Therefore, the highest historical levels of site-related constituents statistically above background were used in this risk assessment. If the concentrations beneath the site actually have decreased, this risk assessment potentially overestimates risk of ingesting ground water as drinking water at this site.
- This document evaluates risks associated with exposures only to inorganic constituents of ground water at the UMTRA Project site near Lakeview. Potential organic constituents (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.
- Constituent 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 constituent 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 constituents 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.

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. Therefore, the following limitations associated with this site are recommended for consideration in future site investigations.

Table 8.1 Constituent maximum concentration limits and health advisories of constituents

| Constituent | UMTRCA MCL (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) | - | - |

^aFrom 40 CFR §192.02.^bExceeded in plume wells.^cEqual 10 mg/L nitrate as nitrogen.^dUnder review, 10 mg/L as nitrogen.

Dash indicates not available.

- The farthest southern extent of the contaminant plume has not been characterized.
- Substantiation of the conceptual model of saline ground water should be considered.
- Dilution and dispersion in conjunction with ground water flow is identified as a major control on the fate and transport of constituents at the Lakeview site. The present rates of ground water transport of site-related constituents, indicated by little or no change in constituent 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 constituent concentrations in individual wells is not understood.
- Three constituents (iron, manganese, and uranium) are identified as above background in the sediments adjacent to the site. These observed constituents could reflect natural variation associated with a different drainage area, such as Warner Creek, rather than site-related constituents. Additionally, surface water data suggest the surface water is not impacted by site-related contaminated ground water. However, the limited database precludes definitive conclusions about the origins of any constituents detected in these media.
- 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, this is an area of uncertainty at this site.
- 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, consideration of this exposure pathway may be warranted.

9.0 LIST OF CONTRIBUTORS

The following individuals contributed to the preparation of this report.

| Name | Contribution |
|------------------------------|---|
| J. Marshall, C. B. Day | Overall document responsibility; authorship |
| W. Brattin | Technical review |
| K. Smith | Statistical evaluation |
| R. Dotson, K. Monks | Hydrogeology |
| J. Marshall, C. H. Day | Ecological assessment |
| T. Jackson | Geochemistry |
| M. Gawthrop-Cooper | Institutional controls |
| B. Malczewska-Toth | Toxicology and peer review |
| L. Sanchez, WordCenter, Inc. | Text processing |
| E. Bond, K. DeGruyter | Graphic design |
| D. Tamez | Technical editing |



10.0 REFERENCES

Alexander et al. (F. W. Alexander, B. E. Clayton, and H. T. Delves), 1974. *Quarterly Journal of Medicine*, Vol. 53, pp. 89-111.

Baes et al. (C. F. Baes, R. D. Sharp, A. J. Sjoreen, and R. W. Shor), 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture*, ORNL-5786, prepared by Oak Ridge National Laboratory, Health and Safety Research Division, Oak Ridge, Tennessee.

Bailey, D., 1994. Oregon Department of Environmental Quality, Salem, Oregon, personal communication with C. Silva, Jacobs Engineering Group Inc., UPDCC File Location No. 13.19.7, Albuquerque, New Mexico, 11 July 1994.

Ballou et al. (J. E. Ballou, R. A. Gies, A. C. Case, D. L. Haggard, R. L. Buschbom, and J. L. Ryan), 1986. "Deposition and Early Disposition of Inhaled Uranium-233 Uranyl Nitrate and Uranium-232 Uranyl Nitrate in the Rat," *Health Physics*, No. 51, pp. 755-772.

Baumgartner, D. J., 1995. Director of Environmental Research Laboratory, University of Arizona, personal communication with R. Saar, Jacobs Engineering Group Inc., 6-10 July 1995.

Berlin, M., and B. Rudell, 1986. "Uranium," in *Handbook on the Toxicology of Metals*, second edition, L. Friberg, G. F. Nordberg, and V. B. Vouk, eds., Elsevier Science Publishers, Amsterdam, the Netherlands, pp. 647-658.

BFEC (Bendix Field Engineering Corporation), 1984. "Radiologic Characterization of the Lakeview, Oregon, Uranium Mill Tailings Remedial Action Site," prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico

Brater, C. D., 1992. "Treatment of Renal Disorders and the Influence of Renal Function on Drug Disposition," in *Clinical Pharmacology*, Vol. 31, pp. 776-785.

Brown et al. (D. E. Brown, N. V. Peterson, and G. D. McLean), 1980. *Preliminary Geology and Resource Potential of the Lakeview Area, Oregon*, State of Oregon Department of Geology and Mineral Industries, Open-File Report O-80-9.

Buchet et al. (J. P. Buchet, R. Lauwerys, and H. Roels), 1981. "Urinary Excretion of Inorganic Arsenic and its Metabolites After Repeated Ingestion of Sodium Meta-Arsenite by Volunteers", in *International Archives of Occupational and Environmental Health*, No. 48, pp. 111-118.

- Buchet et al. (J. P. Buchet, H. Roels, and R. Lauwerys), 1980. "Repeated Surveillance of Exposure to Cadmium, Manganese, and Arsenic in School-Age Children Living in Rural, Urban, and Nonferrous Smelter Areas in Belgium," *Environmental Research*, No. 22, pp. 95-108.
- Calabrese, E. J., and R. W. Tuthill, 1977. "Elevated Blood Pressure and High Sodium Levels in the Public Drinking Water," *Archives of Environmental Health*, September/October 1977, pp. 200-202.
- Carson et al. (B. L. Carson, H. V. Ellis III, and J. L. McCann), 1986. *Toxicology and Biological Monitoring of Metals in Humans*, Lewis Publishers, Inc., New York, New York, pp. 226-230.
- Carter, R., 1994. Oregon Water Resources Department, Salem, Oregon, personal communication with C. Silva, Jacobs Engineering Group Inc., Albuquerque, New Mexico, UPDCC File Location No. 13.19.7, 11 July 1994.
- Casarett, L. J., and J. Doull, 1991. *Toxicology: The Basic Science of Poisons*, fourth edition, M. O. Amdur, J. Doull, and C. D. Klaassen, eds., Pergamon Press, New York, New York.
- Casarett, L. J., and J. Doull, 1986. "Toxic Effects of Metals," in *Toxicology: The Basic Science of Poisons*, third edition, Macmillan Publishing Company, New York, New York.
- CDH (Colorado Department of Health), 1991. *Basic Standards and Methodologies for Surface Water*, 3.1.0., Water Quality Control Commission, Denver, Colorado.
- Church, D. C., 1984. *Livestock Feeds and Feeding*, second edition, McGraw-Hill Publishing Co., New York, New York.
- Cothorn, C. R., and W. L. Lappenbusch, 1983. "Occurrence of Uranium in Drinking Water in the United States," *Health Physics*, No. 45, pp. 89-99.
- Crecelius, E. A., 1977. "Changes in the Chemical Speciation of Arsenic Following Ingestion by Man," *Environmental Health Perspective*, No. 19, pp. 147-150.
- Davis, S. N., and R. J. M. DeWeist, 1966. *Hydrology*, John Wiley and Sons, New York, New York.
- DHHS (U.S. Department of Health and Human Services), 1993. *Toxicological Profile for Arsenic*, prepared by Life Systems, Inc., for Clement International Corporation, U.S. Department of Health and Human Services, Atlanta, Georgia.
- DHHS (U.S. Department of Health and Human Services), 1992a. *Toxicological Profile for Boron and Compounds*, prepared by Clement International Corporation for the U.S. Department of Health and Human Services, Atlanta, Georgia.

DHHS (U.S. Department of Health and Human Services), 1992b. *Toxicological Profile for Manganese*, prepared by Clement International Corporation for the U.S. Department of Health and Human Services, Atlanta, Georgia.

DHHS (U.S. Department of Health and Human Services), 1991. *Toxicological Profile for Nickel*, prepared by Clement International Corporation for the U.S. Department of Health and Human Services, Atlanta, Georgia.

DOE (U.S. Department of Energy), 1995a. *Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project*, final, DOE/EIS-0198, April 1995, prepared by the U.S. Department of Energy, Environmental Restoration Division, UMTRA Project Team, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1995b. "Supplement to the Baseline Risk Assessment of Ground Water Contamination at the Uranium Mill Tailings Site Near Lakeview, Oregon," DOE/AL/62350-145S, prepared by the U.S. Department of Energy, Environmental Restoration Division, UMTRA Project Team, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1994a. *Human Health Risk Assessment Methodology for the UMTRA Ground Water Project*, DOE/AL62350-170, prepared for the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1994b. *Long-Term Surveillance Plan for the Collins Ranch Disposal Site, Lakeview, Oregon*, DOE/AL/62350-19F, Rev. 3, August 1994, prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1993. *UMTRA Water Sampling and Analysis Plan, Lakeview, Oregon*, DOE/AL/62350-69F, September 1993, prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1992. *Remedial Action Plan and Site Design for Stabilization of the Inactive Uranium Mill Tailings Site at Lakeview, Oregon*, UMTRA-DOE/AL-050510.0000, Rev. 1, July 1992, prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.

DOE (U.S. Department of Energy), 1985a. "Processing Site Characterization Report of the Uranium Mill Tailings at Lakeview, Oregon," UMTRA-DOE/AL-050106.0000, draft, prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.

- DOE (U.S. Department of Energy), 1985b. *Environmental Assessment of Remedial Action at the Lakeview Uranium Mill Tailings Site, Lakeview, Oregon*, DOE/EA-0271, April 1985, prepared by the U.S. Department of Energy, UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.
- Eisler, R., 1990. "Boron Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review," U.S. Fish and Wildlife Service, Contaminant Hazard Reviews, Report No. 20, Washington, D.C.
- Eisler, R., 1989. "Molybdenum Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review," U.S. Fish and Wildlife Service, Contaminant Hazard Reviews, Report No. 19, Washington, D.C.
- Elinder, C.G., 1986. "Iron," in *Handbook on the Toxicology of Metals*, L. Friberg, G. F. Nordberg, and V. Houk, eds., second edition, Elsevier Press Science Publishers, Amsterdam, the Netherlands.
- EPA (U.S. Environmental Protection Agency), 1994a. Integrated Risk Information System, on-line database, Office of Research and Development, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1994b. Health Effects Assessment Summary Tables (HEAST), annual update, March 1994, 9200.6-303(93-1), Office of Solid Waste and Emergency Response, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1992a. *Drinking Water Criteria Document for Sulfate*, final, Health and Environmental Criteria Division, Office of Science and Technology, U.S. Environmental Protection Agency.
- EPA (U.S. Environmental Protection Agency), 1992b. Toxic Substances Spreadsheet, U.S. Environmental Protection Agency, Region IV, Atlanta, Georgia.
- EPA (U.S. Environmental Protection Agency), 1991. *Assessment and Control of Bioconcentratable Contaminants in Surface Water*, draft, March 1991, Office of Research and Development, Office of Water, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1989a. *Risk Assessment Guidance for Superfund*, Vol. I, Human Health Evaluation Manual, EPA/540/1-89/002, Office of Emergency and Remedial Response, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1989b. *Risk Assessment Guidance for Superfund*, Vol. II, Environmental Evaluation Manual, EPA/540/1-89/001, Office of Emergency and Remedial Response, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1988. "Interim Sediment Quality Criteria Values for Nonpolar Hydrophobic Organic Contaminants," Office of Water Regulations and Standards, Washington, D.C.

- EPA (U.S. Environmental Protection Agency), 1984. *Health Assessment Document: Inorganic Arsenic*, U.S. Environmental Protection Agency, Washington, D.C.
- EPA (U.S. Environmental Protection Agency), 1972. "Water Quality Criteria," National Academy of Sciences and National Academy of Engineering, Washington, D.C.
- Ferm, V. H., 1971. "The Teratogenic Profile of Sodium Arsenite in the Golden Hamster," *Archives of Environmental Health*, No. 22, pp. 557-560.
- Fisenne et al. (I. M. Fisenne, P. M. Perry, and N. H. Harley, 1988. "Uranium in Humans," *Radiation Protection Dosimetry*, No. 24, pp. 127-131.
- Follett, R. H., and P. N. Soltanpour, 1985. "Irrigation Water Quality Criteria," Colorado State University Extension Service, Publication No. 506.
- Freis, E. D., 1976. "Salt, Volume and the Prevention of Hypertension," *Circulation*, Vol. 53, No. 4, pp. 589-595.
- Friberg et al. (L. Friberg, G. F. Nordberg, and V. B. Vouk), 1986. *Handbook on the Toxicology of Metals, Volume II: Specific Metals*, second edition, Elsevier, New York, New York.
- Hem, J. D., 1970. "Study and Interpretation of the Chemical Characteristics of Natural Water," U.S. Geological Survey Water Supply Paper 1473.
- Hill, S. R., 1989. Manager, Jacobs Engineering Group Inc., Albuquerque Operations Office, personal communication with M. Matthews, U.S. Department of Energy, Albuquerque Operations Office, Albuquerque, New Mexico, UPDCC File Location No. 0.2.2.38.41, Albuquerque, New Mexico, 8 November 1989.
- Hood, R. D., 1972. "Effects of Sodium Arsenite on Fetal Development," *Bulletin of Environmental Contamination and Toxicology*, No. 7, p. 216.
- Hull, R. N., and G. W. Suter, 1994. *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1994 Revision*, ES/ER/TM-95/RI, U.S. Department of Energy, Oak Ridge National Laboratory, Oak Ridge Tennessee.
- Ishinishi et al. (N. Ishinishi, K. Tsuchiya, M. Vahter, and B. A. Fowler), 1986. "Arsenic," in *Handbook on the Toxicology of Metals, Volume II: Specific Metals*, second edition, Elsevier Press Science Publishers B.V., Amsterdam, the Netherlands.
- Israeli, M., and C. B. Nelson, 1992. "Distribution and Expected Time of Residence for U.S. Households," *Risk Analysis*, Vol. 13, No. 1, pp. 65-72.
- Josephs, H. W., 1958. "Absorption of Iron as a Problem in Human Physiology," *Blood*, Vol. 13, pp. 1-54.

- Kabata-Pendias, A., and H. Pendias, 1992. *Trace Elements in Soils and Plants*, second edition, CRC Press, Boca Raton, Florida.
- Kondakis et al. (X. G., N. Makris, M. Leotsinidis, M. Prinou, and T. Papapetropoulos), 1989. "Possible Health Effects of High Manganese Concentration in Drinking Water," *Archives of Environmental Health*, 1989, Vol. 44, No. 3, pp. 175-178.
- Leach et al. (L. J. Leach, R. M. Gelein, B. J. Panner, C. L. Ylie, and C. C. Cox), 1984. *The Acute Toxicity of the Hydrolysis Products of Uranium Hexafluoride (UF6) When Inhaled by the Rat and Guinea Pig*, final report, ISS K/SUB-81-9039-3, National Technical Information Service, Springfield, Virginia.
- Lindgren et al. (A. Lindgren, M. Vahter, and L. Dencker), 1982. "Autoradiographic Studies on the Distribution of Arsenic in Mice and Hamsters Administered ^{74}As -Arsenite or -Arsenate," *Acta Pharmacology and Toxicology*, No. 51, pp. 253-265.
- Lönnerdal et al. (B. Lönnerdal, C. L. Keen, J. G. Bell, and B. Sandstrom), 1987. "Manganese Uptake and Retention," in *Nutritional Bioavailability of Manganese*, C. Kies, ed., American Chemical Society, symposium series 354, American Chemical Society, Washington, D.C., pp. 9-20.
- Marafante, E., and M. Vahter, 1987. "Solubility, Retentions and Metabolism of Intratracheally and Orally Administered Inorganic Arsenic Compounds in Hamsters," *Environmental Research*, No. 42, pp. 72-82.
- Marcus, W. L., and A. S. Rispin, 1988. "Threshold Carcinogenicity Using Arsenic as an Example," in *Advances in Environmental Toxicology*, Volume XV, Risk Assessment and Risk Management of Industrial and Environmental Chemicals, C. R. Cothorn, M. A. Mehlman, and W. L. Marcus, eds., Princeton Scientific Publishing Co., New Jersey, pp. 133-159.
- Maynard, E. A., and H. C. Hodge, 1949. "Study of Toxicity of Various Uranium Compounds When Fed to Experimental Animals," in *Pharmacology and Toxicology of Uranium Compounds*, Vol. 1, National Nuclear Energy Services, Div. VI, C. Voegtlin and H. C. Hodge, eds., McGraw Hill, New York, New York, pp. 309-376.
- Mena et al. (I. Mena, K. Horiuchi, K. Burke, and G. C. Cotzias), 1969. "Chronic Manganese Poisoning, Individual Susceptibility and Absorption of Iron," *Neurology*, No. 19, pp. 1000-1006.
- Meneely, G. R., and H. D. Battarbee, 1976. "Sodium and Potassium," *Nutrition Reviews*, No. 8, pp. 225-235.
- Merritt, R. C., 1971. *The Extractive Metallurgy of Uranium*, Colorado School of Mines Research Institute, Golden, Colorado.

- Morrison et al. (G. M. P. Morrison, G. E. Batley, and T. M. Florence), 1989. "Metal Speciation and Toxicity," *Chem. Br.*, No. 25, p. 791.
- National Research Council, 1989. *Recommended Daily Allowances*, tenth edition, Commission on Life Sciences, National Academy Press, Washington, D.C.,
- National Research Council, 1980. *Drinking Water and Health*, Vol. 3, Safe Drinking Water Committee, National Academy Press, Washington, D.C.
- National Research Council, 1973. *Medical and Biological Effects of Environmental Pollutants: Manganese*, Committee on Biologic Effects of Atmospheric Pollutants, Division of Medical Sciences.
- National Research Council, 1971. *Nutrient Requirements of Dairy Cattle*, Committee on Animal Nutrition, fourth edition, the National Academy of Sciences, Washington, D.C.
- NCRP (National Council on Radiation Protection and Measurements), 1984. *Exposures From the Uranium Series With Emphasis on Radon and its Daughter*, NCRP Report No. 77, Bethesda, Maryland.
- NOAA (National Oceanic and Atmospheric Administration), 1990. "The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program," NOAA technical memo, NOS, OMA 52.
- NOAA (National Oceanic and Atmospheric Administration), 1974. *Climates of the States, Vol. II, Western States*, Water Information Center, Port Washington, New York.
- ODEQ (Oregon Department of Environmental Quality), 1992. *Water Quality Criteria Summary*, Oregon Administrative Rules, 340-41.
- Oregon Water Resources Department, 1994. Administrative Rules for Water Supply Well Construction and Maintenance Standards, Water Law 537.775, January 1994.
- Palisade Corporation, 1992. *@RISK, Risk Analysis and Simulation Add-In for Lotus 1-2-3*, Version 2.01, Newfield, New York.
- Parkhurst et al. (D. L. Parkhurst, D. C. Thorstenson, and L. N. Plummer), 1980. *PHREEQE-A Computer Program For Geochemical Calculations*, U.S. Geological Survey, Water-Resources Investigation 80-96.
- Pershagen, G., and M. Vahter, 1979. *Arsenic*, Swedish Environmental Protection Board, Stockholm, Sweden.
- PNL (Pacific Northwest Laboratory), 1989. *Sorption Distribution Coefficient Data, Multimedia Environmental Pollutant Assessment*, Pacific Northwest Laboratory, Richland, Washington.

- Power, E. A., and P. M. Chapman, 1992. "Assessing Sediment Quality," in *Sediment Toxicity Assessment*, G. A. Burton, Jr., ed., Lewis Publishers.
- Prister, B. S., 1969. *GKIAE Report by Atomizdat*, Moscow (Canadian translation AEC/TR/7178 (1970), and USCEAR Rep. A/AC.82/G/L 1298).
- Robinson et al. (M. F. Robinson, J. M. McKenzie, C. Thomson, and A. L. Van Rijn), 1973. *Br. J. Nutr.*, Vol. 30, pp. 195-205.
- Roseberry, A. M., and D. E. Burmaster, 1992. "Lognormal Distributions for Water Intake by Children and Adults," *Risk Analysis*, Vol. 12, No. 1, pp. 99-104.
- Rosoff, B., and H. Spencer, 1964. *Nature (London)*, Vol. 202, pp. 410-411.
- Saric, M., 1986. "Manganese," in *Handbook on the Toxicology of Metals*, Vol. II, L. Friberg, G. F. Nordberg, V. B. Vouk, eds., Elsevier, New York, pp. 354-386.
- Shacklette, H. T., and J. G. Boerngen, 1984. *Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States*, U.S. Geological Survey Professional Paper 1270.
- Shannon, R. L., and D. S. Strayer, 1989. "Arsenic-Induced Skin Toxicity," *Human Toxicology*, No. 8, pp. 99-104.
- Stevens et al. (W. F. Stevens, W. Bruenger, D. R. Atherton, J. M. Smith, and G. N. Taylor), 1980. "The Distribution and Retention of Hexavalent ^{233}U in the Beagle," *Radiation Research*, No. 83, pp. 109-126.
- Sullivan et al. (M. F. Sullivan, P. S. Ruemmler, J. L. Ryan, and R. L. Bushhom), 1986. "Influence of Oxidizing or Reducing Agents on Gastrointestinal Absorption of U, Pu, Am, Cm, and Pm by Rats," *Health Physics*, No. 50, pp. 223-232.
- Tracy et al. (B. L. Tracy, J. M. Quinn, J. Lahey, A. P. Gilman, K. Mancuso, A. P. Yagdinas, and D. C. Villeneuve), 1992. "Absorption and Retention of Uranium from Drinking Water by Rats and Rabbits," *Health Physics*, Vol. 62, pp. 65-73.
- Tracy et al. (B. L. Tracy, F. A. Prantl, and J. M. Quinn,) 1983. "Transfer of ^{226}Ra , ^{210}Pb , and Uranium from Soil to Garden Produce: Assessment of Risk," *Health Physics*, Vol. 44, p. 469.
- Tseng, W. P., 1977. "Effects and Dose-Response Relationship of Skin Cancer and Blackfoot Disease With Arsenic," *Environmental Health Perspectives*, No. 19, pp. 109-119.
- Venugopal, B., and T. D. Luckey, 1978. "Chemical Toxicity of Metals and Metalloids," in *Metal Toxicity in Mammals*, Plenum Press, New York, New York.

- Walker, G. W., 1980. *Preliminary Report on the Geology of the Lakeview Uranium Area, Lake County, Oregon*, U.S. Geological Survey Open-File Report 80-532.
- Webb, J. L., 1966. *Enzymes and Metabolic Inhibitors*, Academic Press, New York, New York.
- Weigel, F., 1983. "Uranium and Uranium Compounds," in *Kirk-Othmer Encyclopedia of Chemical Technology*, Vol. 23, third edition, M. Grayson, ed., John Wiley and Sons, New York, New York, pp. 502-547.
- Whitney et al. (E. N. Whitney, E. N. Hamilton, and S. R. Rolfs), 1990. *Understanding Nutrition*, fifth edition, West Publishing Company, St. Paul, Minnesota.
- WHO (World Health Organization), 1981. *Arsenic*, Environmental Health Criteria 18, published under the joint sponsorship of the United Nations Environment Programme, the International Labour Organisation, and the World Health Organization, Geneva, Switzerland.
- Wolfin, John, 1985. U.S. Fish and Wildlife Service, Boise, Idaho, personal communication with the Technical Assistance Contractor, Jacobs Engineering Group Inc., UMTRA Project Office, Albuquerque Operations Office, Albuquerque, New Mexico.
- Wrenn et al. (M. E. Wrenn, P. W. Durbin, B. Howard, J. Lipsztein, J. Rundo, E. T. Still, and D. L. Willis), 1985. "Metabolism of Ingested U and Ra," *Health Physics*, No. 48, pp. 601-633.

CODE OF FEDERAL REGULATIONS

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

40 CFR Part 264, *Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities*, U.S. Environmental Protection Agency, Administration, Washington, D.C.

50 CFR Part 17, *Endangered and Threatened Wildlife and Plants*, U.S. Department of the Interior.

FEDERAL REGISTER

45 FR 82480, *Review of Plant Taxa for Listing as Endangered or Threatened Species*, 15 December 1980.

48 FR 53640, *Supplement to Review of Plant Taxa for Listing*, 28 November 1983.

60 FR 2854, *Ground Water Standards for Remedial Actions at Inactive Uranium Processing Sites*; final rule, U.S. Environmental Protection Agency, 11 January 1995.

UNITED STATES CODE

42 USC §7901 *et seq.*, *Uranium Mill Tailings Radiation Control Act*, 8 November 1978.