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

October 1994

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October 1994

**Prepared for
U.S. Department of Energy
UMTRA Project Office
Albuquerque, New Mexico**

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

CITIZENS' SUMMARY

This *Baseline Risk Assessment of Ground Water Contamination at the Uranium Mill Tailings Site in Lakeview, Oregon*, evaluates potential impacts to public health or the environment resulting from ground water contamination at the former uranium mill processing site. Tailings and other contaminated material at this site were placed in an off-site disposal cell by the U.S. Department of Energy's (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project. The remedial activities at the site were completed in 1989. Currently, the UMTRA Project is evaluating ground water contamination. This risk assessment is the first document specific to this site for the Ground Water Project.

This risk assessment evaluates the most contaminated ground water that flows beneath the former processing site. The evaluation focuses on the shallow ground water because this portion of the aquifer in both background and downgradient areas is the most contaminated. The Lakeview area has two discrete types of ground water: geothermal and nongeothermal; the geochemistry of the water indicates that the two water types mix to varying degrees throughout the area. Background ground water quality in the nongeothermal unit is naturally of low quality, with somewhat elevated concentrations of nitrate, phosphate, and ammonium. The background geothermal ground water is of even poorer quality, with elevated concentrations of total dissolved solids, sulfate, chloride, sodium, boron, and fluoride.

The monitor wells that have consistently shown the highest concentrations of most contaminants are used to assess risk. This risk assessment will be used in conjunction with additional activities and documents to determine what remedial action may be needed for contaminated ground water at the site.

Ground water on the former processing site is not currently used, and some residents in the area are supplied with municipal water. Some residents in the vicinity of the processing site reportedly use ground water from private wells as a primary potable source, while other residents drink bottled water but use ground water from private wells for other household uses. Elevated levels of some inorganics have been reported in domestic wells downgradient of the processing site. However, the originally low-lying land surface has been filled over the last 12 years with a mixture of wood chips, boiler ash, and log pond dredgings. Infiltrating precipitation moving through this fill may be leaching natural salts into the ground water. Additionally, the soils in the area have been identified as "sodic and saline" due to excess salt; these salts may be migrating into underlying ground water. The salt leaching has contributed to degradation of the natural ground water quality in the Lakeview area.

This risk assessment will assume current and future use of ground water resources, including the use of ground water for drinking, cooking, and bathing. There is also the potential for exposure to the contaminants in ground water by using the water for domestic animals or irrigation. The potential exists for these animals and plants to accumulate ground water contaminants and for people to be indirectly exposed by drinking milk and eating meat and garden produce.

This risk assessment follows an approach outlined by the U.S. Environmental Protection Agency (EPA). The first step is to evaluate ground water data collected from monitor wells at the site. Evaluation of these data showed that the contaminants of potential concern in the ground water are arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium.

The next step in the risk assessment is to estimate how much of these contaminants people would be exposed to if they drank from a well installed in the contaminated ground water at the former processing site. Contaminant concentrations vary within the plume and within the two discrete types of ground water chemistry present at this site. They also vary each time a well is sampled. In addition, people vary in how much they weigh and drink. Therefore, this risk assessment uses probability distributions to determine how much of each contaminant would likely be ingested if someone installed a well at the site. The probability distributions describe how likely it is that a particular event will happen. For example, based on population survey data, probability distributions can describe what percentage of people drink a half-gallon of water each day and what percentage drink only one cup of water each day.

The estimated amounts of contaminants people might take in from these routes are then compared to the toxic effects these contaminant levels might cause. Based on this analysis, no adverse health effects would be expected following ingestion of garden produce watered with contaminated ground water. However, long-term irrigation with ground water would be toxic to the plants themselves because the contaminants would accumulate in the soil. Ingestion of meat from livestock watered with contaminated ground water would not be expected to represent a hazard to humans. However, direct consumption of the ground water from the former processing site could represent a concern.

If ground water at the Lakeview site were used as drinking water, the most significant health hazards would be posed by high concentrations of sulfate and manganese. Sulfate intake from drinking ground water at the site would cause severe diarrhea in infants that could lead to dehydration. The manganese concentrations at the former processing site are at levels that have been reported in the literature to cause central nervous system problems when ingested over a long period of time.

Other contaminants that occur in the ground water at levels associated with adverse noncancer health effects when ingested over a long period of time are iron, molybdenum, and chloride. The upper 40 percent of simulated exposures for the iron concentrations in ground water are above the dose that could cause pigmentation of the skin, potential disruption of liver and endocrine function, and at very high intakes, cirrhosis of the liver and diabetes. However, almost all the iron in the Lakeview site ground water is in the trivalent form, which is less toxic than iron in the divalent form. The adverse effect associated with molybdenum is mild toxicity manifested primarily by mineral imbalances. The chloride concentrations in ground water could result in hypertension, although most of the exposure range is within normal dietary levels.

The levels of arsenic in ground water at the site would result in an intake level from drinking water above the EPA's maximum acceptable level for additional lifetime cancer risks of 1 in 10,000. The additional cancer risk estimated from ingesting contaminated ground water for a lifetime would be 2 in 1000 for ground water containing the median arsenic concentration measured in monitor wells at the former processing site. The additional cancer risks estimated from ingestion of contaminated ground water containing the median or upper confidence limit concentrations of either polonium-210 or uranium would be within the EPA's potentially acceptable range of 10^{-4} to 10^{-6} .

This risk assessment also evaluated potential effects on livestock if the ground water were used to water them. Based on available information, boron and sulfate are the primary contaminants that could represent a hazard to livestock. The sulfate and boron concentrations are high enough to cause adverse effects in cattle if the ground water were used as the sole source of long-term drinking water. Several contaminants could not be evaluated because information on drinking water concentrations that would be protective of livestock is not available.

An ecological threat could exist to plants at harvestable maturity that have roots in contact with soil saturated with the most contaminated ground water. This threat may exist because reported boron concentrations exceed concentrations associated with phytotoxicity, and concentrations of manganese and molybdenum fall within the range reported to be phytotoxic. The ground water would not be suitable as a body of water for fish to live in if a pond were created or if ground water discharged to an existing surface water body and comprised the bulk of the water. However, available surface water and sediment data from creeks in the site vicinity suggest that contaminated ground water from the site has not adversely affected the surface water and sediment quality.

Potential use of this ground water would not be suitable for continuous irrigation of crops because arsenic, boron, iron, manganese, molybdenum, and sodium could build up to toxic levels in the soil following long-term use.

A long-term ground water monitoring plan has been proposed and is being implemented for the former processing site and vicinity. As additional water quality data are collected and interpreted, the monitoring plan will be updated annually to provide ongoing protection for public health and the environment.

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LIST OF ACRONYMS AND ABBREVIATIONS

<u>Acronym</u>	<u>Definition</u>
ac	acre
ACL	alternate concentration limit
BCF	bioconcentration factor
CDPHE	Colorado Department of Public Health and Environment
cm	centimeter
cm/hr	centimeters per hour
cm/s	centimeters per second
cm ³ /s	cubic centimeters per second
°C	degrees Celsius
°F	degrees Fahrenheit
DHHS	U.S. Department of Health and Human Services
DOE	U.S. Department of Energy
DW	dry weight
EPA	U.S. Environmental Protection Agency
ft	foot
FWS	Fish and Wildlife Service
gpm	gallons per minute
ha	hectare
HEAST	Health Effects Assessment Summary Table
IRIS	Integrated Risk Information System
Kd	soil-to-water distribution coefficient
km	kilometer
km ²	square kilometer
L	liter
L/cm ³	liters per cubic centimeter
L/day	liters per day
L/kg	liters per kilogram
LOAEL	lowest-observed-adverse-effect level
μg	microgram
μg/L	micrograms per liter
μm	micron
m	meter
mi	mile
m ³	cubic meter
m ³ /s	cubic meters per second
MCL	maximum concentration limit
mEq/L	milliequivalents per liter
mg	milligram
mg/kg	milligrams per kilogram
mg/kg-day	milligrams per kilogram per day
mg/kg DW	milligrams per kilogram dry weight
mg/L	milligrams per liter

LIST OF ACRONYMS AND ABBREVIATIONS (Concluded)

<u>Acronym</u>	<u>Definition</u>
mi	mile
mi ²	square mile
NCP	National Contingency Plan
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NRC	U.S. Nuclear Regulatory Commission
ODEQ	Oregon Department of Environmental Quality
pCi	picocurie
pCi/L	picocuries per liter
PNL	Pacific Northwest Laboratory
ppm	parts per million
RAP	remedial action plan
RDA	recommended daily allowance
RfD	reference dose
RRM	residual radioactive materials
SAR	sodium adsorption ratio
SF	slope factor
SQC	sediment quality criteria
TAC	Technical Assistance Contractor
TDS	total dissolved solids
UCL	upper confidence limit
UMTRA	Uranium Mill Tailings Remedial Action
UMTRCA	Uranium Mill Tailings Radiation Control Act
yd ³	cubic yard

1.0 INTRODUCTION

The purpose of this baseline risk assessment is to determine whether ground water contamination at the Lakeview, Oregon, uranium mill tailings processing site has the potential to adversely affect public health or the environment. The Lakeview site is one of 24 abandoned uranium mill tailings sites 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.

Under the UMTRA Surface Project, the DOE stabilized the Lakeview tailings in a disposal cell that meets the requirements of the governing U.S. Environmental Protection Agency (EPA) standards (40 CFR Part 192 (1994)), minimizes radon and other radiation emissions, and minimizes further contamination of ground water at the processing site. At Lakeview, remedial action consisted of cleanup of contaminated buildings at the processing site and excavation and transport of tailings and other contaminated materials to a disposal site approximately 7 miles (mi) (11 kilometers [km]) north of the processing site and the town of Lakeview. Remedial action was completed in October 1989.

Under the UMTRA Ground Water Project, ground water contamination at the 24 UMTRA Project processing sites will be evaluated in more detail to determine what compliance activities are necessary to meet the requirements of Subpart B of the EPA standards. The DOE was authorized to perform ground water restoration as necessary in 1988 amendments to the UMTRCA.

The proposed evaluation strategy and remediation methodology for contaminated ground water at UMTRA Project sites are described in the *Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (DOE, 1994a). This baseline risk assessment is one of the site-specific documents prepared to evaluate potential health and environmental risks and to provide information to assist in determining which ground water compliance activity is appropriate for each site. Based on the findings of the programmatic environmental impact statement, record of decision, and additional ground water characterization, a site-specific environmental assessment will be prepared to evaluate and select the appropriate ground water compliance strategy for the Lakeview site.

This risk assessment is a baseline assessment in the sense that it describes pre-remediation ground water conditions at the site, with ground water contamination only partially characterized. Nonetheless, this document evaluates the potential for imminent public health or environmental risks that may need attention before the site is fully characterized.

The evaluation is based on available ground water data from the most contaminated wells at the processing site, and only major exposure pathways have been thoroughly examined. If future data collection, decisions, or actions at this site cause conditions to change, other pathways will be evaluated in more detail.

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

- Data evaluation.
 - Combining existing data from various site investigations and related reports.
 - Comparing sample results with background and tailings source data.
 - Selecting chemical data for use in the risk assessment.
- Exposure assessment.
 - Characterizing exposure settings.
 - Identifying exposure pathways.
 - Quantifying exposure.
- Toxicity assessment.
 - Identifying toxicity values.
 - Evaluating noncarcinogenic effects from radionuclides and noncarcinogens.
 - Evaluating carcinogenic effects from radionuclides and chemical carcinogens.
- Public health risk characterization.
 - Comparing toxicity ranges to predicted exposure ranges.
 - Combining risks across exposure pathways and multiple contaminants.
 - Characterizing uncertainties.
- Environmental risk.
 - Characterizing potential biota exposure pathways.
 - Identifying potential ecological receptors.
 - Evaluating environmental risk qualitatively.

This framework is incorporated in the methodology used to evaluate current human health risk at UMTRA Project sites and to estimate risks from potential future use of contaminated ground water or surface water near the former uranium processing site. A report describing this methodology is in preparation.

2.0 SITE DESCRIPTION

2.1 SITE BACKGROUND

The processing site is located approximately 1.5 mi (2.4 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. The mill was constructed to produce uranium ore, and 171 tons (155 metric tonnes) 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 were then 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 tonnes) 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 formerly occupied by the almost square-shaped tailings pile, approximately 64 ac (26 ha) were the evaporation ponds, and the remaining area included the former mill structures, ore storage area, and acres of windblown contamination from tailings. Figure 2.2 provides information on the site and land use in the vicinity of the site.

Surface remedial action at the Lakeview processing site consisted of cleanup of contaminated buildings, excavation of all contaminated soils, and transport to the disposal site. Approximately 926,000 cubic yards (yd³) (708,000 cubic meters [m³]) of contaminated materials from the tailings pile, evaporation ponds, windblown and waterborne deposit areas, and windblown-contaminated areas dispersed around the processing site were relocated to the Collins Ranch disposal site approximately 7 mi (11 km) north of Lakeview. Approximately 736,000 tons (667,700 tonnes) 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 U.S. Nuclear Regulatory Commission (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 (°F) (7.8 degrees Celsius [°C]) and ranges from an average

LEGEND

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

5 0 5 10 15 MILES

8 0 8 16 24 KILOMETERS

FIGURE 2.1
LAKEVIEW, OREGON, SITE LOCATION MAP

LEGEND

- PASTURE/ RANCHLAND
- RESIDENTIAL/COMMERCIAL
- LUMBER MILLS
- RANCHLAND/ IRRIGATION FIELDS
- PUBLIC PROPERTY
- COUNTY RACETRACK
- U.S. HIGHWAY
- STATE HIGHWAY

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

temperature of 27°F (-2.8°C) in January to 67°F (19°C) in July. The average annual precipitation is 14 inches (36 centimeters [cm]) (NOAA, 1974). Maximum rainfall occurs during January and February, while the least rainfall occurs from June through October. Thunderstorms are common during the summer. The average annual snowfall accumulation is 48 inches (122 cm), with the largest amount (15.6 inches [40 cm]) falling during January (NOAA, 1974).

2.3 HYDROGEOLOGY

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

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

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

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

Vertical gradients have been noted between the shallow and deep monitor well clusters (DOE, 1993). The background well cluster (501 and 502) shows head 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 remain relatively constant. Static water level elevations in the downgradient monitor well

clusters range from relatively 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 level fluctuations are expected to occur due to a combination of natural discharge, evapotranspiration, and localized pumpage in private wells.

Hydraulic properties at the Lakeview processing site were estimated; the results are presented in the surface remedial action plan (DOE, 1992) and the processing site characterization report (DOE, 1985a). These site characterization efforts included two 48-hour aquifer performance tests (with 24 hours of continuous pumping followed by 24 hours of recovery) to determine the hydraulic properties of the shallow and deep lacustrine zones. In one test, a shallow monitor well (519) was pumped at a rate of 0.60 gallons per minute (gpm) (38 cubic centimeters per second [cm^3/s]) 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} centimeters per second [cm/s]), using the Jacob-Cooper, Theis, and Boulton methods of analysis. In the other aquifer performance test, a deep monitor well (520) was pumped at a rate of approximately 12 gpm (760 cm^3/s) 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/s), 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/s), 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/s) in the shallow zone to 160 ft per year (2.5×10^{-4} cm/s) in the deep zone. These values were determined using the hydraulic conductivities (ranging from 0.78 to 11 ft per day [2.8×10^{-4} to 39×10^{-4} cm/s]) and an assumed effective porosity of 0.15 (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 tailings pile and is approximately 8 mi (13 km) south of the existing 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

Table 2.1 Static ground water levels, Lakeview, Oregon

Location ID	North coordinate (ft)	East coordinate (ft)	Formation of completion	Flow relationship	Casing elevation (ft MSL)	Ground elevation (ft MSL)	Log date	Log time	Depth from top of casing (ft)	Depth from ground (ft)	Ground water elevation (ft MSL)
501	8523.5	7525.3	SP	Upgradient	4771.05	4769.32	08/25/93	15:29	5.28	3.55	4765.77
503	7931.9	3921.1	SP	Downgradient	4747.73	4745.89	08/25/93	15:54	10.99	9.15	4736.74
505	7476.2	3342.9	SP	Downgradient	4744.64	4742.74	08/25/93	15:59	9.88	7.98	4734.76
507	6599.3	6102.0	SP	Crossgradient	4742.77	4741.00	08/25/93	16:35	6.76	4.99	4736.01
509	6897.1	4762.6	SP	Downgradient	4742.14	4740.06	08/25/93	16:47	7.36	5.28	4734.78
518	5914.7	2231.0	SP	Downgradient	4739.79	4737.55	08/25/93	16:18	10.31	8.07	4729.48
540	7340.20	5486.57	AL	Downgradient	4747.89	4745.93	08/25/93	15:20	9.09	7.13	4738.80
541	7938.71	4223.85	AL	Downgradient	4747.56	4745.55	08/25/93	15:10	11.34	9.33	4736.22

Report date: July 1, 1994.

SP - Sand or gravelly sand, poorly graded.

AL - Alluvium.

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

The drainage area for Hammersley Creek is about 3.3 square miles (mi^2) (8 square kilometers [km^2]) with an approximate slope of 0.06. Surface water in an area of approximately 30 mi^2 (80 km^2) drains into Thomas Creek above the Lakeview site. The majority of this drainage basin is located 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 (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 for irrigated 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 the population 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, used for grazing or growing forage crops such as alfalfa. The fields are marshy to the northeast, north, and west of the site because of surface discharge from the geothermal springs 800 to 1800 ft (240 to 550 m) north of the site. These marshy areas, including several small ponds, are associated with Hunters Creek. At the headwaters of Hunters Creek, about 1800 ft (550 m) north of the site (Figure 2.2), 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 pasture land with about seven associated residences and a trailer park on Highway 395, about 1600 feet (480 m) east of the site. East of the southern portion of the site are pasture land, 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. East of Highway 395 are primarily residences. The area southeast of the former mill site is primarily heavy industry (lumber mills). Immediately south of the site is pasture land.

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 in with light industry. The area southwest and west of the site is pasture land and forage crop land irrigated by canals (Figure 2.2).

2.6 WATER USE

Information on domestic, 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 wells (in sections 8 and 9) are within about 1 mi (2 km) downgradient from the former processing site (Figure 2.5).

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

Drinking water is supplied by the city of Lakeview within city limits; residents outside the city limits have private wells. Lakeview obtains its water from two sources. In the winter and spring, water is obtained from springs located in the mountains east of the city. In the summer and fall, as these springs dry up, water is obtained from six water wells located within the city limits. These municipal water supply wells are in sections 16 and 17, downgradient from the former processing site. The closest of these water supply wells is more than 1 mi (2 km) from the processing site. The municipal wells are routinely analyzed 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 standards.

Approximately 10 residences are along Missouri Avenue, about 1100 ft (340 m) south of and downgradient from the site. Another group of up to 15 residences is located 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 residences indicates that while a few homes on Missouri Avenue are connected to city water, the majority use shallow wells as their only source of domestic water, including drinking water. In some Roberta Avenue residences, due to poor water quality, private well water is used only for domestic purposes other than drinking (washing, laundry, etc). Bottled water is used for drinking.

Table 2.2 Private and public wells in the vicinity of Lakeview, Oregon^a

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

Table 2.2 **Compilation of private and public wells in the vicinity of Lakeview, Oregon^a**
(Concluded)

Location, including section number	Number of wells	<u>Range of depths</u>		<u>Screen ranges</u>	
		ft	m	ft	m
<i>Far downgradient</i> Section 16 (continued)					
City well (1)	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 (1)	3	80-360	24-110	22-340	7-104
	1	300	91	280-300	85-91
West of processing site					
<i>Crossgradient</i> Section 5					
	1	165	50	40-95	12-29
	1	130	40	NA	NA
	3	48-223	15-68	48-210	15-64
<i>Downgradient</i> Section 8					
County well (1)	2	5-300	2-91	NA	NA
	2	43-?	13-?	NA	NA

^aThis 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.

LEGEND

- 4800— TOPOGRAPHIC CONTOUR (FT)
- - - FENCE LINE
- 534 WELL ID
• DOMESTIC WELL
- - - - - EPHEMERAL STREAM
- (395) U.S. HIGHWAY

1000 0 1000 2000 FEET

500 0 500 METERS

FIGURE 2.5
LOCATION OF SELECTED DOMESTIC WELLS
IN THE VICINITY OF THE LAKEVIEW PROCESSING SITE
LAKEVIEW, OREGON

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

Ground water quality data from 1984 through 1993 were used in this risk assessment. The data from the 1982 sampling round were not incorporated because the reported concentrations of a number of key constituents were inconsistent with results of all subsequent sampling rounds.

Most of the water quality data for monitor wells were obtained from filtered water samples (0.45-micron filter). These filtered water quality 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 in later sections. In contrast to monitor well data, domestic well data used in this assessment were not filtered prior to laboratory analysis.

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

3.1 BACKGROUND WATER QUALITY SUMMARY

Background water quality is defined as the quality of water if uranium milling activities had not taken place. In Lakeview's case, there are two distinct types of background waters upgradient from the site. One type is a geothermal ground water that 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 is nongeothermal and is found upgradient of the tailings pile area of the site. Monitor well 523, upgradient and off-site, is considered the geothermal background well. Monitor well 501, also upgradient and off-site, is representative of nongeothermal background water.

**Table 3.1 Summary of DOE monitor wells at the Lakeview processing site,
Lakeview, Oregon**

Location ID	Depth		Screen		Sampling dates	Number of rounds
	ft	m	ft	m		
Upgradient						
501	25	8	20-25	6-8	1984-93	20
502	85	26	80-85	24-26	1984-92	18
530	31	9	16-15	5-8	1982-85	3
523	25	8	20-25	6-8	1984-85	3
524	80	24	70-75	21-23	1984-85	3
Downgradient, shallow, evaporation pond area						
503	25	8	20-25	6-8	1984-93	21
505	30	9	23-28	7-8.5	1984-93	21
511	34.5	10.5	27-32	8-10	1984-85	2
513	29.5	9.0	17-22	5-7	1984-85	3
515	24.5	7.5	20-25	6-8	1984-85	2
518	29.5	9.0	24-29	7-9	1984-93	14
519	34.5	10.5	30-40	9-12	1984	1
521	24.5	7.5	16-21	5-6	1984-85	2
525	29.5	9.0	24-29	7-9	1984-92	21
529	26	8	23-37	7-11	1982, 1984, 1985	3
541	30	9	28-33	8.5-10	1990-93	6
Downgradient, shallow, tailings area						
507	25	8	18-23	5.5-7	1984-93	21
527	26.5	8	25-37	8-11	1982, 1984, 1985	3
528	28	8.5	23-29	7-9	1982, 1984, 1985	3
532	28	8.5	22-38	7-11.5	1982, 1984, 1985	4
540	30	9	27-32	8-10	1990-93	6
Downgradient, shallow, between tailings and evaporation ponds						
509	35	11	29-34	9-10	1984-85, 1993	4
531	43	13	38-51	11.5-15.5	1982, 1984, 1985	3
Downgradient, deep, evaporation ponds						
504	80	24	73-78	22-24	1984-92	20
506	85	26	77-82	23.5-25	1984-92	20
512	125	38	82-87	25-26.5	1984-85	2
514	80	24	70-75	21-23	1984-85, 1991	4
516	81	25	72-77	22-23.5	1984-85	2
520	90	27	70-85	21-26	1984	1
522	59	18	52-57	16-17	1984-85	2
526	80	24	63-68	19-21	1984-92	19
Downgradient, deep, tailings						
508	80	24	72-77	22-23.5	1984-92	19
517	80	24	70-75	21-23	1984-85	3
Downgradient, deep, between tailings and evaporation ponds						
510	75	23	70-75	21-23	1984-85	2

Shallow monitor well 530 was also located upgradient from the site within the area of nongeothermal ground water. However, data from this well were not used for statistical evaluation of background 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 waters.

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

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

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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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 540)	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 503)	17/19	<0.1	1.1	1.4
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 503)	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 521)	2/2	55	56	57

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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
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 540)	6/6	525	563	642
Chloride				
Nongeothermal background	20/20	1	2	5
Geothermal background	3/3	12	94	100
Plume (well 513)	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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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 503)	19/19	0.4	3.6	7.0
Iron				
Nongeothermal background	4/20	<0.02	<0.03	0.07
Geothermal background	1/3	<0.03	<0.03	0.03
Plume (well 513)	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 540)	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 540)	5/5	14	34	39
Mercury				
Nongeothermal background	0/8	<0.0002	<0.0002	<0.0002
Geothermal background	0/1	--	<0.0002	--
Plume	1/14	<0.0002	<0.0002	0.0002

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
Molybdenum				
Nongeothermal background	0/9	<0.01	<0.01	<0.01
Geothermal background	1/1	--	0.03	--
Plume (well 521)	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 540)	5/5	0.04	0.09	0.13
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 513)	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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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 529)	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 513)	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 529)	3/3	6100	6700	7300
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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
Uranium				
Nongeothermal background	7/18	<0.001	<0.003	0.002
Geothermal background	0/1	--	<0.003	--
Plume (well 540) ^d	6/6	0.003	0.009	0.014
Vanadium				
Nongeothermal background	3/8	<0.01	<0.01	0.03
Geothermal background	0/1	--	<0.01	--
Plume	6/16	<0.01	<0.01	0.13
Zinc				
Nongeothermal background	9/19	<0.005	<0.005	0.25
Geothermal background	1/2	<0.005	<0.008	0.01
Plume (well 540)	6/6	0.078	0.30	0.40
Radionuclides		pCi/L		
Lead-210				
Nongeothermal background	3/4	0.1	0.5	1.0
Geothermal background	0/1	--	<1.0	--
Plume	6/7	0.0	0.3	1.6
Polonium-230				
Nongeothermal background	3/4	0.0	0.1	0.8
Geothermal background	0/1	--	<0.5	--
Plume (well 503)	5/5	3.8	6.8	9.1
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

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

^aNongeothermal 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.

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

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

^dUranium 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.

mg/L - milligrams per liter.

pCi/L - picocuries per liter.

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

FIGURE 3.1
TRILINEAR DIAGRAM OF MAJOR CHEMISTRIES FOR GROUND WATER
IN THE AREA OF THE LAKEVIEW PROCESSING SITE, OREGON

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
Inorganic constituents				
Aluminum				
Nongeothermal background	0/6	<0.05	<0.05	<0.10
Geothermal background	0/2	<0.10	<0.10	<0.10
Plume	0/14	<0.05	<0.05	<0.10
Ammonium				
Nongeothermal background	6/17	<0.1	<0.1	0.2
Geothermal background	3/3	0.2	0.4	0.4
Plume (well 506)	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 506)	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 506)	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 506)	19/19	0.4	6.0	6.7

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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
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 508)	19/19	10	30	34
Chloride				
Nongeothermal background	13/18	0.5	1.1	13
Geothermal background	3/3	89	94	97
Plume (well 506)	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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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 506)	19/19	2.0	2.5	3.3
Iron				
Nongeothermal background	3/18	<0.02	<0.03	0.10
Geothermal background	1/3	<0.03	<0.03	0.04
Plume (well 506)	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 508)	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 508)	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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
Molybdenum				
Nongeothermal background	0/6	<0.01	<0.01	<0.01
Geothermal background	1/1	--	0.11	--
Plume (well 506)	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
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 506)	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 506)	6/6	55	57	66

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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 506)	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 506)	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 506)	2/16	240	280	331
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

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

Parameter ^a	Frequency of detection ^b	Observed concentration		
		Minimum	Median ^c	Maximum
		mg/L		
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-230				
Nongeothermal background	4	0.0	0.7	1.6
Geothermal background	1	--	<0.5	--
Plume	9	0.1	0.5	0.8
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

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

^aNongeothermal 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.

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

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

3.2 MAGNITUDE AND EXTENT OF SITE-RELATED CONTAMINATION

3.2.1 Tailings and raffinate chemistry

Uranium ores were obtained from the White King and Lucky Lass mines about 15 mi (24 km) northwest of the mill and hauled to the mill by truck for processing. Ores from the White King Mine contained uraninite (UO_2), coffinite $\text{U}(\text{Si}_2\text{H}_4)\text{O}_4$, and various secondary uranium minerals in a silicified volcanic breccia. This ore was associated with molybdenum, arsenic, mercury, lead, and antimony (BFEC, 1984). The Lucky Lass Mine contained primarily secondary uranium minerals with some uraninite or coffinite in tuff and vesicular basalt.

The ore was leached using sodium chlorate and sulfuric acid. The uranium-bearing leachate was then 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 (DOE, 1985a) identified several of these components, including uranium, strontium, chloride, aluminum, calcium, manganese, iron, and arsenic. Chemical profiles of solids, water-soluble extracts, and acid-soluble extracts from tailings and soils located 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 radionuclides and most heavy metals do not occur in the contaminated ground water because they were adsorbed or coprecipitated in the subsoils beneath the source areas of contamination (DOE, 1985a). These conclusions are discussed in this section.

Analyses of the major element water chemistry of the contaminated shallow ground waters beneath the site indicate that there are 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, it is apparent that these two types of contaminated waters have relative proportions of ions similar to those in their respective background waters (Figure 3.1), with the exception that bicarbonate accounts for a larger proportion of the anions in the more dilute background waters. Bicarbonate is held at near-constant levels in all ground waters by equilibrium with calcite. However, while the relative proportions of major ions in the contaminated ground waters are similar to their respective background water types, the absolute concentrations of ions in the contaminated waters are 10 to 20 times greater than those in the background waters.

A hypothesis that explains the similarity of ground water contaminated from tailings-seepage waters to nongeothermal waters, and the similarity of evaporation pond seepage waters to geothermal waters, is that prior to construction of the tailings piles and evaporation ponds, the soils on the site contained evaporite salts derived from the background waters. These salts were then leached from the soils by the 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 would explain why both types of contaminated ground waters contain sulfate, derived from the processing, while only the evaporation pond waters contain elevated concentrations of chloride; the major source of the chloride may have been natural salts in soils formed by the evaporation of the geothermal waters. 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 waters) and relatively low concentrations of magnesium and calcium (constituents that are depleted in the geothermal waters). The hypothesis also explains why the contaminated ground water from the tailings pile area contains relatively high concentrations of calcium and magnesium, which are also relatively high in the nongeothermal waters upgradient of the tailings pile area.

The amount of salt in a saline soil required to produce the observed concentrations of chloride, boron, and fluoride was determined using the following values and assumptions: contamination extends to a depth of 40 ft (12 m) in an aquifer with 20 percent porosity; chloride, boron, and fluoride concentrations are approximately 1200, 42, and 3.6 mg/L, respectively, in the contaminated water; and natural chloride, borate, and fluoride salts were concentrated in the upper 39 inches (1 m) of the soil prior to leaching. The computation indicates that the soil, prior to leaching, would have contained 2 percent sodium chloride, 2 percent borate salt (as borax), and 0.004 percent

calcium fluoride, by weight. Assuming this is present in the saline soil at the site is reasonable. The computation demonstrates that a small amount of salt can have a significant effect on water chemistry.

This model gains support from observations of natural salt deposits at the surface to the south and west of the site, though the composition of the salts remains to be analyzed. Additional support comes from U.S. Department of Agriculture Soil Conservation Service soils maps for the area in the vicinity of the site. Soils in the area of the evaporation ponds and upgradient geothermal springs are described as being "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 waters, 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 domestic wells, as discussed in Section 3.3.

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

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

Table 3.3 provides a summary of filtered ground water quality in the deeper portion of the aquifer. The vertical extent of contamination appears to be limited to depths of less than 80 ft (20 m). One well (522) screened between 52 and 57 ft (16 and 17 m) has elevated sulfate concentrations (880 mg/L), indicating contamination. Deeper wells, screened from about 63 to 87 ft (19 to 27 m), have chemistries similar to the geothermal background wells downgradient of the evaporation ponds, or chemistries similar 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 contamination has reached a depth of 80 ft (24 m) in monitor wells 504, 506, and 520. The downward movement of contaminants is consistent with a downward component to the vertical hydraulic gradient on the site. However, the low concentration of possible contaminants at 80 ft (24 m) suggests, if it is occurring, a very slow downward movement of contaminants.

3.2.3 Statistical comparison of background and plume water quality

After determining the extent of contamination using sulfate as an indicator, water quality data collected between 1984 and 1993 were used to statistically identify the full suite of contaminants elevated above background levels in contaminated shallow portions of the aquifer. 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, 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 located within the shallow portion of the aquifer, the portion most affected by contamination. It was necessary to use filtered water samples because very few unfiltered data were available. Wells 503 and 540 were selected for analysis because they consistently have the highest concentrations of key contaminants among those wells that currently exist on or downgradient from the site.

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

After determining which constituents in the plume were above nongeothermal background levels, each identified constituent was compared to geothermal background levels to see whether 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 being above nongeothermal background levels were found to be within the range of geothermal background water quality: antimony, fluoride, and silica. The remaining constituents identified as above nongeothermal background levels were all, on average, present at levels above the range of geothermal background levels and therefore were identified as contaminants.

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

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

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

Table 3.2 summarizes the water quality measurements from filtered water samples collected between 1984 and 1993 from the contaminated shallow aquifer and compares ground water affected by the former milling-related activities to geothermal and nongeothermal background ground water quality. Contaminated ground water concentrations presented are, in most cases, those observed from well 503 (evaporation pond area) and/or well 540 (tailings area). However, well 503 water quality was, in several cases, supplanted by higher 1982-1985 levels recorded for one or more of the abandoned wells in the evaporation ponds area prior to surface remediation activities. Because there is

no evidence that ground water contamination has decreased significantly during the last 10 years, it is possible that these early high levels may still be present under the site but not accessible to wells currently in existence. Contaminants for which early data are reported in Table 3.2 include boron, chloride, iron, molybdenum, potassium, sodium, strontium, and sulfate.

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

3.3 DOMESTIC WELL WATER QUALITY

Water quality data were collected in 1982, 1984, and 1985 from 5 domestic wells located north and southeast of the site and in 1993 from 14 domestic wells located north, southeast, south, southwest, and west of the site (Table 3.4), including wells on Missouri and Roberta Avenues. Locations of the wells are shown in Figure 2.5. Unfiltered samples were collected and analyzed for major elements and trace elements indicative of the uranium milling process. Analytical data from domestic 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 to the south and southwest of the site were collected to determine whether 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). Results of the latest water quality sampling tests for each well are presented in Table 3.5.

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

Table 3.4 Summary of domestic wells sampled by the DOE, 1982 through 1993, in the vicinity of Lakeview, Oregon

Location ID	Depth		Sampling dates	Number of sampling rounds
	ft	m		
533	Unknown		1984, 1985, 1993	3
534	92	28	1984, 1993	2
535	150	46	1984, 1985, 1993	3
536	Unknown		1993	1
537	50	15	1984, 1985, 1993	3
538	100	30	1982, 1984, 1993	3
542	90	27	1993	1
543	280	85	1993	1
544	Unknown		1993	1
545	71	22	1993	1
546	50	15	1993	1
547	68	21	1993	1
548	85	26	1993	1
549	80	24	1993	1

Table 3.5 Summary of unfiltered water quality for domestic wells in the vicinity of Lakeview, Oregon

Location ID	Date	Alk	As	Ca	Cl	Fe	Mg	Mn	Mo	Ni	pH	K	SiO ₂	Na	SO ₄	Temp	Se	TDS	U	V	Zn
North of site (geothermal)																					
535	8/93	138	0.009	76	40	<0.03	12	<0.01	<0.01	<0.04	7.7	3.2	44	27	74	20	<0.005	380	0.001	<0.01	0.057
538	8/93	NA	0.013	5	96	<0.03	0.2	0.01	0.04	<0.04	NA	6.7	111	196	222	NA	<0.005	1010	<0.001	<0.01	0.066
544	8/93	NA	0.157	13	99	<0.03	<0.1	<0.01	0.04	<0.04	NA	8.6	145	203	258	95	<0.005	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	39	9	16	<0.005	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	17	3	13	<0.005	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	30	8	13	<0.005	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	18	2	15	<0.005	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	33	18	11	<0.005	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	22	13	14	<0.005	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	18	11	19	<0.005	230	<0.001	<0.01	0.277
South of site																					
546	10/93	NA	0.019	45	15	0.86	12.3	1.50	<0.01	<0.04	NA	0.4	46	38	38	NA	<0.005	308	0.002	<0.01	0.027
547	10/93	NA	0.010	294	388	2.28	116	14.60	<0.01	<0.04	NA	5.1	52	96	137	NA	<0.005	2010	<0.001	<0.01	0.085
548	10/93	NA	0.008	810	2160	4.18	254	28.70	<0.01	<0.04	NA	12.0	51	502	713	NA	<0.01	6000	<0.001	<0.01	0.349
549	10/93	NA	0.022	5	67	0.72	2	0.34	0.02	<0.04	NA	2.6	53	125	116	NA	<0.005	452	0.001	<0.01	0.102
Alk	Alkalinity						SiO ₂	Silica						Na	Sodium						
As	Arsenic						Na	Sodium						SO ₄	Sulfate						
Ca	Calcium						SO ₄	Sulfate						Temp	Temperature (°C)						
Cl	Chloride						Temp	Temperature (°C)						Se	Selenium						
Fe	Iron						Se	Selenium						TDS	Total dissolved solids						
Mg	Magnesium						TDS	Total dissolved solids						U	Uranium						
Mn	Manganese						U	Uranium						V	Vanadium						
Mo	Molybdenum						V	Vanadium						NA	Not analyzed						
Ni	Nickel						Zn	Zinc													
K	Potassium						NA	Not analyzed													

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

Domestic wells to the 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. As discussed in Sections 3.1 and 3.2.2, both site-related contamination and natural geothermal water are associated with much higher sulfate and TDS. Thus, the low-sulfate domestic wells are believed to contain uncontaminated, nongeothermal waters. Several of these nongeothermal waters 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 domestic wells is similar to that in DOE monitor wells used to establish nongeothermal background water quality.

Four of five domestic wells due south of the site (downgradient, including locations 546, 547, 548, and 549) have elevated concentrations of sulfate, which could indicate that contamination from the site has reached these wells. However, the areal pattern of sulfate distribution in these wells is inconsistent with site-related contamination. The wells with the highest sulfate (locations 545 and 550, having 945 and 377 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). Sulfate contamination from the site in excess of 300 mg/L appears to be limited to an area within 2400 ft (730 m) of the site, whereas the domestic wells with high sulfate are located about 5000 ft (1500 m) south of the site, placing them outside the apparent area of site-related contamination. Therefore, the domestic wells with high sulfate are probably not affected by site-related contaminants.

The domestic wells with high sulfate 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 are probably 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 her water quality began to deteriorate after the fill was added to the area. It is possible 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.

3.4 CONTAMINANTS OF POTENTIAL CONCERN

The data presented in Sections 3.1 and 3.2 were used to compile a list of contaminants that are of potential concern for the assessment of human health risks at the former Lakeview processing site. A constituent was placed on the

list of contaminants of potential concern (Table 3.6) if reported concentrations in the shallow on-site or downgradient monitor wells were, on average, higher than background levels. The contaminants in column 1 of Table 3.6 (contaminants that exceed background levels) were screened for their impact on human health using the criteria discussed below to develop a final list of contaminants of potential concern for human health (column 4 of Table 3.6).

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

A second screening of the remaining contaminants was made on the basis of exhibiting very low toxicity and relatively high dietary intake by comparison to the values measured. For human health, these screening criteria eliminated ammonium, aluminum, calcium, magnesium, silica, and strontium. The contaminants of potential concern remaining following screening are arsenic, boron, chloride, iron, manganese, molybdenum, nickel, polonium-210, sodium, sulfate, and uranium (Table 3.6). These contaminants form the basis of the human health portion of the risk assessment for the Lakeview site.

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

3.5 CONTAMINANT FATE AND TRANSPORT

Contaminant mobility, uptake, and toxicity depend on the species of ions that exist in the aqueous environment. The types of ion species and complexes depend on the availability of various anions and cations for the formation of complex ions and on pH and Eh conditions. Species of the contaminants of potential concern have been computed using the geochemical speciation code PHREEQE (Parkhurst et al., 1980). The predominant species and their molar percentages are summarized in Table 3.7.

Arsenic

Arsenic concentrations in the contaminated ground waters 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 waters are in the shallow portion of the aquifer.

Arsenic occurs as arsenate species HAsO_4^{2-} in the ground waters. Reaction of ferric hydroxides with arsenate may be an effective control on the concentration

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

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

^aAlthough silica causes toxicity when inhaled, dissolved silica has not been demonstrated to produce adverse health effects in humans or animals following ingestion.

Table 3.7 Predominant ionic species of the contaminants of potential concern

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)

of arsenic (Hem, 1970), especially because the contaminated ground waters are 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 compared to both geothermal surface water and deep geothermal ground water.

Boron

Boron concentrations in the contaminated ground waters are up to about 9 times those of background. Boron was not used in the processing of the ores, and boron is not usually associated with uranium ores but is commonly associated with geothermal waters, including those upgradient of the site. Its presence, along with chloride, 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 will probably play the greatest role in decreasing borate concentrations.

Chloride

Chloride concentrations in the contaminated ground waters range up to about 12 times concentrations reported in background samples. Much of the chloride in the contaminated ground waters 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 waters are several hundred times those reported in background samples of ground water. Iron is soluble under acid or reducing conditions but forms insoluble hydrated oxides under oxidizing conditions at a pH greater than 6. All waters at the site are 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 both precipitation and dilution.

Manganese

Manganese concentrations in the contaminated waters are up to 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 waters. 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 waters throughout the region. However, in the case of the contaminated ground waters, none 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 waters are four times those of the geothermal background ground water. Molybdenum occurs in both the geothermal waters and the contaminated ground waters as molybdate (MoO_4^{2-}), a negatively charged ligand. As with most negatively charged ligands, sorption of molybdenum is most effective under acidic conditions (pH in the 3 to 4 range). Thus, sorption of molybdenum in the contaminated ground waters will be less important than dilution as a mechanism for decreasing concentrations.

Nickel

Nickel concentrations are up to 3 times the detection limit for the analyses (0.01 and 0.04 mg/L) in ground water associated with the tailings pile. Nickel was also 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, having a half-life of 138 days. Thus, its presence in the contaminated ground waters 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 waters 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 analyzed for in ground waters, or one of the other parent isotopes occurring in solid form within the aquifer. Radon-222 is commonly associated with geothermal waters (Davis and DeWiest, 1966) in amounts more than adequate to explain the observed

activity of polonium-210, and this could be the source of the polonium-210 at the Lakeview site. However, the polonium-210 was not observed in the geothermal background waters, so the geothermal radon-222 is probably 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 most likely parent to have migrated into the aquifer in aqueous solution before being adsorbed onto soils.

Since 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 close to its source. The size of this area is dictated by the rate of ground water flow and the rate of polonium-210 decay. Assuming that the rate of shallow horizontal ground water movement is approximately 50 ft/year (8×10^{-5} cm/s) (see Section 2.3), and that the polonium-210 will almost entirely decay within 10 half-lives (3.8 years), then 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 will probably 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 waters. Sodium forms very soluble bicarbonate, chloride, and sulfate salts; much of the sodium in the contaminated ground waters 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 will probably play the greatest role in decreasing sodium concentration.

Sulfate

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

Uranium

Uranium in the contaminated waters 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 waters as a uranyl phosphate complex ($\text{UO}_2(\text{HPO}_4)2^{2-}$). The

low levels of uranium in the contaminated ground water attest to the ability of the soils at the site to strongly adsorb uranium species, and further attenuation by adsorption will probably occur.

3.6 SURFACE WATER MONITORING

Surface water quality has been monitored in the water bodies in the vicinity of the site. Surface water bodies at and adjacent to the processing site include ponds, drainage ditches, and three creeks, as discussed in Section 2.4 and shown in Figure 2.2. Hunters Hot Springs are an important aspect of the surface water hydrology and geochemistry; they feed geothermal water into a creek (Hunters Hot Springs Creek) and associated ponds north of the site.

Surface water has been sampled at six locations in the past (Figure 3.3). Unfiltered surface water samples were collected in October 1993 from two locations along Hunters Hot Springs Creek (608 and 609), one location just downstream of the confluence of Warner Creek and Thomas Creek (605), and one location in Thomas Creek just upstream of the confluence with Hammersley Creek (606). Historic sampling location 607, in Thomas Creek downstream of its confluence with Hammersley Creek, was dry during the October 1993 sampling event. Previously, from one to three sampling rounds were conducted at these five locations 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 in preparation for the development of this risk assessment. Sediment samples were collected in October 1993 from the six locations discussed above; the results are discussed in Section 7.0.

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 identified as exceeding background ground water quality (Table 3.6) only iron and molybdenum were detected at concentrations above background levels at the adjacent locations. However, the concentrations are only slightly higher, and because of the limited data set (i.e., only one recent round of sampling), a statistical evaluation cannot be conducted. An in-depth discussion of surface water contaminants is provided in Section 7.0.

Precipitation and snowmelt may have carried both dissolved and suspended constituents along the surface drainages. Metal constituents transported as dissolved species would have become diluted after discharging to these water bodies. Alternatively, dissolved species could have precipitated, becoming

adsorbed to sediments or absorbed into biota with varying biochemical and geochemical conditions. Constituents transported from the processing site that were sorbed onto soil particles would have been deposited as sediments. Variations in geochemical conditions or biological action could release constituents adsorbed onto sediments into surface waters. Thus, deposited sediments could act as a source of site-related surface water contamination. However, as discussed further in Section 7.0, there is no compelling evidence suggesting that this is occurring.

4.0 EXPOSURE ASSESSMENT

4.1 POTENTIALLY EXPOSED POPULATION

Exposure can occur only if there are both a source of contamination and a mechanism of transport to a receptor population or individual. In this baseline risk assessment, only ground water and ground water following surface expression are addressed. A telephone survey indicated that a number of residents downgradient of the site have domestic wells that could potentially be affected by the plume of contaminated ground water. These residents may use the shallow wells as a potable water source, but some residences use bottled water for drinking and use well water for other domestic purposes. A domestic well constructed in the alluvial/lacustrine aquifer can create an exposure pathway through drinking and bathing. This scenario was evaluated because people in the area use shallow ground water as a drinking water source. The elevated inorganics in the domestic well water may be naturally occurring or a result of the filling of the low-lying land surface. The water from these shallow wells is used for drinking, watering livestock, and irrigating garden plants. Using the livestock for food would create a potential exposure pathway to humans. Plants with roots in the alluvial aquifer or irrigated with water from this zone could take up and concentrate contaminants, forming a potential pathway to humans through plant consumption.

People downgradient of the processing site may currently be exposed to contaminated ground water from downgradient domestic wells. This potential current exposure is also assumed to be a future ground water use scenario at the processing site for purposes of this risk assessment. This scenario evaluates domestic ground water use consistent with current use by the rural population in the region. The potentially exposed population includes individuals of the following age groups: infants (birth to 1 year old); children (1 to 10 years old); and adults (11 to 64 years old). These age groups were selected for the following reasons:

- 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.

4.2 EXPOSURE PATHWAYS

Ground water in the region is primarily used for household purposes such as drinking (although at least some homes use bottled water), cooking, and bathing. Other uses typical of the region that could indirectly lead to human exposure include irrigation and livestock watering. Figure 4.1 provides a

FIGURE 4.1
CONCEPTUAL MODEL
LAKEVIEW, OREGON

conceptual model for potential ground water exposure pathways that could result from these uses.

4.2.1 Drinking water ingestion

Drinking water ingestion is generally the most significant exposure pathway for ground water contaminated with metals and other nonvolatile compounds. For this evaluation, drinking water consumption includes water consumed for drinking, as well as water used directly for food preparation (e.g., reconstituted juices, soup, rice, and beans). For comparison of relative pathway significance, a screening level assessment of drinking water intake is shown in Table 4.1. These calculations are based on estimates of the 95 percent upper confidence limit (UCL) on the mean concentrations of contaminants within the plume (i.e., the 95 percent UCL on the mean contaminant concentration from the most contaminated plume wells).

4.2.2 Dermal absorption

Dermal absorption is the process by which chemicals coming into contact with the skin are absorbed into the blood vessels near the 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 whether a dermal absorption pathway would be significant compared to the drinking water pathway for the contaminants of potential concern. Since chemical-specific absorption factors are not available for the contaminants, it was assumed that they are absorbed across the skin at the same rate as water. This assumption will probably overestimate any potential contribution from dermal absorption. Additionally, the concentration in water was assumed to be the 95 percent UCL on the mean concentration from the most contaminated baseline wells (affected by former uranium milling activities), which also will 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.

4.2.3 Ingestion of ground-water-irrigated produce

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

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

Contaminant of potential concern	Cw	Ground water exposure doses (mg/kg-day)		Ratio of dermal: ingestion ^a
		Ingestion	Dermal contact	
Carcinogenic effects (mg/L)				
Arsenic	0.11	1E-03	3E-06	0.002
Polonium-210	8.5	2E+-5	3E+02	0.002
Uranium	14	4E+02	6E+02	0.002
Noncarcinogenic effects (pCi/L)				
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--5	0.002
Nickel	0.13	4E-03	7E-06	0.002
Polonium-210	8.5	2E-01 ^b	5E-04 ^b	0.002
Sodium	4000	1E+02	2E-01	0.002
Sulfate	6900	2E+02	4E-01	0.002
Uranium ^c	0.02	3E-05 ^b	1E-06 ^b	0.002

Equation definitions for exposure dose calculations

Ingestion of ground water

Chemicals

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

Radionuclides

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

Dermal contact with ground water

Chemicals

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

Radionuclides

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

Table 4.1 Exposure dose calculations and equation definitions for ground water ingestion and dermal contact, Lakeview UMTRA Project site, Lakeview, Oregon (Concluded)

Where:

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

^aThis value is calculated by dividing the dermal contact exposure dose by the ground water ingestion exposure dose.

^bPicocuries per lifetime.

^cUranium-234 and uranium-238 combined.

Table 4.2 Exposure dose calculations and equation definitions for ground-water-irrigated garden produce ingestion, Lakeview UMTRA Project site, Lakeview, Oregon

Contaminant of potential concern	Cw	Kd (L/kg)	Bv	Br	Garden produce ingestion exposure doses ^a (mg/kg-day)	Ratio of produce ingestion: water ingestion ^b
Carcinogenic effects (mg/L)						
Arsenic	0.11	19.4	0.04	0.006	2E-06	0.002
Polonium-210	8.5	14.9	0.0025	0.0004	1E+01	0.00006
Uranium	14	50	0.0085	0.004	3E+02	0.0009
Noncarcinogenic effects (pCi/L)						
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
Polonium-210	8.5	14.9	0.0025	0.0004	2E-05 ^c	0.00008
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 ^c	0.001

Equation definitions for exposure dose calculations

Ingestion of garden produce irrigated with ground water

Chemicals

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

Radionuclides

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

Where:

- Cw = Contaminant concentration in ground water (maximum concentration detected).
- Kd = Soil-water partition coefficient (L/kg); from Pacific Northwest Laboratory (PNL), 1989, except for uranium. The Kd for uranium is a site-specific value.
- Bv = Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
- Br = Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless).
- DF = Dry weight fraction of plant (0.066 unitless).
- IRp = Ingestion rate for garden produce (0.05 kg per day for vegetative parts; 0.03 kg per day for reproductive parts).

Table 4.2 Exposure dose calculations and equation definitions for ground-water-irrigated garden produce ingestion, Lakeview UMTRA Project site, Lakeview, Oregon (Concluded)

FI = Fraction of garden produce ingested from contaminated source (1.0 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).

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

^bThis value is calculated by dividing the garden produce ingestion exposure dose by the ground water ingestion exposure dose.

^cPicocuries per lifetime.

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

^eExposure doses due to vegetative parts and reproductive parts of garden produce are calculated separately, then summed for total intake.

screening show that for the contaminants of potential concern at this site, ingesting garden vegetables and fruit irrigated with contaminated ground water would lead to potential exposures representing 7 percent or less of the exposure associated with drinking water ingestion. Thus, this exposure pathway was eliminated from further consideration.

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

The relative contribution from ingestion of milk from ground-water-fed livestock is 4 percent or less for all constituents with the exception of molybdenum, which contributes 20 percent (Table 4.3). Ingestion of meat from these animals would also contribute 7 percent or less of the exposure anticipated from drinking water for all constituents with the exception of molybdenum, which contributes 20 percent (Table 4.4). This pathway may be important, but there are insufficient data to produce meaningful quantitative estimates at this time. 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.

4.2.5 Ingestion of fish

This pathway was not evaluated because the surface water bodies in the vicinity of the Lakeview site have 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.

4.3 EXPOSURE CONCENTRATIONS

The exposure concentration of a contaminant in ground water is defined as the concentration an individual is assumed to take in over a specific period. In this evaluation, the contaminant concentrations are assumed to be in a steady state, although actual contaminant concentrations (and therefore exposures) are expected to decrease with time after the tailings are removed and the pH of the water returns to a more neutral level. 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 to be exposure for any period longer than 7 years.

For noncarcinogens, exposure concentrations are expressed as probability distributions. Distributions were derived from actual recorded water quality measurements of the well or wells currently in existence that consistently recorded the highest concentrations of each contaminant. For example, the manganese distribution reflects concentrations in well 540, whereas the sulfate distribution reflects well 503 levels. The theoretical distribution was selected to have, to the extent possible, the same average concentration, standard

Table 4.3 Exposure dose calculations and equation definitions for ingestion of milk from ground-water-fed livestock, Lakeview UMTRA Project site, Lakeview, Oregon

Contaminant of potential concern	Cw	Kd ^a (L/kg)	Bv	Fm	Milk ingestion exposure doses (mg/kg-day)	Ratio of milk ingestion: water ingestion ^b
Carcinogenic effects (mg/L)						
Arsenic	0.11	19	0.04	0.00006	9E-07	0.0009
Polonium-210	8.5	15	0.0025	0.00035	6E+02	0.003
Uranium	14	50	0.0085	0.0006	2E+03	0.007
Noncarcinogenic effects (pCi/L)						
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
Polonium-210	8.5	15	0.0025	0.00035	8E-04 ^c	0.004
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 ^c	0.008

Equation definitions for exposure dose calculations

Ingestion of milk from ground water fed livestock

Chemicals

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

Radionuclides

$$\text{Lifetime intake (pCi/lifetime)} = C_m \times IR_m \times FI \times EF \times ED$$

Where:

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

$$C_m = F_m \times ([Q_p \times C_p] + [Q_s \times C_s] + [Q_w \times C_w])$$

where

F_m = Feed-to-milk transfer coefficient (days/kg).

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 = Contaminant concentration in pasture (mg/kg). $C_p = K_d \times C_w \times B_v$.

C_s = Contaminant concentration in soil (mg/kg). $C_s = K_d \times C_w$.

Table 4.3 Exposure dose calculations and equation definitions for ingestion of milk from ground-water-fed livestock, Lakeview UMTRA Project site, Lakeview, Oregon (Concluded)

Cw	=	Contaminant concentration in ground water (maximum concentration detected).
Kd	=	Soil-water partition coefficient (L/kg); from PNL, 1989, except for uranium. The Kd for uranium is a site-specific value.
Bv	=	Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
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.

^bThis value is calculated by dividing the milk ingestion exposure dose by the ground water ingestion exposure dose.

^cPicocuries per lifetime.

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

Table 4.4 Exposure dose calculations and equation definitions for ingestion of meat from ground-water-fed livestock, Lakeview UMTRA Project site, Lakeview, Oregon

Contaminant of potential concern	Cw	Kd (L/kg)	Bv	Ff	Meat ingestion exposure doses	Ratio of meat ingestion: water ingestion ^a
Carcinogenic effects (mg/L)						
Arsenic	0.11	19.4	0.04	0.002	8E-06	0.008
Polonium-210	8.5	14.9	0.0025	0.00025	1E+02	0.005
Uranium	0.14	50	0.0085	0.0002	2E+00	0.0005
Noncarcinogenic effects (pCi/L)						
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
Polonium-210	8.5	14.9	0.0025	0.0003	2E-04 ^b	0.0008
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 ^b	0.0006

Equation definitions for exposure dose calculations

Ingestion of meat from ground water fed livestock

Chemicals

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

Radionuclides

$$\text{Lifetime intake (pCi/lifetime)} = C_b \times IR_b \times FI \times EF \times ED$$

Where:

- Cb = Contaminant concentration in beef (mg/kg), estimated using the following equation:

$$C_b = F_f \times ([Q_p \times C_p] + [Q_s \times C_s] + [Q_w \times C_w])$$

where
- Ff = Feed-to-flesh transfer coefficient (days/kg).
- Qp = The quantity of pasture eaten by cattle per day (19 kg per day).
- Qs = The quantity of soil eaten by cattle per day (0.38 kg per day).
- Qw = The quantity of water consumed by cattle per day (56 L per day).
- Cp = Contaminant concentration in pasture (mg/kg). $C_p = K_d \times C_w \times B_v$.
- Cs = Contaminant concentration in soil (mg/kg). $C_s = K_d \times C_w$.
- Cw = Contaminant concentration in ground water (maximum concentration detected).

Table 4.4 Exposure dose calculations and equation definitions for ingestion of meat from ground-water-fed livestock, Lakeview UMTRA Project site, Lakeview, Oregon (Concluded)

Kd	=	Soil-water partition coefficient (L/kg); from PNL, 1989, except for uranium. The Kd for uranium is a site-specific value.
Bv	=	Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
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).
IRb	=	Ingestion rate of meat (0.075 kg per day).
FI	=	Fraction of meat ingested from contaminated source (1.0 unitless).

^aThis value is calculated by dividing the meat ingestion exposure dose by the ground water ingestion exposure dose.

^bPicocuries per lifetime.

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

deviation (spread), and pattern of occurrences (shape) as occurred in actual water quality data. The normal distribution provided an adequate fit of the data for all constituents.

Recent analytical data were used except in cases where 1982-1985 contaminated concentrations from well(s) in the vicinity of the former evaporation ponds clearly and grossly exceeded levels measured in more recent times. 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 there was no indication that contaminant levels are decreasing over time, and worst-case exposure may exist 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. In order 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 in the case of arsenic. Several rounds of unfiltered arsenic measurements indicated that the unfiltered levels were about 35 percent higher than levels measured in filtered samples. This information was incorporated into the arsenic distribution. For the other contaminants of potential concern, unfiltered results were too limited in quantity to incorporate into the risk assessment.

Uranium concentrations from well 940 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 concentration of uranium as 0.016 mg/L, and the theoretical distribution is centered on this estimated value.

The distributions are all truncated: values below 0 mg/L or above the 99th percentile are not used. The lower limit of 0 mg/L reflects the impossibility of negative exposures. The truncation of the upper end of the distribution places a reasonable limit on possible exposures. For every contaminant, this highest allowable concentration was higher than the maximum observed concentration in the historical water quality data. The original distributions derived from

existing wells were used to calculate the 95 percent UCL for the true mean concentration of that constituent. The software package @RISK (Palisade Corporation, 1992) was used to generate the probability curves for the noncarcinogenic contaminants of potential concern. The results are shown in Figures 4.2 through 4.11.

The concentrations of radionuclides were represented by the same wells used to represent the inorganic constituents. Potential carcinogenicity of ground water contaminants is discussed in Section 5.0.

4.4 ESTIMATION OF INTAKE

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

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

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

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. Therefore, only calculations for these three carcinogens will be presented here. Intake of a radioactive carcinogen is therefore quantified as total exposure to radioactivity throughout the residency period of an individual:

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

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

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

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

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) 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 50 years for adults and 10 years for children was assumed reasonable for the population in the Lakeview assessment, because it is consistent with exposure duration parameters for rural residents. Because the concentrations of arsenic and the radionuclides in ground water at the Lakeview site are remaining constant over time, evaluations of risk based on this exposure duration combined with historic and current median and UCL concentrations of arsenic and the radionuclides should yield reasonable to conservative estimates of carcinogenic risks for this site.

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

FIGURE 4.12
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES

FIGURE 4.13
PROBABILITY DISTRIBUTIONS FOR BODY WEIGHT

sensitivity to a particular constituent. Therefore, simulated intake distributions for 1- to 10-year old children for the contaminants at this site (with the exception of sulfate, where intake by infants was assessed) are presented in Figures 4.14 through 4.23.

4.5 EXPOSURE ASSESSMENT UNCERTAINTIES

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

- Uncertainties resulting from a lack of thorough environmental sampling data (ground water, surface water, sediment, and biota), which could lead to an underestimate or overestimate in the exposure analysis.
- Uncertainties arising from the assumption that the ground water contaminant source term at the site has reached a steady state and that contaminant concentrations at the exposure point will remain constant for chronic periods of exposure (generally greater than 7 years). Because the source of contamination at Lakeview has been removed, the assumption of a constant source will probably lead to an overestimation of risk.
- Uncertainties associated with the model used to estimate uptake of contaminants into plants for the irrigated garden produce pathway. Site-specific plant uptake factors could vary substantially from the default literature estimates. As with environmental sampling, the net effect on risk estimates of this uncertainty cannot be predicted.

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

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

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

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

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

5.0 TOXICITY ASSESSMENT

Several contaminants that have the potential for causing adverse human health effects have been detected in ground water at the site. This section summarizes the toxicological effects of the chemical contaminants and carcinogenic potentials of the radionuclides. The following source materials were used in developing these toxicological profiles: when available, EPA's Integrated Risk Information System (IRIS) (EPA, 1994a); EPA's Health Effects Assessment Summary Tables (HEAST) (EPA, 1994b); the Agency for Toxic Substances Disease Registry Toxicological Profiles published by the Department of Health and Human Services (DHHS); the *Handbook on the Toxicology of Metals* (Friberg et al., 1986); and peer-reviewed scientific literature when these review documents were not available for certain constituents. By basing toxicity information on the standardized review documents cited above, the evaluation of risks 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 data base, 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 data base.

The toxicity profiles presented here focus on drinking-water-based toxicity source material for humans whenever available. Animal information is included only if human data are not available. Animal information on the toxicity range graphs is represented by the use of widely spaced dotted lines. When uncertainty exists about the beginning or ending points of a range of exposures that produces specific toxic effects, closely spaced dots will be used at the appropriate end of the line denoting range.

5.1 CONTAMINANT TOXICITY SUMMARIES

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

5.1.1 Arsenic

Absorption

Arsenic is effectively absorbed through the gastrointestinal tract and via 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 via the blood to most tissues. In humans as well as in most animal species, exposure to either arsenite or arsenate leads to an initial accumulation in the liver, kidneys, and lungs. The clearance from these tissues is very rapid, and a long-term retention of arsenic is seen in organs rich in sulfhydryl-containing proteins, such as the 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 retention of arsenic occurs after exposure to trivalent arsenic than to 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 has also been demonstrated to enter both cow and human milk (Marcus and Rispin, 1988).

In the human body, where methylcobalamine acts as a major methyl group donor in the biotransformation process, inorganic arsenic is converted to methylated compounds. It has been demonstrated that the major site of arsenic methylation is the liver (Marcus and Rispin, 1988). 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 one order of magnitude per methyl group) and increases the rate of arsenic excretion from the body.

The major route of excretion following human exposure to inorganic arsenic is via the kidneys (Ishinishi et al., 1986). Only a few percent is excreted in feces. The rate of excretion 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). In the case of 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 a rate

of excretion 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 arsenic's widespread occurrence, the general human population is exposed to it primarily from drinking water and foodstuffs. Certain target groups are exposed to arsenic from industrial and agricultural uses. Medicinal use has also 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. The concentration of arsenic in fish and seafood, particularly shellfish, is generally one or two 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 the valency form of the element. 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 exists 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 intoxications result from exposure to small doses of arsenic over a long period of time. These intoxications are frequently caused by arsenic content in drinking water and in food. Changes of the skin leading to skin cancer are commonly 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 were also 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 the methylation of arsenic. This can result in an increased deposition of the element in the target organs (e.g., lung or skin).

The EPA has classified 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 prevalence in populations exposed through consumption of drinking water containing high concentrations of arsenic. The EPA is currently 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. The health effects from exposure to arsenic as a function of dose are summarized in Figure 5.1.

FIGURE 5.1
ARSENIC TOXICITY RANGES

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).

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 can also be absorbed from cosmetics or medical preparations through mucous membranes or damaged skin.

Toxicity

No human epidemiological studies on chronic ingestion of boron are available. 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, were also reported in some of the infants following ingestion of 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 ingestion of 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 two 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.

The toxicity of boron is summarized in Figure 5.2.

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 the newborn and 1920 mg/kg in the adult (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

▲ ORAL RfD (0.09 mg/kg-day)

|| AVERAGE DIETARY INTAKE (FOOD AND WATER, 0.14-0.35 mg/kg-day)

.....
NO ADVERSE EFFECTS, MICE, DOGS, AND RATS

.....
TESTICULAR ATROPHY AND SPERMATOGENIC ARREST (ANIMAL STUDIES)

.....
ACUTE GASTROINTESTINAL EFFECTS IN INFANTS,
NAUSEA, VOMITING, DIARRHEA (184 mg/kg-day) →

FIGURE 5.2
BORON TOXICITY RANGES

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 between 3018 and 8875 mg. Excretion of chloride 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. 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 intakes of chloride vary largely with intake of salt. Estimates range from 2,400 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 waters. Ground water usually contains larger quantities of chloride than surface water. Some public supply wells may contain 100 mg chloride/L (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/kg-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/kg-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. 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.

The loss of chloride from the body generally parallels that of sodium. 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 daily allowance (RDA) has been determined. The estimated chloride minimum requirement of healthy persons 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 levels of chloride (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.

The toxicity of chloride is summarized in Figure 5.3 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 (Goyer, 1991; Whitney et al., 1990). Normally, 10 to 15 percent of dietary iron is absorbed, but this percentage varies to compensate for the level of iron in the body (Elinder, 1986). For example, patients with iron-deficiency anemia can absorb as much as 60 percent of an oral dose of iron (Josephs, 1958).

FIGURE 5.3
CHLORIDE TOXICITY RANGES

Iron absorption also is influenced by factors such as 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 is decreased in the presence of high levels of phosphate, cobalt, copper, and zinc (Elinder, 1986). Excess manganese can significantly decrease 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 (Goyer, 1991).

Normally, the adult human body contains about 3 to 5 grams of iron. Two-thirds of this amount is found in the blood, bound to hemoglobin. Less than 10 percent of the body iron is found in myoglobin and iron-requiring enzymes. About 20 to 30 percent of the remaining iron in the body pool is bound to iron-storage proteins in liver, bone marrow, and spleen (Elinder, 1986).

Under normal conditions, the total elimination of iron from the body is limited to 0.6 to 1.0 mg 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 rates of elimination, 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 on the order of 30 to 150 mg/kg fresh weight. Grains and fruits are low in iron, usually ranging from 1 to 20 mg/kg. In both human and cow's milk, iron concentration is about 0.5 mg/L (Elinder, 1986).

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 in children may occur following ingestion of more than 0.5 gram (approximately 22 mg/kg) of iron, about 2.5 grams (approximately 110 mg/kg) as 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 (Goyer, 1991). Shock and metabolic acidosis can develop. If the patient survives the initial crisis, liver damage with hepatitis and coagulation defects often occur within a couple of days. Renal failure and cirrhosis of the liver may occur as delayed effects (Elinder, 1986).

Long-term intake of iron in a form that is readily absorbed and in doses exceeding 50 to 100 mg of iron per day (0.7 to 1.4 mg/kg-day for a 70-kg adult male) (Elinder, 1986) results in an increased body burden of iron because iron is removed from the body at a much slower rate than it is absorbed. As the body burden of iron increases to between 20 and 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 concentrations of iron in the liver, pancreas, endocrine organs, and heart. This increased tissue iron can produce cirrhosis of the liver, disturbances in endocrine and cardiac function, and diabetes mellitus (Goyer, 1991) (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 (Goyer, 1991).

FIGURE 5.4
IRON TOXICITY RANGES

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, absorption of manganese is by diffusion only (Saric, 1986). High levels of dietary calcium and phosphorus have been shown to increase the requirements for manganese in several species (Lönnerdal et al., 1987).

Tissue accumulation and clearance

Manganese is widely distributed throughout the body. 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 (Goyer, 1991). Normal concentrations in the brain are low, but the half-time in the brain is longer and the metal may accumulate in the brain with excessive absorption (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 (Goyer, 1991; Saric, 1986).

Environmental sources of manganese

On the whole, food constitutes the major source of manganese intake for humans. The highest manganese concentrations are found in plants, especially wheat and rice. Drinking water generally contains less than 0.1 mg/L. Manganese levels in soil 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 to be between 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 would result in toxic effects with lower ingested doses of manganese in drinking water than in food (EPA, 1994a).

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

Limited information is available on the effects of manganese ingestion. Because effects from drinking water seem to differ from those from food sources, only studies on water consumption will be considered here. A Japanese study of 25 people drinking well water with manganese concentrations of 14 mg/L (0.4 mg/kg-day estimated intake) reported symptoms of intoxication, including a

mask-like face, muscle rigidity and tremors, and mental disturbances. There were two cases (8 percent) of death among intoxicated people. A Greek study of over 4000 individuals drinking 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) showed varying degrees neurological effects in individuals drinking from 0.007 to 0.07 mg/kg-day, but no effects 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 be more rapidly cleared. Exposure to insoluble forms results in lower manganese absorption, but higher chronic tissue levels and therefore greater toxicity (EPA, 1994a). Only limited information is available on the effects of various forms of manganese.

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

The toxicity of manganese is summarized in Figure 5.5.

5.1.6 Molybdenum

Absorption

Absorption of molybdenum 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 (Tipton et al., 1969; 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 route of excretion in humans is in 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 two orders of magnitude, from 0.1 mg/kg to 10 mg/kg, leading to

FIGURE 5.5
MANGANESE TOXICITY RANGES

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 from 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. 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 soil concentrations of molybdenum and subsequent increased molybdenum concentrations in food would lead to molybdenum intakes in the range of 0.14 to 0.21 mg/kg-day. These health effects are summarized in Figure 5.6 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 but was 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

FIGURE 5.6
MOLYBDENUM TOXICITY RANGES

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

In humans, the majority of 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, 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 ingestion of 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 survey of United States ground water, 97 percent of all samples (total of 2053 samples) contained less than 20 $\mu\text{g}/\text{L}$ of nickel and 80 percent had less than 10 $\mu\text{g}/\text{L}$, although in areas near nickel mining operations, levels as high as 200 $\mu\text{g}/\text{L}$ have been reported (Friberg et al., 1986).

Toxicity of nickel

Acute exposure to high levels of nickel 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 have not been 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 were also 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 persons exposed to the same level of nickel in the environment. Chemical exposure history, genetic make-up, developmental state, health, and nutritional status result in decreased function of the detoxification and excretory processes (mainly hepatic and renal) or the preexisting compromised function of target organs. For these reasons, it is expected that the elderly (with declining function) and the youngest of the population (with immature and developing organs) will generally be more vulnerable to toxic substances than healthy adults.

Exposure to nickel may lead to sensitization. Available data indicate that 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 do. The threshold for elicitation of the response has been suggested to be 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. These health effects are summarized in Figure 5.7 as a function of dose.

FIGURE 5.7
NICKEL TOXICITY RANGES

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. It travels in the blood, where it ultimately passes through the kidneys. The kidneys filter all the sodium out; then, with great precision, 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 the regulation of 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.

Excretion of sodium is mainly urinary, with appreciable amounts also excreted in feces, sweat, and tears (Venugopal and Luckey, 1978). Mammalian renal 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 human adults. 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. From the Chemical Analysis of Interstate Carrier Water Supply Systems, 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

Acute toxicity symptoms from sodium chloride in healthy adult males 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/kg-day, since sodium is 39 percent of the weight of sodium chloride) (Meneely and Battarbee, 1976). The mean lethal dose of sodium for humans was reported to be 3230 mg/kg (Venugopal and Luckey, 1978).

Epidemiological studies have indicated that 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 that there appear to be critical levels of sodium ingestion that cause blood pressure to rise with age and some people to develop hypertension. It has been reported that with sodium intake below 227 mg per day (3 mg/kg-day for a 70-kg adult), hypertension was absent (Freis, 1976). 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. Average daily intakes in the United States from dietary sources are often within a range where hypertensive effects occur. Drinking water generally contains relatively low levels of sodium and therefore does not significantly contribute 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 kidney is the major organ 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 America 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.

The potential sodium health effects as a function of dose are summarized in Figure 5.8.

5.1.9 Sulfate

Absorption

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

Tissue accumulation and retention

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

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 of sulfates. 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 western United States supplies. No estimates are available on intake of sulfates from food sources.

FIGURE 5.8
SODIUM TOXICITY RANGES

Toxicity of sulfate

The acute and chronic effects of sulfate toxicity differ more in severity than in symptoms or mechanisms. Therefore, this discussion will combine acute and chronic toxicity. As mentioned above, there are no data to indicate a bioaccumulation of sulfate 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 is the basis of the toxic effects as well.

Toxicity in humans is primarily manifested in diarrhea; the severity of the diarrhea is dose-dependent. Chronic ingestion of sulfate can result in persistent diarrhea, leading to ionic imbalances and dehydration similar to that seen with extremely high acute doses. In the case of drinking water 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 most susceptible population for sulfate-induced diarrhea. Also, there are data to indicate diabetic and elderly populations with compromised kidney function may be more sensitive than healthy adults to the effects of sulfates (EPA, 1992a). These health effects are summarized in Figure 5.9 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 from sampling their drinking water. Therefore, these data do not represent well-controlled studies where dosage ranges can be readily defined.

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 in the form of uranium-238 (Cothorn 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. The radioactive decay chain of uranium-238 and uranium-234 is summarized in Figure 5.10. 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.

FIGURE 5.9
SULFATE TOXICITY RANGES

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

Absorption

Absorption of uranium in the gastrointestinal tract depends on the solubility of the uranium compounds. The hexavalent uranium compounds, especially the uranyl salts, are water-soluble, while tetravalent compounds generally are not (Weigel, 1983). Even with soluble compounds, only a small fraction is absorbed. Human gastrointestinal absorption rates of 0.76 to 7.8 percent have been determined (Wrenn et al., 1985).

Tissue accumulation and clearance

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

In body fluids, uranium tends to be converted into water-soluble hexavalent uranium (Berlin and Rudell, 1986). Approximately 60 percent of the uranium in plasma complexes with low-molecular-weight anions (e.g., bicarbonates, citrates), while the remaining 40 percent binds to the plasma protein transferrin (Stevens et al., 1980). Following oral exposure in humans, more than 90 percent of uranium is excreted in the feces and not absorbed in the gastrointestinal tract. Of the small percent that is absorbed (typically less than 5 percent), approximately 60 percent is excreted in the urine within 24 hours and 98 percent is excreted within 7 days, based on animal studies (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). It is absorbed from soil into plant tissues to an extent that depends on the plant species and the depth of its root system (Berlin and Rudell, 1986). Plant concentrations of uranium averaged 0.075 $\mu\text{g/kg}$ 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 $\mu\text{g/kg}$ (Prister, 1969). The total dietary intake of uranium from the consumption of average foods is approximately 1 μg 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 intake of uranium (Berlin and Rudell, 1986).

Toxicity of uranium

Exposure of the general public to natural uranium is unlikely to pose an immediate lethal threat to humans. 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 to be as low as 14 mg/kg-day following 23-day oral exposures, depending on the solubility of the uranium compound tested (higher solubility compounds have greater toxicity), route of exposure, and animal species. High doses of uranium cause complete kidney and respiratory failure.

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

The 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). The health effects for uranium are summarized in Figure 5.11 as a function of dose.

5.2 CONTAMINANT INTERACTIONS

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

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

FIGURE 5.11
URANIUM TOXICITY RANGES

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 bioligand cofactor facilitating the gastrointestinal absorption of iron. This interaction occurs only when the iron is present in the form of a ferric ion (DHHS, 1991). In the Lakeview ground water, most of the detected iron exists in the form of ferric ions (Table 3.7). Therefore, nickel in the soil at this site may affect iron absorption.

In animal studies, iron status affected uranium absorption (EPA, 1989b). No other information on uranium interactions with other metals has been found. However, the common target organ suggests interaction with nickel in the production of kidney toxicity.

The carcinogenic effects of the radionuclides and arsenic probably combine at least additively.

5.3 CONTAMINANT RISK FACTORS

The most recent oral RfDs for the noncarcinogenic contaminants of concern are summarized in Table 5.1. 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), being 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 whether cancer risks at low doses are dose proportional (i.e., the linear dose-response hypothesis) or whether 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 intake of carcinogens over a lifetime with the increased excess cancer risk from that exposure. The most recent cancer SFs for arsenic, polonium-210 and the uranium-234/-238 radioactive decay series are given in Table 5.2.

Two potential sources of uncertainty are the following:

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

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	Water/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	1.6E-11	A	Note ^c	Water/HEAST
Uranium-238	1.6E-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.

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

6.0 HUMAN RISK EVALUATION

To evaluate human health risks to an individual or population, the results of the exposure assessment are combined with the results of the toxicity assessment. As discussed in Section 5.0, potential adverse health effects are a function of how much of the contaminant an individual takes into his or her body. Indeed, at lower levels many of the contaminants associated with the mill tailings are beneficial to health, since they are essential nutrients. At higher levels, these same elements can cause adverse health effects or, at very high levels, death. In this section, the expected intake, if ground water within the plume were used as drinking water, is correlated to potential health effects from these levels of exposure.

6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

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

The primary concerns for human health from ingestion of contaminated ground water at the Lakeview site are from exposure to sulfate and manganese, and to a lesser extent, chloride, iron, and molybdenum.

The potential range of sulfate exposure from drinking contaminated ground water is mostly above the range expected to produce mild toxicity exhibited by laxative effects in adults and diarrhea in infants (Figure 6.1). The majority (approximately 75 percent) of the potential exposure range falls within the range of exposures that could result in severe persistent diarrhea in infants. Diarrhea resulting from the sulfate exposures could alter the excretion or reabsorption of manganese, 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.

For manganese, nearly all simulated exposures are greater than the oral RfD derived by the EPA and are greater than the levels reported to produce mild neurological symptoms often similar to early symptoms of Parkinson's disease. Approximately 85 percent of the distribution falls above levels that have been reported to potentially produce more fully developed Parkinson's-like effects following 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 produces chronic iron

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

BACKGROUND INTAKE LEVELS

NO OBSERVED EFFECTS

MILD TOXICITY RANGING FROM LAXATIVE (ADULTS) TO DIARRHEA IN INFANTS

SEVERE TOXICITY - PERSISTENT INFANT DIARRHEA LEADING TO DEHYDRATION - - - -> DEATH

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

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

▲ RDA FOR ADULT MEN (0.14 mg/kg-day)

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

...
INCREASED IRON IN ORGANS, SKIN PIGMENTATION,
CIRRHOSIS WITH INCREASED DOSES

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

toxicity, including skin pigmentation, potential disruption of liver and endocrine 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 ground water is expected to be in the trivalent form, which is less readily absorbed, and therefore less toxic, than the divalent form.

For sodium, nearly the entire range (99 percent) of the simulated intakes 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 would be associated with mild toxicity manifested primarily by mineral imbalances (increased copper and uric acid excretion) (Figure 6.5). If these exposures were increased by 20 percent due to contribution from milk ingestion and by 20 percent due to contribution from meat ingestion, the doses would still be within the dose range expected to result in mild toxicity. Even with these increased dose contributions, the predicted intake of molybdenum would still be well below the range reported to result in gout-like symptoms.

For chloride, the majority (approximately 90 percent) of the simulated 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 of the simulated exposure range for nickel is below the oral RfD and would not be expected to result in adverse effects in 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). However, the potential exposure to uranium would not be expected to be associated with toxicity in animals (Figure 6.8). Although animal data are not always predictive of human toxicity, the lack of available human data to evaluate oral toxicity of uranium at these concentrations should not be seen as an indication that no toxic effects will occur. RfDs incorporate safety and uncertainty factors and therefore are generally conservative values designed to protect human health. However, they are based on careful evaluation of existing data bases; therefore, exposures that significantly exceed the RfD should be considered as potential problems.

For the noncarcinogenic effects of arsenic, nearly all of the simulated exposure range for ingestion of contaminated ground water is within the range associated with normal background intake levels (Figure 6.9). A very small portion (less

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

BACKGROUND INTAKE FROM DRINKING WATER

▲ NUTRITIONAL REQUIREMENT

▲ NRC-RECOMMENDED LIMIT FOR DIETARY INTAKE

...
DIETARY INTAKE LEVELS

...
HYPERTENSION AND KIDNEY IMPAIRMENT MAY OCCUR IN HUMANS

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

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

LAKEVIEW, OREGON, SITE

HEALTH EFFECTS OF POTENTIAL NICKEL EXPOSURE RANGES FOR CHILDREN

FIGURE 6.7

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 WOULD RESULT IN ALLERGIC DERMATITIS IN SENSITIZED INDIVIDUALS.

DIETARY INTAKE

▲ 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 REPORTED
AT ANY OF THE POTENTIAL INTAKES IN CHILDREN.

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

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

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

SKIN PATHOLOGY OCCURS IN HUMANS (HYPERPIGMENTATION, HYPERKERATOSES)
WITH INCREASED DOSES
ARTERIAL THICKENING IN CHILDREN AND ADULTS

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 (Figure 6.10), even though a large portion of the intake range exceeds the oral RfD.

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

As discussed in Section 5.2, physiologic interactions among the numerous metal contaminants at this site are likely. These interactions are likely to alter 1) the absorption of some metals in the presence of others, resulting in competition for binding sites once absorbed, 2) the concentrations of binding proteins that often transport metals in the bloodstream and through cell membranes, and 3) the tissue distribution of individual metals. The complexity of these interactions and the number of metal contaminants at Lakeview make it impossible to predict whether the toxicity of a given metal will be increased or decreased by concurrent exposure to multiple 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. Although the carcinogenic potential of arsenic is currently under evaluation by the EPA, the cancer SF previously assigned to arsenic is used in this risk assessment. Table 6.1 presents estimates of the excess lifetime cancer risk predicted to result from exposure to arsenic and the two radionuclide contaminants of potential concern (polonium-210 and uranium) through ingestion of contaminated ground water in the Lakeview site area. These estimates are based on the cancer SFs developed by the EPA; however, natural uranium has not been demonstrated to cause cancer in humans or animals following ingestion exposures. The excess lifetime cancer risk estimates for the two radionuclides identified as exceeding background (polonium-210 and uranium) are within the potentially acceptable range which, according to the National Contingency Plan (NCP), is between 10^{-4} and 10^{-6} . However, the estimated carcinogenic risk associated with the median and the UCL arsenic concentrations are within the range of 10^{-3} and 10^{-4} , which exceeds the NCP criteria. Summing the risks for the radionuclides calculated in Table 6.1 to one significant figure, median and UCL excess cancer risk estimates from ground water ingestion would be 4×10^{-5} and 6×10^{-5} , respectively, which is within the potentially acceptable range. Summing the risks for all carcinogenic

▲ ORAL RfD (0.09 mg/kg-day)

□ AVERAGE DIETARY INTAKE (FOOD AND WATER, 0.14-0.35 mg/kg-day)

.....
NO ADVERSE EFFECTS IN MICE, DOGS, AND RATS

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

FIGURE 6.10

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

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

Contaminant of potential concern	Concentration in ground water ^a		Ground water exposure doses ^b		Oral SF (mg/kg-day; pCi) ⁻¹	Excess lifetime cancer risk ^c	
	Median	UCL	Median	UCL		Median	UCL
Arsenic	0.06	0.11	1.2E-03	2.0E-03	1.8E+0	2E-03	4E-03
Polonium-210	6.8	8.5	2.4E+05	3.0E+05	1.5E-10	4E-05	5E-05
Uranium	6.2	14	2.2E+05	4.9E+05	1.6E-11	4E-06	8E-06
Total radionuclide risk:						4E-05	6E-05
Total carcinogenic risk:						2E-03	4E-03

^aConcentrations in mg/L for arsenic and in pCi/L for polonium-210 and uranium.

^bExposure doses in mg/kg-day for arsenic and in pCi/lifetime for polonium-210 and uranium.

^cExcess lifetime cancer risk is calculated by multiplying the exposure dose by the SF.

SF - slope factor.

UCL - upper 95 percent confidence limit on the mean.

Ingestion rate: 2 L per day.

Exposure frequency: 350 days per year.

Exposure duration: 50 years.

chemicals results in 2×10^{-3} and 4×10^{-3} for median and UCL cancer risk estimates, respectively. These cumulative cancer risks exceed the NCP criteria.

6.3 LIMITATIONS OF RISK EVALUATION

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

- This risk assessment evaluates only risks related to inorganic ground water contamination. Potential contamination with any of the few organic constituents used in uranium processing has not been addressed.
- With the exception of individuals sensitized to nickel due to previous nickel exposures, subpopulations that might have increased sensitivity to specific contaminants are not specifically addressed on the graphs.
- Some individuals may be more sensitive to the toxic effects of certain constituents for reasons that have not been determined.
- There are uncertainties associated with the derivation of the toxicity values (RfDs and SFs). These values are intentionally derived in a way that is more likely to overestimate than underestimate risk.

- Data available to interpret potential adverse health effects are not always sufficient to allow accurate determination of all health effects (i.e., lack of testing in humans or testing of dose ranges other than those expected at this site).
- Although plume movement is evaluated hydrologically and geochemically, the monitoring locations sampled may not be in the most contaminated portion of the plume.
- Only the drinking water exposure pathway has been considered in depth, although other pathways were screened to determine their contribution.

The evaluation presented here has considered these limitations and compensated wherever possible by presenting toxicity ranges rather than point estimates to incorporate as much variability as could be reasonably defined. The impact 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 the risk assessment is to determine whether contaminants detected at the site have the potential to adversely affect the existing biological community at or in the area surrounding the site. Currently, the EPA has no guidance for quantifying potential impacts to ecological receptors but has developed a qualitative approach generally used for ecological evaluation (EPA, 1989b). With the qualitative approach, the EPA recommends that ambient environmental media concentrations be compared to water quality, sediment quality, or other relevant criteria to determine whether any of the concentrations that the ecological receptors are expected to encounter exceed these criteria.

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

7.1 EXPOSURE CHARACTERIZATION

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

Currently, impacted media at the site include contaminated ground water and potentially contaminated surface water and sediments. Since the tailings pile and contaminated soil have been removed from the processing site, some direct exposure pathways (such as incidental ingestion of soil, dermal contact with soil, and inhalation of air containing particulates) do not represent an ecological concern and will not be evaluated in this baseline risk assessment. Other direct exposure pathways (such as ingestion of surface water and sediments potentially affected by contaminated ground water or particulate transport and bioconcentration of contaminants in surface water by aquatic organisms) and indirect exposure pathways (such as consumption of previously exposed organisms) are possible at the site.

The net accumulation by organisms of a constituent directly from the surrounding environment is known as bioconcentration. Net accumulation by organisms as a result of all routes of exposure, including the diet, is known as bioaccumulation. Generally, bioconcentration is measured for uptake of chemicals from water by aquatic organisms. Bioconcentration factors (BCF) 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 vicinity of the site are Hunters Hot Springs, Warner Creek, Thomas Creek, and Hammersley Creek. These surface water bodies 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. These exposure pathways were evaluated in this risk assessment.

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

For the purposes of this baseline risk assessment, it was assumed as a conservative measure that a domestic well could, at some point in the future, be placed at a location that intercepts the most contaminated ground water in the plume. The water from this hypothetical well could be used for a livestock watering pond (which could also be stocked with fish) or for irrigation of agricultural crops.

7.2 ECOLOGICAL RECEPTORS

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

The following information on ecological receptors is based primarily on surveys done before tailings removal began (DOE, 1985b) and is provided as a historical perspective. 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 located in the ecoregion known as the Intermountain Sagebrush Province. This semiarid region is situated in the rain shadow of the Cascade Mountains to the west. The region is located in the ponderosa shrub forest natural vegetation type (DOE, 1985b).

The site is characterized primarily by introduced grassland vegetation. The site has also been colonized by desert saltgrass (*Distichlis spicata* var. *stricta*). Hammersley Creek is distinguished by a narrow riparian zone as it 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 potentially present in the vicinity of the site 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 of the use by waterfowl 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 a number of other species present as migrants or vagrants.

Reptiles of 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

In the state of Oregon, there are two species of plants on the U.S. Fish and Wildlife Service (FWS) listing 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 (1994)) (45 FR 82480 (1980); 48 FR 53640 (1983)).

The Oregon Natural Heritage data base, maintained for the state by the Oregon Chapter of the Nature Conservancy, indicates that neither of the two plant species listed as endangered nor the one plant species proposed for protected status occurs in or near the Lakeview site (Wolfen, 1985).

The only wildlife species listed by the FWS (50 CFR Part 17 (1994)) potentially present at the site are the American peregrine falcon (endangered) (*Falco peregrinus anatum*) and the bald eagle (currently proposed as a threatened species) (*Haliaeetus leucocephalus*), both of which could occur as vagrants or migrants because of the proximity to suitable habitat along nearby lakes and rimrocks.

7.2.3 Aquatic organisms

No quantitative surveys of aquatic organisms occurring in the water bodies near the site have been conducted to date as part of this baseline risk assessment.

A brief qualitative survey of the aquatic organisms was conducted in the vicinity of historic surface water sampling locations 605, 606, 607, 608, 609, and the 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 of the water sampling locations: dragonfly and damselfly nymphs (*Odonata*); water striders (*Gerridae*); backswimmers (*Notonectidae*); midge larvae (*Chironomidae*) and unidentified water fleas (*Cladocera*). No fish were observed at any location, which is not surprising due to the limited habitat in the streams. All of the streams, with the exception of the open water marshy areas in Hunter Hot Springs, had shallow water (less than 4 inches [10 cm] in depth) and were channelized with no riffle/run areas or undercut banks.

7.3 CONTAMINANTS OF ECOLOGICAL CONCERN

The complete list of ground water contaminant levels that exceed background levels (Table 3.6) was considered in assessing contaminants of potential concern for ecological receptors potentially exposed to ground water (for example, plant uptake, livestock watering, and fish stocked in a pond). This list of contaminants was then compared with the available surface water data collected in October 1993. Table 7.1 presents the surface water data for those contaminants analyzed in the surface water samples and above background levels in ground water.

If a contaminant was not detected in surface water (for example, nickel) or the concentration detected adjacent to the site was less than or equal to the concentration upstream of the site (the background level), it was excluded as a contaminant of potential concern for ecological receptors. Uranium was detected only at the upstream location. Additionally, iron and molybdenum are excluded as contaminants of potential concern because the difference between the adjacent location concentrations versus the upstream concentration was minimal. The concentrations used in these comparisons to background levels were from unfiltered samples.

After these comparisons were made, none of the detected constituents were identified as potential ecological contaminants of concern for the water bodies in the site vicinity.

No sediment samples were collected from the water bodies in the site vicinity before the October 1993 sampling. Only 10 constituents were analyzed at that time (arsenic, iron, manganese, molybdenum, nickel, sulfate, tin, uranium, vanadium, and zinc). This list of analytes was selected prior to development of the risk assessment. Table 7.2 presents the sediment data for those contaminants analyzed in the sediment samples and above background levels in ground water (tin, vanadium, and zinc are not considered site-related constituents).

Table 7.1 Occurrence of constituents analyzed for in surface water collected from water bodies in the site vicinity, Lakeview UMTRA Project site, Lakeview, Oregon

Constituent	Upstream of site location ID	Adjacent to site location ID			Pond location ID
	609	608	605	606	613
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 mg/L for unfiltered samples collected October 26, 1993.

Table 7.2 Occurrence of constituents analyzed for in sediment collected from surface water bodies in the site vicinity, Lakeview UMTRA Project site, Lakeview, Oregon

Constituent	Upstream of site location ID	Adjacent to site location ID			Downstream of site location ID	Pond location ID
	609	608	605	606	607	613
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 (mg/kg DW). The samples were collected October 26, 1993.

The contaminants of potential concern in sediments were selected in the same manner used for the surface water data. Arsenic, nickel, and sulfate were excluded as contaminants of potential concern because the differences between the adjacent and downstream concentrations versus the upstream concentration were minimal. The contaminants of potential concern selected for sediment are iron, manganese, and uranium.

7.4 POTENTIAL IMPACTS TO WILDLIFE AND PLANTS

7.4.1 Ground water

Concentrations of the contaminants of potential concern in plant tissue were estimated using soil-to-plant BCFs. Soil concentrations in the saturated zone were estimated by multiplying the ground water concentration by the soil-water distribution coefficient (K_d). The methodology and parameters used to estimate root uptake and plant tissue concentrations at harvestable maturity for the contaminants of potential concern are presented in Table 7.3. This methodology is described in detail elsewhere in the literature (Baes et al., 1984) and therefore will not be presented here.

The estimated tissue concentrations for the contaminants of potential concern in the vegetative portions (for example, stems and leaves) and in the nonvegetative portions (for example, fruits and tubers) of plants at harvestable maturity were compared to approximate concentrations (in mature leaf tissue) that have been reported to be toxic to plants (phytotoxic) (Table 7.3). Data on soil concentrations of metals that have been reported to be phytotoxic 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 7.3, few available data relate tissue concentrations to phytotoxicity. The reported phytotoxic concentrations are not representative of very sensitive or highly tolerant plant species because the bulk of the data is from 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 the phytotoxicity data. No comparison data were available for chloride, iron, polonium-210, sodium, sulfate, and uranium. Thus, it is not possible with available information to evaluate whether the estimated tissue concentrations could result in adverse effects to plants.

Bioaccumulation in terrestrial organisms as a function of contaminants of potential concern in ingested plants or animals (for example, birds eating fish) is a potential exposure pathway at the site. Birds and other vertebrates consuming these plants and animals can bioaccumulate some of the contaminants of potential concern from this diet if the amount ingested exceeds the amount eliminated. This is often a function of the areal extent of

Table 7.3 Comparison of estimated plant concentrations to phytotoxic concentrations, Lakeview UMTRA Project site, Lakeview, Oregon

Contaminant of potential concern	UCL concentration in ground water (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factors		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	0.0085	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 pCi/L was converted to mg/L.

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.

mg/kg—milligrams per kilogram.

mg/L—milligrams per liter.

NC—value cannot be calculated because Kd is zero.

L/kg—liters per kilogram.

NA—not available.

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

contamination versus the areal extent of the animals' feeding range. In the case of small contaminated areas, the amount of food in the diet usually exceeds the impacted food, and bioaccumulation is not a concern. Therefore, exposure via the diet for all food chain species is possible in certain areas (for example, wetland areas), but the potential for bioaccumulation is not always a concern.

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

To evaluate the potential impact that use of contaminated ground water in a livestock pond might have on wildlife (that is, animals drinking from the pond or fish stocked in the pond), the UCL ground water concentrations for the contaminants of potential concern were compared to available comparison water quality criteria (Table 7.4). There are no available federal or state criteria or standards established for the protection of terrestrial wildlife via water exposure. Therefore, it is difficult to evaluate the potential hazards to terrestrial receptors without additional information. However, available surface water quality values for the protection of freshwater aquatic life do exist and include 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 7.4), indicating that 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 the ground water in the area is for irrigating agricultural crops. Table 7.4 compares the approximate concentrations in water used for irrigation purposes that should be protective of plants with the ground water concentrations (EPA, 1972). The EPA developed these approximate irrigation water concentrations to protect agricultural crops from toxicity associated with buildup of a particular constituent in the soil. Six of the contaminants of potential concern—arsenic, boron, iron, manganese, molybdenum, and nickel—have comparison values. The UCL ground water concentrations for all of these inorganics exceed the comparison values, with the exception of nickel. No comparison values are available for the remaining contaminants of potential concern. Thus, it is not possible to evaluate the

Table 7.4 Comparison of contaminants of potential concern in ground water with available water quality values, Lakeview UMTRA Project site, Lakeview, Oregon

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

^aValue obtained from the Water Quality Criteria Summary, 340-41, Oregon Administrative Rules, Oregon Department of Environmental Quality (ODEQ, 1992), unless specified otherwise. These values are standards protective of aquatic life via chronic exposure.

^bFrom EPA (1972), unless specified otherwise. Irrigation water 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 (FWQC) 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 (mg/L) 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 (CDPHE, 1991). This is a hardness-dependent standard.

Concentrations reported in milligrams per liter unless otherwise noted.

NA—not available.

potential for these compounds to adversely affect plants when applied in irrigation water.

Based on the available information, use of the alluvial ground water near the site (containing the UCL concentrations) as a continuous source of irrigation water may result in deleterious effects to crops, primarily because of the elevated concentrations of arsenic, boron, iron, manganese, and molybdenum.

7.4.2 Surface water

Aquatic life in surface water bodies in the site vicinity may be exposed to contaminants associated with the site. It is possible that contaminants may have washed into these drainages through surface water runoff before the tailings were removed.

Terrestrial wildlife can be directly exposed to contaminants by ingesting surface water, sediments, aquatic organisms, and plants that may have bioconcentrated contaminants. However, good information on generic BCFs for terrestrial wildlife is currently not available in the scientific literature.

7.4.3 Risk to aquatic life

Surface water data collected from the upstream location (location 609) were compared with data from locations adjacent to the site (locations 605, 606, and 608) as part of the selection process for the contaminants of ecological concern (Section 7.3). The results of the comparison indicated that the constituent concentrations detected adjacent to the site either did not exceed background concentrations or were similar to (within the same order of magnitude) the background concentrations. This suggests that past activities at the site have not affected the surface water quality in the vicinity. However, the surface water quality data base is limited, and additional data would be required to confirm conditions.

Many of the contaminants identified as exceeding background ground water quality have not been measured recently in surface water from the water bodies in the site vicinity. Thus, the potential for these other contaminants to represent a hazard to aquatic life cannot be evaluated.

7.4.4 Sediments

Surficial sediment samples (0 to 4 inches [0 to 10 cm]) were collected on October 26, 1993, from the same locations at which surface water samples were collected (Figure 3.3). Prior to this sampling event, no sediment samples had been collected near the site. The sediment samples were analyzed for arsenic, iron, manganese, molybdenum, nickel, sulfate, tin, uranium, vanadium, and zinc.

There are no established state or federal sediment quality criteria (SQC) for the protection of aquatic life for the contaminants of potential concern at this site (EPA, 1988; NOAA, 1990).

The concentrations of the contaminants of potential concern (iron, manganese, and uranium) detected at the adjacent and downstream locations were compared with the upstream concentrations. The concentrations of iron, manganese, and uranium generally increase at the adjacent and downstream locations relative to the upstream location (Table 7.2).

Because there are no guidelines for iron, manganese, and uranium, it is not possible to evaluate the potential for these concentrations to represent a hazard to ecological receptors without further study.

Many of the contaminants identified as exceeding background ground water quality have not been measured in sediments from the water bodies in the site vicinity. Thus, the potential for these other contaminants to represent a hazard to aquatic life cannot be evaluated at this time.

Another basic criterion required to evaluate the suitability of water quality for irrigation, in addition to the concentrations of metals that may be toxic to plants, is the sodium hazard. 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 the physical characteristics of soils (e.g., increase in colloiddally 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 calcium plus magnesium in the water. Using the 95 percent UCL ground water concentrations for sodium, calcium, and magnesium, a 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.

7.5 POTENTIAL IMPACTS TO LIVESTOCK

Ingestion by livestock of vegetation that may have bioconcentrated contaminants from alluvial ground water is a potential pathway. However, without additional data (for example, actual plant tissue concentrations or exposure and toxicity information for livestock), it is difficult to evaluate this exposure pathway.

It is possible that ground water could be used in the future to provide water for livestock watering. To evaluate the potential impact to livestock in this future hypothetical scenario, the UCL ground water concentrations for the contaminants of potential concern were compared to approximate drinking water concentrations considered to be protective of livestock (EPA, 1972) (refer

to Table 7.4). The comparison water quality guidelines for boron and sulfate are exceeded by the UCL ground water concentrations, while the UCL concentration for arsenic is below the guideline. The ground water concentration for boron (58 mg/L) is more than 10 times the livestock drinking water value (5 mg/L).

Sulfate was detected in ground water at a UCL concentration of 6860 mg/L. This level is above the value (1000 mg/L) reported to cause diarrhea in cattle (National Research Council, 1971; Church, 1984).

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

7.6 LIMITATIONS OF THE ECOLOGICAL RISK ASSESSMENT

The qualitative evaluation of potential ecological risks presented here is a screening level assessment of the risks associated with potential exposure of plants and animals to contaminated ground water, surface water, and sediment at the 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 (for example, drought), biotic interactions, behavior patterns, biological variability (differences in physical conditions, nutrient availability, etc.), 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 was collected during this screening.
- Little is known about site-specific intake rates for wildlife or amounts of contaminants taken up by plants. General literature values were used in many cases.
- Only limited ecotoxicological reference data are available.
- Considerable uncertainty is associated with the toxicity of mixtures of contaminants.

7.7 SUMMARY

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

A limited data set currently exists of the sediment quality in the surface water bodies near the site. The available data suggest that the site is not acting as a source of sediment-bound metals.

Based on available data and criteria, a potential threat exists 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 manganese and molybdenum, could cause phytotoxicity following accumulation in plant tissue. This ground water would not be suitable for continuous long-term use as irrigation water for crops due to arsenic, boron, iron, manganese, molybdenum, and sodium. Water from the most contaminated wells in this aquifer would not be suitable as a source of water for fish to live in. 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.

The potential for the contaminants of concern detected in media at the site to represent a food chain hazard to terrestrial and aquatic wildlife (via bioaccumulation and biomagnification) is considered low, based on available surface water, sediment, and ground water data. However, no tissue analysis from biota (for example, invertebrates and plants) has been conducted.

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

8.0 INTERPRETATIONS AND RECOMMENDATIONS

8.1 RISK SUMMARY

The UMTRA Project is required by the UMTRCA 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 processing site to evaluate the presence or probable absence of these hazards.

Although some domestic wells downgradient of the processing site have high concentrations of sulfate and chloride, these wells are probably not affected by site-related contaminants. However, as a conservative measure, the most contaminated wells at the processing site were used in this risk assessment to evaluate potential future use of ground water. This assessment is conservative because the contaminant concentrations are expected to decrease over time due to removal of the source of contamination (that is, the tailings), and because the contaminant concentrations decrease with depth in the upper aquifer and distance from the site due to the effects of dilution and dispersion. Health risks would be associated with potential exposure from drinking contaminated ground water at the site.

If ground water at the processing site were ingested, serious adverse health effects would 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. Excess lifetime cancer risks associated with drinking water exposure to 95 percent UCL on the mean concentration of arsenic in contaminated ground water is at a level that exceeds the NCP's maximum criterion of 1 in 10,000 (5 in 1000).

Use of contaminated ground water from current or potential future wells downgradient of the site for irrigating crops or gardens and watering livestock is not anticipated to result in human health risks. Adverse human health effects would not be expected following ingestion of milk and meat from animals grazed and watered on the pastureland downgradient of the site or from ingestion of garden produce watered with the contaminated ground water.

The contaminated ground water near the processing site would not be acceptable as a source of water for fish to live in or as a source of continuous irrigation water for agricultural crops. In addition, 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.

Available surface water and sediment data from water bodies in the site vicinity suggest that contaminated ground water from the site has not adversely affected surface water and sediment quality. None of the contaminant concentrations detected in the surface water and sediments exceeded the

available water quality criteria. No sediment quality values are available to evaluate the potential for adverse effects.

8.2 LIMITATIONS OF THIS RISK ASSESSMENT

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

- This document evaluates risks associated with exposures only to inorganic contaminants of ground water at the UMTRA Project site near Lakeview. Potential organic contaminants (those few related to uranium processing) have not been considered.
- In general, the results presented in this document are based on filtered (0.45-micron) water samples. Unfiltered data were used for arsenic. The effect of filtration differs for different elements. Although the difference on UMTRA Project sites is usually not large, filtered samples can have somewhat lower or equal concentrations than unfiltered samples for some constituents. Constituents in suspension may be lost with filtration but can still produce toxic effects if ingested and broken down in the acidic environment of the stomach.
- The toxicity of any contaminant varies both from person to person and in the exposure level. For example, normal variability in biochemical factors among individuals, differences in medical history, previous exposure to toxicants, and dietary and exercise habits can all affect susceptibility to chemical toxicity. Chemicals exert different toxic effects, depending on the receptor's exposure level. In presenting ranges of exposures that can produce toxic effects, this assessment tries to emphasize that variability. However, it is not possible to account for all sources of variability and still present useful and meaningful analyses. Cases in which specific subpopulations of individuals are known to be more sensitive to toxic effects of given constituents have been noted. Using ranges for expected toxic effects provides the reader with a better understanding of the likelihood that toxicity will occur.
- To assess toxicity, standardized reference values developed by agencies such as the EPA are used to determine plant uptake, tissue concentrations in livestock, and toxic effects in humans. These reference values themselves have limitations, including the following:
 - Toxicity, uptake, and bioconcentration data are not available for all constituents elevated above background 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 are generally based on exposure to only the constituent of concern. In reality, exposures generally occur simultaneously to several chemicals. The interactive effects of multiple constituents and the impact of these interactions on expected toxicity generally cannot be accurately assessed from existing data.
- Considerable effort has been directed at determining plume movement and placing monitor wells in locations that capture maximal contamination. Nevertheless, physical systems and models used to determine contaminant plume migration vary widely and may result in well placements that do not measure the highest contaminant concentrations or determine the fullest extent of plume impact.
- Variability can be introduced through sampling and analytical processes. 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. However, there is not a high degree of confidence in the reliability of the data for the Lakeview site.
- 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. When a measurable contribution from other pathways could increase expected exposure significantly enough to alter the predicted toxicity, the alternate source contribution is noted in Section 6.0.

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 (1994)); in 1987, the EPA proposed revised ground water standards (52 FR 36000 (1987)). The UMTRA Project is required to comply with the 1987 proposed ground water standards until final standards are published. The ground water standards consist of 1) ground water protection standards to evaluate disposal cell performance, and 2) ground water cleanup standards for existing contamination at processing sites. The contaminants that have a proposed maximum concentration limit (MCL) are

summarized in Table 8.1. Because an MCL has not been established for every contaminant, the proposed standard requires meeting background levels for those contaminants that do not have an MCL.

While the proposed standards apply only to the UMTRA Project, the EPA has also published drinking water health advisory levels for both long- and short-term exposures. These advisories are shown in Table 8.1.

8.4 RISK MITIGATION MEASURES/INSTITUTIONAL CONTROLS

A potential for adverse health effects exists following long-term use of contaminated ground water below the Lakeview former uranium processing site. The city of Lakeview and numerous private wells are near the processing site in Lake County, Oregon. Surface water bodies exist at and near the processing site, but sampling has indicated that surface water is not contaminated from ground water affected by the processing site. This section presents possible ways to restrict access to contaminated ground water to mitigate potential risks.

In the preamble to the proposed 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 (52 FR 36000 (1987)). Although the preamble refers to institutional controls for long periods of time (e.g., up to 100 years during natural flushing), this concept can also be applied to short-term restrictions of access to ground water. Because not all 24 UMTRA Project sites can be evaluated simultaneously, institutional controls may be needed pending remedial action decisions or during implementation of remedial action for 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 discontinue a well's use if it were determined to be contaminated (Carter, 1994).

The Oregon Department of Environmental Quality (ODEQ) has an official notification policy to restrict use of privately owned ground water wells if they present a major contamination problem to other wells in the vicinity. ODEQ rules can be implemented if ground water well contamination is proven to be a major source of contamination. The ODEQ also notifies in writing any private well owner if there is a known contamination problem (Bailey, 1994).

Table 8.1 Concentration limits of constituents

Constituent	UMTRCA MCL 40 CFR §192.02 (mg/L)	Health advisories 10-kg child, 10-day (mg/L)	Health advisories 70-kg adult lifetime (mg/L)
Chemicals (inorganic)			
Antimony	-	0.015	0.003
Arsenic	0.05	-	-
Barium	1.0	-	2
Boron	-	0.9	0.6
Cadmium	0.01	0.04	0.005
Chromium	0.05	1.0	0.1
Cobalt	-	-	-
Copper	-	-	-
Fluoride	-	-	-
Iron	-	-	-
Lead	0.05	-	0.015
Manganese	-	-	-
Mercury	0.002	-	-
Molybdenum	0.1	0.08	0.04
Nickel	-	1.0	0.1
Nitrate	44 ^{a,b}	44 ^c	-
Selenium	0.01 ^a	-	-
Silver	0.05	0.2	0.1
Strontium	-	25	17
Sulfate	-	-	-
Thallium	-	0.007	0.0004
Vanadium	-	0.08	0.02
Zinc	-	6.0	2
Radionuclides			
Lead-210	-	-	-
Polonium-210	-	-	-
Radium-226/-228	5 pCi/L	-	-
Thorium-230	-	-	-
Uranium (U-234/-238)	30 pCi/L ^a (0.044 mg/L)	0.03 ^d mg/L	0.1 ^d mg/L

^aExceeded in plume wells.

^bEqual 10 mg/L nitrate as nitrogen.

^cUnder review.

^dProposed values, under review; expected revision 1995.

8.5 RECOMMENDATIONS

The proposed ground water standards consisting of MCLs or background concentrations are generally sufficient to protect human health and the environment. However, in some cases, a risk assessment may identify site-specific factors that suggest these standards are too restrictive, there may be no potential for exposure, and a less restrictive alternate concentration limit (ACL) may be acceptable. In other cases, the standards may not be sufficiently protective (e.g., if many contaminants are near the MCL with additive or synergistic adverse health effects).

At Lakeview, no permanent physical barriers exist to prevent access to contaminated ground water at the uranium processing site. Therefore, ACLs are not likely to be justified for those constituents with MCLs. However, for constituents that exceed background levels and do not have MCLs, this risk assessment suggests that background levels are more restrictive than necessary. This includes contaminants that were screened on the basis of their concentrations falling within nutritional levels (e.g., potassium and zinc) and other contaminants, such as aluminum, magnesium, and strontium, that were demonstrated to be at concentrations well below adverse health effect levels. ACLs should be sought for these contaminants.

Because no permanent physical barriers exist, measures should be implemented to restrict access to contaminated ground water to protect human health and the environment. Three chemicals are reported at concentrations that could potentially pose a health risk if ground water were to be used. The levels of manganese reported in ground water, if ingested, could produce Parkinson-like symptoms. The levels of sodium and sulfate could present a serious health risk if contaminated ground water at this site were used. The levels of sodium within the contaminant plume are reportedly within the ranges causing hypertension and kidney dysfunction. The sulfate levels reported cause severe diarrhea in infants. This situation (i.e., levels of manganese, sodium, and sulfate within the range of reported adverse health effects) suggests that it is advisable to implement institutional controls as soon as possible, regardless of the remedial action strategy pursued.

Most site characterization activities conducted to date at the former Lakeview processing site were intended to provide sufficient preliminary site characterization information to design and implement a surface remediation plan (for relocating the residual radioactive materials to the disposal site). The results of the hydrogeologic testing were not intended to provide a conclusive hydrogeologic and geochemical characterization of the former Lakeview processing site. There are no ground water sampling points between the former processing site and closest downgradient residences to verify the presence or absence of impacts. Additional site characterization activities should be conducted during the ground water compliance phase of the UMTRA Project. These site characterization activities may include additional aquifer performance

testing, installation and sampling of additional monitor wells (specifically between the former processing site and the downgradient residences), geochemical analyses of saline soil, or geophysical analyses. Data collected at the Lakeview site indicate that geophysical analysis is a viable method of data collection, and the data that can be obtained may eliminate the need to install additional monitor wells to fill data gaps.

In the near term, it is recommended that selected DOE monitor wells continue to be monitored for the following chemicals: arsenic, boron, manganese, nickel, sodium, sulfate, and uranium. Selected wells should be monitored for polonium-210 as ground water chemistry indicates the polonium-210 could be a result of naturally-occurring radon, and efforts should be made to identify the source of the polonium-210. Additionally, it is recommended that the presence of antimony, bromide, and cobalt be monitored in the ground water. Antimony was eliminated as a contaminant of potential concern because the reported levels fell within the range of geothermal background, but high levels of antimony have been measured in lysimeters. Bromide and cobalt data suggest an elevation above background concentrations, but too few rounds of analytical results are available to statistically evaluate whether this elevation exists.

As future ground water work plans are developed, additional characterization may be warranted to further evaluate surface water and sediment conditions of the water bodies in the vicinity of the Lakeview site and potential ecological receptors near the site.

9.0 LIST OF CONTRIBUTORS

The following individuals contributed to the preparation of this report.

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