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UMTRA GROUND WATER PROJECT

BASELINE RISK ASSESSMENT OF GROUND WATER CONTAMINATION AT THE URANIUM MILL TAILINGS SITE NEAR RIVERTON, WYOMING

September 1995

Prepared by the

U.S. Department of Energy
Albuquerque, New Mexico

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September 1995

Prepared for
U.S. Department of Energy
Environmental Restoration Division
UMTRA Project Team
Albuquerque, New Mexico

Prepared by
Jacobs Engineering Group Inc.
Albuquerque, New Mexico

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CITIZENS' SUMMARY

INTRODUCTION

The Uranium Mill Tailings Remedial Action (UMTRA) Project consists of two phases: the Surface Project and the Ground Water Project. At the UMTRA Project site near Riverton, Wyoming, Surface Project cleanup occurred from 1988 to 1990. Tailings and radioactively contaminated soils and materials were taken from the Riverton site to a disposal cell in the Gas Hills area, about 60 road miles (100 kilometers) to the east. The surface cleanup reduces radon and other radiation emissions and minimizes further ground water contamination.

The UMTRA Project's second phase, the Ground Water Project, will evaluate the nature and extent of ground water contamination at the Riverton site that has resulted from the uranium ore processing activities. Such evaluations are used at each site to determine a strategy for complying with UMTRA ground water standards established by the U.S. Environmental Protection Agency (EPA) and if human health risks could result from exposure to ground water contaminated by uranium ore processing. Exposure could hypothetically occur if drinking water were pumped from a well drilled in an area where ground water contamination might have occurred. Human health and environmental risks may also result if people, plants, or animals are exposed to surface water that has mixed with contaminated ground water.

Risk assessment is the process of describing a source of contamination and showing how that contamination may reach people and the environment. The amount of contamination people and the environment are exposed to is calculated and used to characterize possible health or environmental effects that may result from this exposure.

This document is a risk assessment report, the first site-specific document prepared for the Riverton site under the Ground Water Project. What follows is an evaluation of the current and possible future impacts that exposure to ground water contaminated by uranium ore processing might have upon public health and the environment. Further site characterization and the conclusions reached in this evaluation will be used to determine the measures needed to protect human health and the environment and to comply with EPA standards.

RISK SUMMARY

Because no one is drinking site-contaminated ground water, no human health risks are currently associated with the affected ground water. Based on existing data, no human or livestock health problems are expected from watering livestock or crops with ground water from two shallow private wells near the site. This favorable risk situation will continue if land and water use near the site does not change. Changes in land use may or may not create future risks. Therefore, when any specific land uses are determined for the site area, these uses should be evaluated to identify potential health and environmental risks that might result from the contaminated ground water or surface water.

Although site-affected ground water discharges to the surface, the site impact has not yet been determined. Vegetation growing over the contaminated shallow ground water may or

may not be taking up contaminants through roots extending into the aquifer. Contaminated ground water also discharges into the Little Wind River. However, the amount of possible contamination would not result in human health risks from swimming or fishing in the Little Wind River because of the river's great dilution capability.

To evaluate future risks, this risk assessment looks at the worst-case possibility of drinking water from a well placed in an area where ground water is most contaminated by site activities. Water from this hypothetical well is then evaluated as the sole source of drinking water for people and livestock or as a source of irrigation water for crops. In addition, this assessment considers the possible impacts this water may have on plants and animals. Under this hypothetical scenario, those people, livestock, plants, and wildlife could experience certain health problems; the worst ground water from the shallow aquifer should not be used; and the less contaminated water from other areas of the site should be evaluated prior to any use.

GROUND WATER QUALITY AND USES

Background ground water quality

Background ground water quality is the quality that ground water in the area would have if uranium milling had not occurred. Three aquifers are present in the vicinity of the Riverton site: an unconfined surficial aquifer at approximately 6 to 10 feet (ft) (2 to 3 meters [m]) below the land surface, a shallow semiconfined aquifer at a depth of about 20 to 25 ft (6 to 7.5 m), and a deep confined aquifer at a depth of about 60 to 80 ft (18 to 24 m). The deep aquifer is a source of potable domestic water, livestock and irrigation water, and municipal water in the Riverton site area.

The natural ground water from the shallow aquifers is not used for human consumption in the site area. Sulfates from natural sources in the unconfined surficial aquifer make the water taste and smell unpleasant. Ground water in the shallow aquifers does not move into the deep confined aquifer because of upward ground water pressure in the deep aquifer.

Although the natural ground water in the deep, confined aquifer contains sulfate and other minerals that exceed drinking water quality standards, many of the area residents reportedly are accustomed to its saline properties and find the water palatable.

Site-related ground water quality

The main contaminants in the unconfined surficial aquifer that resulted from the uranium processing are arsenic, lead-210, manganese, molybdenum, nickel, polonium-210, sulfate, thorium-230, uranium, and vanadium. The semiconfined aquifer also appears to have been affected by the uranium processing. However, levels of site-related contamination in the semiconfined aquifer are notably lower for all contaminants except molybdenum. Only one well completed in the surficial aquifer shows molybdenum concentrations similar to those in the unconfined surficial aquifer. Contamination in both aquifers apparently extends southeast and discharges to the Little Wind River, where it is quickly diluted. The river is likely a barrier for the contamination, which may discharge to the surface in the form of wetlands.

HUMAN HEALTH RISK ASSESSMENT

Methods

The risk assessment process begins by identifying the ground water contaminants that have resulted from uranium ore processing. This is done by comparing water quality data from wells drilled on the site to water quality data from background wells. The next step is to examine possible human health problems that could result from drinking the ground water containing these contaminants, eating meat and drinking milk from cattle that have drunk the ground water, eating domestic garden produce irrigated with the water, incidental ingestion of water and sediment while swimming in the Little Wind River, and consumption of fish from the river.

Both current and possible future human health risks are evaluated. To evaluate current risks from the site-affected ground water, it is first necessary to determine whether anyone is currently drinking the ground water. To evaluate possible future risks, it is assumed both that a drinking water well is drilled into the most contaminated area under the site in the unconfined surficial aquifer and that people are getting all their drinking water from this well. To evaluate possible future risks from consumption of meat and milk from cattle that had drunk the water and from eating vegetables from a domestic garden irrigated with the ground water, it is assumed that ground water from the unconfined surficial aquifer is the sole source of livestock and irrigation water.

Noncarcinogenic health risks were evaluated for children because the ratio of contamination to body weight is greatest for children. Therefore, children 1 to 10 years old would be the most likely group to experience health problems from drinking contaminated ground water. Infants (0 to 1 year) generally drink less tap water than children, but they are sensitive to contaminants like sulfate. For cancer, which takes a long time to develop, risks are evaluated for adults (age group 11 years and up). Health risks associated with eating the meat and drinking the milk from cattle watered with the ground water, eating vegetables from domestic gardens irrigated with ground water, and swimming and fishing in the Little Wind River are also evaluated.

The seriousness of possible health effects varies for several reasons. The levels of contaminants in ground water vary from one well sampling round to the next. People vary in body weight and water consumption and in their reactions to chemical exposure. This risk assessment considers these differences whenever possible and presents the most current information on the types of health effects that may result from hypothetical exposures.

Results

No drinking water wells exist in the unconfined surficial aquifer at or near the Riverton site. Because no one uses the ground water for drinking or other domestic purposes, there are currently no health problems from it. Using water from the deep confined aquifer for drinking is safe because upward pressure between the two aquifers keeps contamination out of the deep confined aquifer. This situation will continue if water use near the site does not change and the integrity of the deep wells is maintained. However, two shallow

livestock wells northeast and east of the most contaminated portion of the unconfined aquifer are sometimes used to water livestock or irrigate crops.

It is unlikely that people will use ground water from the surficial aquifer for drinking in the future because the water is poor and the yield of the shallow aquifers is low. Generally better quality water is available from a deep confined aquifer at depths greater than 300 ft (90 m). Other residents choose to haul drinking water from other local sources. Risks, if any, associated with future changes in land and water use need to be evaluated at the time they are identified.

Based on current concentrations of contaminants found in the most contaminated wells, there are possible health risks if people were to start drinking this ground water. Table CS-1 provides information on the nature of possible health problems that could be expected.

Table CS-1 Human health effects from hypothetical future use of contaminated ground water from the Riverton site as drinking water

Contaminant	Possible health problems associated with drinking water from surficial aquifer	
	Short-term effects	Long-term effects
Arsenic	None	Excess lifetime cancer risk exceeds the maximum recommended by EPA as acceptable; other health problems not expected.
Manganese	None	Mild neurological symptoms include memory loss, irritability, and muscle rigidity.
Molybdenum	None	Mild effects include copper deficiency anemia.
Sulfate	Severe diarrhea, particularly in infants	Unknown.
Uranium	None	Health problems not expected from chemical toxicity; excess lifetime cancer risk exceeds the maximum recommended by EPA as acceptable.

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

Note that only the people who drink all their water from a well placed in the most contaminated portion of the unconfined surficial aquifer (a small area) would be expected to experience the health problems described in Table CS-1. The table therefore provides the upper limit of possible risks; real future risks would probably be lower.

The most important possible health hazard in this ground water is sulfate. Sulfate ingestion in the amounts in this ground water, particularly by infants, could cause severe

diarrhea in a short time. However, high sulfate levels make water taste and smell unpleasant, which could discourage use.

Less severe health problems could result from the other contaminants if people drank ground water from the most contaminated portion of the unconfined surficial aquifer for a long time. The manganese levels could affect the nervous system and the molybdenum levels could result in copper deficiency. These health problems could be greater in people with kidney problems. In addition, the increased lifetime cancer risk from arsenic and uranium may exceed the maximum acceptable value recommended by the EPA.

Drinking ground water from the semiconfined aquifer could be associated with health problems caused by the levels of manganese, sulfate, and molybdenum. Compared to the possible future health risks from drinking contaminated ground water from the unconfined surficial aquifer, notably lower health risks would be expected from the levels of sulfate and manganese in the semiconfined aquifer. However, the levels of molybdenum in the ground water from the semiconfined aquifer could result in the same type and degree of health problems as those expected from drinking ground water from the unconfined surficial aquifer. These health problems could result in copper deficiency anemia after drinking the water over a long time.

Human health would not be affected by eating meat or drinking milk from cattle watered with the most contaminated ground water from either shallow aquifer and from eating garden produce irrigated with this water. Furthermore, available data indicate no health problems would be expected from swimming in the Little Wind River or eating fish caught from the river.

ECOLOGICAL AND LIVESTOCK RISK ASSESSMENT

Methods

The ecological risk assessment presented in this document is a screening level assessment that relies on limited environmental sampling and data from literature. In the field of ecological risk assessment, many uncertainties result from limited data on the impact of chemicals and chemical mixtures on plants and animals. Furthermore, due to limited environmental sampling at the Riverton site, there is no information on the site-specific levels of contamination in plants or animals.

Therefore, the possible effects of the contaminants on grazing livestock and agricultural crops are evaluated by comparison:

- The contaminant concentrations in the ground water, surface water, and sediment that could be affected by contaminated ground water are compared to available guidelines from regulatory agencies and literature values.

- Surface water and sediment are evaluated from different areas of the Little Wind River and of the wetlands in the site vicinity.

- Ground water from the unconfined surficial aquifer is also evaluated, assuming plant roots extend into the aquifer and take up the most contaminated ground water. This risk assessment also uses this water as a sole source of drinking water for livestock and for crop irrigation, and examines possible impacts on plants.

Livestock and agricultural results

Currently, no health risks to livestock and crops are associated with the contaminated ground water in the unconfined surficial aquifer because livestock are not watered with the most affected ground water, and agricultural crops are not irrigated with this ground water. Based on existing data, livestock could safely drink water from the two shallow private wells in the site vicinity and water from these wells could also be safely used for irrigation.

In the future, if the ground water from the most contaminated portion of the aquifer were the sole source of drinking water for livestock, sulfate levels could cause diarrhea in these animals. Also, this water is not recommended for irrigation because sulfate, manganese, and molybdenum in the water could harm crops.

Ecological results

No apparent ecological risks exist to plants at harvestable maturity that may have roots in contact with soil saturated with the most contaminated ground water in the unconfined surficial aquifer. However, limited environmental sampling and available guidelines from regulatory agencies were not sufficient to fully evaluate the possible long-term impacts of the affected ground water or surface water on plants or animals.

CONCLUSIONS

Ground water contamination beneath the Riverton site is limited to the unconfined surficial aquifer and the semiconfined aquifer; the Little Wind River is likely a barrier for the contamination. No drinking water wells are placed in these aquifers on or near the site. Although there are two shallow livestock wells near the site that might be affected by the site activities, the water from these wells is safe for both livestock and crops. Therefore, currently no health risks are associated with site ground water. Also, based on limited sampling, no human health risks are associated with ground water discharging to the Little Wind River.

This risk assessment has determined that in the future, if contaminated ground water were used as a sole source for drinking, livestock and some people could experience health problems; it could also be harmful to plants irrigated with the contaminated ground water. Therefore, contaminated ground water should not be used for irrigation or for drinking water.

The Riverton site evaluation is ongoing and will include further characterization of ground water quality, levels, and movement. This and future investigations will be used to determine how to deal with the contaminated ground water. In addition, if specific plans for land use at the site are developed in the future, the possible risks should be evaluated.

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LIST OF ACRONYMS AND ABBREVIATIONS

Acronym	Definition
ACL	alternate concentration limit
BCF	bioconcentration factor
BIA	Bureau of Indian Affairs
DHHS	U.S. Department of Health and Human Services
DOE	U.S. Department of Energy
EA	environmental assessment
EPA	U.S. Environmental Protection Agency
ER-L	effects range-low
FWQC	Federal Water Quality Criteria
HEAST	Health Effects Assessment Summary Tables
ICRP	International Commission on Radiological Protection
IRIS	Integrated Risk Information System
LOAEL	lowest-observed-adverse-effect level
MCL	maximum concentration limit
MEQ/L	milliequivalent per liter
MSL	mean sea level
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
PEIS	program environmental impact statement
RfD	reference dose
SF	slope factor
SQC	sediment quality criteria
SS	sandstone
TDS	total dissolved solids
UMTRA	Uranium Mill Tailings Remedial Action
UMTRCA	Uranium Mill Tailings Radiation Control Act
USC	United States Code
USGS	United States Geological Survey
WDEQ	Wyoming Department of Environmental Quality
WHO	World Health Organization

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1.0 INTRODUCTION

The purpose of this baseline risk assessment is to determine whether ground water contamination at the Riverton, Wyoming, uranium mill tailings site could adversely affect human health or the environment. The Riverton site is one of 24 designated uranium mill tailings sites undergoing remediation in accordance with the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA) (42 USC §7901 *et seq.*) under the oversight of the U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project. The remediation of surface contamination at the Riverton processing site was completed in 1990 with relocation and stabilization of the tailings in a disposal cell near the Gas Hills plant.

This risk assessment is a baseline assessment in that it describes existing ground water conditions at the site. This document evaluates the potential for public health or environmental risk related to ground water contamination that may need attention before further site characterization. The risk assessment is based on available ground water data from wells and surface water at the processing site. Major exposure pathways relevant to these uses have been identified and examined for this risk assessment.

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

- *Data evaluation.*
  - Combining existing data from various site investigations.
  - Comparing sample results with background and tailings source data.
  - Selecting appropriate chemical data for the risk assessment.
- Exposure assessment.
  - Characterizing exposure settings.
  - Identifying exposure pathways.
  - Quantifying exposure.
- Toxicity assessment.
  - Identifying toxicity values.
  - Evaluating noncarcinogenic effects.
  - Evaluating carcinogenic effects from radionuclides and chemical carcinogens.
- Public health risk characterization.
  - Comparing toxicity ranges to predicted exposure ranges.
  - Combining risks across exposure pathways and multiple contaminants.
  - Characterizing uncertainties.

- Environmental risk assessment.

    - Characterizing potential biota exposure pathways.
    - Identifying potential ecological receptors.
    - Evaluating environmental risk qualitatively.

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

This risk assessment will support decisions made for the UMTRA Ground Water Project. The DOE is authorized to remediate ground water under the 1988 UMTRCA Amendments Act (42 USC §7922 *et seq.*) and will determine site-specific ground water compliance strategies for each site. This baseline risk assessment provides information to assist in determining the site-specific ground water compliance strategy for the Riverton site.

2.0 SITE DESCRIPTION

The Riverton site is in Fremont County, Wyoming, 2 miles (mi) (3 kilometers [km]) southwest of the city of Riverton (Figure 2.1). The site is on private land, within the boundaries of the Wind River Indian Reservation (Arapaho and Shoshone Indian tribes).

2.1 SITE BACKGROUND

The mill at the Riverton tailings site was constructed in 1958 and shut down in 1963. Fremont Minerals, Inc., initially operated the mill, treating a variety of uranium ores from the surrounding area. The company name subsequently was changed to Susquehanna-Western, Inc. During its 4 years of operation, the mill processed approximately 900,000 tons (800,000 metric tons) of ore. Waste solids from the uranium ores were transferred to a tailings pile. This rectangular pile covered 70 acres (ac) (30 hectares [ha]) and contained approximately 1 million cubic yards (yd^3) (800,000 cubic meters [m^3]) of tailings. The mill property also included a sulfuric acid plant, which is still in operation.

Between 1988 and 1990, the uranium mill tailings pile and contaminated soils were removed from the site. Approximately 1.8 million yd^3 (1.4 million m^3) of contaminated material were relocated to a disposal cell in the Gas Hills area. The excavated areas were backfilled with clean fill; the finished land surface was graded to form a crown and planted with rye grass.

2.2 CLIMATE

The Riverton area is arid, due largely to the influence of the Wind River mountain range to the west that hinders the passage of moisture. The average annual precipitation in Riverton from 1951 to 1980 was 8 inches (20 centimeters [cm]). The greatest amount of precipitation and ground water infiltration occurs in April, May, and June in the form of late spring snows, snow melt, and showers.

The Riverton area experiences wide and sudden changes in temperature because of the high elevation (approximately 4940 feet (ft) (1500 meters [m]) and the presence of dry air. Riverton is influenced both by cold masses from Canada and by prevailing warm, westerly winds. The highest and lowest temperatures recorded in Riverton from 1951 to 1980 were 104 degrees Fahrenheit (°F) (40 degrees Celsius [°C]) and -46°F (-43°C), respectively. During the same period, an average of 207 days per year had minimum temperatures less than or equal to 32°F (0°C), and an average of 37 days per year had maximum temperatures of 90°F (32°C) or greater.

2.3 SURFACE WATER FLOW PATTERNS

The Riverton site is on a nearly level surficial terrace in the Wind River basin, approximately 13,000 ft (4000 m) upstream of the confluence of the Wind and Little Wind Rivers. The Wind River is approximately 4000 ft (1000 m) north of the site and the Little Wind River is approximately 3000 ft (900 m) southeast of the

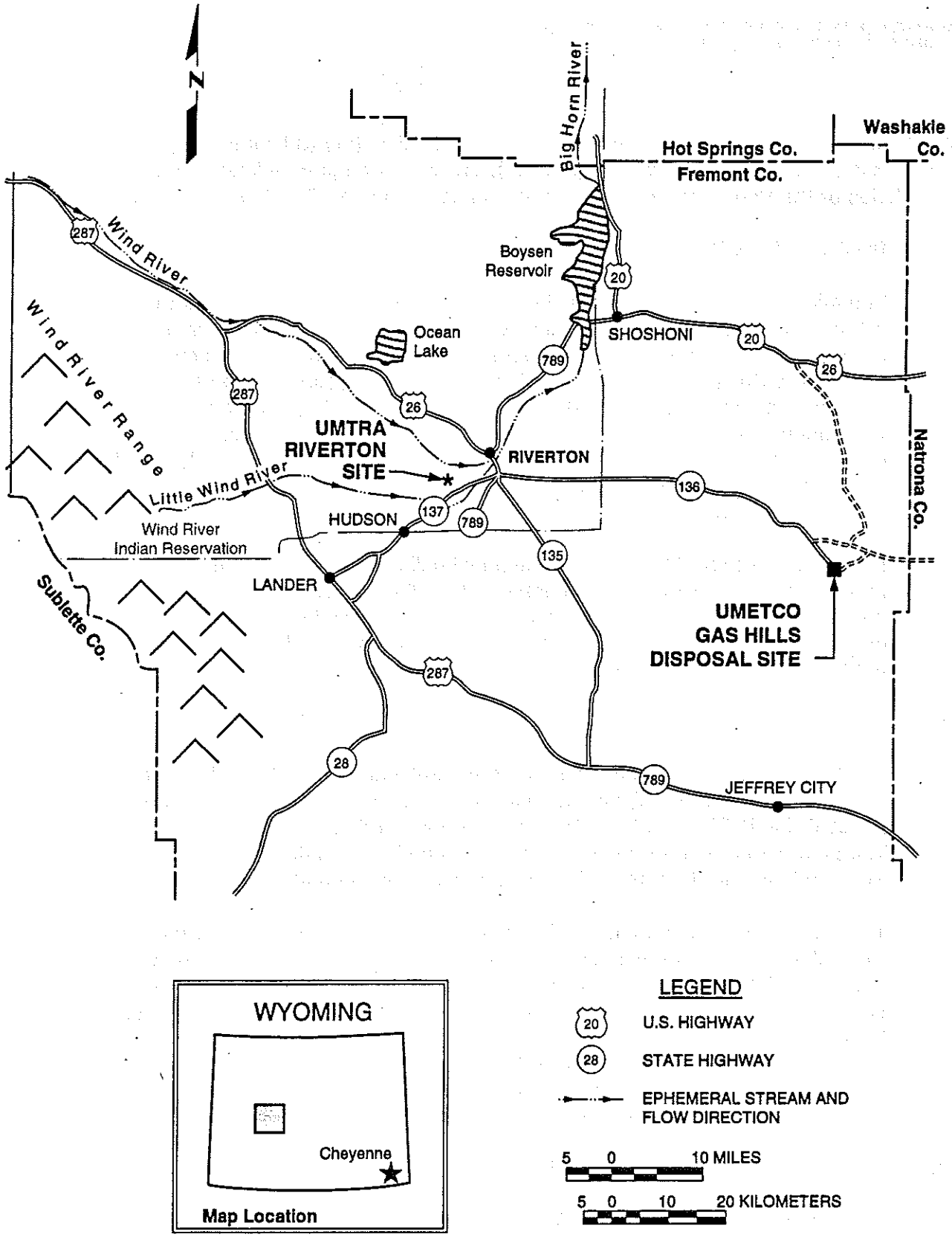


FIGURE 2.1
RIVERTON, WYOMING, SITE LOCATION MAP

site (Figure 2.2). A wetland area lies approximately 1000 ft (300 m) north of the northern property line.

This stream joins the Wind River approximately 5500 ft (1700 m) east of the northeastern corner of the site.

A system of irrigation canals along the northern and eastern sides of the property carry water from the Wind River, discharging into the wetlands area east of the site. The irrigation system operates from June to October. The flow in the canal where it enters the site and where it exits the site was approximately 1 cubic foot (ft^3) (0.028 cubic meters [m^3]) per second in June 1994.

A man-made, unlined drainage channel carries discharge water from the sulfur processing plant on the western side of the site. This channel joins the regional irrigation canal system and natural wetlands areas before passing under State Highway 137 where it joins the Little Wind River approximately 3500 ft (1100 m) due south of the property. Another wetland area is east of the site. Drainage from this wetland passes under Highway 137 approximately 2500 ft (760 m) east of the southeast corner of the property. A former meander in the Little Wind River has been cut off, leaving an oxbow lake southeast of the site.

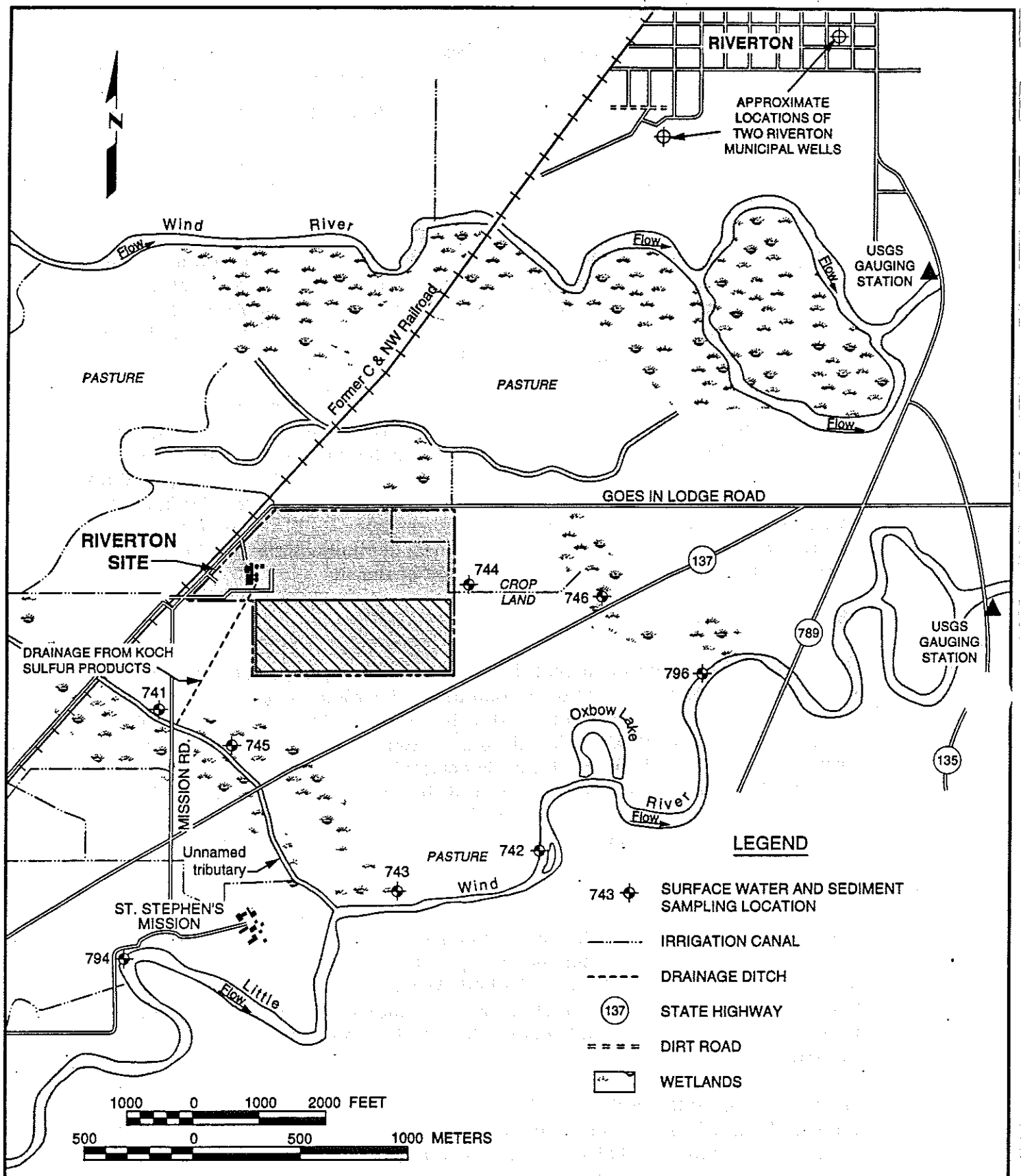
2.4 HYDROGEOLOGY

This baseline risk assessment considers the upper five hydrogeologic units in this area. They are, in descending order from the land surface, an unconfined surficial alluvial and sandstone aquifer 15 to 20 ft (5 to 6 m) thick; a leaky shale aquitard 5 to 10 ft (2 to 3 m) thick; a semiconfined sandstone aquifer 15 to 30 ft (5 to 9 m) thick; a second, more impermeable shale aquitard approximately 40 ft (12 m) thick; and a confined sandstone aquifer at least 50 ft (17 m) thick (Figure 2.3). The unconfined surficial aquifer, semiconfined sandstone aquifer, and confined sandstone aquifer are discussed in more detail in the following paragraphs.

2.4.1 Unconfined surficial aquifer

The unconfined surficial aquifer consists of 15 to 20 ft (5 to 6 m) of alluvial sand and gravel underlain by a discontinuous layer of sandstone. The alluvium is continuous from the Wind River to the Little Wind River. The sandstone layer exists both north and south of the former tailings pile, but pinches out and is absent for approximately 1500 ft (500 m) south of the southern edge of the pile. There is no aquitard between the alluvial sediments and this sandstone layer.

The water table in the surficial aquifer in March 1993 ranged from approximately 7.5 ft (2.3 m) below the ground surface (elevation 4936 ft [1504 m] above mean sea level [MSL]) at the middle of the northern edge of the property (monitor well 728) to about 5.5 ft (1.7 m) below ground surface (elevation 4930 ft [1503 m], MSL) near the southeast corner of the site (monitor well 716). The ground water flow was east-southeast with a horizontal gradient of approximately 0.0024, as shown in Figure 2.4.



MODIFIED FROM THE RIVERTON WEST & ARAPAHOE 7.5 MIN. USGS QUADRANGLES TOWNSHIPS 1N AND 1S RANGE 4E.

FIGURE 2.2
SURFACE WATER FLOW PATTERN AND SAMPLING LOCATIONS
RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-SURF&SAMPLOCS

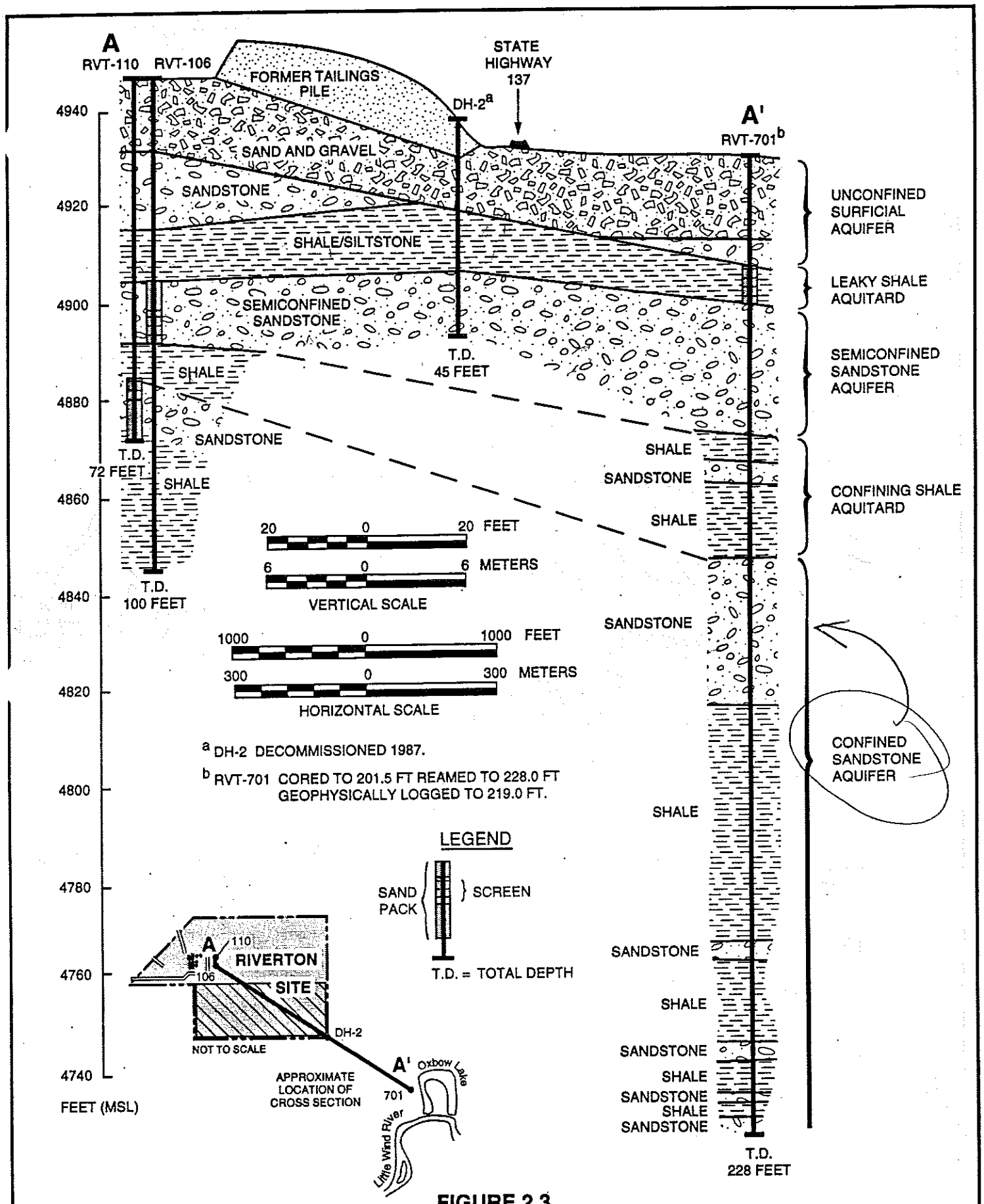


FIGURE 2.3
GEOLOGIC CROSS SECTION
RIVERTON, WYOMING, SITE

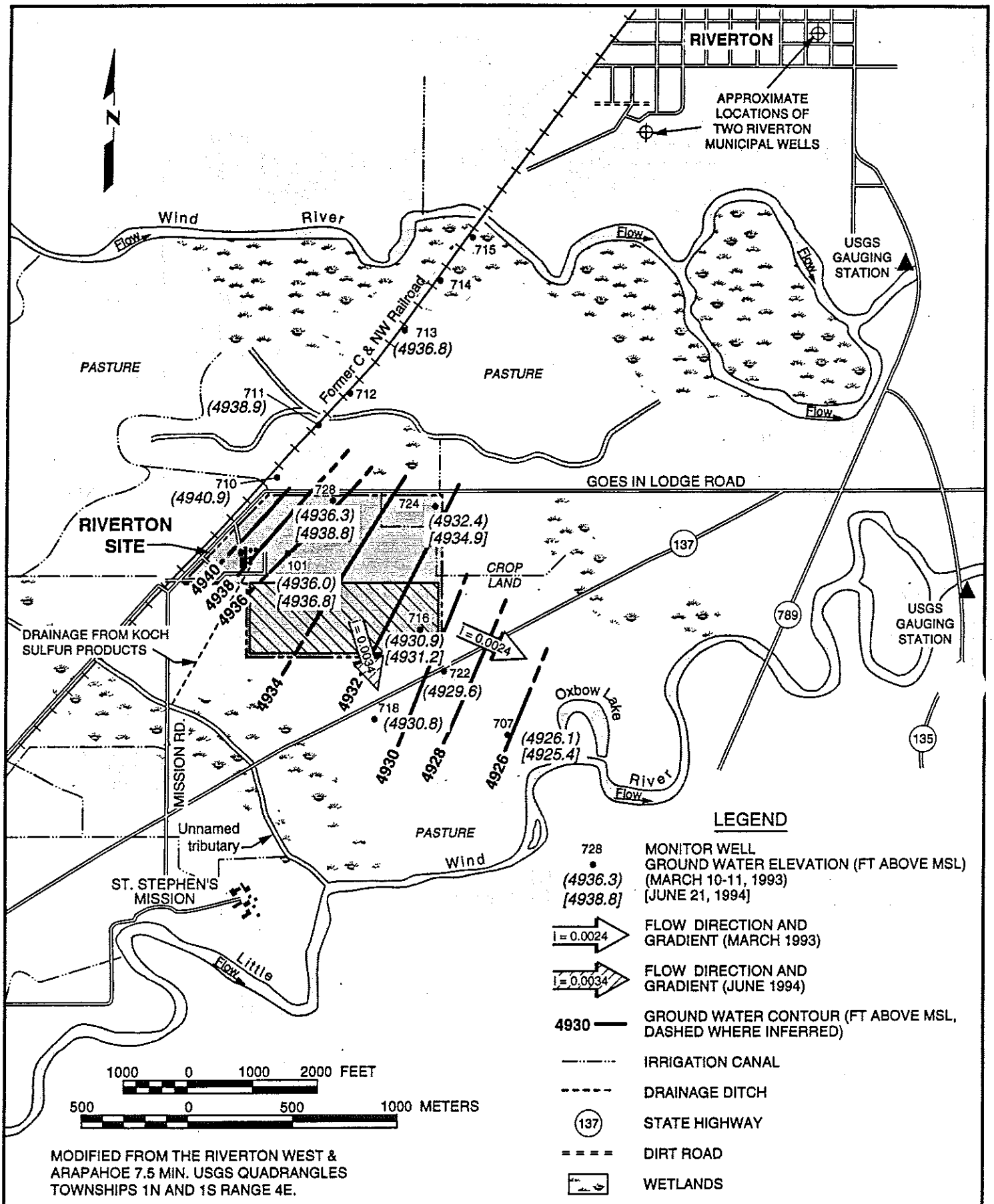


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FIGURE 2.4
GROUND WATER ELEVATIONS IN THE SURFICIAL AQUIFER RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-GWELEVSSURFAQUIF

In June 1994, the ground water level in both the surficial and semiconfined aquifers (monitor wells 728 and 727, respectively) at the northern edge of the site had risen 2 ft (0.6 m) above the March 1993 level (Table 2.1), while the water level in both aquifers near the middle and at the southeastern corner were about the same as in March 1993. The ground water flow in June 1994 was almost due south and the horizontal gradient beneath the site increased by about 30 percent to 0.0034 (Figure 2.4). The rise in the water table is interpreted to be in response to infiltration from the nearby irrigation canal. A similar rise was seen in a well completed in the semiconfined aquifer at the northwest corner of the property, indicating hydraulic connection between the surficial and semiconfined aquifers in this area. Infiltration from the canals may also form hydraulic boundaries limiting the movement of ground water to the east during the irrigation season.

Ground water in the surficial aquifer discharges predominantly to the Little Wind River along approximately 1 mile (1.6 km) of its course, including into the newly formed oxbow lake. When the irrigation canals are not in use, some of the ground water in the surficial aquifer may also discharge into the wetlands to the east. When the irrigation canals are in use, the water level in the wetlands may be high enough to reverse the flow direction and recharge the surficial aquifer.

A 24-hour, 5-gallon (gal) (19-liter [L]) per minute aquifer pumping test was conducted in the surficial aquifer in monitor well 112 near the northwest corner of the former tailings pile as part of the surface remediation environmental assessment (EA) (DOE, 1987). Measurable drawdown was not observed in either the semiconfined or confined aquifers during the test. The test demonstrated a hydraulic conductivity of approximately 56 ft (17 m) per day. Using Darcy's Law and estimated effective porosity of 0.30 (DOE, 1987) for the alluvium and unconfined sandstone and the March 1993 gradient of 0.0024, the calculated average linear ground water flow velocity is approximately 160 ft (50 m) per year (DOE, 1995). During the irrigation season, the steeper gradient of 0.0034 could increase the flow velocity to 230 ft (80 m) per year (DOE, 1995).

The calculated rate of ground water flow predicts that approximately 12,600 ft^3 (350 m^3) of ground water will enter the river each day along a 1-mi (1.6-km) section of the Little Wind River's course (DOE, 1995). The volume equals approximately 0.15 ft^3 (4×10^{-3} m^3) per second. For comparison, the average annual flow in the river over the last 41 years has been 579 ft^3 (16.4 m^3) per second and the historic minimum flow was 41 ft^3 (1.16 m^3) per second on 7 August 1960 (USGS, 1993). The ground water contribution is a minimal part of the total flow in the Little Wind River.

2.4.2 Semiconfined aquifer

A semiconfining shale unit underlies the surficial aquifer. This leaky aquitard ranges in thickness from 5 to 10 ft (2 to 3 m). A semiconfined sandstone unit underlies this shale layer. The top of the semiconfined sandstone is approximately 20 to 25 ft (6 to 7.6 m) below land surface. This unit ranges in thickness from 15 to 30 ft (5 to 9 m) and is continuous throughout the Riverton site. A 24-hour, 18-gal

Table 2.1 Ground water monitor well information, Riverton, Wyoming, site

Well ID	Ground elevation	Borehole depth	Borehole diameter	Casing elevation	Casing diameter	Screened interval	Filter pack	Aquifer
101	4946.2	17.5	6.0	4946.58	2.0	10.5-15.5	5.5-17.5	Surficial
104	4945.3	15.5	6.3	4945.90	2.0	8.5-13.5	6.5-15.5	Surficial
108	4946.2	56.0	6.0	4946.02	2.0	48.5-53.5	35.5-56.5	Semiconfined SS
110	4946.2	72.0	6.0	4946.44	2.0	61.3-66.5	59.0-72.0	Confined SS
705	4930.1	50.0	10.0	4930.80	6.0	38.0-48.0	35.5-50.0	Semiconfined SS
706	4931.1	21.5	6.0	4932.00	2.0	14.5-19.5	12.8-21.5	Surficial
707	4930.4	18.0	6.0	4931.00	2.0	9.8-14.8	7.5-16.8	Surficial
709	4930.2	111.0	10.0	4930.70	6.0	85.0-105.0	84.0-111.0	Confined SS
710	4947.2	20.0	6.0	4947.90	2.0	11.2-16.2	8.0-20.0	Surficial
711	4943.5	21.5	6.0	4944.50	2.0	10.8-15.8	6.0-21.5	Surficial
712	4943.5	19.5	6.0	4944.50	2.0	10.6-15.6	10.0-19.5	Surficial
716	4936.4	12.5	6.0	4939.12	2.0	7.5-12.5	5.5-12.5	Surficial
717	4936.4	50.0	6.0	4938.80	2.0	37.5-47.5	29.0-49.5	Semiconfined SS
718	4937.0	18.0	6.0	4937.18	2.0	13.0-18.0	10.0-18.0	Surficial
719	4936.8	40.0	6.0	4936.94	2.0	28.0-38.0	23.0-40.0	Semiconfined SS
720	4937.9	10.5	6.0	4940.46	2.0	5.5-10.5	3.5-10.5	Surficial
721	4937.9	49.0	6.0	4940.47	2.0	37.0-47.0	27.0-49.0	Semiconfined SS
722	4935.2	18.0	6.0	4935.35	2.0	6.0-16.0	4.0-18.0	Surficial
723	4935.0	49.0	6.0	4935.26	2.0	35.5-45.5	31.0-47.5	Semiconfined SS
724	4939.4	16.0	6.0	4941.36	2.0	11.0-16.0	6.0-16.0	Surficial
725	4939.4	38.0	6.0	4941.36	2.0	24.5-34.5	19.5-36.5	Semiconfined SS
726	4939.5	133.0	6.0	4942.00	2.0	121.0-131	80.0-133.0	Confined SS
727	4949.5	40.0	6.0	4951.69	2.0	27.0-37.0	21.5-39.0	Semiconfined SS
728	4943.9	24.0	6.0	4946.01	2.0	12.0-22.0	9.0-24.0	Surficial
729	4932.1	17.0	6.0	4932.07	2.0	9.0-14.0	8.0-14.0	Surficial
730	4932.5	40.0	6.0	4932.48	2.0	28.0-38.0	21.0-40.0	Semiconfined SS

NOTE: Elevations in feet relative to mean sea level; depths in feet; diameters in inches.

SS - sandstone.

(68-L) per minute aquifer test in the semiconfined sandstone yielded a calculated hydraulic conductivity of about 30 ft (10 m) per day (DOE, 1995).

The shale aquitard does not appear to completely separate the two units. The piezometric heads in the surficial and semiconfined aquifers are nearly identical in most areas (compare Figures 2.4 and 2.5), suggesting a hydraulic connection.

Using an estimated effective porosity of 15 percent (typical of sandstones according to Freeze and Cherry, 1987), a hydraulic conductivity of 30 ft (10 m) per day, and the March 1993 gradient of 0.0024, it is projected that the ground water will move with an approximate velocity of 175 ft (60 m) per year (DOE, 1995). The ground water in the semiconfined sandstone also appears to discharge into the Little Wind River, although additional data will be needed to confirm this. The rate of discharge from the semiconfined area is approximately 0.1 ft^3/s (3×10^{-3} m^3/s) (DOE, 1995).

2.4.3 Confined sandstone

Approximately 10 to 25 ft (3 to 8 m) of shale aquitard with discontinuous sandstone lenses underlies the semiconfined sandstone. The confined sandstone aquifer underlies the shale aquitard beginning 60 to 80 ft (18 to 24 m) below ground surface. The sandstone is at least 525 ft (160 m) thick (completion record for private well 436 in Table 2.2). Water level data from the monitor wells completed in the confined sandstone indicate ground water in this unit flows south with a gradient of approximately 0.002 (Figure 2.6). No aquifer tests were performed in the confined aquifer; therefore, ground water flow velocity cannot be calculated.

Water levels observed in the monitor wells do not conclusively define the vertical ground water gradient between the aquifers. For example, in the well cluster south of the site near the Little Wind River, the piezometric head in deep monitor well 703 (4928.31 ft [1503.14 m], MSL) exhibits an upward gradient from the lower confined sandstone aquifer to the shallower aquifers (4926.05 ft [1501.46 m], MSL) in monitor well 707). In contrast, the piezometric head in the nearby deep monitor well 709 (4924.65 ft [1501.03 m], MSL) exhibits a downward gradient even though it is screened at approximately the same depth. While vertical gradients between the surficial and semiconfined aquifers and the confined sandstone aquifer have not been conclusively defined, observation wells screened in the confined aquifer did not fluctuate during the aquifer test in the overlying semiconfined aquifer, indicating the two aquifers are not hydraulically connected.

2.5 LAND USE

Land in the immediate site vicinity is used predominantly for agriculture. Much of the area is pasture for cattle and horses, and the primary crop is hay (Figure 2.7) (DOE, 1987). Several residences surrounding the site have vegetable gardens (DOE, 1995). The St. Stephens Mission School, a contract school for the Bureau of Indian Affairs (BIA), is 0.6 mi (0.9 km) south of the site.

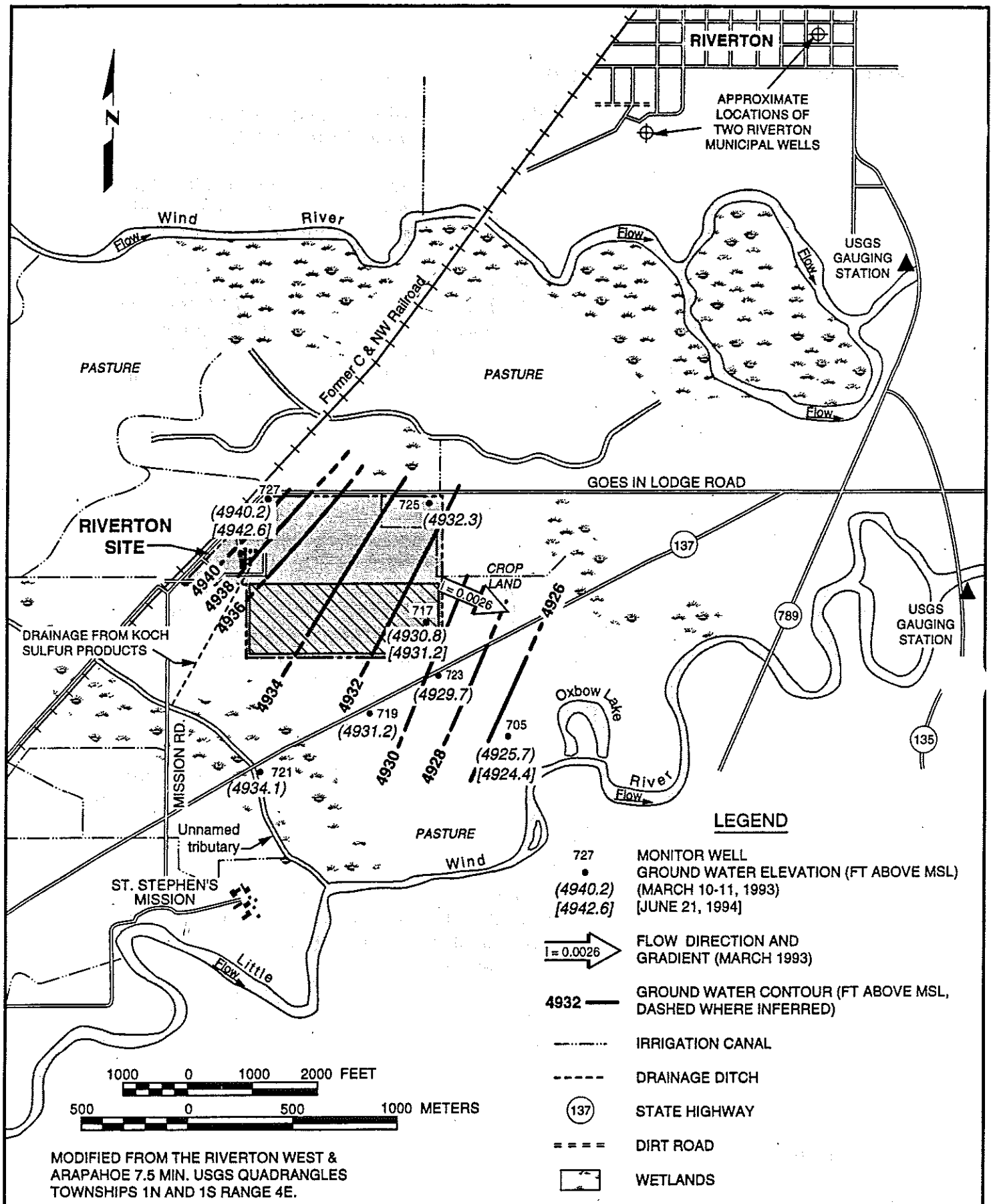

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FIGURE 2.5
GROUND WATER ELEVATIONS IN THE SEMICONFINED SANDSTONE AQUIFER RIVERTON, WYOMING, SITE

Table 2.2 Private well sampling, construction, and water use, Riverton, Wyoming, site

TAC ID	Date sampled	Total/casing depth (ft)	Aquifer	Water use
405	1981, 1983, 1984(2x) 1985, 1990, 1991(3x) 1992(2x), 1993	274/a	Confined SS bedrock	Potable
406	1981, 1990, 1991(3x) 1992(2x), 1993	350/a	Confined SS bedrock	Potable
410	1982, 1983, 1984(3x) 1990(2x), 1992(2x) 1993	100/a	Confined SS bedrock	Domestic
411	1985, 1988, 1990 1991(3x), 1992(2x) 1993	270/261	Confined SS	Domestic
417	1981, 1990, 1991 1992, 1993	360/a	Confined SS bedrock	Potable[a]
420	1981, 1983, 1984(3x) 1985, 1990(2x) 1991(3x), 1992(2x)	273/228	Confined SS bedrock	Potable
421	1981, 1985	200/a	Confined SS bedrock	Potable
423	1984, 1985, 1988 1990, 1991(3x) 1992(2x), 1993	290/a	Confined SS bedrock	Potable
430	1981, 1983, 1984(2x) 1985, 1990, 1991(2x) 1992(2x), 1993	284/320	Confined SS bedrock	Potable
431	1984, 1985, 1992 1993	approx. 15/a (installed with back-hoe)	Surficial	Stock
436	1982, 1991(3x) 1992(2x), 1993	525/a	Confined SS bedrock	Potable
440	1984, 1985(2x), 1988 1990(2x)	267/a	Confined SS bedrock	Potable[a]
441	1985	100/a	Confined SS bedrock	NA windmill
442	1994	405/a	Confined SS bedrock	Domestic
443	1994	397/356.5	Confined SS bedrock	Potable
444	1994	a/a	a	Unused
445	1994	35/a	Semiconfined SS bedrock	Stock

Table 2.2 Private well sampling, construction, and water use, Riverton, Wyoming, site (Concluded)

TAC ID	Date sampled	Total/casing depth (ft)	Aquifer	Water use
446	1994	410/370	Confined SS bedrock	Potable
448	1985	405/a	Confined SS bedrock	Potable
451	1994	360/338	Confined SS bedrock	Potable
452	1994	a	a	Potable
453	1994	a	a	Potable
460	1993	450/a	Confined SS bedrock	Potable
951	1988, 1992 (2x)	273/246	a	Potable[a]

[a]Information needs to be collected or confirmed. Where water use is not certain, potable use was assumed.

Notes: 1. See Figure 2.7 for well locations.
2. Water uses: Potable = Drinking and other uses.
Domestic = Bathing, washing dishes and other uses, but not drinking.
Stock = Watering livestock and irrigation, but not drinking or domestic.
Process = Industrial use.

NA - not available.
SS - sandstone.

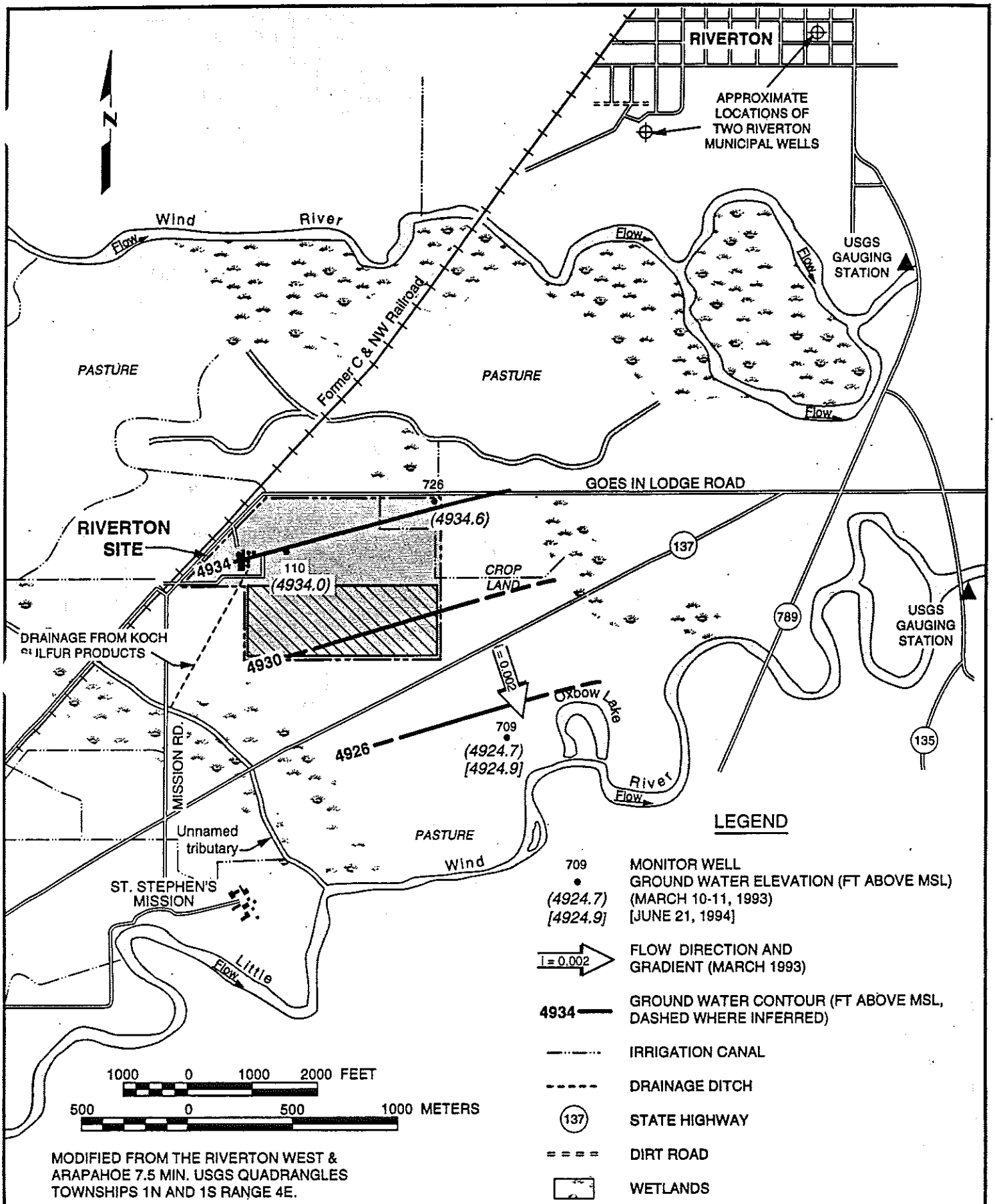

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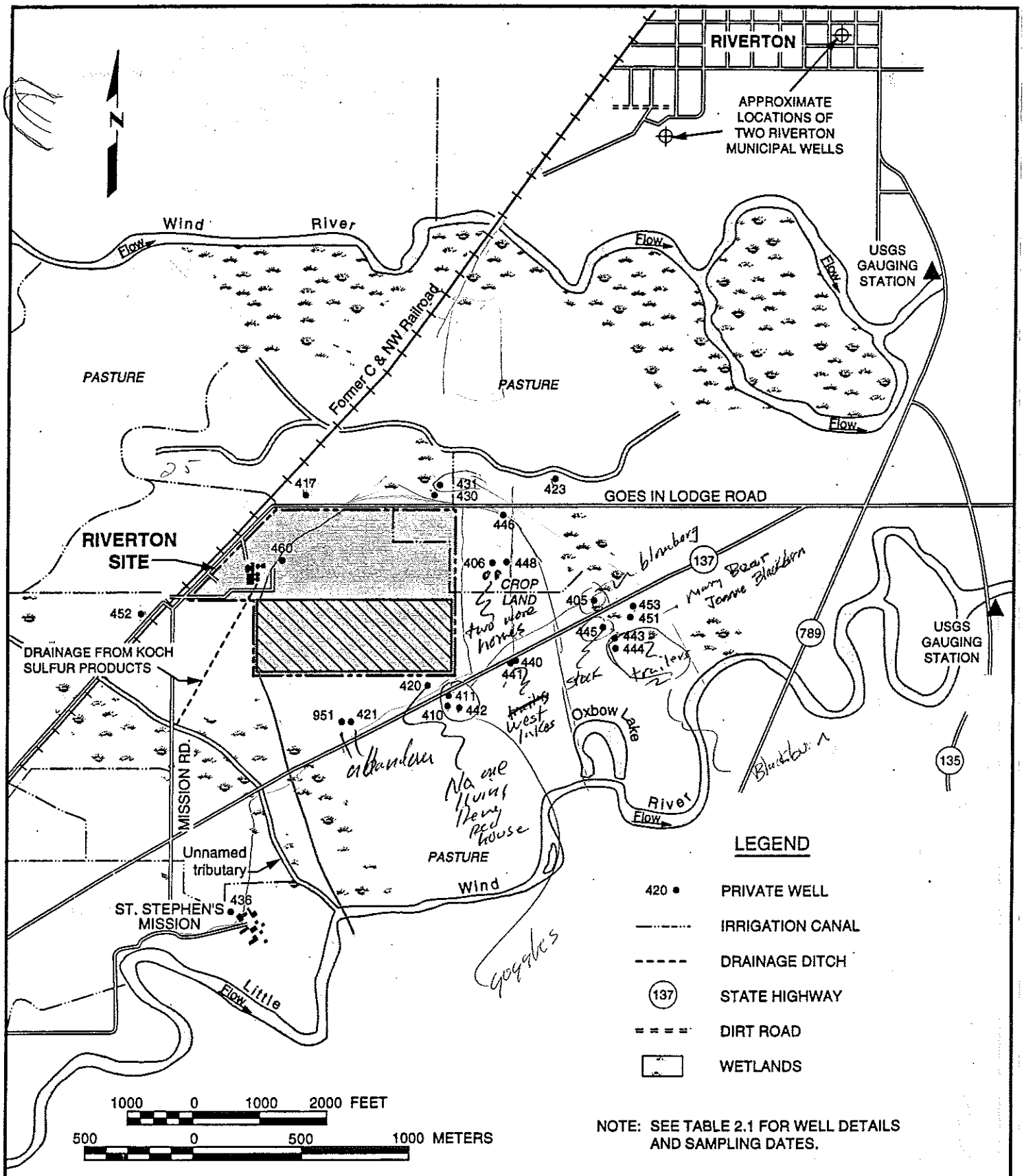
FIGURE 2.6
GROUND WATER ELEVATIONS IN THE CONFINED SANDSTONE AQUIFER RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-GWELEVS-CONF.



NOTE: SEE TABLE 2.1 FOR WELL DETAILS AND SAMPLING DATES.

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FIGURE 2.7
LAND USE AND PRIVATE WATER WELL LOCATION MAP
RIVERTON, WYOMING

MAC: SITE/RVT/BRA/BASE-LANDUSEMAP

The Arapaho and Shoshone Indian tribes own most of the land surrounding the tailings site, although there are scattered parcels of privately owned fee land as well (DOE, 1987). Indian tribal lands, administered by the Arapaho-Shoshone Joint Business Council, and Indian allotted lands, reserved for use by individuals, are also in the site vicinity. Most allottees live on the land rather than leasing it for other purposes (e.g., grazing). A tribal zoning ordinance administered by the Arapaho-Shoshone Joint Business Council governs area land-use planning and policies and controls all area development.

2.6 WATER USE

The Wind River is the main source of Riverton municipal water during the spring and summer (April through October). During the fall and winter, the city's only water source is its well system, which taps the confined aquifer of the Wind River Formation. Municipal water is not available to residents near the site. In the Riverton area, natural ground water in the confined aquifer (at depths greater than 100 ft [30 m]) is a sodium-sulfate type, with a high mineral content and high concentrations of total dissolved solids (TDS) (Durum, 1959). Consequently, the water contains minerals in excess of the recommended drinking water quality standards (federal and state of Wyoming maximum contaminant levels). Nevertheless, this water is used for drinking as well as other purposes. Many residents in the Riverton area have become accustomed to the saline properties of the water and find the water palatable (Durum, 1959). Locally, the water from some wells at depths of about 300 ft (90 m) is reported to have a moderately low mineral content but it is unsatisfactory for drinking because of the strong hydrogen sulfide odor and the precipitation of sulfur (Durum, 1959). The waters of both the Wind and Little Wind Rivers are used locally for irrigation and livestock watering.

Door-to-door surveys of the residences near the Riverton UMTRA Project site were completed in March 1993 and January 1994 (DOE, 1995). Well locations are shown on Figure 2.7; construction details are listed in Table 2.1. Almost all these wells are at depths greater than 100 ft (30 m), placing them in the confined sandstone.

Water for livestock, agricultural, and domestic use is drawn from wells completed in the confined aquifer. These wells are typically installed by the Indian Health Service at depths greater than 300 ft (90 m) to reach the higher-quality water from the deeper units. Some residences use well water for drinking; others choose to haul drinking water from other local sources. Two shallow private wells (wells 431 and 445 in Figure 2.7 and in Table 2.1) near the site are sometimes used to water livestock and for irrigation.

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3.0 EXTENT OF CONTAMINATION

Surface and ground water quality data were collected from 1983 to 1993 at the Riverton site (DOE, 1995). At present, more than 40 monitor wells are screened in the three uppermost water-bearing zones in the vicinity of the abandoned mill site. Surface water quality has been monitored from five locations upstream and downstream from the former mill site. Data collected from 1987 to 1993 are used in this risk assessment. Water samples were collected in accordance with applicable standard operating procedures described in the Albuquerque Operations Manual (JEG, n.d.).

Figure 3.1 shows the locations of current monitor wells. The wells that make up the monitor network and their completion information are listed in Table 2.1.

3.1 BACKGROUND WATER QUALITY SUMMARY

Background water quality is defined as the water quality that could be expected if uranium milling activities had not occurred. Because contamination was detected in the unconfined surficial aquifer and the semiconfined sandstone aquifer, this document discusses background ground water quality for both aquifers. Contamination has not been detected in the confined sandstone aquifer. Therefore, this aquifer is not further evaluated in this risk assessment.

3.1.1 Unconfined surficial aquifer

Background ground water quality for the unconfined surficial aquifer was determined using ground water collected from monitor wells 710 and 711. Background locations are north and upgradient from the former processing site (Figure 3.1).

Background ground water quality can be described as a moderately oxidizing, calcium-sulfate-bicarbonate ground water with a slightly basic pH (up to 7.6). Figure 3.2 shows a trilinear plot of major cation and anion chemistry. Trilinear plots graphically illustrate similarities and/or differences in the water chemistry of samples from different locations. Distributions of cations and anions in samples from related waters plot close together in the two-dimensional fields of the diagrams. The water chemistry plots in Figure 3.2 show similarities in the chemical composition of ground water drawn from monitor wells 710 and 711. The chemical composition of samples from these wells represents ground water that has not been affected by uranium milling. Table 3.1 presents a statistical summary of surficial aquifer ground water quality data, including background water quality data.

Water quality data from background wells 710 and 711 indicate the maximum levels of iron and sulfate exceed national secondary drinking water standards in the surficial aquifer. This is in agreement with the results of a United States Geologic Survey (USGS) study for the Riverton irrigation project. In that study of ground

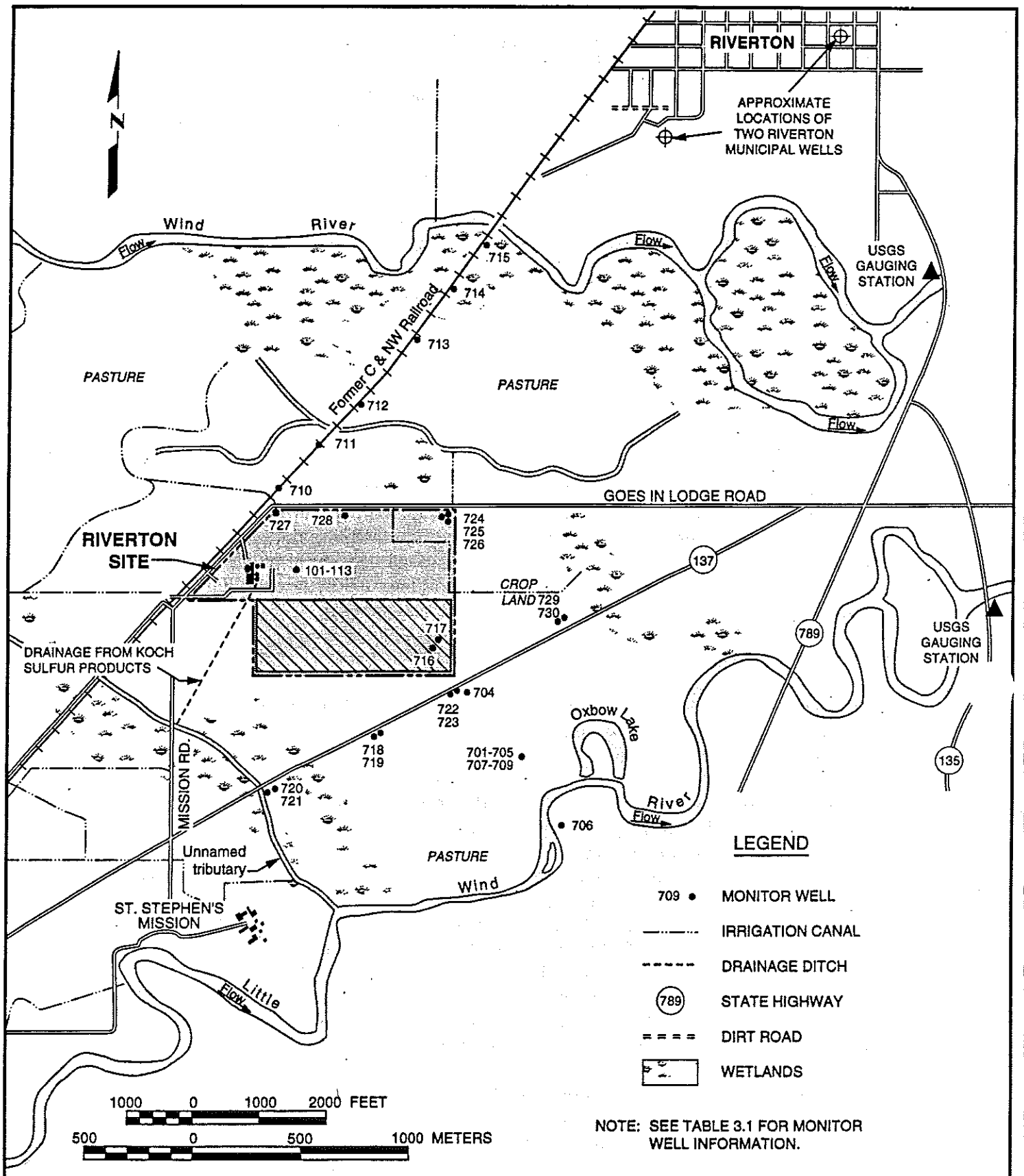

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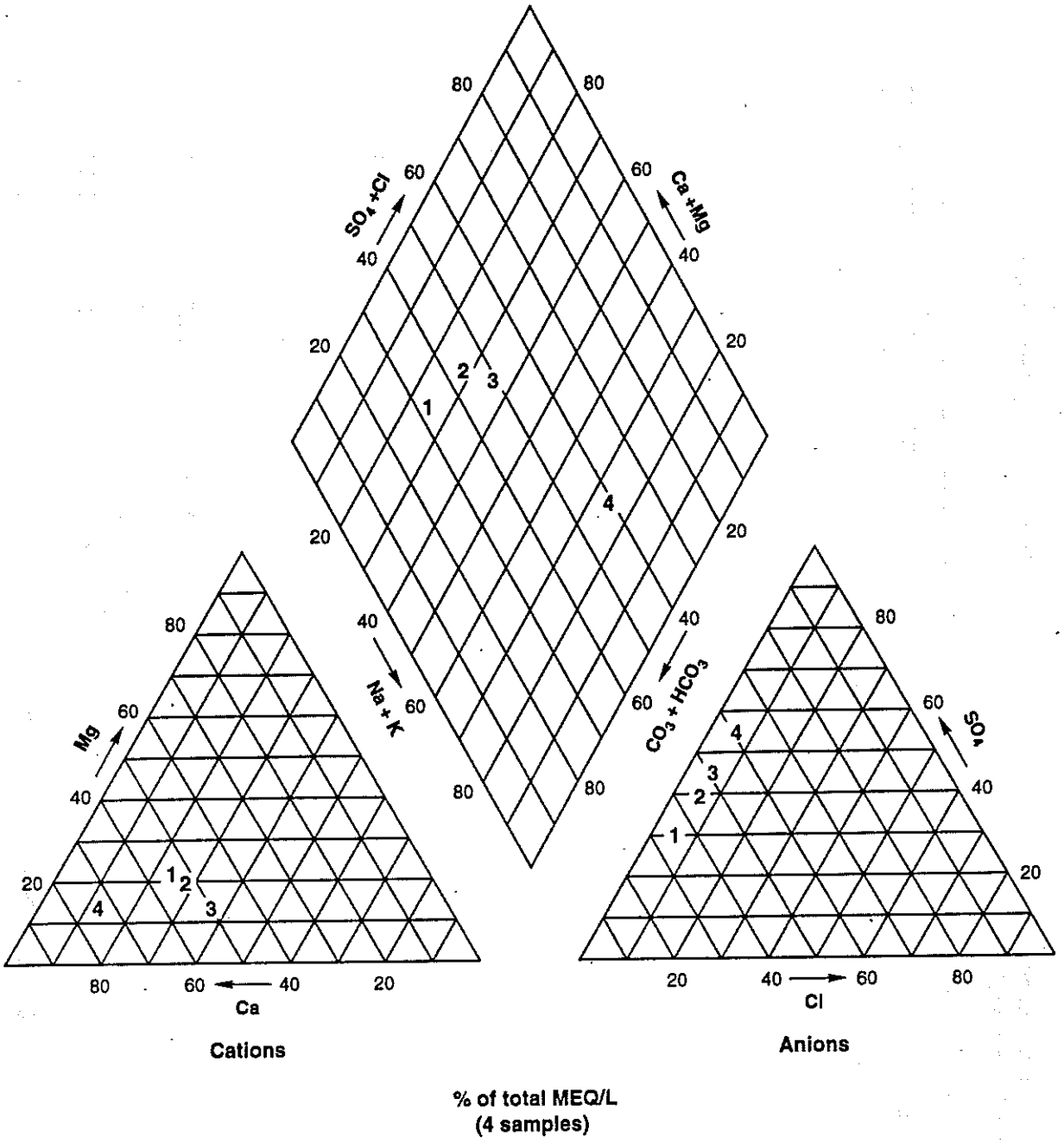
FIGURE 3.1
MONITOR WELL LOCATIONS
RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-MONITORWELLLOC



NOTES:

1. UPGRADIENT ALLUVIAL AQUIFER: MONITOR WELLS 710 AND 711.
2. SEMICONFINED AQUIFER: MONIOR WELLS 725 AND 727.
3. WATER QUALITY DATE: MARCH 1993.

LEGEND

1 MONITOR WELL 710
2 MONITOR WELL 711
3 MONITOR WELL 725
4 MONITOR WELL 727

FIGURE 3.2
TRILINEAR PLOT OF COMPOSITION OF BACKGROUND GROUND WATER
RIVERTON, WYOMING, SITE

Table 3.1 Comparison of background and downgradient ground water quality data for the surficial aquifer, Riverton, Wyoming, site (filtered samples, 1987 to 1993)

Constituent	Number of samples	Detection limit (mg/L)	% above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Aluminum						
Background	19	0.05-0.10	21	-	-	0.20
Plume	15	0.05-0.10	33	-	-	0.39
Ammonium						
Background	6	0.10	50	<0.10	-	1.40
Plume	8	0.10	38	<0.10	-	0.42
Antimony						
Background	18	0.001-0.003	17	<0.001	-	0.005
Plume	13	0.001-0.003	23	<0.001	-	0.036
Arsenic						
Background	20	0.001-0.010	30	0.002	-	0.007
Plume	14	0.001-0.050	43	<0.001	-	0.032
Barium						
Background	20	0.01-0.10	70	0.02	0.03-0.06[b]	0.14
Plume	16	0.01-0.10	69	0.01	0.02	0.05
Beryllium						
Background	11	0.005-0.010	0	-	-	-
Plume	7	0.005-0.010	0	-	-	-
Boron						
Background	2	0.10	0	-	-	-
Plume	2	0.10	100	0.20	0.20	0.20
Bromide						
Background	4	0.10	0	-	-	-
Plume	3	0.10	67	<0.10	0.30	0.30
Cadmium						
Background	20	0.0001-0.005	0	-	-	-
Plume	16	0.0001-0.005	25	<0.0001	-	0.005

Table 3.1 Comparison of background and downgradient ground water quality data for the surficial aquifer, Riverton, Wyoming, site (filtered samples, 1987 to 1993) (Continued)

Constituent	Number of samples	Detection limit (mg/L)	% above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Calcium						
Background		0.01-1.00		53.2	71.7	271
Plume		0.01-1.00		345	403	605
Chloride						
Background		0.50-1.00		4.50	11.2	125
Plume		0.50-1.00		85.0	118	207
Chromium						
Background		0.01		<0.01	-	0.06
Plume		0.01		<0.01	-	0.13
Cobalt						
Background		0.03-0.05		-	-	-
Plume		0.03-0.05		-	-	-
Copper						
Background		0.01-0.02		-	-	-
Plume		0.01-0.02		<0.01	-	0.02
Cyanide						
Background		0.01		-	-	-
Plume		0.01		-	-	-
Fluoride						
Background		0.10		<0.10	0.20	0.40
Plume		0.10-1.00		0.70	0.80	1.00
Iron						
Background		0.02-0.03		<0.03	0.10	1.56
Plume		0.02-0.03		0.04	0.20	3.05
Lead						
Background		0.001-0.010		<0.001	-	0.004
Plume		0.001-0.030		<0.001	-	0.020

Table 3.1 Comparison of background and downgradient ground water quality data for the surficial aquifer, Riverton, Wyoming, site (filtered samples, 1987 to 1993) (Continued)

Constituent	Number of samples	Detection limit (mg/L)	% above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Magnesium						
Background	20	0.001-1.00	100	13.6	17.3	68.0
Plume	16	0.001-1.00	100	179	198	291
Manganese						
Background	20	0.01	75	<0.01	0.70	3.56
Plume	16	0.01	100	4.26	4.78	6.40
Mercury						
Background	16	0.0001-0.0002	6	<0.0001	-	0.0004
Plume	13	0.0001-0.0002	8	<0.0001	-	0.0004
Molybdenum						
Background	20	0.001-0.010	25	0.003	-	0.020
Plume	16	0.001-0.010	100	0.52	0.78	1.02
Nickel						
Background	20	0.01-0.04	10	<0.01	-	0.03
Plume	16	0.01-0.04	94	<0.04	0.17	0.28
Nitrate						
Background	16	0.10-1.00	56	<0.10	0.15-1.0[b]	7.10
Plume	14	0.10-1.00	57	<0.10	0.25-0.95[b]	8.40
Phosphate						
Background	6	0.10	50	<0.10	-	0.30
Plume	8	0.10	50	<0.10	-	1.80
Potassium						
Background	20	0.01-0.50	100	1.86	2.50	6.40
Plume	16	0.01-0.50	100	10.4	14.5	18.0
Selenium						
Background	18	0.001-0.005	6	<0.001	-	0.005
Plume	14	0.001-0.030	43	<0.001	-	0.077

Table 3.1 Comparison of background and downgradient ground water quality data for the surficial aquifer, Riverton, Wyoming site (filtered samples, 1987 to 1993) (Continued)

Constituent	Number of samples	Detection limit (mg/L)	% above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Silica						
Background	4	2.0	100	15.0	17.8	19.0
Plume	3	2.0	100	26.0	30.0	30.3
Silver						
Background	18	0.01	0	-	-	-
Plume	15	0.01	13	<0.01	-	0.02
Sodium						
Background	20	0.002-5.00	100	32	53	167
Plume	16	0.002-20.00	100	697	856	1360
Strontium						
Background	19	0.01-0.10	100	0.29	0.37	1.51
Plume	15	0.01-0.10	100	1.82	2.47	3.40
Sulfate						
Background	20	0.10-20.0	100	84.7	152	854
Plume	16	0.10-10.0	100	2570	3010	4430
Sulfide						
Background	9	0.10-4.00	44	<0.10	-	226
Plume	7	0.10-4.00	0	-	-	-
Thallium						
Background	7	0.01	0	-	-	-
Plume	5	0.01	0	-	-	-
Tin						
Background	12	0.005-0.030	0	-	-	-
Plume	7	0.005-0.030	14	<0.005	-	0.340
Uranium						
Background	20	0.0003-0.003	80	<0.001	0.003	0.008
Plume	16	0.0003-0.003	100	0.719	1.152	1.970

Table 3.1 Comparison of background and downgradient ground water quality data for the surficial aquifer, Riverton, Wyoming, site (filtered samples, 1987 to 1993) (Concluded)

Constituent	Number of samples	Detection limit (mg/L)	% above detection	Observed Minimum (mg/L)	Observed Median[a] (mg/L)	Observed Maximum (mg/L)
Vanadium						
Background	20	0.01	30	<0.01	-	0.04
Plume	16	0.01-0.05	50	<0.01	-	0.14
Zinc						
Background	20	0.005-0.01	25	<0.005	-	0.021
Plume	16	0.005-0.03	38	<0.005	-	0.019
Radionuclides (pCi/L)						
Lead-210						
Background	4	NA	100	0.00	0.15	0.90
Plume	3	NA	100	0.20	3.00	4.00
Polonium-210						
Background	4	NA	100	0.10	0.20	0.30
Plume	3	NA	100	0.10	0.90	2.40
Radium-226						
Background	19	NA	100	0.00	0.20	2.40
Plume	13	NA	100	0.00	0.10	2.20
Thorium-230						
Background	7	NA	100	0.00	0.20	3.80
Plume	5	NA	100	0.00	0.60	6.82

[a]The median or 50th percentile of the sample data cannot be determined unless more than 50 percent of the data are above detection. A dash ("-") in the median column indicates the median cannot be calculated.
[b]The median cannot be precisely determined because the detection limit is inflated. However, the median is known to be within the range presented in this table.
NA - Detection limits are not applicable.
Note: Background wells are 710 and 711; plume well is 707.

water within alluvial deposits in the Riverton area, the sulfate and iron contents in over half the wells sampled were above the levels acceptable for drinking water. Also, the high proportion of radium to other major elements was reported to limit the use of ground water for irrigation (Dana, 1962).

3.1.2 Semiconfined aquifer

To determine background concentrations for ground water in the semiconfined water-bearing unit water quality data from monitor wells 725 and 727 were used. These wells are upgradient from the remediated tailings pile on the northern perimeter of the site boundary. Table 3.2 summarizes water quality data for the semiconfined aquifer from the March 1993 sampling round.

Background ground water quality in the semiconfined aquifer can be described as a moderately oxidizing, sulfate-bicarbonate-calcium sodium type with a slightly basic pH (about 8). Figure 3.2 shows that calcium and bicarbonate are the dominant cation and anion in monitor well 725 but sodium and sulfate are the predominant cation and anion in monitor well 727. This variability is difficult to explain; only one sampling round precludes the use of statistical analysis. Ground water accessed from these wells probably represents background chemistry for the semiconfined aquifer.

3.2 MAGNITUDE OF SITE-RELATED CONTAMINATION

Ground water in both the surficial and semiconfined aquifers appears to be affected by milling activities. Uranium mill tailings were deposited directly above the surficial aquifer. Therefore, ground water is impacted to a greater extent in the surficial aquifer than the semiconfined aquifer because of the greater intimacy between ground water in the surficial aquifer and tailings leachate that had drained into underlying soils. The magnitude of contamination present in both aquifers is discussed below.

3.2.1 Surficial aquifer

Ground water in the surficial aquifer downgradient from the abandoned processing site is characterized by elevated TDS and other constituents associated with uranium milling (e.g., uranium, molybdenum, and sulfate). Isopleths denoting the concentrations of uranium, molybdenum, and sulfate in ground water sampled from the surficial aquifer are shown in Figures 3.3 through 3.5. From the concentration isopleth maps, contamination is migrating to the southeast.

Samples from monitor well 707 represent ground water with the greatest concentration of contaminants in the surficial aquifer (Table 3.1). This well is approximately 2000 ft (600 m) southeast of the southeast corner of the site boundary. Presently, the center of the plume appears to be in the vicinity of monitor well 707, traveling less than half the calculated linear velocity of ground water (165 ft [50 m] per year). This difference occurs because the interaction

Table 3.2 Comparison of background and downgradient ground water quality data for the contaminants of concern in the semiconfined aquifer, Riverton, Wyoming, site (March 1993 sampling, filtered samples)[a]

Constituent	Minimum-maximum	Median[b]
Inorganics (mg/L)		
Aluminum		
Background	<0.05	-
Downgradient	<0.05	-
Arsenic		
Background	<0.005	-
Downgradient	<0.005	-
Barium		
Background	<0.1	-
Downgradient	<0.1	-
Bromide		
Background	<0.1-0.1	<0.1
Downgradient	<0.1-0.28	0.2
Cadmium		
Background	<0.001	-
Downgradient	<0.001	-
Calcium		
Background	30-98	64
Downgradient	106-401	181
Chloride		
Background	16-20	18
Downgradient	55-73	61
Chromium		
Background	<0.01	-
Downgradient	<0.01	-
Fluoride		
Background	0.2	0.2
Downgradient	0.1-0.2	0.2
Iron		
Background	<0.03-0.04	<0.04
Downgradient	<0.03-0.39	0.16

Table 3.2 Comparison of background and downgradient ground water quality data for the contaminants of concern in the semiconfined aquifer, Riverton, Wyoming, site (March 1993 sampling, filtered samples)[a] (Continued)

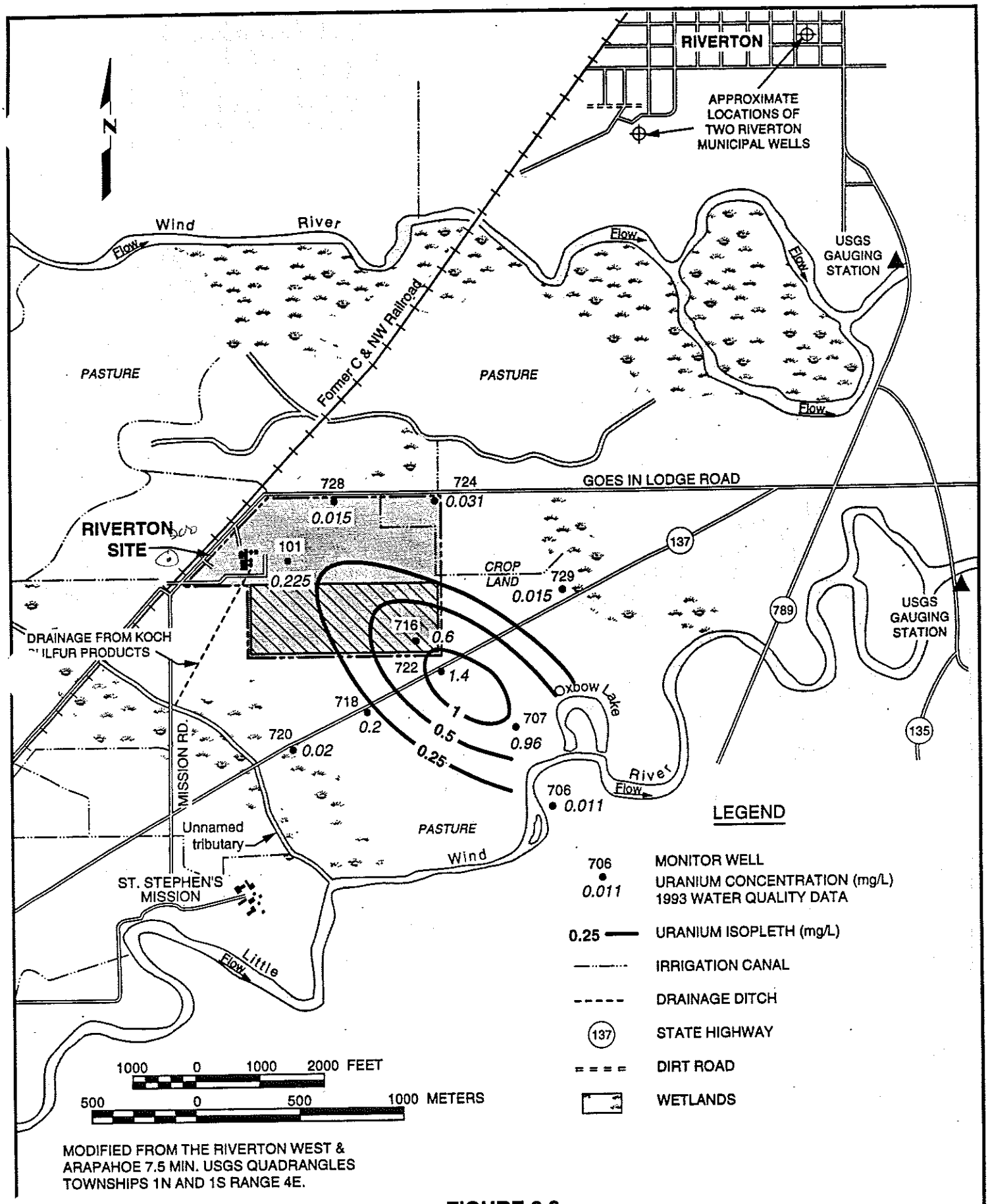
Constituent	Minimum-maximum	Median[b]
Lead		
Background	<0.003	-
Downgradient	<0.003	-
Magnesium		
Background	2.8-17	10
Downgradient	7.5-32	13
Manganese		
Background	0.02-0.06	0.04
Downgradient	0.13-0.91	0.26
Molybdenum		
Background	<0.01	-
Downgradient	<0.01-1.27	0.03
Nickel		
Background	<0.04	-
Downgradient	<0.04	-
Potassium		
Background	1.0-1.3	1.2
Downgradient	1.2-2.7	1.9
Selenium		
Background	<0.005	-
Downgradient	<0.005	-
Sodium		
Background	79-118	99
Downgradient	108-665	288
Strontium		
Background	0.2-0.5	0.3
Downgradient	0.6-2.7	0.9
Sulfate		
Background	170-207	189
Downgradient	592-1860	686
Uranium		
Background	0.002-0.008	0.005
Downgradient	0.002-0.017	0.011

Table 3.2 Comparison of background and downgradient ground water quality data for the contaminants of concern in the semiconfined aquifer, Riverton, Wyoming, site (March 1993 sampling, filtered samples)[a] (Concluded)

Constituent	Minimum-maximum	Median[b]
Vanadium		
Background	<0.01	-
Downgradient	<0.01	-
Zinc		
Background	0.008-0.012	0.010
Downgradient	0.008-0.013	0.010
Radionuclides (pCi/L)		
Lead-210		
Background	0.3-1.5	0.9
Downgradient	0.0-1.5	0.7
Polonium-210		
Background	0.0	0.0
Downgradient	0.0-0.6	0.1
Radium-226		
Background	0.0-0.8	0.4
Downgradient	0.0-4.0	0.9
Thorium-230		
Background	0.1-1.6	0.9
Downgradient	0.0-0.6	0.2

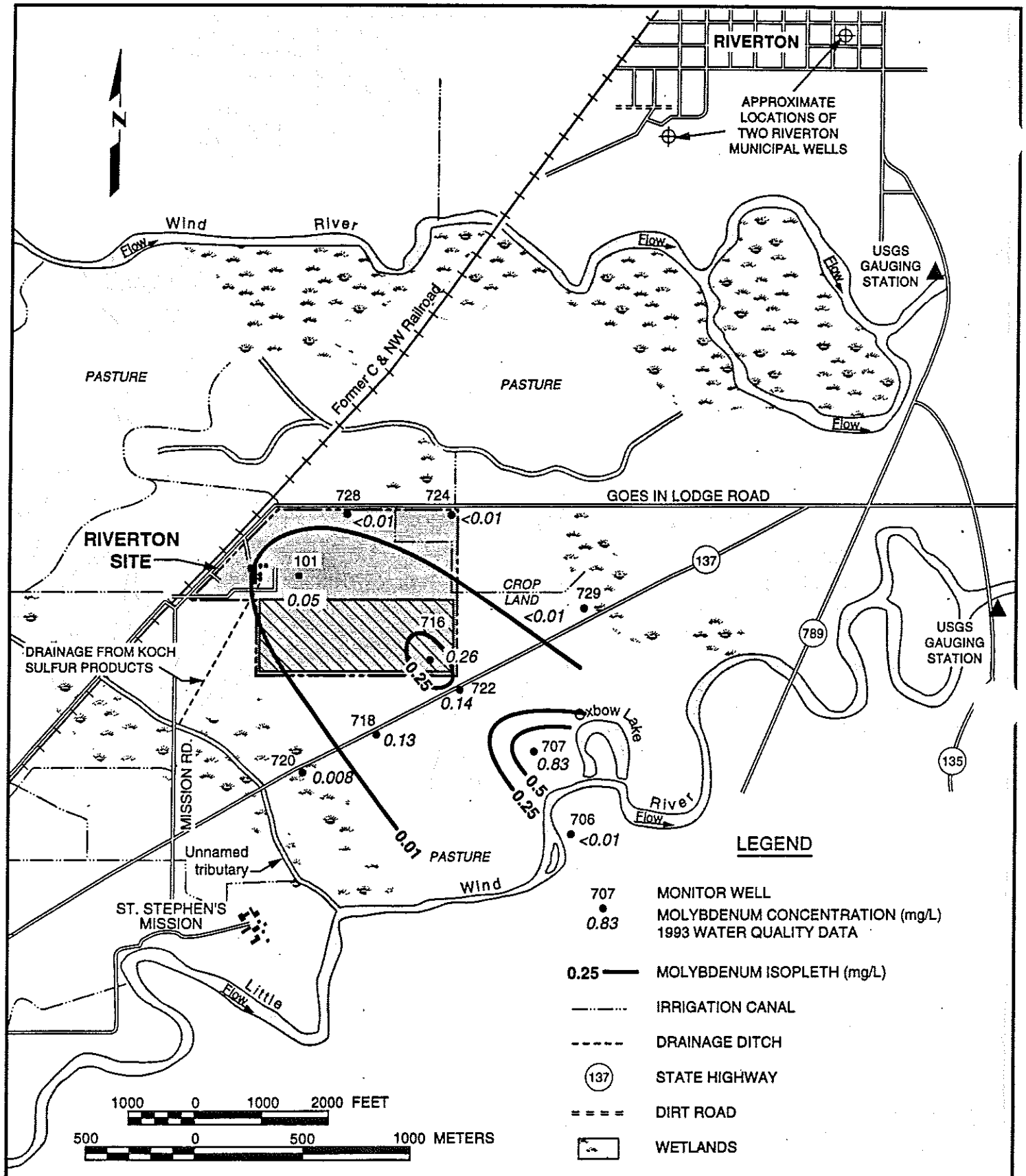
[a]Background, defined by monitor wells 725 and 727, consist of two data values for each constituent listed. Downgradient wells 108, 717, 719, and 723 provide four data values for each constituent listed.
[b]The median is the 50th percentile of the data. Based on two data values, the median is the arithmetic average. For four data values, the median is the arithmetic average of the two middle values.

A dash ("-") in the median column indicates the median cannot be calculated.



MODIFIED FROM THE RIVERTON WEST & ARAPAHOE 7.5 MIN. USGS QUADRANGLES TOWNSHIPS 1N AND 1S RANGE 4E.

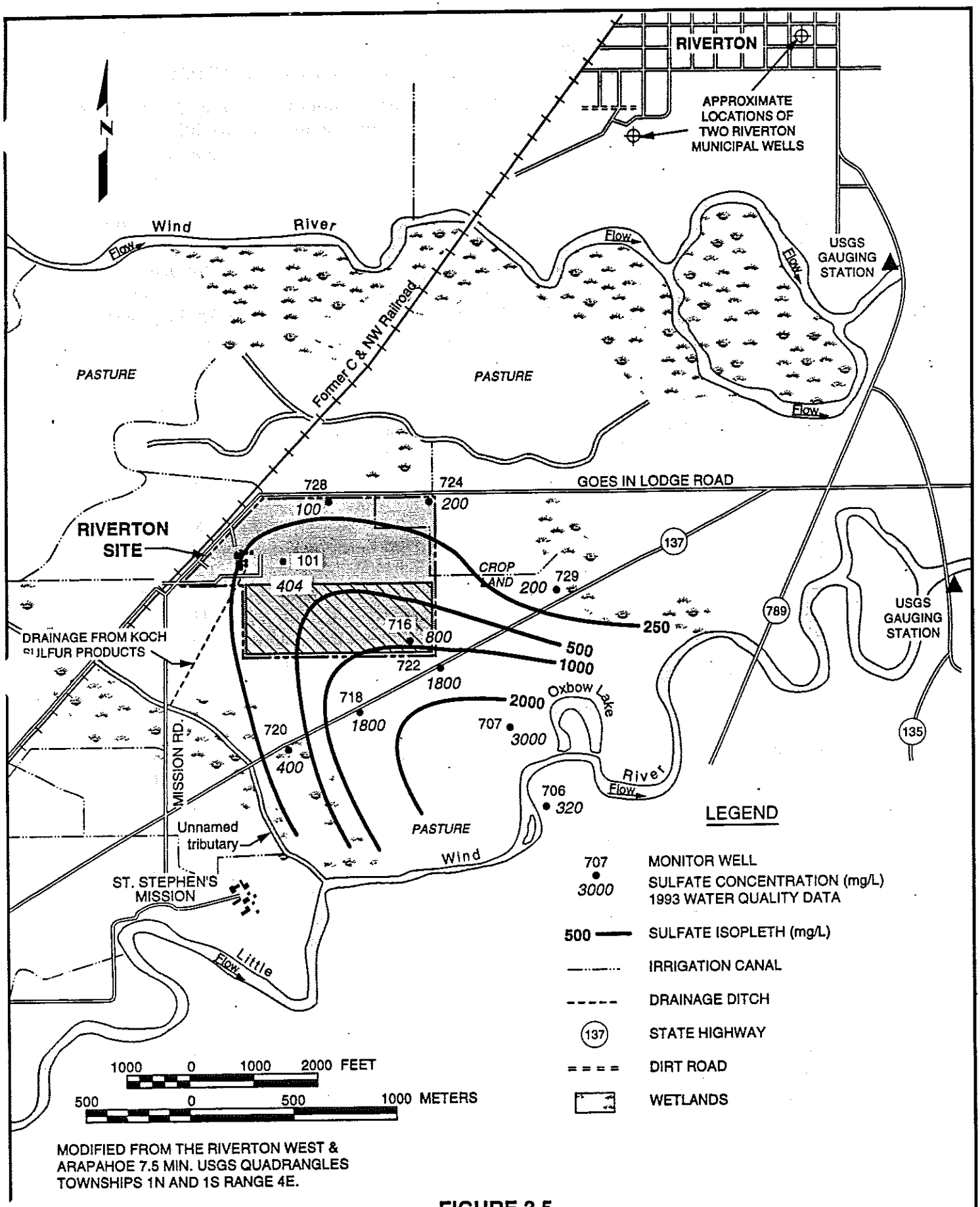
FIGURE 3.3
ISOPLETHS SHOWING URANIUM CONCENTRATIONS IN THE SURFICIAL AQUIFER RIVERTON, WYOMING, SITE



MODIFIED FROM THE RIVERTON WEST & ARAPAHOE 7.5 MIN. USGS QUADRANGLES TOWNSHIPS 1N AND 1S RANGE 4E.

FIGURE 3.4
ISOPLETHS SHOWING MOLYBDENUM CONCENTRATIONS IN THE SURFICIAL AQUIFER RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-ISOMOLY



MODIFIED FROM THE RIVERTON WEST & ARAPAHOE 7.5 MIN. USGS QUADRANGLES TOWNSHIPS 1N AND 1S RANGE 4E.

FIGURE 3.5
ISOPLETHS SHOWING SULFATE CONCENTRATIONS IN THE SURFICIAL AQUIFER RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-ISOSUL

between contaminants and the aquifer matrix retards contaminants at different rates as they travel through the aquifer. This phenomenon contributes to different contaminant plume shapes and positions with respect to average linear ground water flow.

To generate a list of inorganic constituents related to milling activities, water quality data from monitor well 707 were statistically compared to background ground water quality data from monitor wells 710 and 711. Water quality data have been collected from wells in the path of the plume only since the mid-1980s. Although temporal trends of water quality data from monitor wells show weakly increasing or decreasing concentrations, the observed center of the plume is apparently southeast of the site boundary.

Because the alluvium is laterally thin (5 to 10 ft [1.5 to 3.0 m] thick) and relatively porous, lateral dominates vertical dispersion. These conditions allow the ground water to flush contaminants from the shallow system. Figure 3.5 shows the probable maximum extent of the plume.

Two private wells producing water from the surficial aquifer near the site appear to be influenced by site-related contamination. These are wells 431 and 445.

Private well 431 is located adjacent to the northeast corner of the site and is crossgradient. This well is reported to be about 15 ft (4.6 m) deep. Uranium levels (0.008 to 0.016 mg/L) in ground water sampled from this well are 1 to 2 times the maximum observed levels in background. All other measured constituents in well 431 are within the range of background indicating that contamination in this well is largely diluted by background ground water.

The second site-affected private well (445) is southeast and downgradient of the site. Ground water sampled from this well during January 1994 had detectable levels of arsenic (0.008 mg/L), uranium (0.016 mg/L) and vanadium (0.02 mg/L). The well is completed in the semiconfined aquifer at a depth of approximately 36 to 40 ft (11 to 12 m). Comparison to nearby monitor wells completed in the semiconfined aquifer (well 729) and the surficial aquifer (well 730) shows the concentrations of alkalinity, sulfate, TDS, and uranium in the private well are very similar to those in the surficial aquifer, and notably different from those in the semiconfined. This indicates private well 445 is producing water from the surficial aquifer.

3.2.2 Semiconfined aquifer

The potentiometric surface of the semiconfined water-bearing unit is equal to or lower than the water table surface of the surficial aquifer in the vicinity of the abandoned mill site, suggesting a vertical ground water flow.

Water quality data have been collected from the semiconfined aquifer since 1987. These data suggest that ground water in the semiconfined aquifer is contaminated directly beneath and downgradient from the processing site. Monitor wells 108,

705, 717, 719, and 723 represent the plume wells in this aquifer (Table 3.3). Data from March 1993, the most recent sampling round, show that ground water levels in this region are elevated with respect to background ground water levels of manganese, strontium, sulfate, molybdenum, uranium, calcium, chloride, iron, and sodium. The sulfate concentration isopleth is shown in Figure 3.6. Contamination is moving southeast from the site, probably at a slower rate than the linear velocity of ground water.

3.3 CONTAMINANTS OF POTENTIAL CONCERN

The data presented in Sections 3.1 and 3.2 were used to compile a list of contaminants of potential concern in assessing human health and environmental risks at the site. Because contamination levels are higher in the surficial aquifer, it will be evaluated quantitatively. The less-contaminated and less-characterized semiconfined aquifer will be evaluated qualitatively (Section 6.0).

In general, constituents were listed among contaminants of potential concern when the site was considered a likely source of the contamination and when the constituent average concentrations (measured in plume wells) exceeded average background levels at the 0.05 level of significance (DOE, 1995). Column 1 of Table 3.3 shows which constituents exceed surficial ground water background levels.

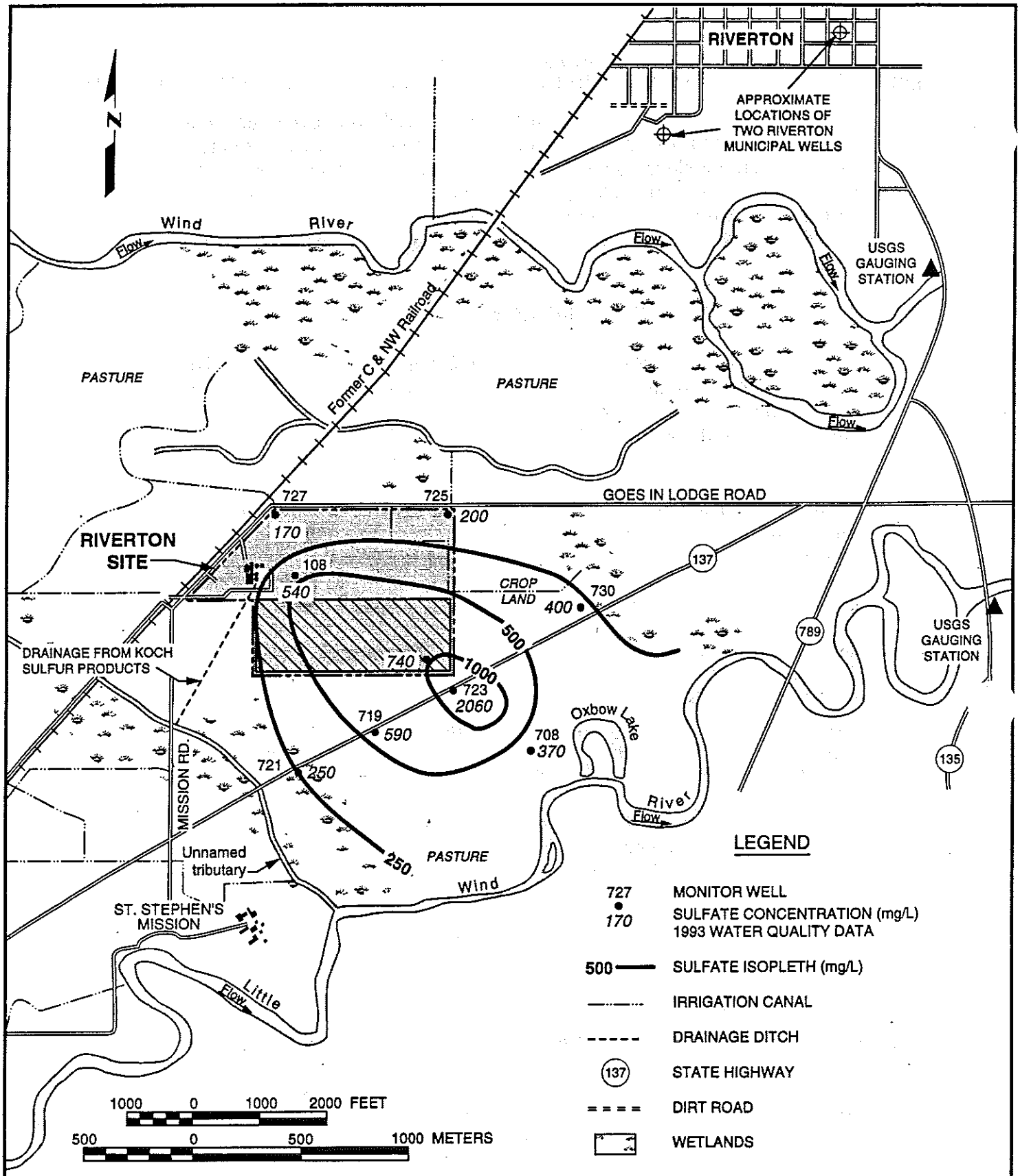
Several of the chemical species detected above background levels are essential nutrients. These constituents were not evaluated further because they were measured within nutritional ranges (column 2) (DOE, 1995). These constituents include calcium, chloride, fluoride, iron, potassium, and zinc. Several other contaminants in the surficial aquifer were screened for their low toxicity and/or low concentrations compared to high dietary intakes (column 3) (DOE, 1995).

For the surficial aquifer, these screening criteria eliminated all contaminants but arsenic, manganese, molybdenum, nickel, sulfate, uranium, vanadium, and several radioactive progeny of the uranium decay series, including lead-210, polonium-210, and thorium-230. Column 4 of Table 3.3 lists the final contaminants of potential concern.

For the semiconfined aquifer, the contaminants of potential concern are manganese, molybdenum, sulfate, and uranium.

3.4 CONTAMINANT FATE AND TRANSPORT

Milling activities have influenced the surficial aquifer downgradient from the processing site. Since 1963, when uranium ore processing ceased, natural flushing has moved the plume centroid southeast toward the Little Wind River; the most contaminated well, monitor well 707, is approximately 250 ft (80 m) from the Little Wind River.



MODIFIED FROM THE RIVERTON WEST & ARAPAHOE 7.5 MIN. USGS QUADRANGLES TOWNSHIPS 1N AND 1S RANGE 4E.

NOTE: WATER QUALITY DATE MARCH 1993 FOR ALL WELLS EXCEPT 108 (AUGUST 1992).

FIGURE 3.6
ISOPLETHS SHOWING SULFATE CONCENTRATIONS IN THE SEMICONFINED AQUIFER RIVERTON, WYOMING, SITE

MAC: SITE/RVT/BRA/BASE-ISOSULSEMI

Table 3.3 Contaminants of potential concern in surficial ground water, Riverton, Wyoming, site

Contaminants that exceed background	Contaminants in nutritional range	Contaminants of low toxicity and/or high dietary range	Contaminants of potential concern
Aluminum	Calcium	Aluminum	Arsenic
Arsenic	Chloride	Boron	Lead-210
Boron	Fluoride	Bromide	Manganese
Bromide	Iron	Magnesium	Molybdenum
Calcium	Potassium	Selenium	Nickel
Chloride	Zinc	Silica	Polonium-210
Fluoride		Sodium	Sulfate
Iron		Strontium	Thorium-230
Lead-210			Uranium
Magnesium			Vanadium
Manganese			
Molybdenum			
Nickel			
Polonium-210			
Potassium			
Selenium			
Silica			
Sodium			
Strontium			
Sulfate			
Thorium-230			
Uranium			
Vanadium			
Zinc			

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

Contaminant levels in ground water collected from monitor well 707 are elevated in comparison to background levels of arsenic, manganese, molybdenum, nickel, sulfate, uranium, and vanadium. These constituents, their predominant aqueous species, and their molar percentages are listed in Table 3.4. Aqueous species were calculated with the geochemical computer codes MINTEQA2 (Allison et al., 1991) and PHREEQE (Parkhurst et al., 1980).

The chemical species of an element is determined by the chemical composition of the ground water, including pH, redox potential (Eh), temperature, availability of ions and complexes, and concentration of the element. The mobility of a constituent depends upon its species. Toxicokinetics (contaminant absorption, distribution, metabolism, and excretion) and toxicity also depend on the concentration and type of species present in ground water.

According to geochemical modeling MINTEQA2 (Allison et al., 1991) and PHREEQE (Parkhurst et al., 1980), monitor well 707 reaches ground water that is slightly oversaturated with barite and gypsum and moderately oversaturated with the uranium phase schoepite, hydroxides and oxyhydroxides of iron, jarosite (iron sulfate), nickel hydroxide, barium arsenate, and calcium and magnesium vanadate phases (DOE, 1995). Because kinetically favorable uranium, vanadium, nickel, arsenic, and sulfate phases precipitate and adsorb to the aquifer matrix, these constituents are removed from ground water and their concentrations are reduced.

Contaminant concentration in the plume will be influenced by the dispersive effects of dilution and by chemical reactions such as oxidation/reduction, sorption onto the aquifer matrix, coprecipitation with other mineral phases, microbial reactions, and advective transport of ground water. With distance from the source, dispersive effects will decrease the concentration of all the contaminants in the plume. The following reactions can be expected for individual contaminants:

- Arsenic is probably removed by precipitation and adsorption onto the aquifer matrix.
- Manganese is removed by coprecipitation with iron phases; when pH rises, manganese may precipitate as an oxyhydroxide.
- Nickel is oversaturated with respect to nickel hydroxide and probably coprecipitates with ferric oxyhydroxide phases.
- Sulfate is probably removed by the precipitation of gypsum and possibly other sulfate mineral phases.
- Uranium is probably attenuated because of the precipitation of schoepite and adsorption onto the aquifer matrix.

Table 3.4 Predominant aqueous species and molar percentage of contaminants of potential concern in ground water in the surficial aquifer downgradient from the Riverton, Wyoming, site

Contaminant of potential concern		Identity of species in ground water	Approximate molar %
Arsenic[a]	Arsenate	$HAsO_4^{2-}$	61
	Arsenate	$H_2AsO_4^-$	39
Manganese[a]	Manganese	Mn^{2+}	62
	Manganese sulfate	$MnSO_{4\,AQ}$	35
	Manganese carbonate	$MnCO_3^+$	3
Molybdenum[b]	Molybdate	MoO_4^{2-}	100
Nickel[a]	Nickel carbonate	$NiCO_{3\,AQ}$	59
	Nickel	Ni^{2+}	22
	Nickel sulfate	$NiSO_{4\,AQ}$	12
	Nickel bicarbonate	$NiHCO_3$	7
Sulfate[a]	Sulfate	SO_4^{2-}	75
	Calcium sulfate	$CaSO_{4\,AQ}$	12
	Magnesium sulfate	$MgSO_{4\,AQ}$	9
	Sodium sulfate	$NaSO_4^-$	4
Uranium[a]	Uranium tricarbonate	$UO_2(CO_3)_3^{4-}$	54
	Uranyl dicarbonate	$UO_2(CO_3)_2^{2-}$	43
	Uranium carbonate	$UO_2(CO_3)_{AQ}$	3
Vanadium[a]	Vanadate	$H_2VO_4^-$	73
	Vanadium oxide	$HV_2O_7^{3-}$	22
	Vanadate	HVO_4^{2-}	5

[a]Aqueous species calculated using MINTEQA2 (Allison et al., 1991).
[b]Molybdenum speciation calculated using the geochemical code PHREEQE (Parkhurst et al., 1980).

- Vanadium is oversaturated with respect to calcium and magnesium vanadate mineral phases and may be removed if the kinetics are favorable (little is known about the precipitation kinetics of these phases). Vanadium also may be removed through adsorption onto the aquifer matrix.

3.5 SURFACE WATER MONITORING

The surface water bodies in the site vicinity that are potentially impacted by site contaminants include the Little Wind River, wetland areas south and east of the site, and the drainage ditch and stream southwest of the site. These areas were sampled in June 1993 (DOE, 1995). Potential impacts to livestock and the environment are discussed in Section 7.0.

Surface water samples from the Little Wind River were collected at locations 794, 742, and 796 in 1993 (Table 3.5, Figure 2.2). Location 794 is upstream of the site. Downstream location 742 has a concentration of uranium about 12 times greater than upstream location 794, indicating contaminated ground water discharge is influencing surface water uranium concentrations at sampling location 742.

Table 3.5 Summary of unfiltered surface water quality data from the Little Wind River for samples collected June 1993, Riverton, Wyoming, site

Constituent	Upstream location 794	Downstream location 742	Downstream location 796
Calcium	41	41	42
Iron	2.0	1.4	3.1
Lead	0.004	0.004	0.004
Magnesium	14	14	14
Manganese	0.09	0.12	0.14
Molybdenum	<0.01	<0.01	<0.01
Uranium	0.002	0.025	0.002

Notes: 1. All data expressed as milligrams per liter.
2. < indicates less than the given limit of detection.

Further downstream at location 743, all constituent levels are very similar to background. This suggests that complete mixing of contaminated ground water discharge with ambient surface water and/or uranium adsorption and sedimentation has returned the surface water uranium levels to near-ambient conditions.

Surface water quality data for samples collected from a drainage ditch, a tributary drainage, a flooded pasture on the Little Wind River floodplain, and a wetland are presented in Table 3.6. Sampling location 741 is upstream of the site on a tributary drainage to the Little Wind River. Sampling location 745 is downstream of the site along the same drainage (Figure 2.2). Calcium, magnesium, and uranium are slightly elevated in the downstream sample compared with the upstream location; however the water quality is still within the range of background ground water

Table 3.6 Unfiltered surface water quality data for surface drainage, wetlands, and floodplain samples collected June 1993, Riverton, Wyoming, site

Constituent	Upstream tributary drainage location 741	Downstream tributary location 745	Downstream drainage ditch location 744	Floodplain location 743	Wetlands location 746
Calcium	70	81	58	160	107
Iron	0.46	0.36	0.13	0.24	0.09
Lead	<0.003	<0.003	<0.003	<0.003	<0.001
Magnesium	20	22	14	38	39
Manganese	0.05	0.05	0.02	0.12	0.06
Molybdenum	<0.01	<0.01	<0.01	0.01	<0.01
Uranium	0.006	0.008	0.016	0.013	0.008

Notes: 1. All data expressed as milligrams per liter.
2. < indicates less than the given limit of detection.

quality within the surficial aquifer and it is not clear if the downstream change in chemistry within the tributary is due to contaminated ground water discharge, natural ground water discharge, or surface water evaporation.

In surface water collected from the drainage ditch east of the site (location 744), uranium concentration is greater than it is in the tributary and in the background ground water from the surficial aquifer. This may indicate contaminated ground water discharge is slightly influencing surface water in this drainage.

One sample of surface water was collected from a wetland location downgradient of the site (location 746) (Figure 2.2). No background sample was collected for comparison. Uranium, calcium, and magnesium concentrations in the wetland sample are greater than those in the upstream tributary sample (Table 3.6), but within the range of background ground water quality in the surficial aquifer. Thus, the concentrations of these constituents may reflect natural conditions rather than contamination. However, more data are needed to rule out the possibility of site-related contamination of this wetland from ground water discharge.

Surface water was collected from a flooded pasture on the floodplain of the Little Wind River at location 743. Calcium, magnesium, manganese, molybdenum, and uranium exceeded concentrations in the upstream sampling locations along the Little Wind River and the tributary drainage; only uranium slightly exceeded concentrations found in background ground water within the surficial aquifer. Because of possible variations in suspended sediments, these comparisons may not be valid. More information is needed to determine if this area is influenced by contaminated ground water discharge.

Sediment samples were collected in June 1993 at the surface water sampling locations. The results are presented in Tables 3.7 and 3.8. Because of the possible variations in grain size, organic content, and exchange capacities, it is not possible to fully evaluate these data. Along the Little Wind River, elevated concentrations of

Table 3.7 Sediment chemistry data from the Little Wind River for samples collected June 1993, Riverton, Wyoming, site

Constituent	Upstream location 794	Downstream location 742	Downstream location 796
Iron	5,860	21,200	8,530
Lead	4.5	14	3.9
Manganese	167	618	214
Molybdenum	<1	<4	4
Uranium	2.0	10.4	2.3

Note: All data expressed as milligrams per kilogram.

Table 3.8 Summary of sediment chemistry data from surface drainage, wetlands, and floodplain samples collected June 1993, Riverton, Wyoming, site

Constituent	Upstream tributary drainage location 741	Downstream tributary location 745	Downstream drainage ditch location 744	Floodplain location 743	Wetlands location 746
Iron	10,500	3150	12,200	18,000	11,000
Lead	11	4.8	6	22	68
Manganese	455	54	187	235	314
Molybdenum	11	1	<1	2	9
Uranium	7.6	1.7	5.5	5.0	11.3

Note: All data expressed as milligrams per kilogram.

iron, lead, manganese, and uranium at downstream location 742 may indicate these constituents have accumulated in sediments due to the contaminated ground water discharge in this area. However, concentrations of these constituents and of molybdenum are also elevated in sediments collected upstream of the site in the tributary drainage. Thus, the results from all sediment samples are ambiguous and more information is needed to evaluate the possibility that contaminants could accumulate in sediments from ground water discharge.

4.0 EXPOSURE ASSESSMENT

4.1 POTENTIALLY EXPOSED POPULATION

Ground water is withdrawn at numerous points within a 2-mi (3-km) radius of the site. Most of these are residential wells used for drinking and other household purposes. All but two of these wells generally pump water from the deeper confined sandstone, which is not affected by site-related contamination. These two shallow wells are sometimes used for livestock watering and/or irrigation. One of these two wells showed signs of low-level contamination.

This risk assessment evaluates shallow ground water as a future source of drinking water for residents near the Riverton site. Contaminant concentrations in monitor well 707 (in the sampling cluster southeast of the site) are specifically considered because the contaminant levels were generally higher there.

The evaluation of potentially contaminated surface water and sediment relates to both current and future use. Maximum detected concentrations in surface water and sediment from the Little Wind River are evaluated for current and future residents.

This assessment evaluates domestic ground water uses and recreational surface water uses consistent with those of this region's rural population. The potentially exposed population is divided into the following age groups: infants (birth to 1 year), children (1 to 10 years), and adults (11 to 65 years). These age groups were selected for the following reasons:

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

- Similarity of toxicological variables, including responsiveness of sensitive subgroups (infants and children) to the contaminants of potential concern, toxicant intake-to-body-weight ratios, and toxicokinetics (contaminant absorption, distribution, metabolism, and excretion from an individual's body).

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

4.2 EXPOSURE PATHWAYS

An exposure pathway describes the course a contaminant takes from the source to the exposed individual or population. Therefore, an exposure pathway can be completed only if there are a source of contamination, a point of contact with a population or individual, and a route of exposure (e.g., water ingestion). Because the tailings piles and soils contaminated from uranium milling operations at the site were removed and relocated to a disposal cell, soil or air exposure pathways (such

as accidental soil ingestion, dermal contact with soil, or inhalation of particulates) are not considered.

Ground water in the region is used primarily for household purposes such as drinking, cooking, and bathing. Other uses typical of the region that could indirectly lead to human exposure are irrigation and livestock watering. Bodies of surface water in the area are other potential sources of human exposure. Recreational use of the river and other water bodies (i.e., drainage ditches, stream, and wetland areas) could lead to direct human exposure to surface water and sediment and to indirect exposure if fish caught from the river are eaten. Figure 4.1 provides a conceptual model for potential exposure pathways. To determine which pathways contribute substantially to exposure, the following pathways are screened using adult default parameters.

4.2.1 Drinking water ingestion

Drinking water ingestion is generally the dominant exposure route for ground water contaminated with metals and other nonvolatile compounds. In this evaluation, the definition of drinking water includes water used for drinking and for food preparation (e.g., reconstituted juices, soup, rice, beans). For a comparison of the significance of relative pathways, Table 4.1 shows a screening level assessment of drinking water intake for adults. Section 4.4 contains a further probability evaluation.

4.2.2 Dermal absorption

In the dermal absorption process, chemicals in contact with the skin are absorbed by blood vessels near the skin surface. Although some compounds are easily absorbed, the chemical properties of metals are not generally conducive to skin absorption.

To evaluate this exposure route, a screening calculation was used to compare the dermal absorption to the drinking water pathway. Because chemical-specific absorption factors are not available for these contaminants, they are assumed to absorb across the skin at the rate of water. This assumption probably overestimates the potential contribution of dermal absorption.

Although the dermal dose is an absorbed dose and the ingested dose is a total dose of which only some percentage will be absorbed, the very low contribution of dermal absorption (0.2 percent) is assumed to be insignificant compared to ingestion. Based on these results, dermal absorption is eliminated from more detailed evaluation and is discussed qualitatively in Section 6.0.

4.2.3 Ingestion of ground water-irrigated produce

This exposure route was evaluated for its relative significance in comparison to the drinking water ingestion route. Table 4.2 shows results of the screening calculation. The assumptions for this evaluation are likely to overestimate the

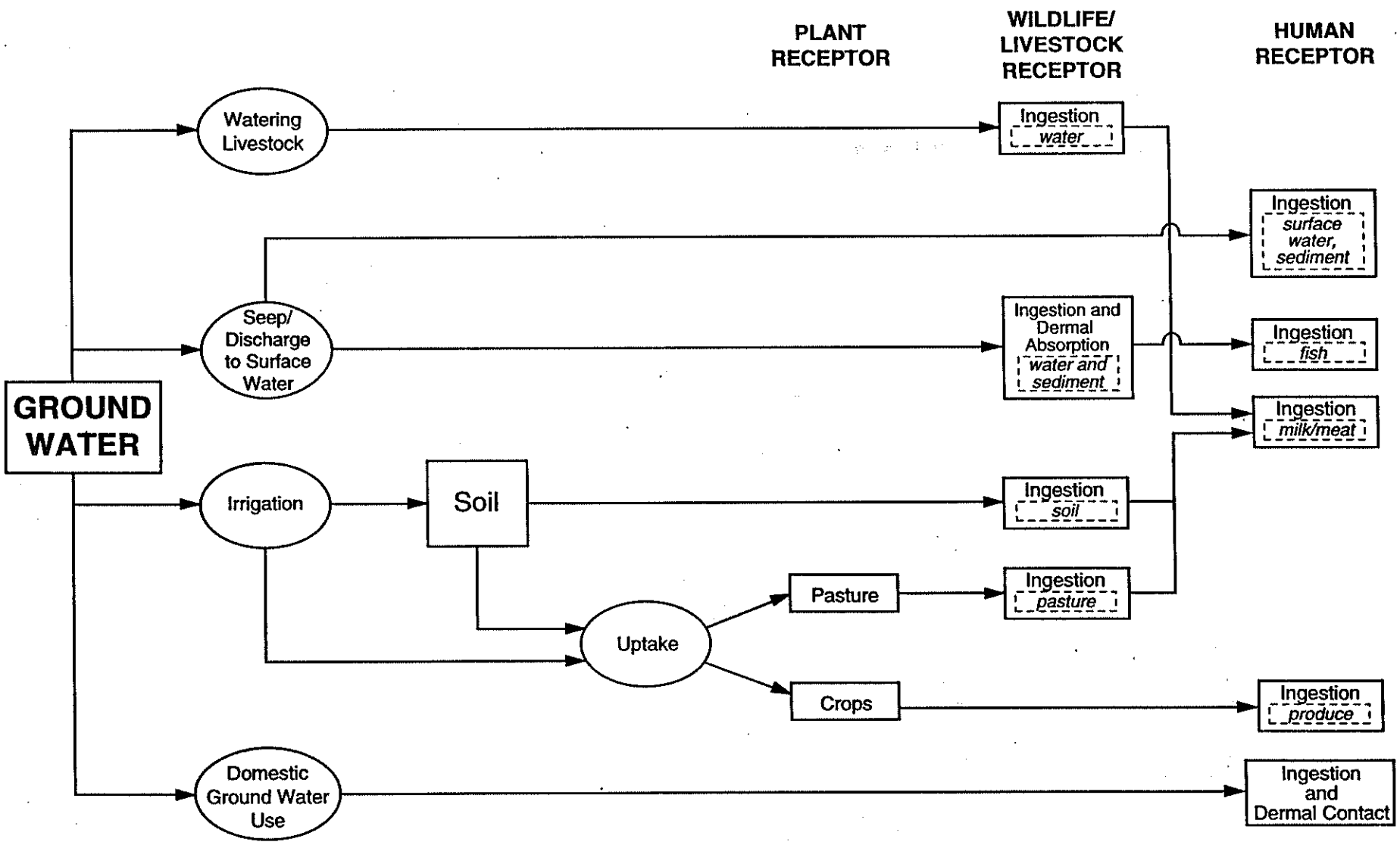


FIGURE 4.1
SITE CONCEPTUAL MODEL

PREPARATION DATE: 9/13/93

MAC: SITE/RVT/BRA/CONCEPTMOD

Table 4.1 Exposure dose calculations and equation definitions for ground water ingestion and dermal contact, Riverton, Wyoming, site

Contaminant of potential concern	Cw (mg/L)	Ground water exposure doses (mg/kg-day)		Ratio of dermal contact to ingestion
		Ingestion	Dermal contact	
Noncarcinogenic effects				
Arsenic	1.4E-02	4E-04	8E-07	0.002
Manganese	4.9E+00	1E-01	3E-04	0.002
Molybdenum	7.9E-01	2E-02	4E-05	0.002
Nickel	1.6E-01	5E-03	9E-06	0.002
Sulfate	3.1E+03	9E+01	2E-01	0.002
Uranium	1.2E+00	3E-02	7E-05	0.002
Vanadium	3.3E-02	9E-04	2E-06	0.002
Carcinogenic effects				
Arsenic	1.4E-02	2E-04	3E-07	0.002
Uranium	8.2E+02[a]	2E+07[b]	4E+04[b]	0.002

[a]In picocuries per liter.
[b]In picocuries per lifetime.

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

Table 4.1 Exposure dose calculations and equation definitions for ground water ingestion and dermal contact, Riverton, Wyoming, site (Concluded)

Equation definitions for exposure dose calculations

Ingestion of ground water - chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

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

Dermal contact with ground water - chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

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

Where:

Cw	=	Arithmetic mean of concentrations in filtered water samples from well 707, collected from 1987-93 (milligrams per liter or picocuries per liter).
IRw	=	Ingestion rate for water (L per day) (2 L per day for an adult).
EF	=	Exposure frequency (365 days per year).
ED	=	Exposure duration (30 years).
BW	=	Body weight (70 kg for an adult).
AT	=	Averaging time (365 days per year x ED for noncarcinogens, 365 days x 70 years for carcinogens).
SA	=	Skin surface area (19,400 square centimeters [cm^2]).
Pc	=	Dermal permeability constant (0.001 cm per hour).
Cf	=	Conversion factor (0.001 L/cm^3).
ET	=	Exposure time (0.2 hour per day).

Table 4.2 Exposure dose calculations for ingesting ground water-irrigated garden produce, Riverton, Wyoming, site

Contaminant of potential concern	Cw (mg/L)	Kd (L/kg)	Bv	Br	Garden produce ingestion exposure doses (mg/kg-day)		Ratio of produce ingestion to water ingestion
					Vegetative parts	Reproductive parts	
Noncarcinogenic effects							
Arsenic	1.4E-02	200	0.04	0.006	5E-06	5E-07	0.01
Manganese	4.9E+00	65	0.25	0.05	4E-03	4E-04	0.03
Molybdenum	7.9E-01	20	0.25	0.06	2E-04	3E-05	0.009
Nickel	1.6E-01	150	0.06	0.06	7E-05	4E-05	0.02
Sulfate	3.1E+03	NA	NA	NA	a	--	--
Uranium	1.2E+00	450	0.0085	0.004	2E-04	6E-05	0.008
Vanadium	3.3E-02	1000	0.0055	0.003	9E-06	3E-06	0.01
Carcinogenic effects							
Arsenic	1.4E-02	200	0.04	0.006	2E-06	2E-07	0.01
Uranium	8.2E+02[b]	450	0.0085	0.004	1E+05[c]	3E+04[c]	0.005

[a]Value cannot be calculated because no Kd, Bv, and Br values are available.
[b]In picocuries per liter.
[c]In picocuries per lifetime.

mg/L - milligrams per liter.
L/kg - liters per kilogram.
mg/kg - milligrams per kilogram.
NA - not available.

0.45 mg/g

see reference

Table 4.2 Exposure dose calculations for ingesting ground water-irrigated garden produce, Riverton, Wyoming, site (Concluded)

Equation definitions for exposure dose calculations

Ingestion of garden produce irrigated with ground water - chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

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

Where:

Cw	=	Arithmetic mean of concentrations in filtered water samples from well 707, collected from 1987 to 1993.
Kd	=	Soil-water partition coefficient (liter per kilogram) (Baes et al., 1984).
Bv	=	Soil-to-plant concentration ratio for vegetative portions of plants (unitless) (Baes et al., 1984).
Br	=	Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless) (Baes et al., 1984).
*	=	Exposure doses due to vegetative portions and reproductive portions of garden produce are calculated separately.
DF	=	Dry weight fraction of plant (0.066; unitless).
IRp	=	Ingestion rate for garden produce (0.05 kg per day for vegetative parts; 0.03 kg per day for reproductive parts).
FI	=	*Fraction of garden produce ingested from contaminated source (1.0; unitless).*
EF	=	Exposure frequency (365 days per year).
ED	=	Exposure duration (30 years for an adult).
BW	=	Body weight (70 kg for an adult).
AT	=	Averaging time (365 days x ED for noncarcinogens, 365 x 70 years for carcinogens).

potential for exposure from this route, since it is assumed that this garden would be the source of all garden produce in the diet. The results of this screening show that for contaminants of potential concern, ingestion of vegetables and fruit irrigated with contaminated ground water would lead to potential exposures of less than 3 percent of the levels associated with drinking contaminated water. Therefore, this pathway is eliminated from detailed evaluation and is discussed qualitatively in Section 6.0.

4.2.4 Ingestion of meat from ground water-fed livestock

The meat-ingestion pathway was evaluated in relation to the drinking water-ingestion route. The results of the screening (Table 4.3) show that for the contaminants of potential concern in ground water, the beef-ingestion pathway would produce a potential exposure of less than 6 percent of that associated with the ground water-ingestion pathway. Therefore, this pathway is eliminated from further evaluation. The potential increment of meat-ingested sulfate to drinking water exposure is discussed in detail in Section 6.3. Further evaluation of the direct toxicity to livestock is presented in Section 7.0.

4.2.5 Ingestion of milk from ground water-fed livestock

The milk-ingestion pathway was similarly evaluated for its relative significance to the ground water-ingestion route. The results of the screening calculation (Table 4.4) indicate that for the contaminants of potential concern in ground water, the milk-ingestion pathway would lead to a potential exposure of less than 4 percent of that associated with the drinking water-ingestion pathway. Therefore, this pathway is eliminated from detailed evaluation and is evaluated qualitatively in Section 6.0.

4.2.6 Incidental ingestion of surface water while swimming and ingestion of fish from the Little Wind River

These pathways were evaluated using the maximum concentration for each contaminant of potential concern detected in the Little Wind River. Surface water samples from the stream, ditches, and wetland areas near the tailings site are not considered in this evaluation because swimming and fishing are considered unlikely in these areas. Table 4.5 identifies the maximum concentration detected for each constituent in the Little Wind River and the sampling location where these concentrations were found. Table 4.5 also presents potential exposure doses for incidental ingestion of surface water and fish. When ingestion exposure doses calculated for surface water and fish ingestion scenarios are compared with nutritional and dietary intake ranges and acceptable daily intakes, the comparison indicates that the ingestion doses fall more than 2 orders of magnitude below these recommended intakes. Thus, these exposure routes were eliminated from further quantitative evaluation.

Table 4.3 Exposure dose calculations for ingesting meat from ground water-fed cattle, Riverton, Wyoming, site

Contaminant of concern	Cw (mg/L)	Kd (L/kg)	Cs (mg/kg)	Bv	Cp (mg/kg)	Ff (kg/day)	Cb (mg/kg)	Beef ingestion exposure dose (mg/kg-day)	Ratio of beef ingestion to ground water ingestion
Noncarcinogenic effects									
Arsenic	1.4E-02	200	2.8E+00	0.04	1.1E-01	2.0E-03	8E-03	9E-06	0.02
Manganese	4.9E+00	65	3.2E+02	0.25	7.9E+01	4.0E-04	7.6E-01	8E-04	0.008
Molybdenum	7.9E-01	20	1.6E+01	0.25	3.9E+00	6.0E-03	7.5E-01	8E-04	0.04
Nickel	1.6E-01	150	2.4E+01	0.06	1.4E+00	6.0E-03	2.7E-01	3E-04	0.06
Sulfate	3.1E+03	NA	a	NA	--	NA	--	--	--
Uranium	1.2E+00	450	5.4E+02	0.0085	4.6E+00	2.0E-04	7.2E-02	8E-05	0.003
Vanadium	3.3E-02	1000	3.3E+01	0.0055	1.8E-01	2.5E-03	4.5E-02	5E-05	0.05
Carcinogenic effects									
Arsenic	1.4E-02	200	2.8E+00	0.04	1.1E-01	2.0E-03	8E-03	4E-06	0.02
Uranium	8.2E+02[b]	450	3.7E+05	0.0085	3.1E+03	2.0E-04	4.9E+01	4E+04[c]	0.002

[a]Value cannot be calculated.
[b]In picocuries per liter.
[c]In picocuries per lifetime.
NA - not available.

Table 4.3 Exposure dose calculations for ingesting meat from ground water-fed cattle, Riverton, Wyoming, site (Concluded)

Equation definitions for exposure dose calculations

Ingestion of meat from ground water-fed cattle

Chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

$$\text{Lifetime intake (pCi/lifetime)} = Cb \times IRb \times FI \times EF \times ED$$
$$Cb = Ff\,([Cw \times Qw] + [Cs \times Qs] + [Cp \times Qp])$$
$$Cs = Cw \times Kd$$
$$Cp = Cs \times Bv$$

Where:

- Cb = Contaminant concentration in beef (milligrams per kilogram or picocuries per kilogram).
- IRb = Ingestion rate for beef (0.075 kg per day for an adult).
- FI = Fraction ingested from contaminated source (1.0 unitless).
- EF = Exposure frequency (365 days per year).
- ED = Exposure duration (30 years for an adult).
- BW = Body weight (70 kg for an adult).
- AT = Averaging time (365 days per year x ED for noncarcinogens; 365 days per year x 70 for carcinogens).
- Ff = Chemical-specific ingestion-to-beef transfer coefficient (unitless) (Baes et al., 1984).
- Cw = Arithmetic mean of concentrations in filtered water samples from well 707, collected from 1987 to 1993.
- Qw = Quantity of ground water consumed daily by cattle (56 L per day).
- Cs = Contaminant concentration in soil (milligrams per kilogram or picocuries per kilogram).
- Qs = Quantity of soil ingested daily by cattle (0.38 kg per day).
- Cp = Contaminant concentration in plant material consumed by cattle (milligrams per kilogram or picocuries per kilogram).
- Qp = Quantity of plant material consumed daily by cattle (19 kg per day).
- Kd = Chemical-specific soil-to-water distribution coefficient (unitless) (Baes et al., 1984).
- Bv = Chemical-specific soil-to-plant concentration factor (unitless) (Baes et al., 1984).

Table 4.4 Exposure dose calculations for ingesting milk from ground water-fed cattle, Riverton, Wyoming, site

Contaminant of concern	Cw (mg/L)	Kd (L/kg)	Cs (mg/kg)	Bv	Cp (mg/kg)	Fm (kg per day)	Cm (mg/kg)	Milk ingestion exposure dose (mg/kg-day)	Ratio of milk ingestion to ground water ingestion
Noncarcinogenic effects									
Arsenic	1.4E-02	200	2.8E+00	0.04	1.1E-01	6.0E-05	2.4E-04	1E-06	0.003
Manganese	4.9E+00	65	3.2E+02	0.25	7.9E+01	3.5E-04	6.7E-01	3E-03	0.03
Molybdenum	7.9E-01	20	1.6E+01	0.25	3.9E+00	1.5E-03	1.9E-01	8E-04	0.04
Nickel	1.6E-01	150	2.4E+01	0.06	1.4E+00	1.0E-03	4.5E-02	2E-04	0.04
Sulfate	3.1E+03	NA	a	NA	--	NA	--	--	--
Uranium	1.2E+00	450	5.4E+02	0.0085	4.6E+00	6.0E-04	2.2E-01	9E-04	0.03
Vanadium	3.3E-02	1000	3.3E+01	0.0055	1.8E-01	2.0E-05	3.6E-04	2E-06	0.001
Carcinogenic effects									
Arsenic	1.4E-02	200	2.8E+00	0.04	1.1E-01	6.0E-05	2.4E-04	4E-07	0.003
Uranium	8.2E+02[b]	450	3.7E+05	0.0085	3.1E+03	6.0E-04	1.5E+02	5E+05[c]	0.025

[a]Value cannot be calculated.
[b]In picocuries per liter.
[c]Picocuries per lifetime.
NA - not available.

Table 4.4 Exposure dose calculations for ingesting milk from ground water-fed cattle, Riverton, Wyoming, site (Concluded)

Equation definitions for exposure dose calculations

Ingestion of milk from ground water-fed cattle

$$\text{Chemical carcinogens and noncarcinogens (mg/kg-day)} = \frac{Cm \times IRm \times FI \times EF \times ED}{BW \times AT}$$

Radionuclide carcinogens

$$\text{Lifetime intake (pCi/lifetime)} = Cm \times IRm \times FI \times EF \times ED$$
$$Cm = Fm\,([Cw \times Qw] + [Cs \times Qs] + [Cp \times Qp])$$
$$Cs = Cw \times Kd$$
$$Cp = Cs \times Bv$$

Where:

Cm	=	Contaminant concentration in milk (milligrams per liter or picocuries per liter).
IRm	=	Ingestion rate for milk (0.30 kg per day for an adult).
FI	=	Fraction ingested from contaminated source (0.1 unitless).
EF	=	Exposure frequency (365 days per year).
ED	=	Exposure duration (30 years for an adult).
BW	=	Body weight (70 kg for an adult).
AT	=	Averaging time (365 days per year x ED for noncarcinogens; 365 days per year x 70 for carcinogens).
Fm	=	Chemical-specific ingestion-to-milk transfer coefficient (unitless) (Baes et al., 1984).
Cw	=	Arithmetic mean of concentrations in filtered water samples from well 707, collected from 1987 to 1993.
Qw	=	Quantity of ground water consumed daily by cattle (56 L per day).
Cs	=	Contaminant concentration in soil (milligrams per kilogram or picocuries per kilogram).
Qs	=	Quantity of soil ingested daily by cattle (0.38 kg per day).
Cp	=	Contaminant concentration in plant material consumed by cattle (milligrams per kilogram or picocuries per kilogram).
Qp	=	Quantity of plant material consumed daily by cattle (19 kg per day).
Kd	=	Chemical-specific soil-to-water distribution coefficient (unitless) (Baes et al., 1984).
Bv	=	Chemical-specific soil-to-plant concentration factor (unitless) (Baes et al., 1984).

Table 4.5 Exposure dose calculations for ingesting incidental surface water and fish from the Little Wind River, Riverton, Wyoming, site

Constituent	Csw (mg/L)	Location ID	BCF (L/kg)	Cf (mg/kg)	Surface water ingestion exposure dose (mg/kg-day)	Fish ingestion exposure dose (mg/kg-day)
Noncarcinogenic effects						
Calcium	4.2E+01	796	NA	NA	2E-03	NA
Iron	3.1E+00	796	100	3.1E+02	1E-04	6E-02
Magnesium	1.4E+01	796	NA	NA	5E-04	NA
Manganese	1.4E-01	796	NA	NA	5E-06	NA
Uranium	2.5E-02	742	2.0	5.0E-02	9E-07	1E-05
Carcinogenic effects						
Uranium	1.7E+01[a]	742	2.0	3.4E+01[b]	5E+02[c]	5E+03[c]

[a]In picocuries per liter.
[b]In picocuries per kilogram.
[c]In picocuries per lifetime.
BCF—bioconcentration factor (EPA, 1992a; NUREG, 1986).
NA—BCF not available for this contaminant.

Equation definitions for exposure dose calculations

Incidental surface water ingestion during swimming

Chemical carcinogens and noncarcinogens

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

$$\text{Lifetime intake (pCi/lifetime)} = Csw \times CR \times ET \times EF \times ED$$

Where:

Csw = Maximum detected constituent concentration in unfiltered surface water at downstream locations from the 1993 sampling round (milligrams per liter or picocuries per liter).
CR = Contact rate (amount ingested) during swimming (0.05 L per hour).
ET = Exposure time per swimming event (2.6 hours per event).
EF = Exposure frequency (7 days per year).
ED = Exposure duration (30 years for an adult).
BW = Body weight (70 kg for an adult).
AT = Averaging time (365 days per year x ED for noncarcinogens; 365 days per year x 70 for carcinogens).

Table 4.5 Exposure dose calculations for ingesting incidental surface water and fish from the Little Wind River, Riverton, Wyoming, site (Concluded)

Ingestion of fish caught in the Little Wind River

Chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

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

Where:

- Cf = Constituent concentration in fish tissue (milligrams per kilogram or picocuries per kilogram). Calculated by multiplying the contaminant concentration in surface water (Csw) by the chemical-specific bioconcentration factor.
- IR = Fish ingestion rate (kilograms per day). Based on average consumption of two 8-ounce servings per week (0.054 kg per day for an adult).
- FI = Fraction ingested from contaminated source (0.25 unitless).
- EF = Exposure frequency (365 days per year).
- ED = Exposure duration (30 years for an adult).
- BW = Body weight (70 kg for an adult).
- AT = Averaging time (365 days per year x ED for noncarcinogens, 365 days per year x 70 for carcinogens).

4.2.7 Incidental ingestion of sediment during recreational activities

This exposure route was evaluated using the maximum concentration detected for each contaminant of potential concern in sediment sampled from either the Little Wind River or the other water bodies in the site vicinity. Table 4.6 identifies the maximum sediment concentration detected and the location where each concentration was measured. Table 4.6 also presents potential exposure doses for these concentrations. The incidental sediment ingestion route was eliminated from further evaluation because the calculated ingestion doses fall more than 3 orders of magnitude below nutritional and dietary ranges or acceptable daily intakes. Therefore, no adverse health effects would be expected from these levels.

4.3 EXPOSURE CONCENTRATIONS FOR DRINKING WATER

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, contaminant concentrations are assumed to stay in a steady state, even though site remediation should actually cause a decrease in exposure concentrations over time. Nonetheless, these estimates are reasonable for chronic exposure soon after remediation. (Chronic exposure for noncarcinogens is considered to be any period greater than 7 years.)

Exposure concentrations are evaluated as a probability of occurrence based on ground water data collected from monitor well 707 in the surficial aquifer between the Little Wind River and the former processing site (DOE, 1995). This well is used in this assessment because it has generally shown the highest contamination levels over the past 7 years of monitoring.

The probability distribution selected for a contaminant had approximately the same mean, median, standard deviation, and shape as were observed in the historical water quality data (DOE, 1995). The upper tail of the distributions were truncated at the 99th percentile. For every contaminant, the truncated upper limit concentration was higher than the maximum observed concentration in the historical water quality data. The software package @RISK was used to generate probability curves for the contaminants of potential concern. Results are shown in Figures 4.2 through 4.8. The expected values shown on the probability curves are the mean of the 10,000 iterations used to generate them and can be considered an average.

4.4 ESTIMATED DRINKING WATER INTAKE

Individuals within the population of future residents are expected to vary in water consumption habits, stable body weight, and length of time they reside in the affected area. Consequently, health risks associated with ground water consumption will also vary among members of this population. To adequately describe the range of potential risks to the future population, naturally occurring variabilities in daily water intake and body weight were incorporated into this assessment through probability distributions selected from published United States

Table 4.6 Exposure dose calculations for ingesting incidental sediment during recreational use of the Little Wind River and surrounding water bodies, Riverton, Wyoming, site

Constituent	Cs (mg/kg)	Location ID	Sediment ingestion exposure dose (mg/kg-day)
Noncarcinogenic effects			
Iron	2.12E+04	742	4E-05
Lead	6.77E+01	746	1E-07
Manganese	6.18E+02	742	1E-06
Molybdenum	9.00E+00	746	2E-08
Uranium	1.13E+01	746	2E-08
Carcinogenic effects			
Uranium	7.68E+03[a]	746	1E+01[b]

[a]In picocuries per kilogram.
[b]In picocuries per lifetime.

Equation definitions for exposure dose calculations

Chemical carcinogens and noncarcinogens

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

Radionuclide carcinogens

$$\text{Lifetime intake (pCi/lifetime)} = Cs \times IR \times EF \times ED \times CF \times ABS$$

Where:

- Cs = Maximum detected constituent concentration in sediments from the 1993 sampling round (milligrams per kilogram or picocuries per kilogram).
- IR = Ingestion rate of sediments (6.25 mg per day).
- EF = Exposure frequency (7 days per year).
- ED = Exposure duration (30 years for an adult).
- CF = Conversion factor (1.0E-06 kg per mg).
- ABS = Chemical-specific gastrointestinal absorption factor (default value of 1.0 for all metals).
- BW = Body weight (70 kg for an adult).
- AT = Averaging time (365 days per year x ED).

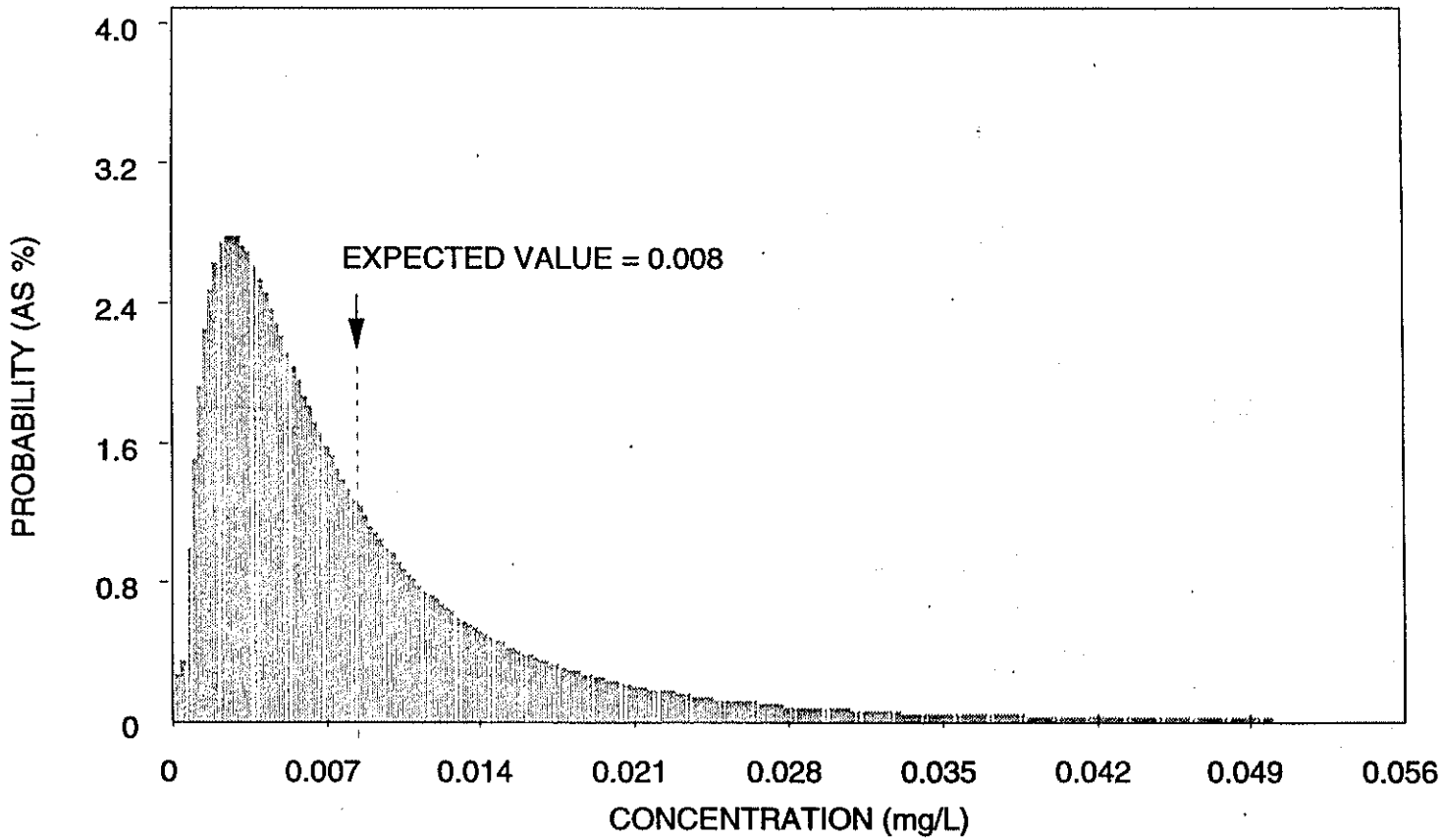


FIGURE 4.2
SIMULATED DISTRIBUTION OF ARSENIC CONCENTRATIONS
RIVERTON, WYOMING, SITE

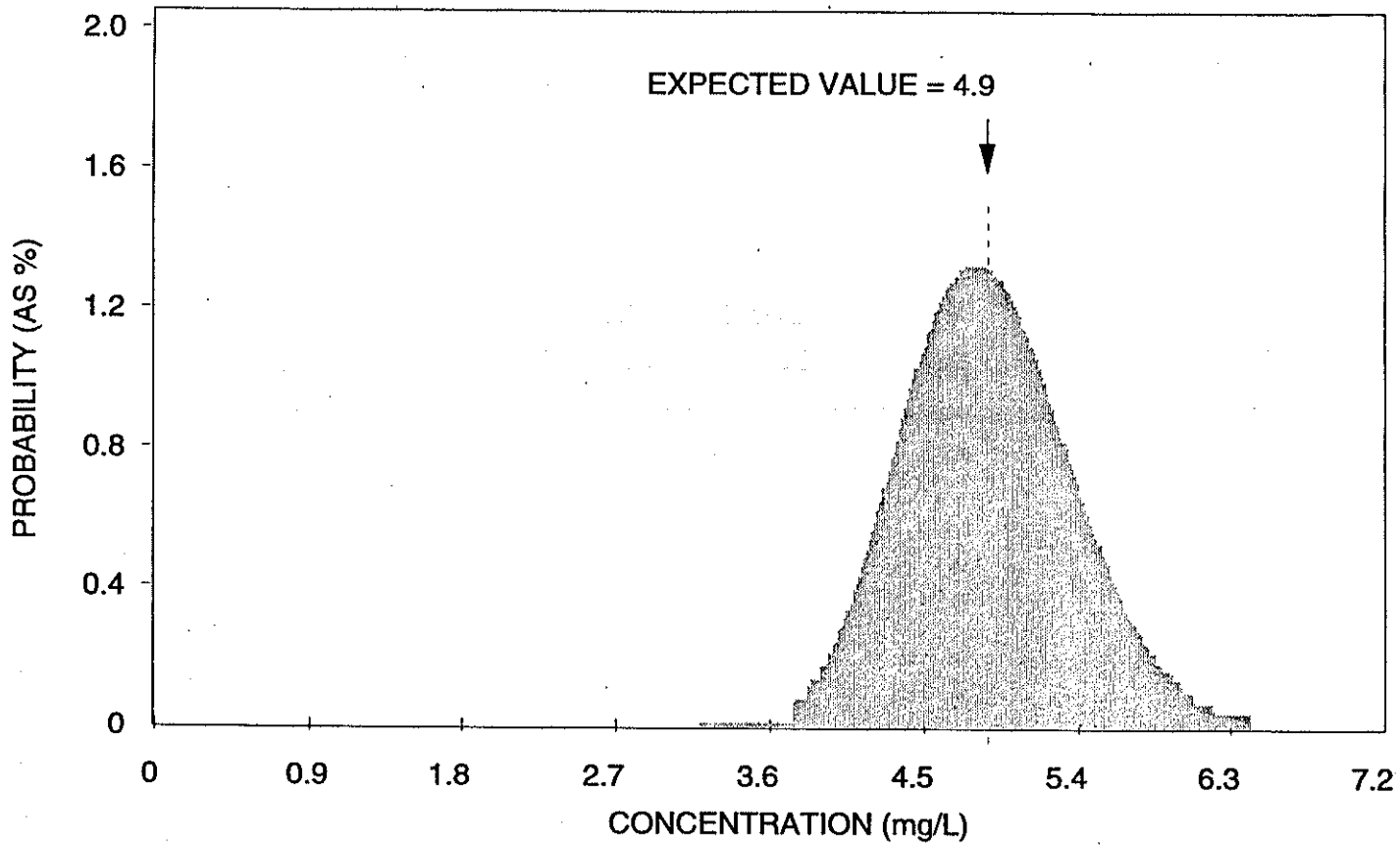


FIGURE 4.3
SIMULATED DISTRIBUTION OF MANGANESE CONCENTRATIONS
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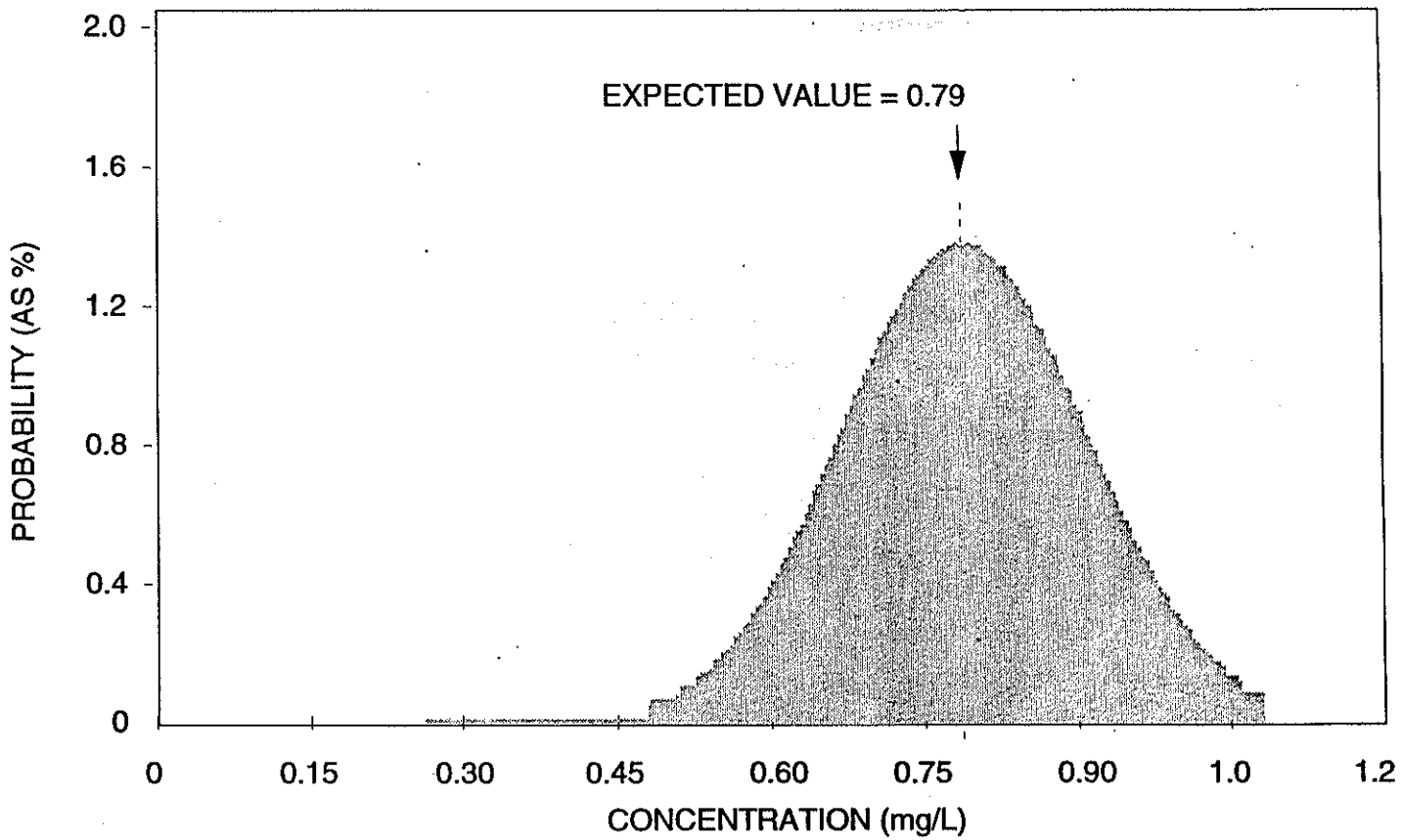

FIGURE 4.4
SIMULATED DISTRIBUTION OF MOLYBDENUM CONCENTRATIONS
RIVERTON, WYOMING, SITE

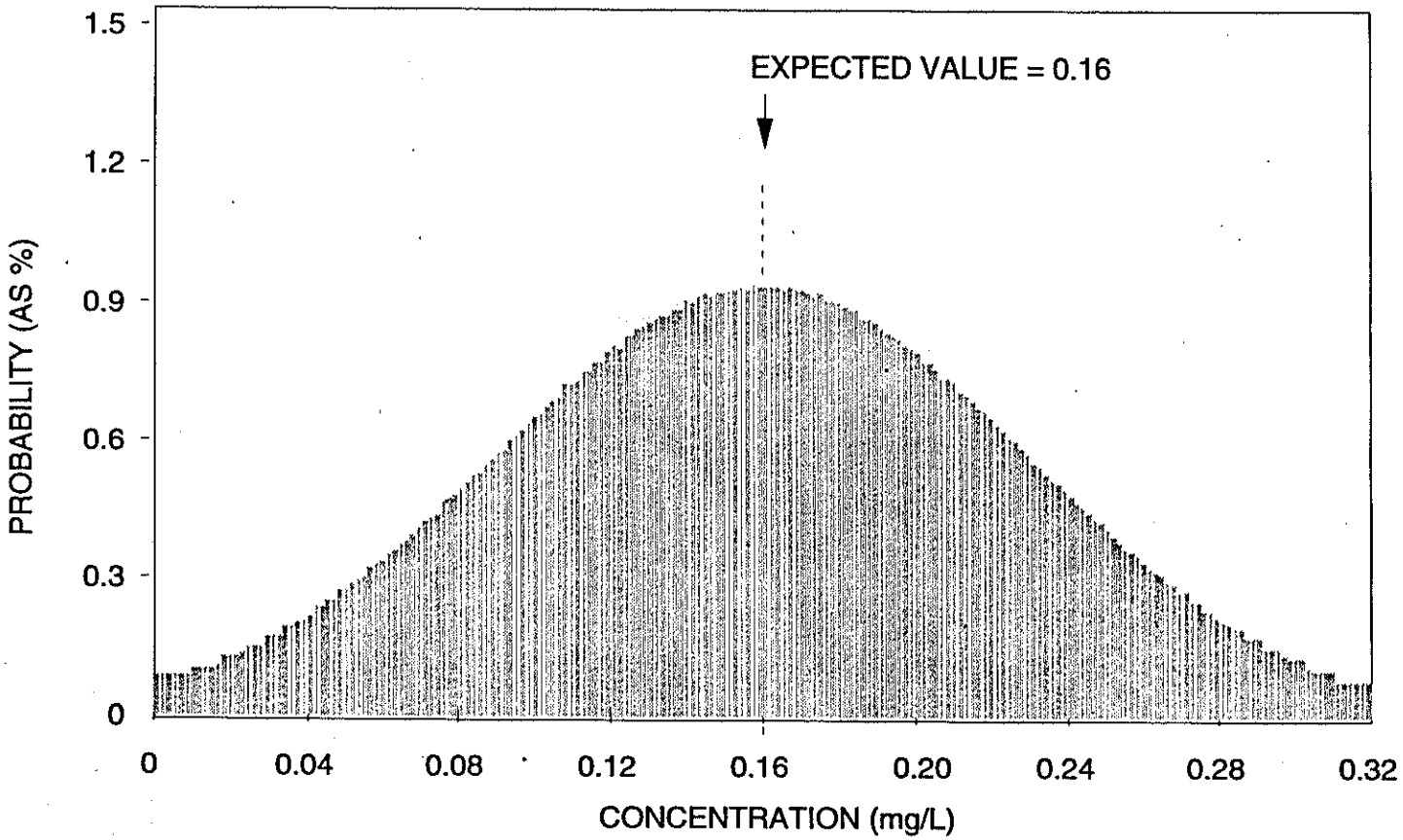

FIGURE 4.5
SIMULATED DISTRIBUTION OF NICKEL CONCENTRATIONS
RIVERTON, WYOMING, SITE

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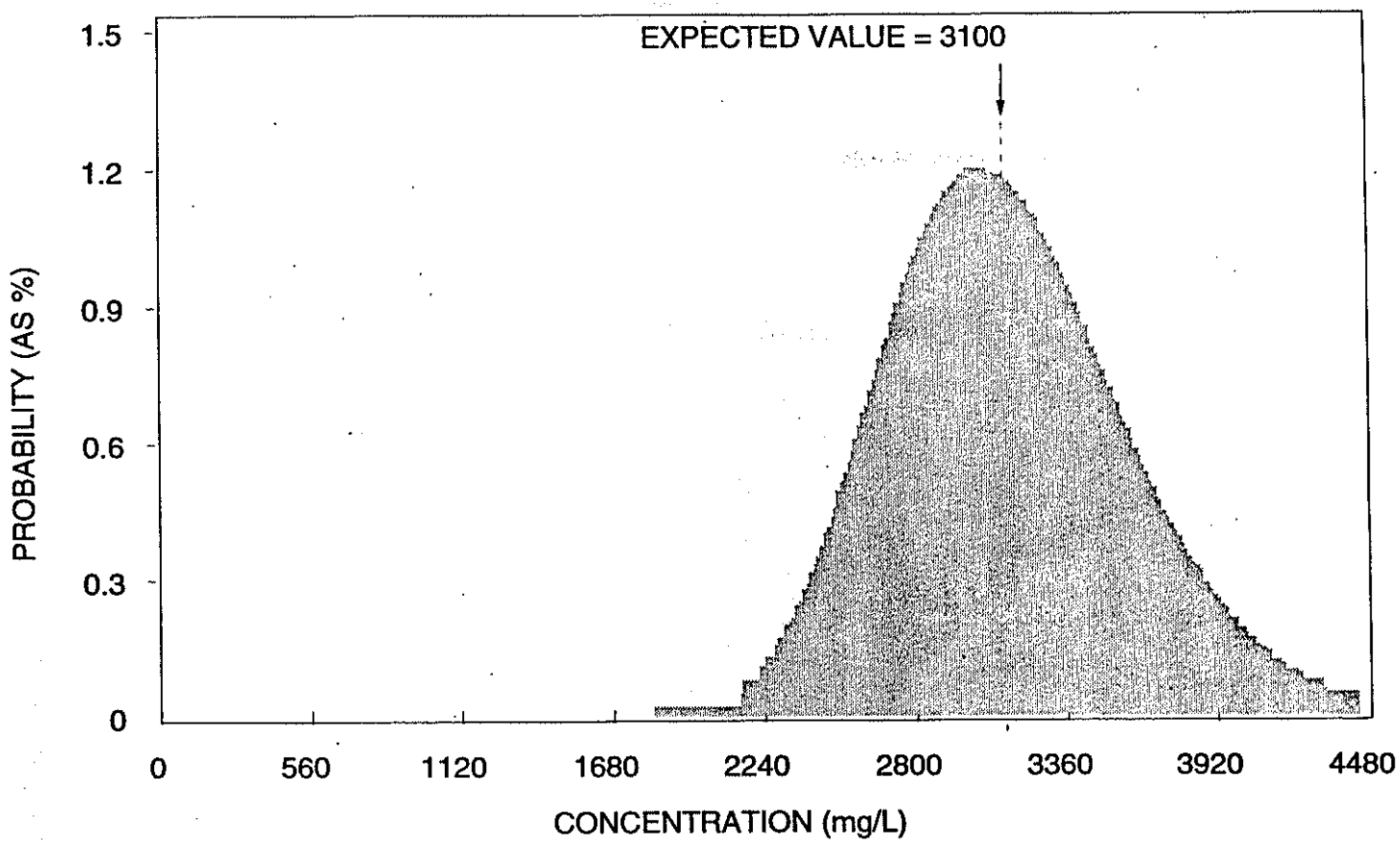

FIGURE 4.6
SIMULATED DISTRIBUTION OF SULFATE CONCENTRATIONS
RIVERTON, WYOMING, SITE

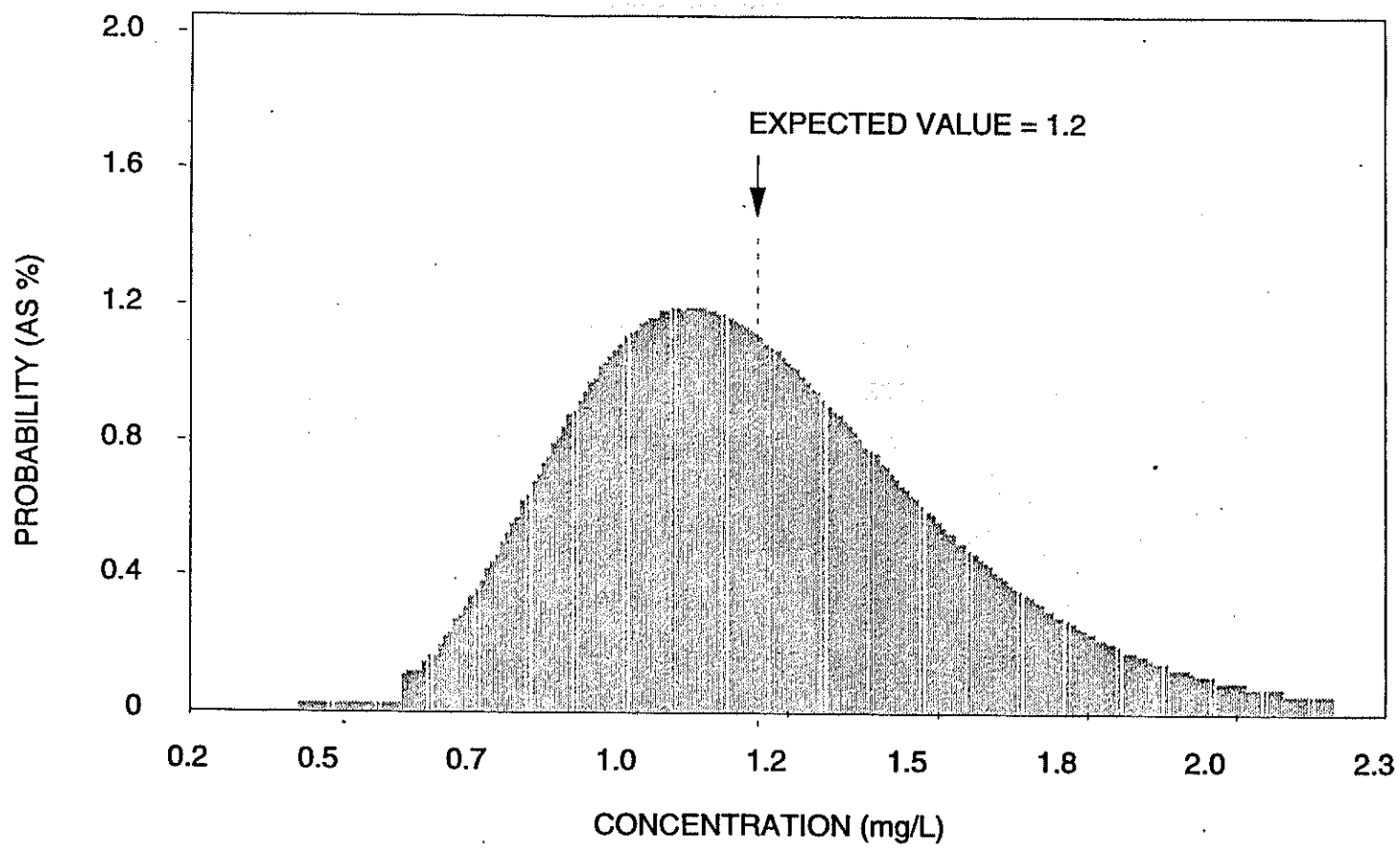

FIGURE 4.7
SIMULATED DISTRIBUTION OF URANIUM CONCENTRATIONS
RIVERTON, WYOMING, SITE

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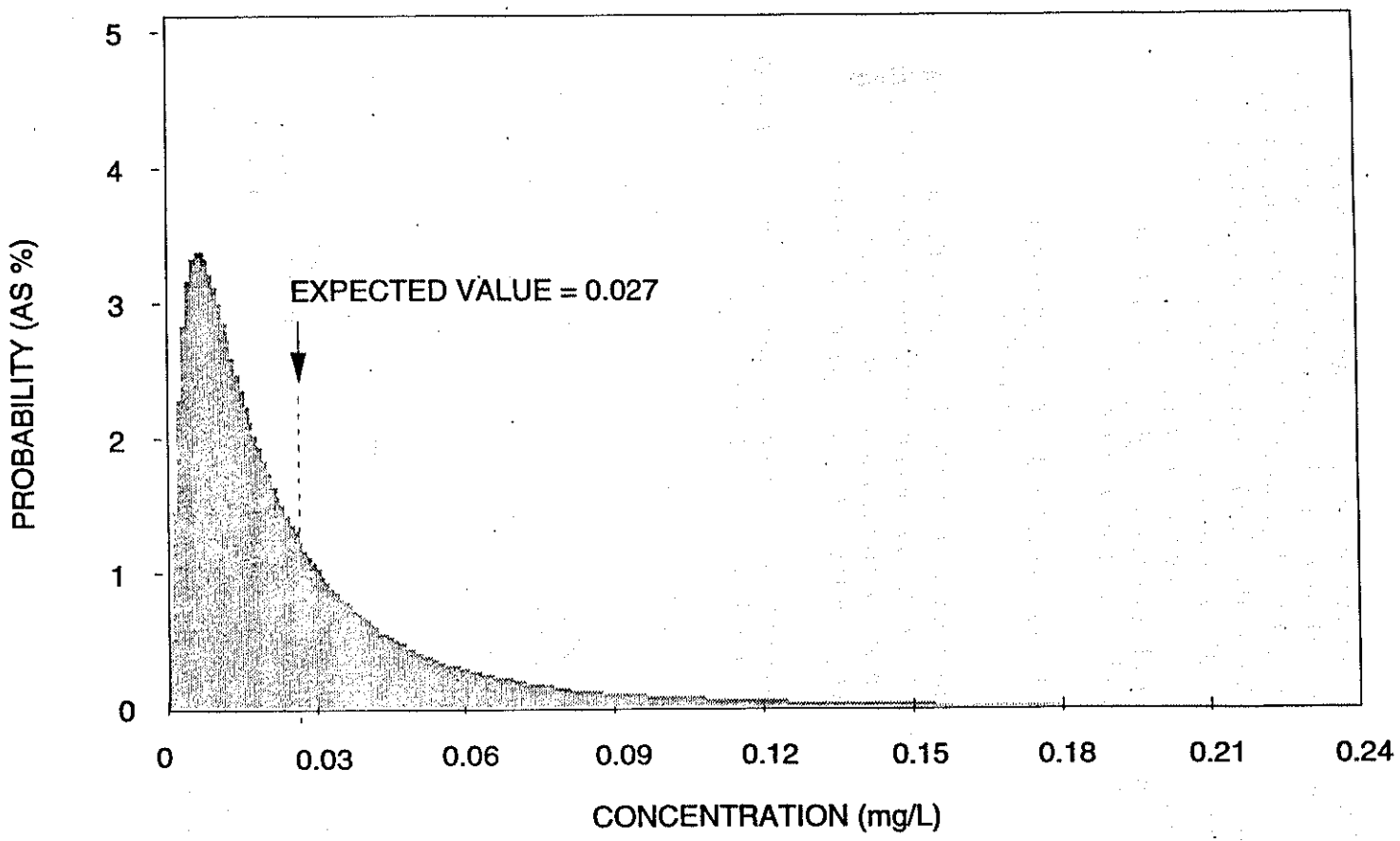


FIGURE 4.8
SIMULATED DISTRIBUTION OF VANADIUM CONCENTRATIONS
RIVERTON, WYOMING, SITE

public health and census documents. All distributions were truncated at the upper and lower 0.01 percentile (DOE, 1995). Within the hypothetical population, the probability of occurrence for values disallowed through this truncation is less than 1 in 10,000.

The potential toxicity of noncarcinogenic contaminants in drinking water and the potential carcinogenicity of arsenic depend primarily on long-term average daily consumption of the contaminant per kilogram of body weight (measured in milligrams per kilogram per day [mg/kg-day]). "Long-term" is defined as at least 7 years for noncarcinogens and 30 years for arsenic and radionuclide carcinogenicity. For noncarcinogens, exposure frequency multiplied by exposure duration in the numerator is assumed to cancel out with averaging time in the denominator; therefore, chronic daily intake is calculated as follows:

$$\text{Intake (mg/kg - day} = \frac{\text{Concentration (mg/L) x ingestion rate (L per day)}}{\text{Body weight (kg)}}$$

The potential carcinogenic risk from radionuclides increases with total intake over time. Also, the body weight factor is relatively insignificant in determining risk from exposure. Intake of a radioactive carcinogen is therefore quantified as total exposure to radioactivity throughout the individual's period of residence.

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

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

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

Average daily intake (L/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 National Food Consumption Survey conducted by the U.S. Department of Agriculture (DOE, 1995). During the survey, total tap water consumption during a 3-day period was recorded for 26,081 survey participants nationwide (Figure 4.9).

Exposure frequency (days per year)

Individuals are typically exposed fewer than 365 days per year because of time away from home (e.g., vacation). Exposure frequency is expected to be higher among potential future residents of Riverton than among the United States

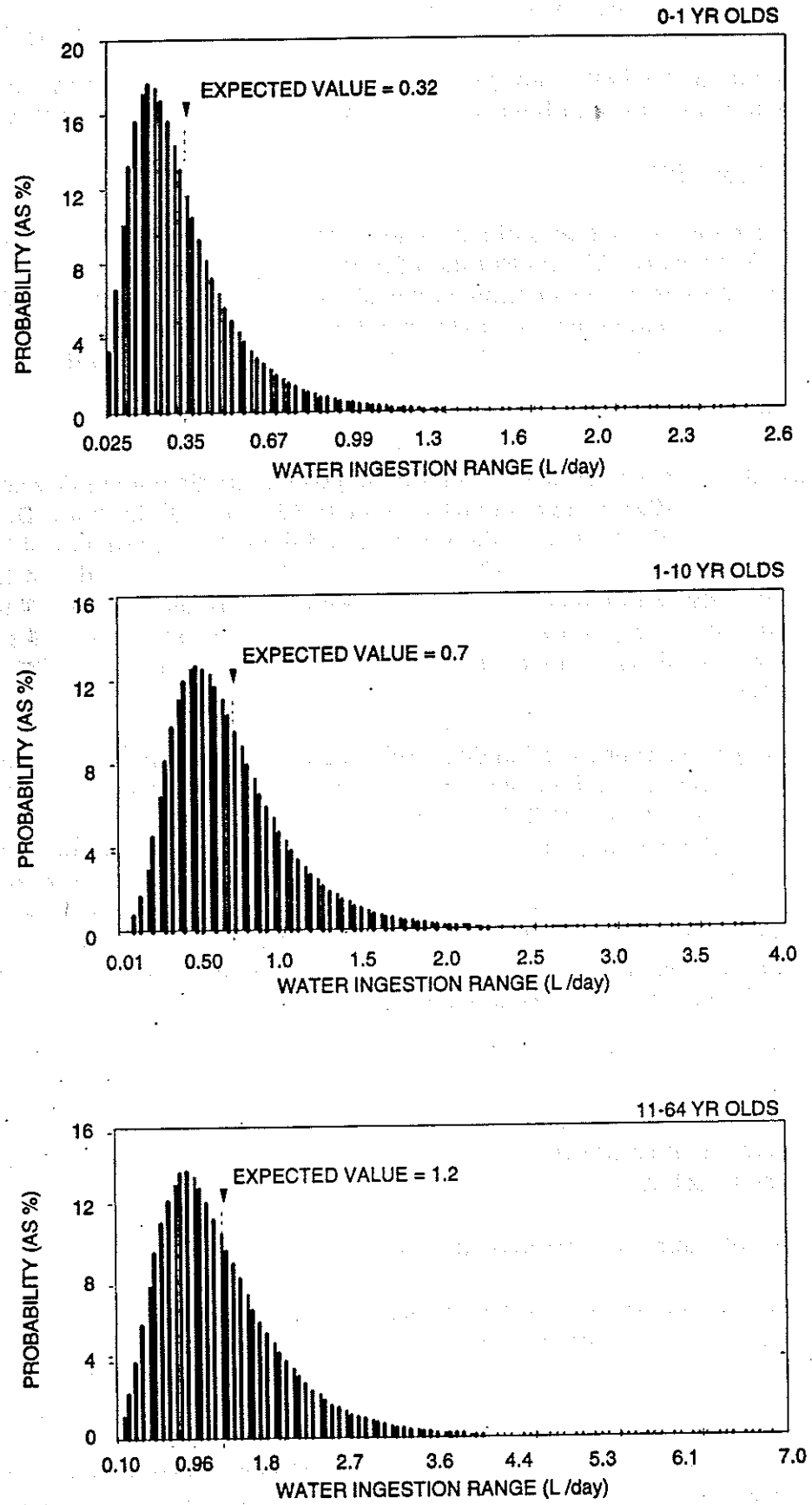


FIGURE 4.9
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES

population as a whole. However, in the absence of reliable information on this variable, exposure of 365 days per year was assumed throughout this assessment.

Body weight (kg)

Extensive data on weights of males and females, by age, were collected nationwide 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 and by sex. The distributions for males and females were combined, using census data on the national ratio of males to females within each age group (Figure 4.10).

Exposure duration (years)

Distributions of total residence time (or exposure duration) were developed by Israeli and Nelson (1992) using data collected in 1985 and 1987 by the U.S. Department of Commerce, the Bureau of the Census, and the U.S. Department of Housing and Urban Development (DOE, 1995). Israeli and Nelson determined that farm residents move less frequently than urban or rural residents. To estimate exposure to carcinogens, a 30-year exposure is assumed to be reasonable for the population in the Riverton risk assessment, as this future population is expected to be primarily agricultural.

Using exposure concentration distributions and the intake parameter distributions from Section 4.3, total uranium intake distributions were derived for the three populations (Figure 4.11) (DOE, 1995). These results indicate that the exposure group with the highest intake rate is children 1 to 10. Because this group receives the highest exposure per body weight, it is used to evaluate risks from all the contaminants of potential concern except sulfate. Because infants are the most susceptible receptors to sulfate toxicity, the intake distribution for this age group is used for sulfate. The simulated intakes for noncarcinogenic contaminants of potential concern are shown in Figures 4.12 through 4.18.

4.5 EXPOSURE ASSESSMENT UNCERTAINTIES

In any phase of an exposure assessment, several potential sources of error can lead to underestimation or overestimation of intake, including the meaningful sources of uncertainty below:

- Lack of thorough environmental sampling data.
- 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).
- The model used to estimate contaminant uptake by plants for the irrigated garden produce pathway. Plant uptake factors could vary substantially from the default estimates under the Riverton site conditions.

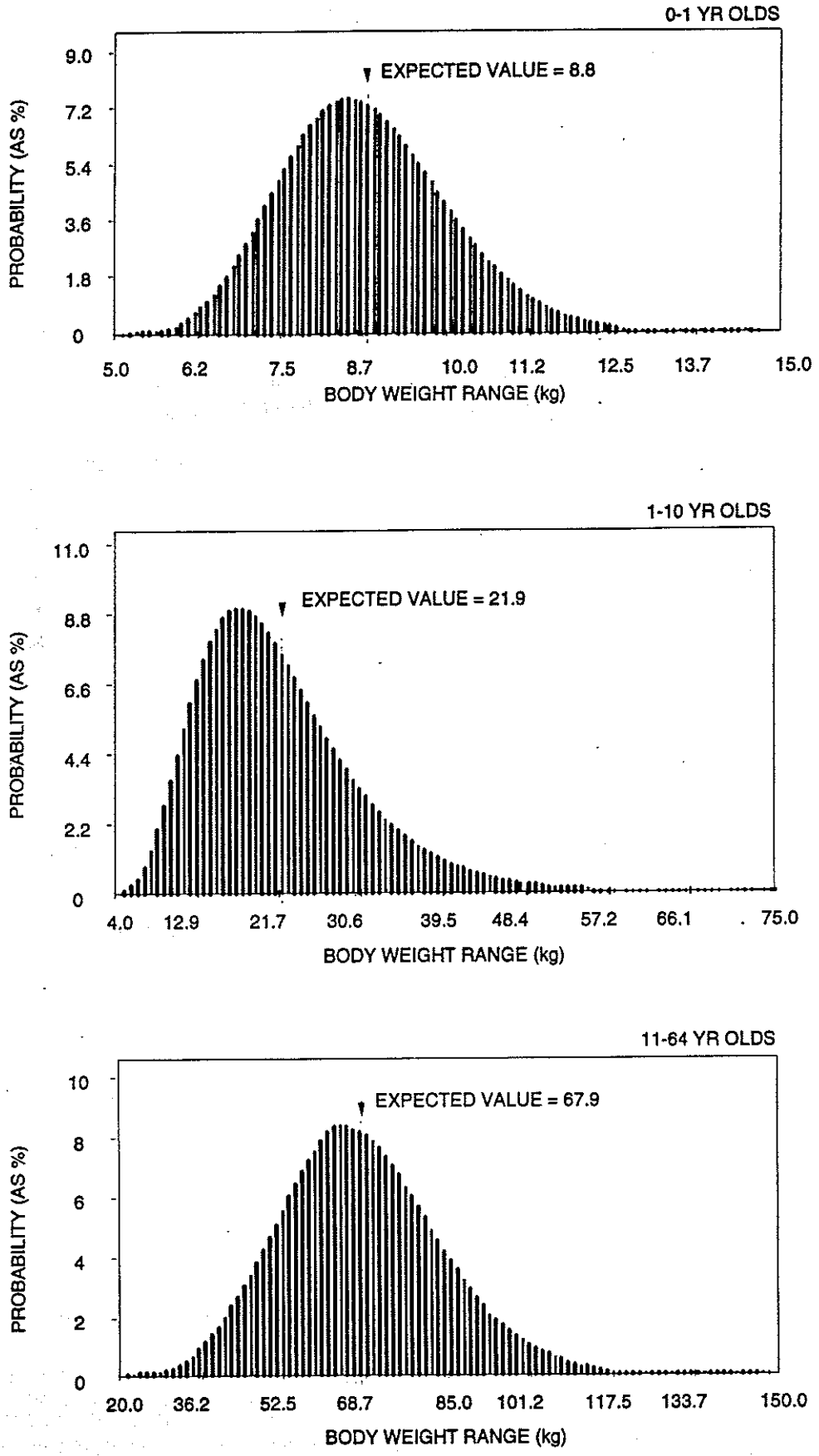
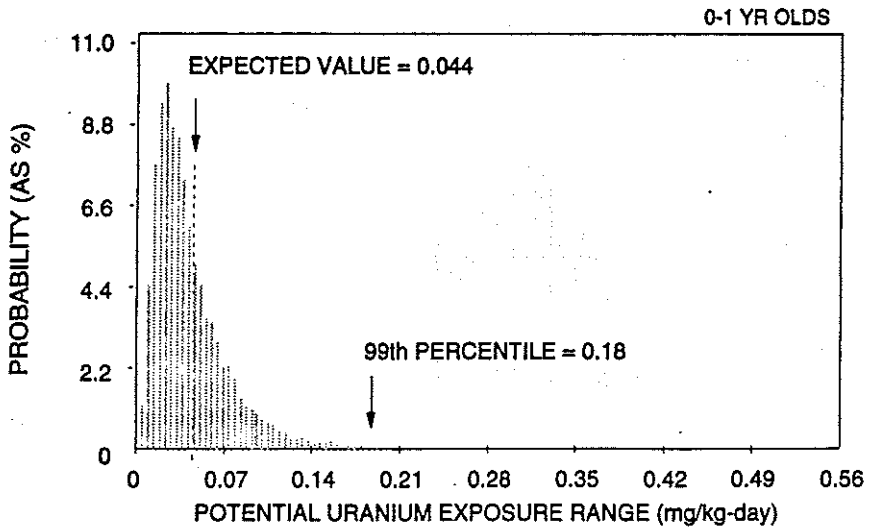
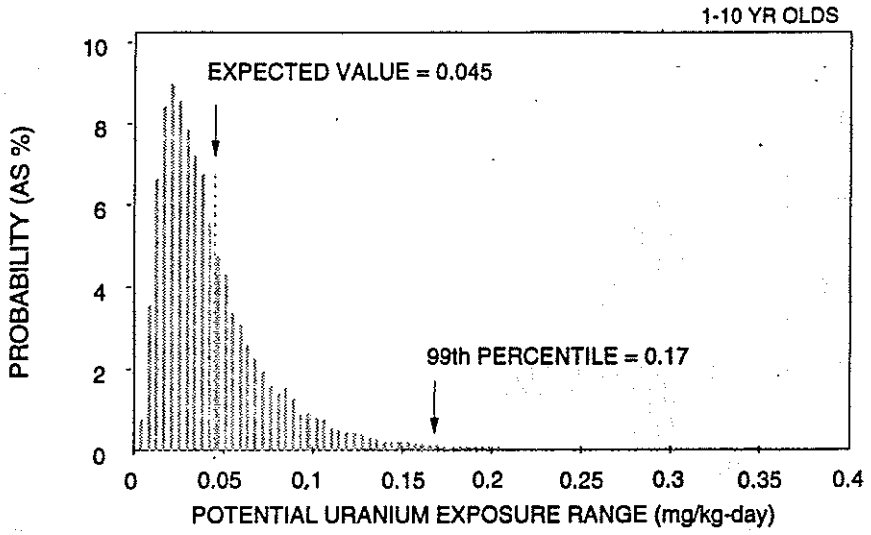
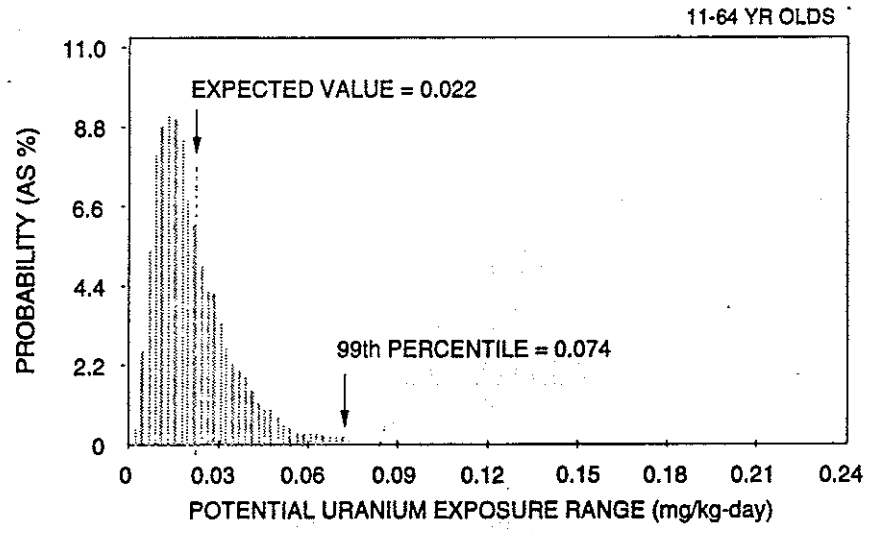


FIGURE 4.10
PROBABILITY DISTRIBUTIONS FOR BODY WEIGHT







NOTE: HIGHEST POTENTIAL EXPOSURE OCCURS IN CHILDREN AGE 1-10 (AVERAGE POTENTIAL EXPOSURE OF 0.045 mg URANIUM/kg BODY WEIGHT/DAY).

FIGURE 4.11

COMPARISON OF DISTRIBUTIONS OF POTENTIAL URANIUM EXPOSURES AS A RESULT OF DRINKING CONTAMINATED GROUND WATER FOR THE THREE AGE GROUPS EXAMINE. RIVERTON, WYOMING, SITE

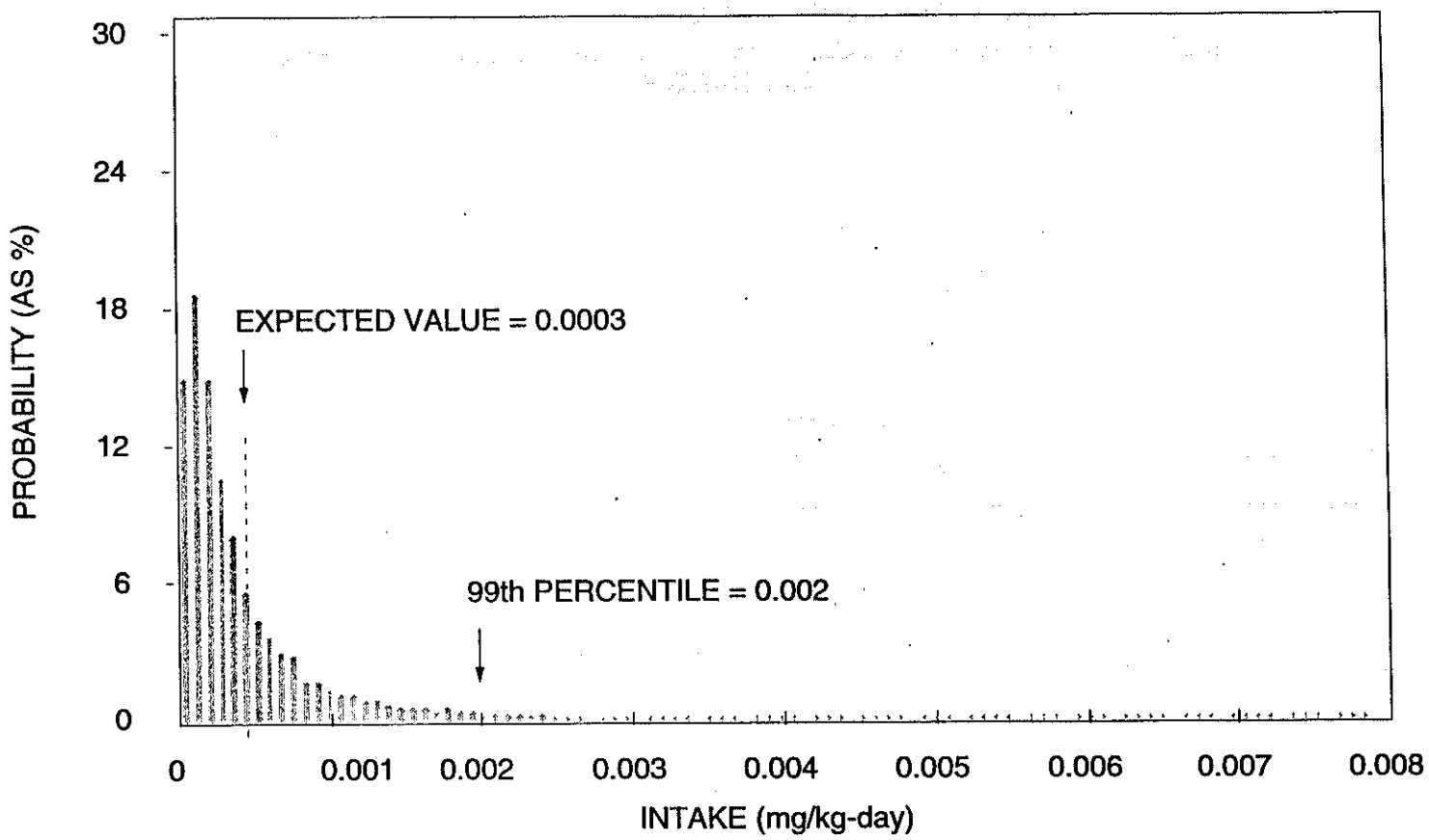


FIGURE 4.12
PROBABILITY DISTRIBUTION OF ARSENIC INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

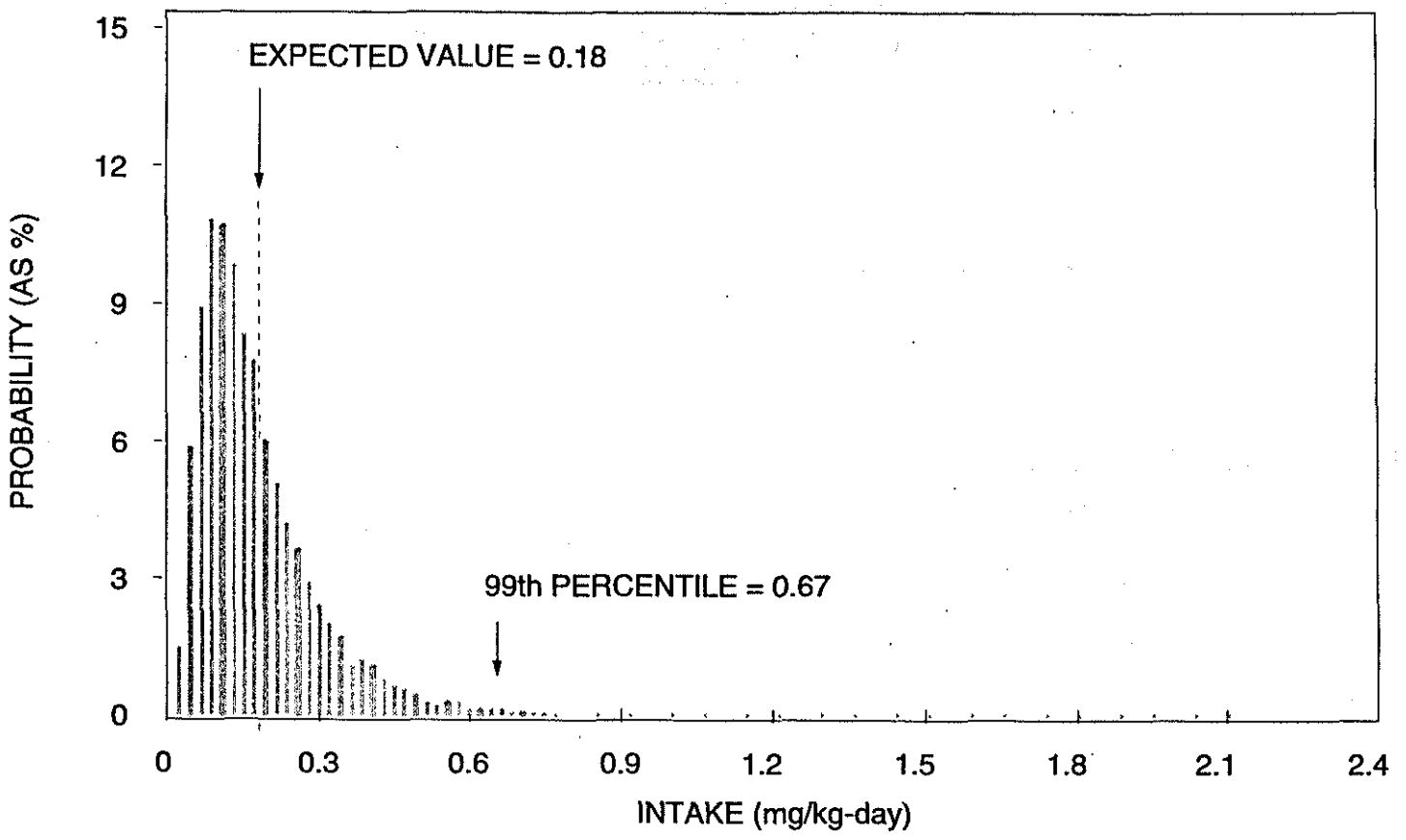


FIGURE 4.13
PROBABILITY DISTRIBUTION OF MANGANESE INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

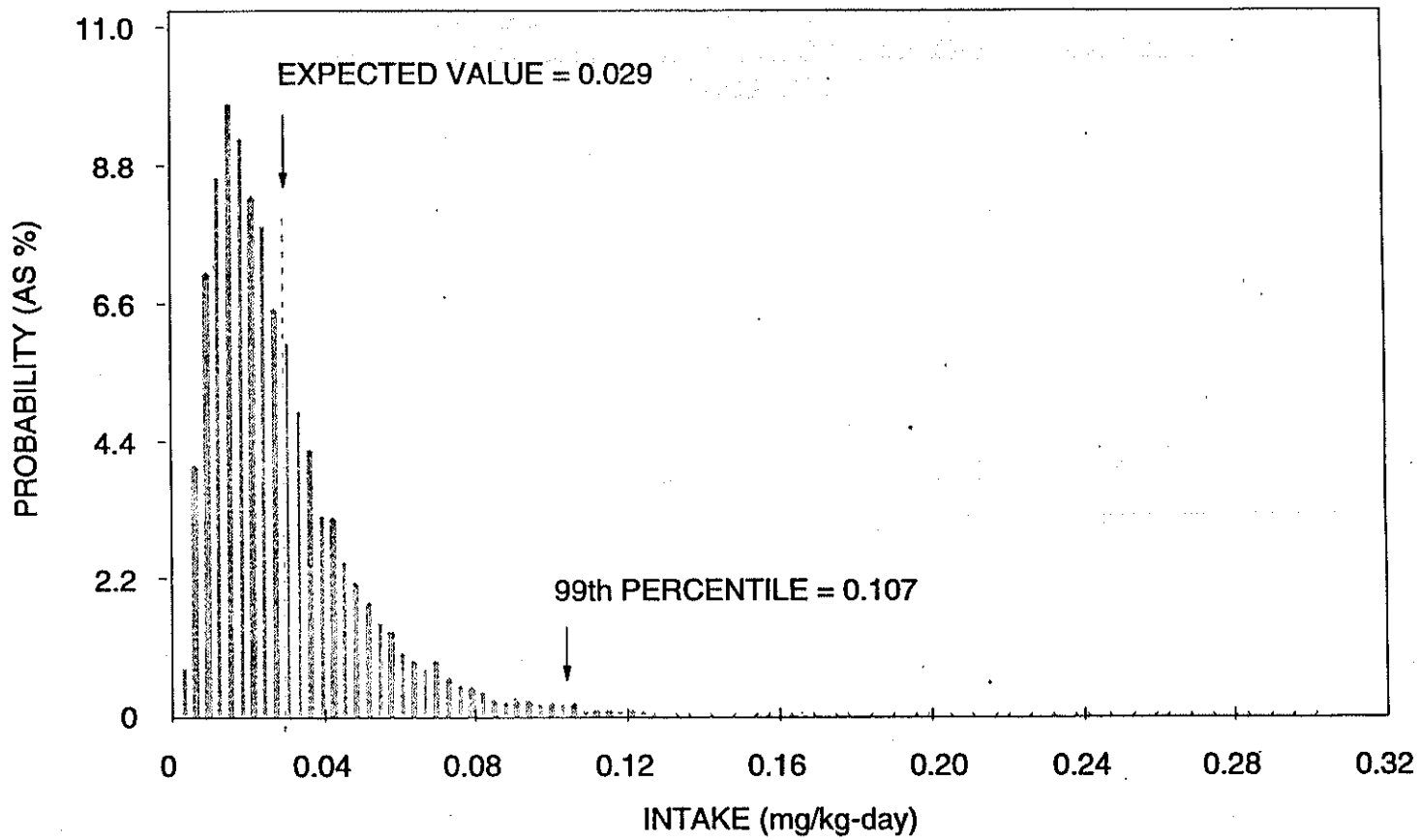


FIGURE 4.14
PROBABILITY DISTRIBUTION OF MOLYBDENUM INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

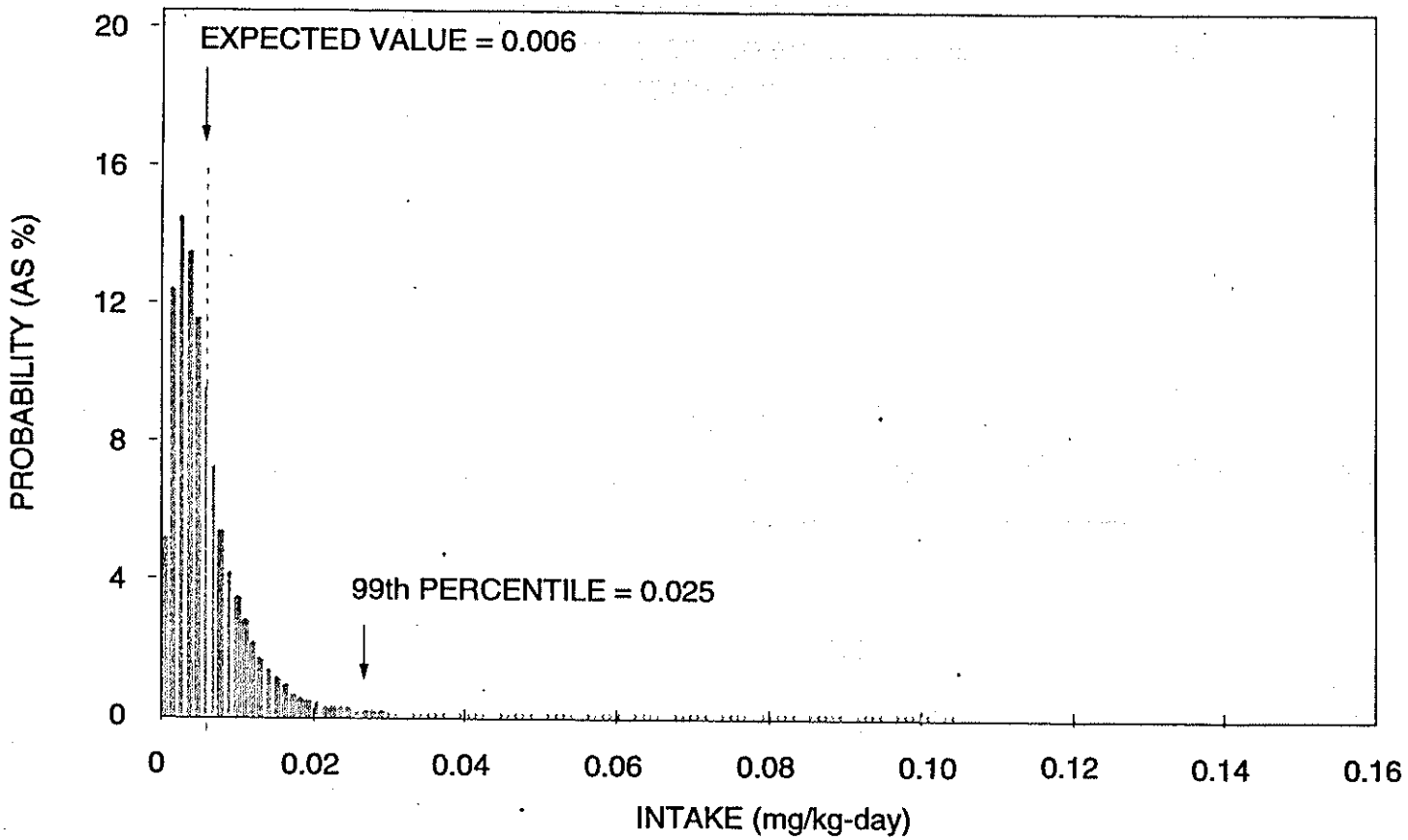


FIGURE 4.15
PROBABILITY DISTRIBUTION OF NICKEL INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

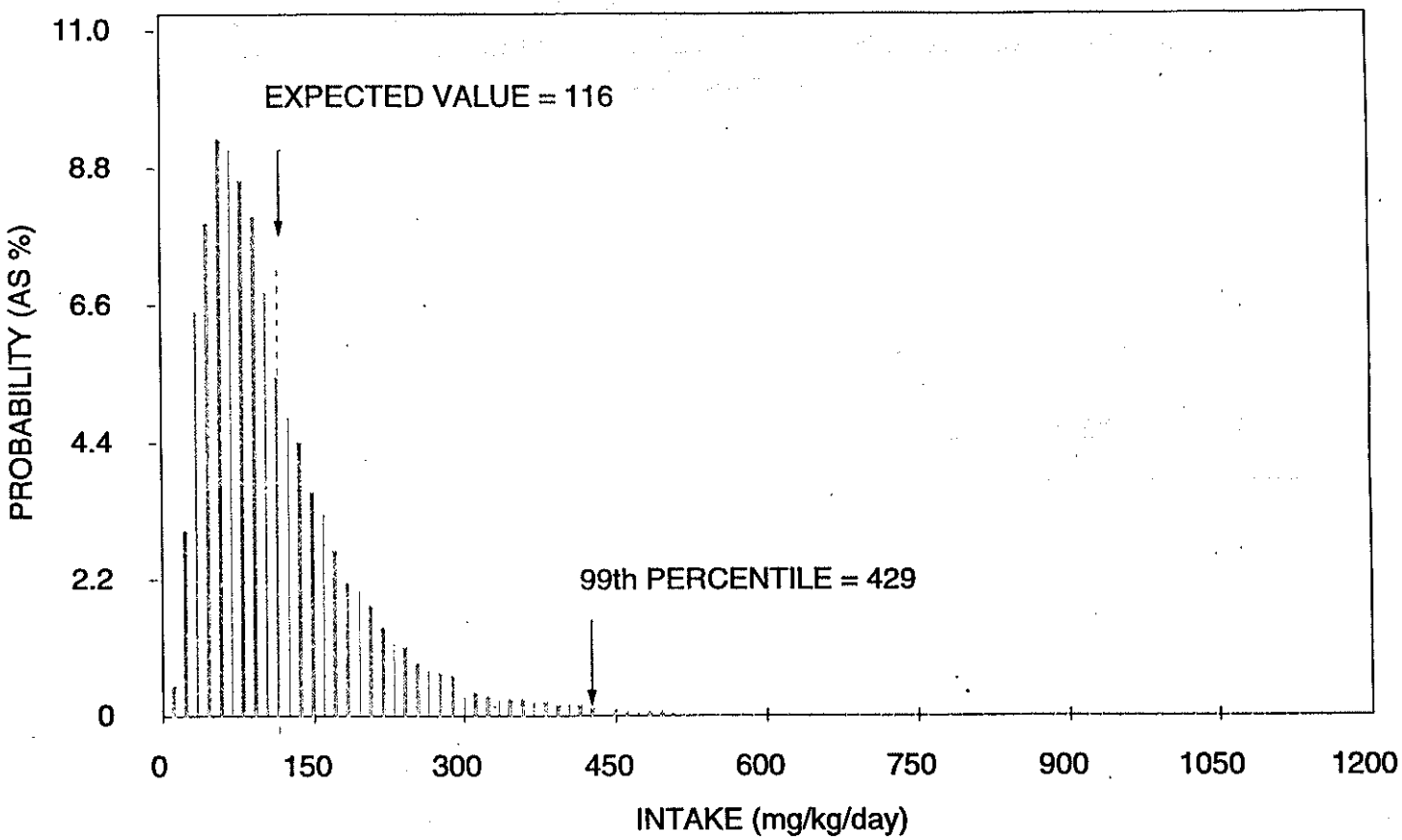


FIGURE 4.16
PROBABILITY DISTRIBUTION OF SULFATE INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

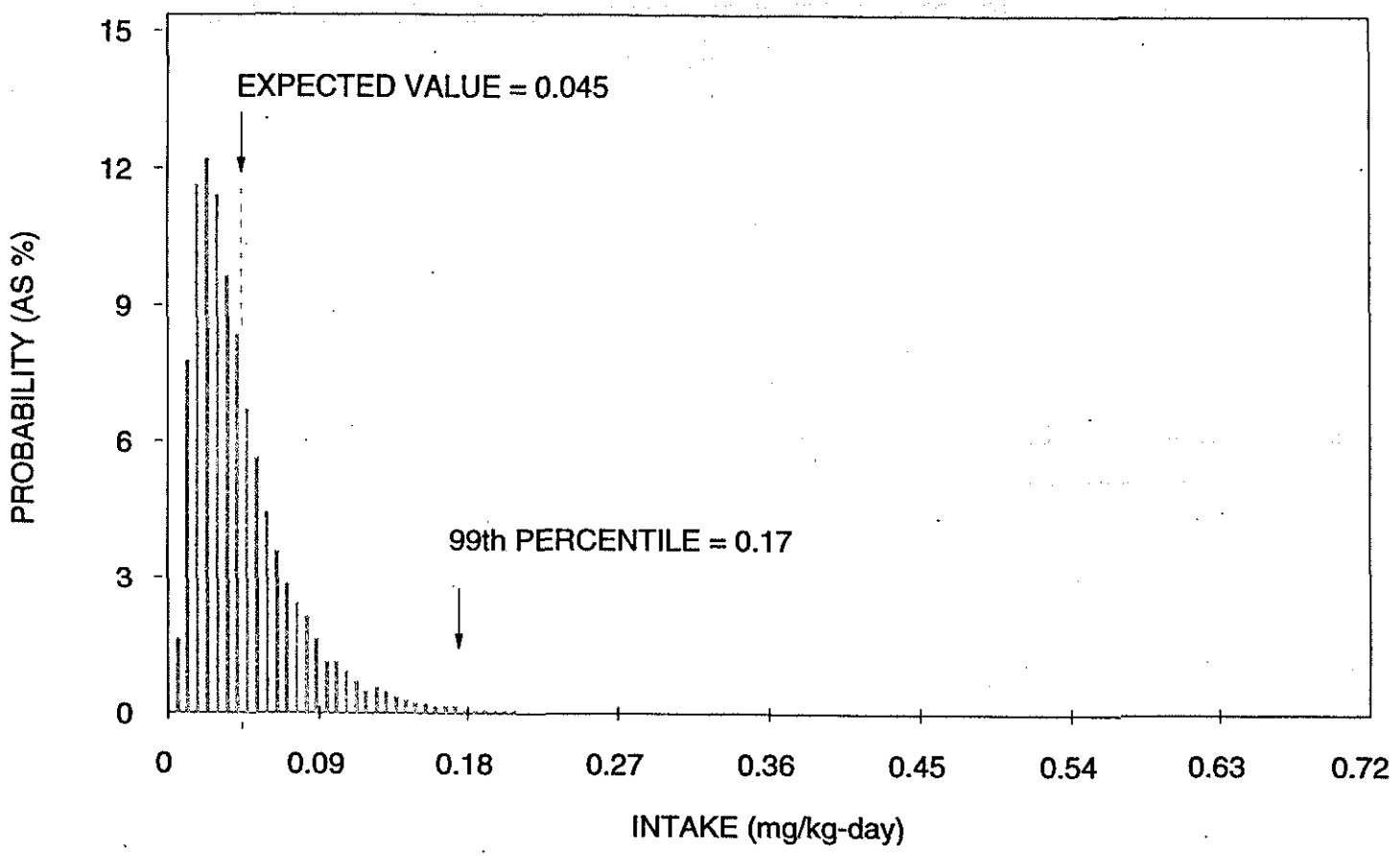


FIGURE 4.17
PROBABILITY DISTRIBUTION OF URANIUM INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

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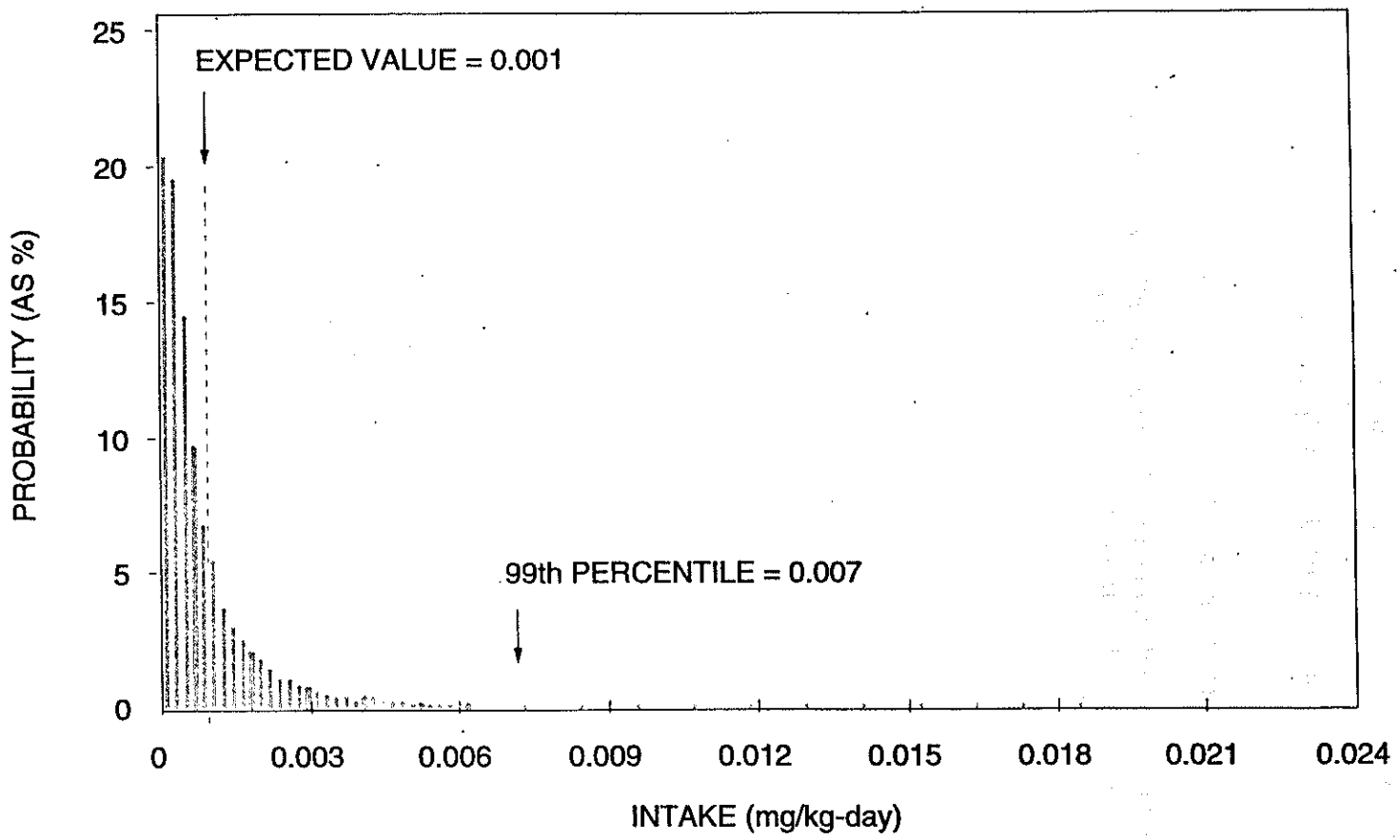

FIGURE 4.18
PROBABILITY DISTRIBUTION OF VANADIUM INTAKES FOR CHILDREN AS A RESULT OF GROUND WATER INGESTION RIVERTON, WYOMING, SITE

- The relationship between an applied and absorbed dose and the effective toxic dose for dermal absorption.

- The incidental rates of surface water and sediment ingestion from the river and other water bodies by humans during recreational use.

- The levels of contaminant intake by cattle and the extent of transfer to milk and muscle.

- The levels of contaminant uptake by fish.

Despite these uncertainties, probability distributions incorporating all definable sources of variability should provide a representative picture of the potential exposure range.

5.0 TOXICITY ASSESSMENT

Several contaminants that could cause adverse human health and environmental effects were detected in ground water at the Riverton site. This section summarizes the toxicological effects of the chemical contaminants and the carcinogenic potentials of arsenic and the radionuclides. Source materials used to develop these toxicological profiles include the EPA's Integrated Risk Information System (IRIS); 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 (cited when toxicity data were not available in standard review documents). By basing toxicity information on the standard review documents cited above, evaluations at UMTRA Project sites should be consistent with evaluations at sites regulated under different legislation.

The toxicity profiles presented here focus on toxicity data for drinking water in humans. Animal data, noted on the toxicity range graphs by widely spaced dotted lines, are included only when human data are not available. Uncertainty about the beginning or ending point of an exposure range producing specific toxic effects is indicated at the appropriate end of the range by closely spaced dots.

5.1 CONTAMINANT TOXICITY SUMMARIES

The following summaries address the basic toxicokinetics and toxicity of the seven inorganic contaminants of potential concern at Riverton (arsenic, manganese, molybdenum, nickel, sulfate, uranium, and vanadium). Although these contaminants have a wide range of toxic effects depending on the exposure levels, the following discussions focus on toxic effects observed in the exposure range most relevant to contamination at Riverton. The carcinogenicity associated with radionuclides is discussed in Section 5.2.

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 depends on the form of arsenic. Higher retention of arsenic occurs after exposure to trivalent arsenic than to the pentavalent form and tissue distribution is altered (Webb, 1966; Casarett and Doull, 1991).

In humans and rats, inorganic arsenic passes through the placental barrier. It 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 1 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 depends on the chemical form of arsenic, the duration of exposure, 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 arsenic transport under natural conditions. In oxygenated water, arsenic occurs in a pentavalent form; under reducing conditions, the trivalent form predominates.

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; DHHS, 1993a). Surveys of drinking water in the United States have revealed that over 99 percent of the public water supplies have arsenic levels below 0.05 mg/L (DHHS, 1993a) (0.05 mg/L is an equivalent to 0.001 mg/kg-day for a 70-kg adult drinking two liters of water per day). 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).

Seafood, meats, and grains contain the highest levels of arsenic. 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 Riverton site, arsenic exists primarily in the pentavalent form in ground water (Table 3.4). For arsenic trioxide, the reported estimated acute oral lethal dose in humans ranges from 70 to 300 mg (1 to 4 mg/kg) (EPA, 1984). Acute exposure to inorganic arsenic compounds may lead to severe inflammation of the gastrointestinal tract, encephalopathy, and acute renal failure after ingestion.

Increasing chronic doses of orally ingested arsenic progressively produce 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 (0.08 mg/kg-day) (DHHS, 1993a).

Chronic arsenic intoxications result from exposure to even 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). The lowest dose of arsenic associated with Blackfoot disease in continuously exposed individuals has been determined to be 0.014 mg/kg-day (DHHS, 1993a).

Hyperpigmentation, hyperkeratosis, 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. However, hyperkeratosis and hyperpigmentation were observed at an exposure level as low as 0.014 mg/kg-day (DHHS, 1993a).

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). Teratogenic effects, also referred to as birth defects, are defined as effects resulting in structural or functional anomalies in live offspring.

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 via inhalation) and of increased skin cancer prevalence (in populations exposed by consuming drinking water containing high concentrations of arsenic). The current slope factor (SF) for oral exposure to arsenic is given in Table 5.1. This SF is currently under review by the EPA with respect to recent data suggesting arsenic ingestion may result in increased cancers in internal organs as well as skin cancers. Figure 5.1 summarizes the health effects of arsenic exposure as a function of dose.

5.1.2 Manganese

Absorption

Following ingestion, manganese absorption is homeostatically controlled: the absorption rate depends on both the amount ingested and the existing manganese levels in tissue. Adults absorb approximately 3 to 4 percent of dietary manganese (Saric, 1986). Manganese can be absorbed following exposure by inhalation, ingestion, and dermal contact. Available data indicate that humans absorb only 3 percent of an ingested dose of manganese chloride (Mena et al., 1969). Manganese in water appears to be more efficiently absorbed than manganese in foodstuff (EPA, 1994a). The absorption rate is influenced by iron and other metals. In states of iron deficiency, manganese is actively absorbed from the intestine. Individuals with anemia can absorb more than twice the percentage of an ingested dose. However, in states of excess iron, manganese absorption is by diffusion only (Saric, 1986). High levels of dietary calcium and phosphorus are 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 after absorption. The highest concentrations are found in the liver and kidney. The biological half time in humans is 2 to 5 weeks, depending on body stores. Manganese readily crosses the blood-brain barrier and is more slowly cleared from the brain than from other tissue

Table 5.1 Toxicity values: potential carcinogenic effects

Parameter	Oral SF[a] $(pCi)^{-1}$ $(mg/kg\text{-}day^{-1})$	Weight of evidence classification	Type of cancer	SF source[b]
Arsenic (inorganic)	1.8E+0[c]	A	Skin	IRIS[d]
Lead-210[e]	1.01E-09	A	Bone	HEAST
Polonium-210	3.26E-10	A	Liver, kidney, spleen	HEAST
Radium-226[e]	2.96E-10	A	Bone	HEAST
Thorium-230	3.75E-11	A	f	HEAST
Uranium-238[e]	6.20E-11	A	g	HEAST
Uranium-234	4.44E-11	A	g	HEAST

[a]For each individual radionuclide listed, oral SFs correspond to the risks per unit intake (risk/pCi) for that radionuclide, except as noted.
[b]From EPA (Supplement Number 2, November 1994b).
[c]Oral SF based on oral unit risk of 5×10^{-5} (EPA, 1994a).
[d]EPA (1994a).
[e]Oral SF includes the contributions from short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal nuclide in the environment.
[f]Target organ systems have not been identified for oral exposure to thorium.
[g]No human or animal studies have shown a definite association between oral exposure to uranium and development of cancer.

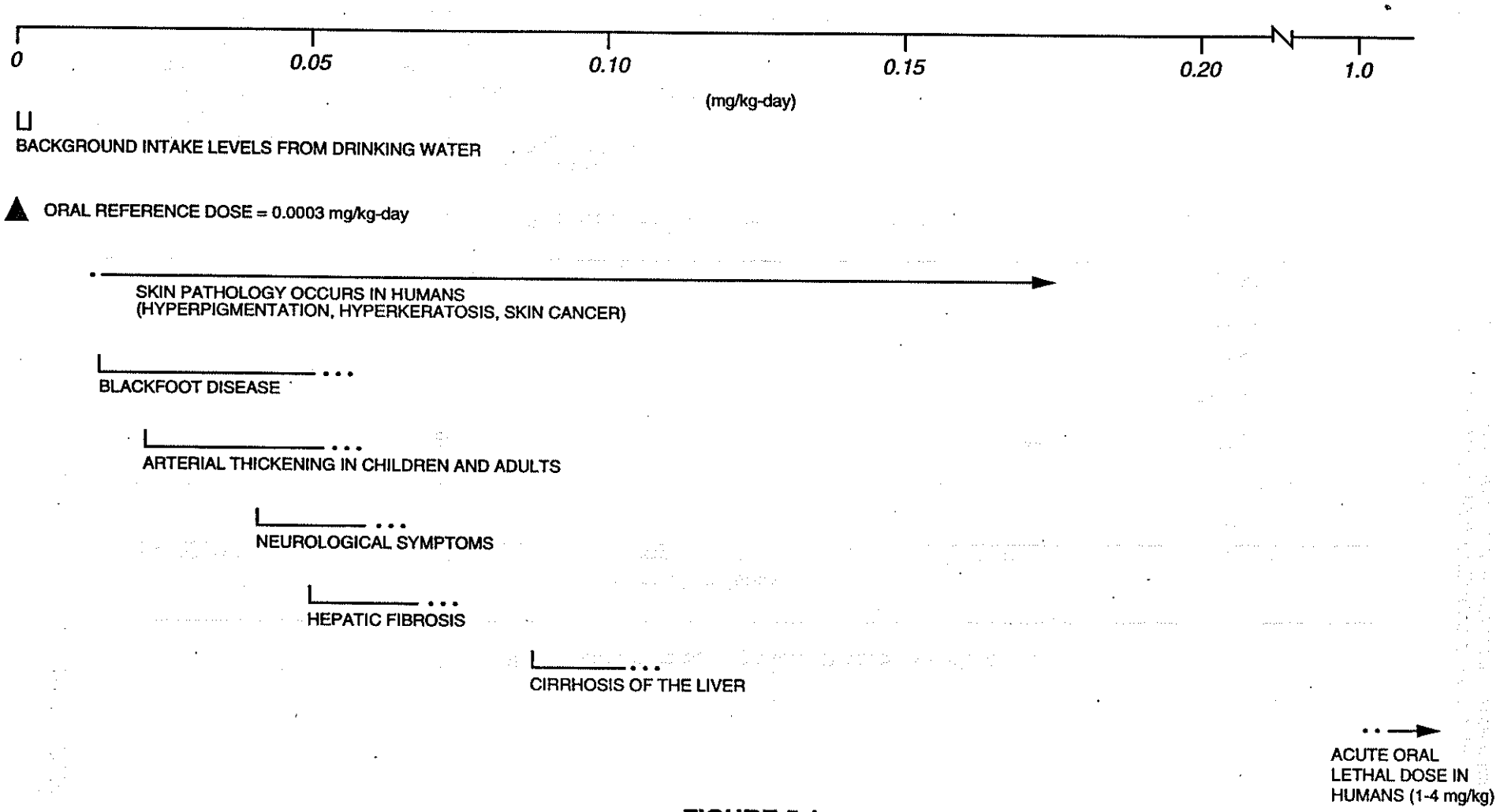

FIGURE 5.1
ARSENIC TOXICITY RANGES

MAC: SITE/Rv ...RA/ARS-GEN

(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 after excessive absorption (National Research Council, 1973).

Absorbed manganese is eliminated rapidly 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 adults. It is secreted into milk.

Absorbed manganese is almost totally secreted in bile and reabsorbed from the intestine as needed 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 is 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 variations 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 using cereals and rice as main food sources, the intake will be higher than in areas where meat and dairy products are a larger part of the diet. The average daily intake has been estimated as 2.0 to 8.8 mg per day (0.03 to 0.13 mg/kg-day) (EPA, 1994a), but intakes as high as 12.4 mg (about 0.2 mg/kg-day) are 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 threefold (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. The adult estimated safe and adequate daily dietary intake ranges from 0.03 to 0.07 mg/kg-day (Saric, 1986). The EPA no-observed-adverse-effect level (NOAEL) for drinking water is set at 0.005 mg/kg-day, while the lowest-observed-adverse-effect level (LOAEL) for drinking water is 0.06 mg/kg-day (EPA, 1994a). The EPA RfD for drinking water is 0.005 mg/kg-day. The RfD for ingested food is 0.14 mg/kg-day. Manganese in drinking water may be more bioavailable (i.e., more readily absorbed) than

manganese in dietary food sources. This bioavailability would result in toxic effects at lower ingested doses in drinking water than in food (EPA, 1994a).

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

Information on the effects of manganese ingestion is limited. Because effects from drinking water seem to differ from effects 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. Two deaths (8 percent) occurred among the intoxicated people. A Greek study of more than 4000 adults at least 50 years old drinking water with manganese concentrations varying from 0.081 to 2.3 mg/L (estimated intakes at 2 L per day for a 70-kg individual range from 0.002 to 0.07 mg/kg-day) showed varying degrees of neurological effects in individuals drinking from 0.007 to 0.07 mg manganese/kg-day, but no effects in individuals drinking less than 0.005 mg/kg-day (Kondakis et al., 1989). However, the many limitations to these studies make data interpretation difficult. Among the limitations is uncertainty regarding the exposure level or whether the effects seen were solely attributable to manganese. Despite these limitations, the similarity of the effects seen in the cases of oral exposure compared with those associated with inhalation exposure suggests that excess manganese intake by humans might lead to neurological injury (DHHS, 1992b).

The chemical form of manganese has complex effects on its toxicity. Although the 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). Information on the effects of various forms of manganese is limited.

Few data are available on manganese toxicity in infants, but infants are probably more susceptible to manganese toxicity due to greater absorption and greater penetration into the central nervous system (EPA, 1994a; Saric, 1986). Figure 5.2 summarizes the toxicity of manganese as a function of dose.

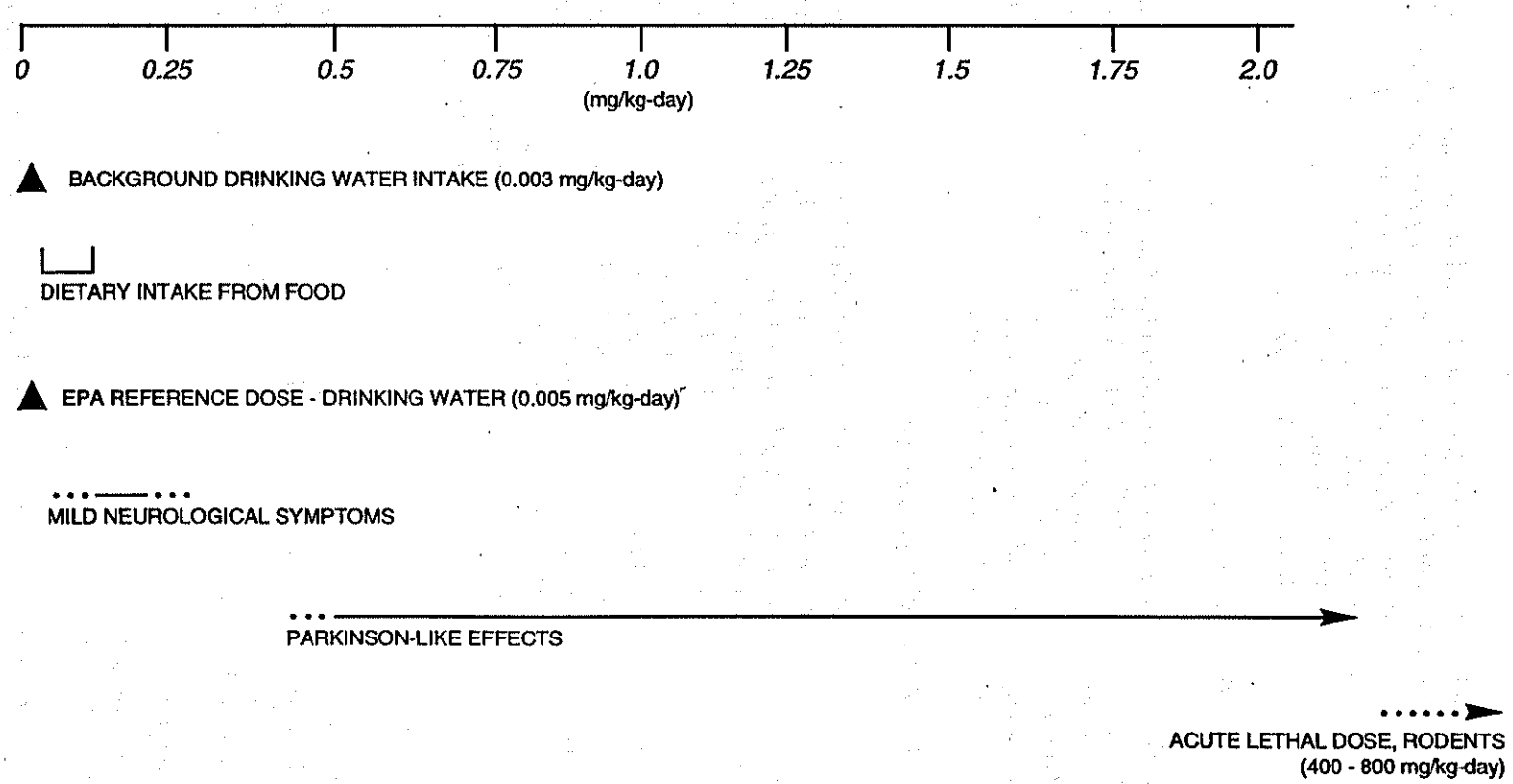

FIGURE 5.2
MANGANESE TOXICITY RANGES

MAC: SITE/RVT/BRA/MAN-GEN

5.1.3 Molybdenum

Absorption

Absorption of molybdenum in the gastrointestinal tract depends on the species of the metal. Inorganic 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 and Cook, 1963; Robinson et al., 1973; Alexander et al., 1974).

Tissue accumulation and clearance

In humans, the highest concentrations of molybdenum occur in the liver, kidneys, 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 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

Molybdenum occurs naturally in combination with other metals, including uranium, lead, iron, cobalt, and calcium. Native soil concentrations can vary by as much as 2 orders of magnitude, from 0.1 mg/kg to 10 mg/kg, leading to large variations in molybdenum concentrations in plant materials. Natural concentrations in ground water have been reported from 0.00011 to 0.0062 mg/L. Human dietary intake of molybdenum has been estimated at 0.05 to 0.24 mg per day (0.0007 to 0.003 mg/kg-day). The contribution of drinking water is estimated to range from 0 to 95 percent. The nutritional intake range 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 molybdenum levels also increase the levels of xanthine oxidase, which is responsible for the production of uric acid. 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 imbalances as a result of increased copper excretion. Excretion of copper 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 hemoglobin synthesis and function. A copper deficiency resulting from excess excretion will impair the oxygen-carrying capacity of the blood, and severe copper deficiencies can lead to hypochromic microcytic anemia. In humans, gout-like symptoms and joint deformities have been reported in regions of Russia where elevated molybdenum concentrations in soil and subsequent increased molybdenum concentrations in food would lead to molybdenum intakes in the range of 0.14 to 0.21 mg/kg-day. Figure 5.3 summarizes the health effects of molybdenum as a function of dose.

5.1.4 Nickel

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 (measured by serum nickel levels) increased by 80 µg/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, 1993b). Other human studies show that generally less than 10 percent of ingested nickel is absorbed by the gastrointestinal tract. This finding is consistent with studies reporting from 1 to 10 percent oral absorption in several animal species (Friberg et al., 1986). Absorbed nickel is transported in the plasma bound to serum albumin and various organic ligands, amino acids, or polypeptides (Casarett and Doull, 1991). Nickel has been found to affect gastrointestinal absorption of iron, but only when iron was administered as ferric sulfate (DHHS, 1993b).

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, the mean serum nickel half-life was 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 orally exposed to nickel compounds, suggesting that nickel crosses the placental barrier (DHHS, 1993b).

In humans, most ingested nickel is excreted in the feces, due to limited absorption. Nickel absorbed in the gastrointestinal tract is excreted in the urine. Excretion of a given dose of nickel is nearly complete in 4 or 5 days (Casarett and Doull, 1991), with approximately 26 percent of the dose excreted in the urine and the remainder eliminated in the feces (DHHS, 1993b).

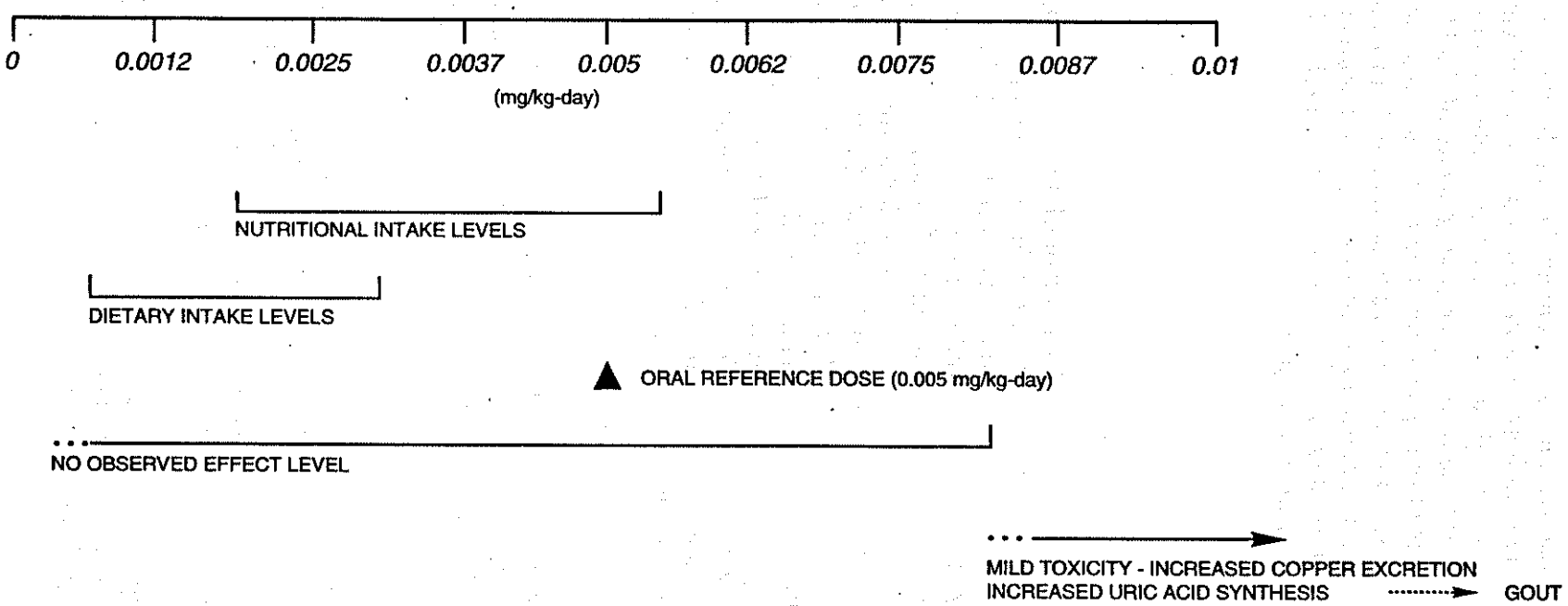

FIGURE 5.3
MOLYBDENUM TOXICITY RANGES

MAC: SITE/RV. √MOL-GEN

Environmental sources of nickel

Exposure to nickel can occur through inhaling ambient air and tobacco smoke and ingesting water and food. Most intake occurs through the diet (DHHS, 1993b). In grains, fresh weight nickel concentrations reportedly range from 0 to 6.45 micrograms per gram (µg/g). In vegetables and fruits, levels range from 0 to 2.59 µg/g and in seafood from 0.3 to 107 µg/g. Average daily dietary intake is approximately 165 µg (Friberg et al., 1986). The drinking water daily intake averages 2 µg (DHHS, 1993b).

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 micrograms per liter (µg/L) of nickel and 80 percent had less than 10 µg/L, although in areas near nickel mining operations, levels as high as 200 µg/L have been reported (Friberg et al., 1986).

Toxicity of nickel

Acute exposure to high 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, muscle 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, 1993b). 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 affect the detoxification and excretory processes (mainly hepatic and renal). For these reasons, it is expected that the elderly (with declining organ function) and the youngest of the population (with immature and developing organs) are generally more vulnerable to toxic substances than are 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 individuals (DHHS, 1993b). Epidemiologic studies indicate that blacks have a higher sensitivity than whites and that women of either racial group have higher reaction rates than do men (DHHS, 1993b). The incidence of allergic reactions may be higher in women because they wear more metal jewelry than men. The response

threshold may be approximately 0.007 mg/kg-day following oral challenge. Cross-sensitivity of nickel and other metals (e.g., cobalt) is also reported (DHHS, 1993b).

For the rat, a NOAEL of 100 parts per million (ppm) of nickel in the diet (5 mg/kg-day) was reported. The EPA chronic RfD for human exposure to nickel was derived based on this NOAEL. 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 which is not expected to produce adverse health effects in humans. Figure 5.4 summarizes the potential health effects from exposure to nickel as a function of dose.

5.1.5 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 therefore can 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 apparently have not 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 up to 2.9 mg/kg-day from drinking water in the concentration range found in supplies in the western United States. No estimates are available on sulfate intake from food.

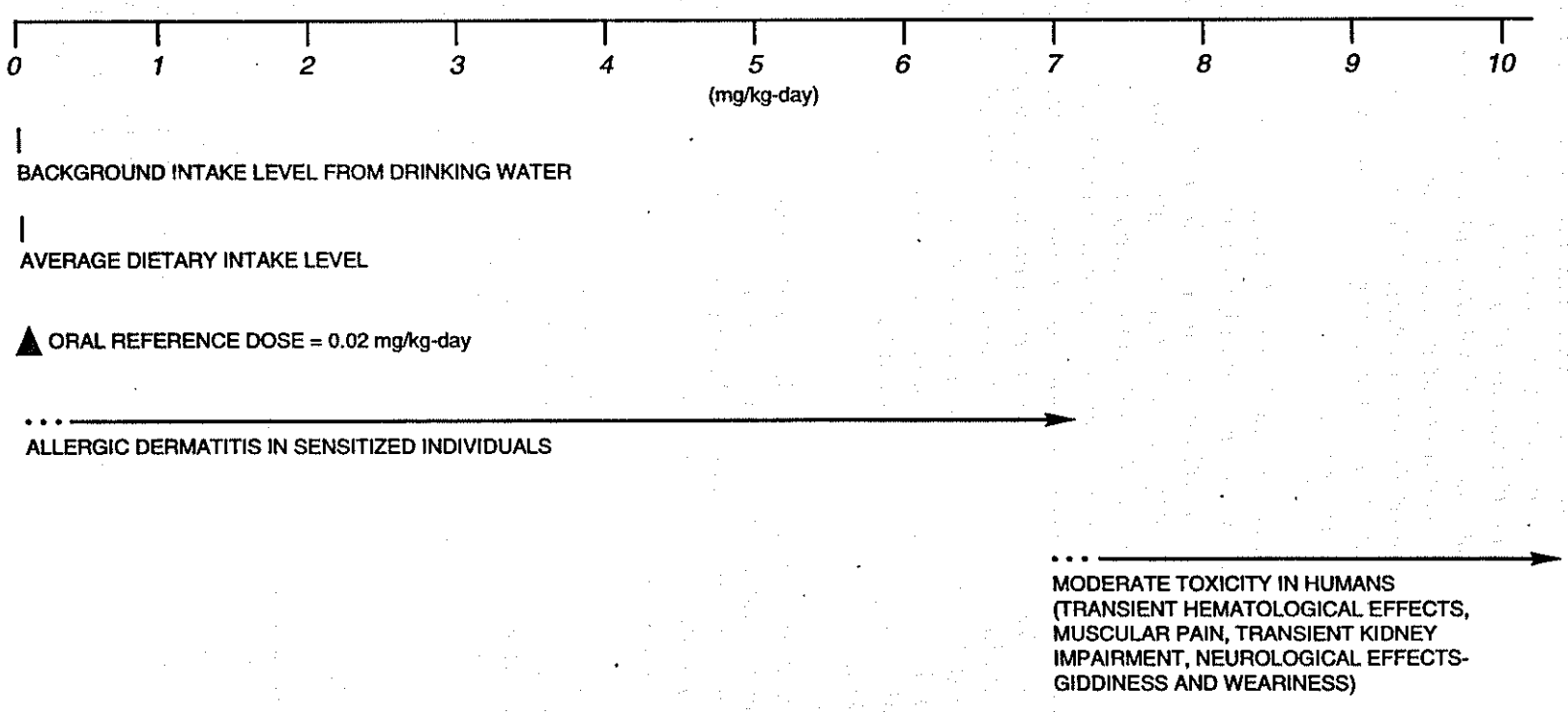

FIGURE 5.4
NICKEL TOXICITY RANGES

MAC: SITE/RVT/BRA/NIC-GEN

Toxicity of sulfate

As with nitrate toxicity, 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 as diarrhea; the severity of the diarrhea is dose-dependent. Chronic sulfate ingestion can result in persistent diarrhea, leading to ionic imbalances and dehydration similar to that seen with extremely high acute doses. When drinking water is contaminated with sulfate, the taste of the water may make it unpalatable and reduce consumption. However, this is not necessarily 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, 1992b). When consumption is reduced, 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, some data indicate diabetic and elderly populations with compromised kidney function may be more sensitive to the effects of sulfates than healthy adults (EPA, 1992b).

In cattle, high sulfate intake has resulted in sulfhemoglobinemia, a condition similar to the methemoglobinemia induced by nitrate ingestion (EPA, 1992b). No reports of sulfhemoglobinemia have been reported following ingestion of sulfate by humans, although the condition has been reported in humans following inhalation of hydrogen sulfide.

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. Figure 5.5 summarizes potential toxic effects from sulfate ingestion as a function of dose.

5.1.6 Uranium

The uranium that occurs naturally at UMTRA Project sites consists of three radioactive isotopes: uranium-234, uranium-235, and uranium-238. More than 99 percent of natural uranium occurs in the form of uranium-238 (Cothern and Lappenbusch, 1983). Uranium-238 undergoes radioactive decay by emitting alpha particles to form uranium-234, thorium-230, radium-226, radon-222, polonium-210, lead-210, and other radioisotopes. Figure 5.6 summarizes the radioactive decay chain of uranium-238 and uranium-234. Because all natural uranium isotopes are radioactive, the hazards of a high uranium intake are from both its chemical toxicity

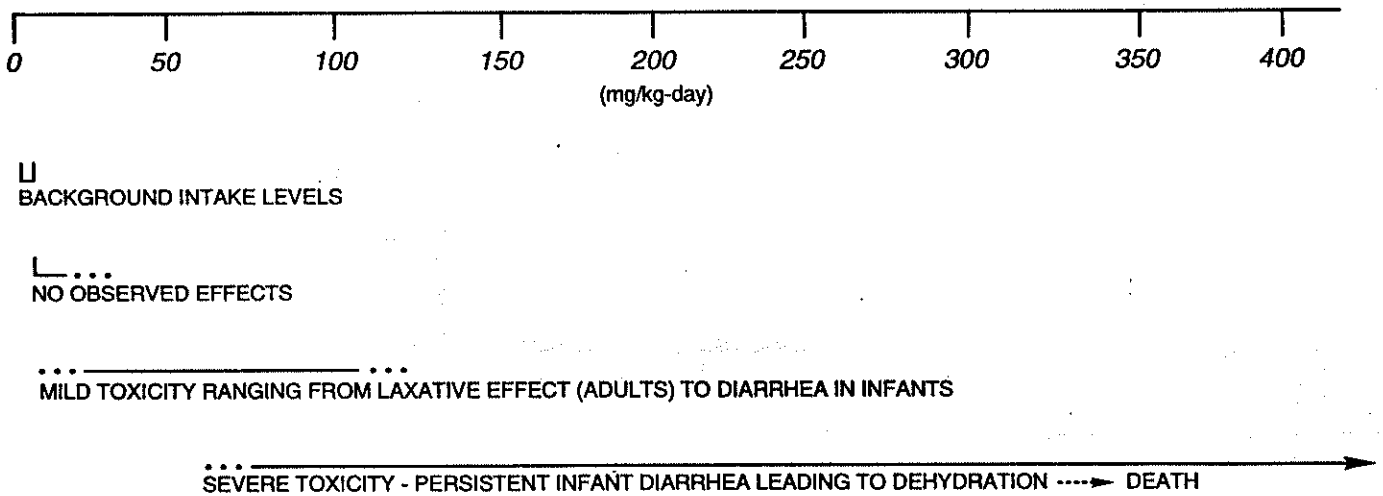

FIGURE 5.5
SULFATE TOXICITY RANGES

MAC: SITE/RVT/BRA/SUL-GEN

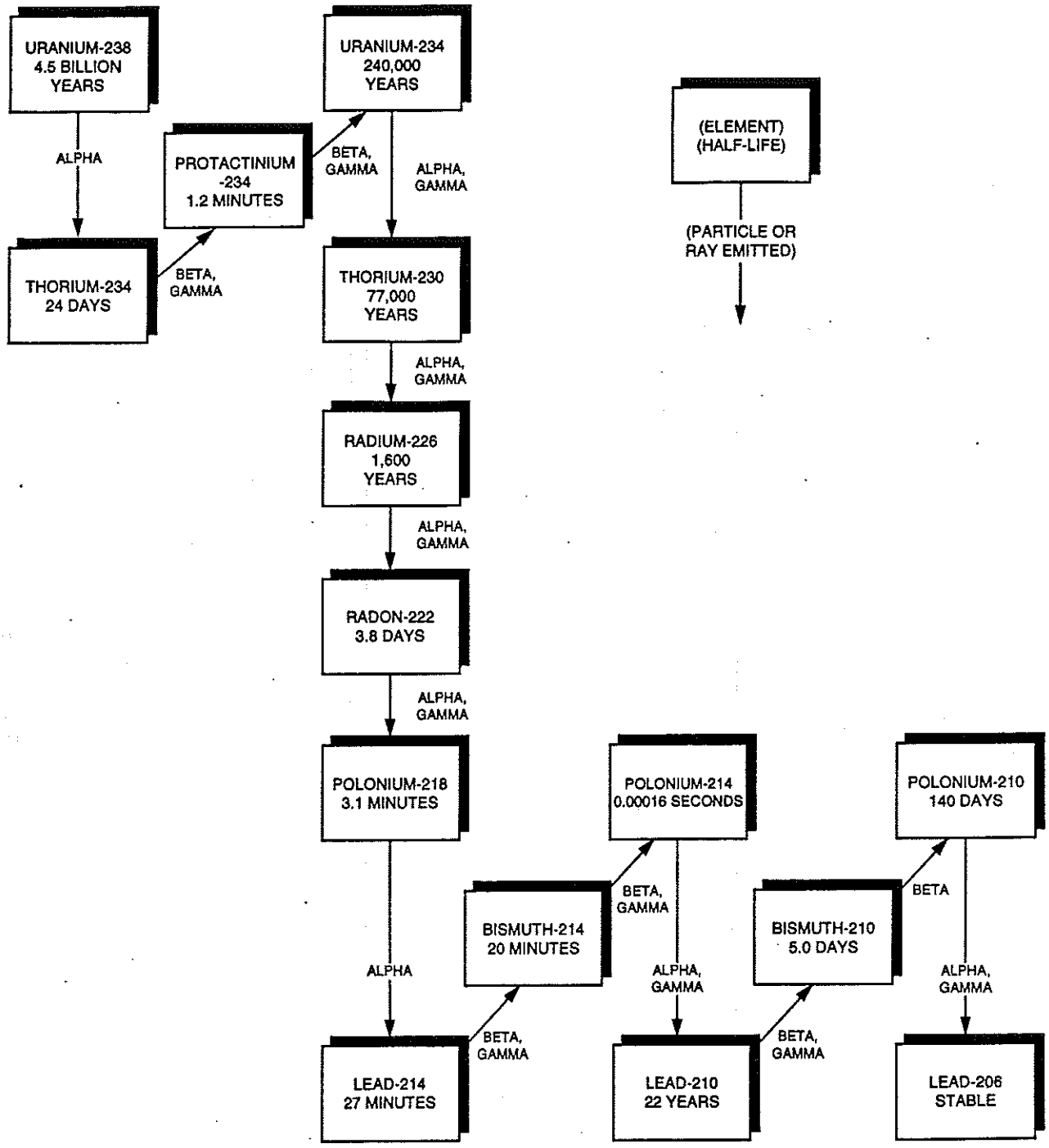

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FIGURE 5.6
HALF-LIVES AND EMISSIONS FROM DECAY CHAIN OF URANIUM-238

and its potential radiological damage. The chemical toxicity of natural uranium is discussed here; the carcinogenic potential associated with exposure to radioactive isotopes of natural uranium is discussed in Section 5.2.

Absorption

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

Tissue accumulation and clearance

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

In body fluids, uranium tends to convert into water-soluble hexavalent uranium (Berlin and Rudell, 1986). Approximately 60 percent of the uranium in plasma complexes with low-molecular-weight anions (e.g., bicarbonates, citrates), while the remaining 40 percent binds to the plasma protein transferrin (Stevens et al., 1980). Following oral exposure to uranium, humans excrete more than 90 percent of the dose in the feces. Of the small percent that is absorbed (typically less than 5 percent), approximately 60 percent in animals is excreted through the urine within 24 hours, and the remainder is distributed to the skeleton and soft tissue; 98 percent of that amount is excreted within 7 days (Ballou et al., 1986; Leach et al., 1984; Sullivan et al., 1986). A small portion of the absorbed uranium is retained for a longer period.

Environmental sources of uranium

Uranium is a ubiquitous element present in the earth's crust at approximately 4 ppm. Uranium concentrations in ground water and surface water average 1 picocurie per liter (pCi/L) and 3 pCi/L, respectively (NCRP, 1984). The extent of uptake from the soil into plant tissues depends on the plant species and the depth of its root system (Berlin and Rudell, 1986). Tracy et al. (1983) report plant uranium concentrations averaging 0.075 µg/kg of fresh plant material.

The main dietary source of natural uranium for the general population is food (e.g., potatoes, grains, meat, and fresh fish) that may contain uranium concentrations between 10 and 100 µg/kg (Prister, 1969). The total uranium dietary intake from consuming 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, probably contribute most to daily uranium intake (Berlin and Rudell, 1986).

Toxicity of uranium

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

No chronic toxic effects are reported in humans following oral exposure to uranium. Data from populations occupationally exposed to high concentrations of uranium compounds through inhalation and studies on 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 effects 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 EPA or a reference dose of 0.003 mg/kg-day was based on this study (EPA, 1994a). Figure 5.7 summarizes the health effects of uranium as a function of dose.

5.1.7 Vanadium

Absorption

Absorption of vanadium from the gastrointestinal tract is low. The International Commission on Radiological Protection estimate for the absorption of soluble vanadium compounds is 2 percent (ICRP, 1960), but the World Health Organization states that absorption of even very soluble forms of vanadium is less than 1 percent from the gastrointestinal tract (WHO, 1988). Limited human data (from three individuals) have suggested that as much as 10 percent of a repeated oral dose may be absorbed (Proescher et al., 1917; Tipton et al., 1969). Soluble vanadium compounds that are inhaled and deposited are more readily absorbed (about 25 percent) (WHO, 1988). Although soluble forms of vanadium may be absorbed through the skin, absorption via this route is probably minimal (EPA, 1977; WHO, 1988).

Tissue accumulation and clearance

Vanadium is found in all body tissues in concentrations ranging from 0.08 µg per gram wet weight in spleen tissue to 0.14 µg per gram in brain and heart tissue and 0.33 µg per gram in aorta tissue (Yakawa and Suzuki-Yasumoto, 1980). Vanadium concentrations in human blood serum are reported to be 0.016 to 0.939 nanograms

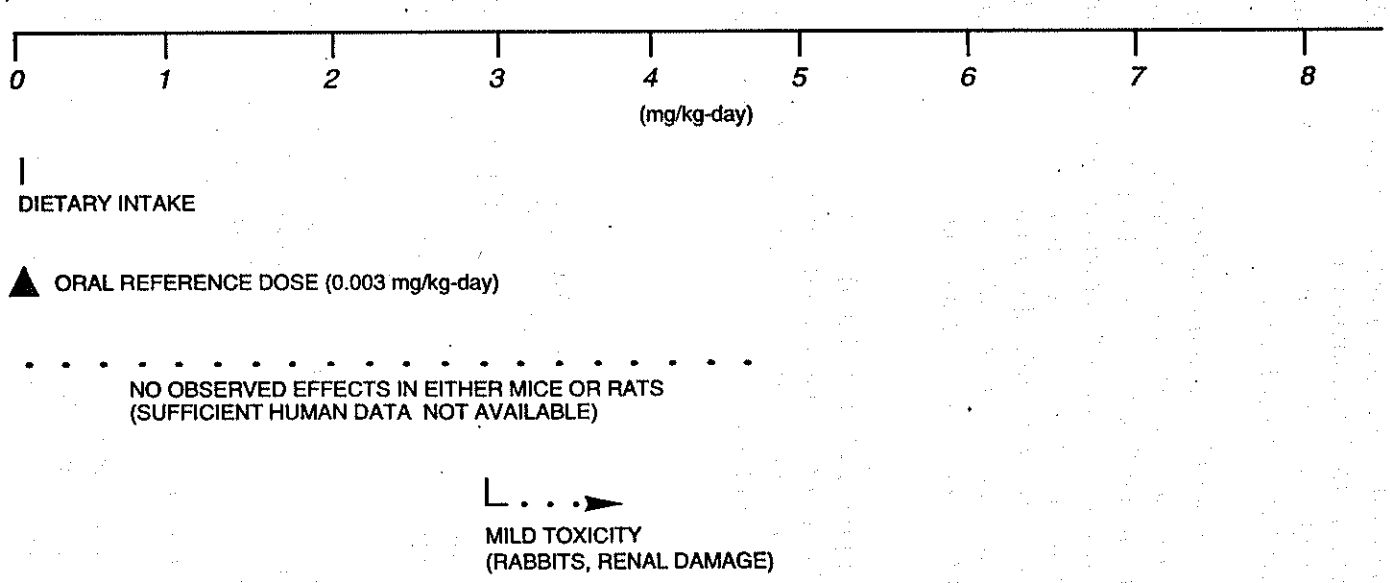

FIGURE 5.7
URANIUM TOXICITY RANGES

MAC: SITE/RVT/BRA/URN-GEN

per milliliter (ng/mL). In hair, vanadium concentrations ranging from 20 to 60 ng per gram have been reported by different authors, with higher values found in manic-depressive patients (57 ng per gram) than in normal control groups (29 ng per gram).

Vanadium distribution in humans following oral exposure may be extrapolated from animal studies. In acute-duration exposures, vanadium is rapidly distributed, primarily in the bones. After intermediate-duration exposure, vanadium concentrations reaching the tissues are low, with the kidneys, bones, liver, and lungs initially showing the highest levels.

Vanadium is an element and is not metabolized. However, in the body, there is an interconversion of two oxidation states of vanadium: vanadyl and vanadate. Vanadium can reversibly bind to the protein transferrin in the blood and then be taken up into erythrocytes. There is a slower uptake of vanadyl into erythrocytes compared to the vanadate form, possibly due to the time required for the vanadyl form to be oxidized to vanadate. Initially, vanadyl leaves the blood more rapidly than vanadate, possibly because of the slower vanadyl uptake into cells (Harris et al., 1984). Five hours after administration, blood clearance is essentially identical for the two forms.

Because vanadium is poorly absorbed in the gastrointestinal tract, a large percentage of vanadium in rats is excreted unabsorbed in the feces following oral exposure. In rats, the principal route of excretion of the small absorbed portion of vanadium is through the kidneys. The mean urinary output per 24 hours is reported to be 10 µg.

<u>Environmental sources of vanadium</u>

Elemental vanadium does not occur in nature, but its compounds exist in more than 50 different mineral ores and in association with fossil fuels. The single largest release of vanadium to the atmosphere occurs through the combustion of fossil fuels, particularly residual fuel oils. The largest amount of vanadium released to soil and water occurs through natural weathering of geological formations (Byerrum et al., 1974; Van Zinderen Bakker and Jaworski, 1980).

Food constitutes the major source of exposure to vanadium for the general population (Lagerkvist et al., 1986). As a whole, dietary intake is estimated to be 6 to 18 µg per day (Pennington and Jones, 1987), although other estimates from older studies using different (and possibly less sensitive) analytical methods have been as high as 2 mg per day (Schroeder et al., 1963).

Drinking water is not considered an important source of vanadium exposure for the general population. Water samples taken from across the United States show 92 percent with values below 10 µg/L. Typical values appear to be around 1 µg/L (Lagerkvist et al., 1986). The estimated daily intake of vanadium by inhalation is 1 µg (Byrne and Kosta, 1978).

Although vanadium is considered an essential element for chickens and rats, there is no certainty about human dietary requirements. For animals, the daily requirement is about 10 to 25 µg per day (Pennington and Jones, 1987).

Toxicity of vanadium

The major adverse health effect to humans from vanadium is seen in workers exposed to large amounts of vanadium pentoxide dusts. The probable oral lethal dose of vanadium pentoxide for humans is between 5 and 50 mg/kg (Gosselin et al., 1976).

Systemic effects of vanadium exposure have been observed in the liver, kidneys, nervous and cardiovascular systems, and blood-forming organs. Metabolic effects include interference with the biosynthesis of cystine and cholesterol, depression and stimulation of phospholipid synthesis, and at higher concentrations, inhibition of serotonin oxidation. Other effects of vanadium on mammalian metabolism include depression of phospholipid synthesis (Snyder and Cornatzer, 1958), reduction of coenzyme Q levels in mitochondria (Aiyar and Sreenivasan, 1961), and stimulation of monoamine oxidase, which oxidizes serotonin (Perry et al., 1955).

Vanadium salts were given to patients in several studies to reduce cholesterol levels (Curran et al., 1959; Somerville and Davies, 1962; Dimond et al., 1963; Schroeder et al., 1963). The doses of vanadium in these studies varied from 7 to 30 µg per day (equivalent to 0.1 to 0.4 mg/kg-day for a 70-kg individual). Transient decreases in serum cholesterol levels were observed in some patients, as were loosened stool or diarrhea and intestinal cramps. Green tongue, a hallmark of vanadium exposure, was observed in all patients.

A relationship between the concentration of vanadium in drinking water and the incidence of dental caries in children is reported by Tank and Storvick (1960). Dental caries incidence in children aged 7 to 11 years was reduced three times (compared to controls) by applying ammonium vanadate in glycerol to the teeth (Belehova, 1969). This relationship was not found in other studies (Hadjimarkos, 1966; 1968).

It has been suggested that raised tissue levels of vanadium are important in the etiology of manic-depressive illness. Improvement after treatment with ascorbic acid or reduced vanadium intake was seen both in manic and depressed patients.

Although animal studies have reported impaired conditioned reflexes following doses of vanadium from 0.05 mg/kg-day (after 6 months of exposure) to 0.5 mg/kg-day (after 21 days of exposure), effects on the nervous system have not been observed following repeated oral administration of vanadium in humans. Workers exposed by inhalation to fairly high concentrations of vanadium compounds have reported nonspecific symptoms, including headache, weakness, vomiting, nausea, and ringing of the ears (WHO, 1988).

Available data on vanadium toxicity are insufficient to evaluate its effect on cholesterol levels, iron metabolism, blood-cell production, and mutagenesis. However, due to poor absorption from the gut, the metal is not considered very toxic following oral administration (WHO, 1988). The EPA oral RfD of 0.007 mg/kg-day was obtained from a chronic drinking water study with vanadium sulfate in rats (EPA, 1987; EPA, 1994b). Figure 5.8 summarizes the toxicity of vanadium.

5.2 CONTAMINANT RISK FACTORS

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 considerable portion of a lifetime. The RfDs include safety factors to account for uncertainties associated with limitations of the toxicological database, including extrapolating results from animal studies to humans and accounting for variability in response from sensitive individuals. These values are updated quarterly and are published in the Health Effects Assessment Summary Tables (HEAST) and also are provided through the EPA's IRIS database. Table 5.2 summarizes the most recent oral RfDs for the noncarcinogenic contaminants of concern.

The EPA currently classifies all radionuclides as Group A, or known human carcinogens, based on their property of emitting ionizing radiation and on the evidence provided by epidemiological studies of radiation-induced cancer in humans. The EPA currently classifies all radionuclides as Group A, or known human carcinogens, based on their property of emitting ionizing radiation and on the evidence provided by epidemiological studies of radiation-induced cancer in humans. At sufficiently high doses, ionizing radiation acts as a complete carcinogen (both initiator and promoter), capable of increasing the probability of cancer development. However, the actual risk is difficult to estimate, particularly for the low doses and dose rates encountered in the environment. Most reliable data were obtained under conditions of high doses delivered acutely. It is not clear whether cancer risks at lower doses are dose-proportional (i.e., the linear dose-response hypothesis) or whether the risk is greatly reduced at low doses (i.e., the threshold hypothesis). A conservative assumption rejects the threshold hypothesis and assumes that any dose or dose rate adds to the risk of cancer. Risk factors are published in HEAST and IRIS for correlating carcinogen intake over a lifetime with the increased excess cancer risk from that exposure. Table 5.1 gives the most recent cancer SF for arsenic and the uranium-234/-238 radioactive decay series.

5.3 CONTAMINANT INTERACTIONS

Although some information is available on potential interactions between contaminants found at UMTRA Project sites, potential interactions can generally be evaluated 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 experiment concentrations, and 2) additional ground water constituents that

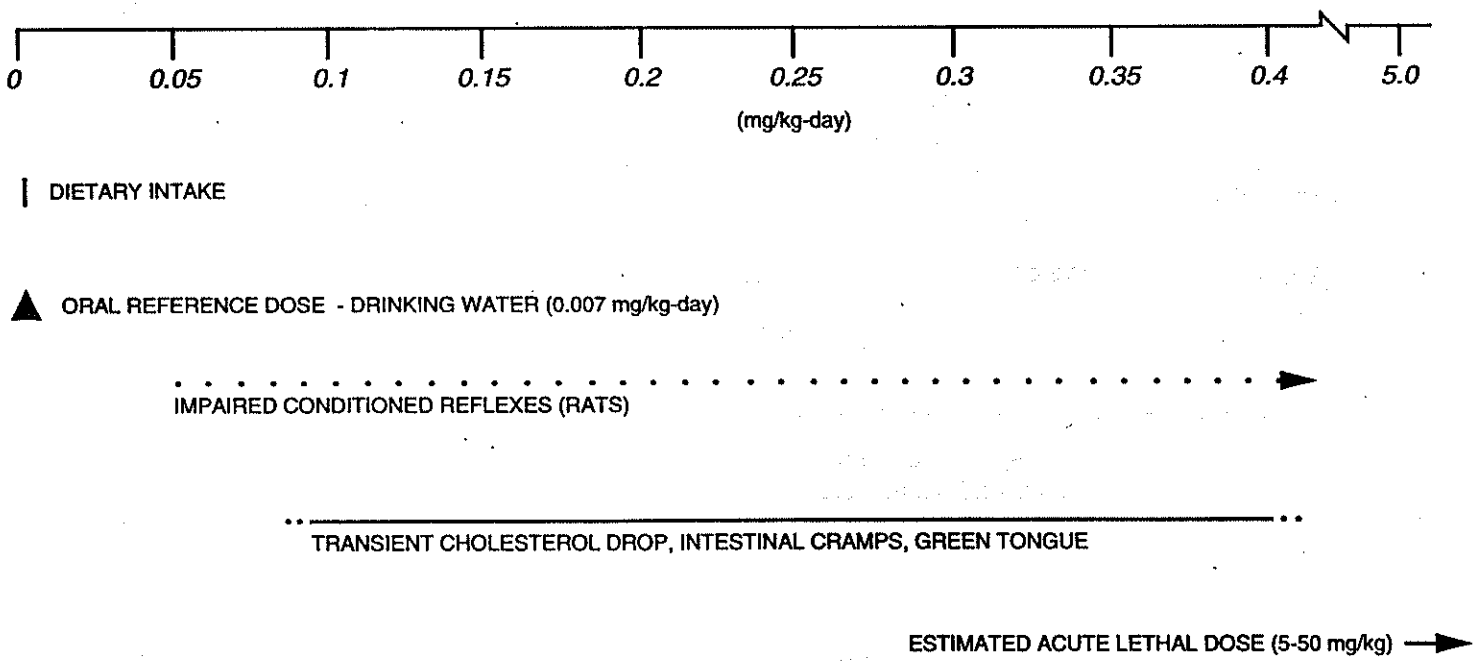

FIGURE 5.8
VANADIUM TOXICITY RANGES

MAC: SITE/RVT/BRA/VAN-GEN

Table 5.2 Toxicity values: potential noncarcinogenic effects

Chemical	Chronic oral RfD[a] (mg/kg-day)	Confidence level	Critical effect/organ	RfD basis/RfD source	Uncertainty factor
Arsenic (inorganic)	3E-4	Medium	Hyperkeratosis, hyperpigmentation, vascular complications	Water/IRIS[b]	3
Manganese	5E-3	NA	Central nervous system effects	Water/IRIS[b]	1
Molybdenum	5E-3	Medium	Increased uric acid production; joints (pain, swelling); blood (decreased copper levels)	Diet/IRIS[b]	30
Nickel (soluble salts)	2E-2	Medium	Decreased body and organ weights	Diet/IRIS[b]	300
Sulfate	ND	NA	Diarrhea	NA	NA
Uranium (soluble salts)	3E-3	Medium	Nephrotoxicity, decreased body weight	Water/IRIS[b]	1000
Vanadium	7E-3	Low	Decreased cysteine	Water/HEAST[c]	100

[a]These doses are adopted as subchronic oral reference doses (RfD) with the exception of uranium.
[b]From EPA (1994a).
[c]From EPA (1994b).
ND - not determined.
NA - not applicable.

may be present in sufficient quantities to modify predicted toxicities even when they are not considered contaminants of concern for human health. Therefore, the interactions described below should be recognized as factors that can influence predicted toxicity, although the precise nature and magnitude of that influence cannot be determined.

Molybdenum can produce a functional copper deficiency, but only in the presence of sulfate. Laboratory animal studies show molybdenum toxicity is more pronounced when dietary copper intake is low (EPA, 1994). In ruminants, which are very susceptible to imbalances between these elements, copper prevents the accumulation of molybdenum in the liver and may antagonize absorption of molybdenum into the bloodstream. It has also been suggested that sulfur can displace molybdate in the body (Casarett and Doull, 1991). Both copper and sulfates can protect against molybdenum toxicity in ruminants, and molybdenum and sulfur can block copper toxicity.

Interactions between several similar metals can alter the predicted absorption, distribution in the body, metabolism, toxicity, or clearance of a metal of interest. For example, manganese absorption can be considerably increased under conditions of low calcium or iron (DHHS, 1992a).

Administered in combination to pregnant mice, vanadium and manganese caused alterations in behavioral development compared to either metal administered alone (DHHS, 1992a; 1992b). Oral administration of vanadium may interfere with the intestinal absorption of copper.

Nickel may interact with other heavy metals such as iron, manganese, zinc, and cadmium (DHHS, 1993b). Nickel toxicity can be mitigated by high levels of zinc. Under iron deficiency status, nickel may facilitate the passive diffusion of iron by stabilizing the transport ligand, increasing its gastrointestinal absorption, but only when the iron is present in the form of ferric ion (DHHS, 1993b). There is an interrelationship between nickel and cobalt sensitization in individuals exposed to the two metals (DHHS, 1992c; 1993b).

No information on uranium interaction with other metals was found in preparation for this risk assessment. However, the common target organ suggests interaction with arsenic and nickel to produce kidney toxicity.

The carcinogenic effects of the radionuclides probably combine at least additively. Arsenic carcinogenicity is a distinct mechanism and may not necessarily be truly additive with radionuclide cancer risks.

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6.0 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 a contaminant an individual takes into his or her body. Because many of the contaminants associated with the mill tailings are essential nutrients, they are beneficial to health at lower levels. At higher levels, these same elements can cause adverse health effects or, at very high levels, death. This section correlates the expected contaminant intake (if ground water within the plume were used as drinking water) to the potential health effects of these levels of exposure.

6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

Surficial aquifer

The results from the exposure assessment for either the highest intake-to-body-weight ratios (or highest doses) or the toxicologically most sensitive group are used to evaluate the potential health effects of noncarcinogens. For arsenic, manganese, molybdenum, nickel, uranium, and vanadium, the highest intake-per-body-weight group is children 1 to 10 years old. The 0- to 1-year-old age group was used for sulfate because this age group is more susceptible to its toxicity. Figures 6.1 through 6.7 show the intake distributions for these age groups, with toxicity information for each contaminant of potential concern.

The exposures estimated for arsenic are comparable to dietary intake levels (Figure 6.1). Noncarcinogenic health effects are not associated with these intakes; the expected value is the acceptable intake level (RfD of 0.0003 mg/kg-day). The potential carcinogenicity of arsenic is described in Section 6.2.

Potential health effects are associated with almost the entire exposure distributions for manganese, molybdenum, and sulfate. Manganese exposures are in the range reported to cause neurological effects that include muscle rigidity and mental disturbances; in the upper 10th percentile of the exposure distribution, effects include a Parkinson-like syndrome. The entire distribution is above the EPA acceptable intake level (RfD of 0.005 mg/kg-day), with the expected value occurring at greater than 30 times the RfD (Figure 6.2).

Nearly all the exposure distribution for molybdenum is associated with increased copper excretion. At the upper range of the distribution, gout-like disease may occur in humans (Figure 6.3). The expected value is about 6 times the acceptable intake level recommended by EPA (RfD of 0.005 mg/kg-day).

The sulfate exposure distribution is well above the range shown to cause severe diarrhea and dehydration in infants (Figure 6.4). However, these high sulfate concentrations probably would affect the taste of the water, thereby reducing exposure potential. This reduced water intake could in turn exacerbate dehydration.

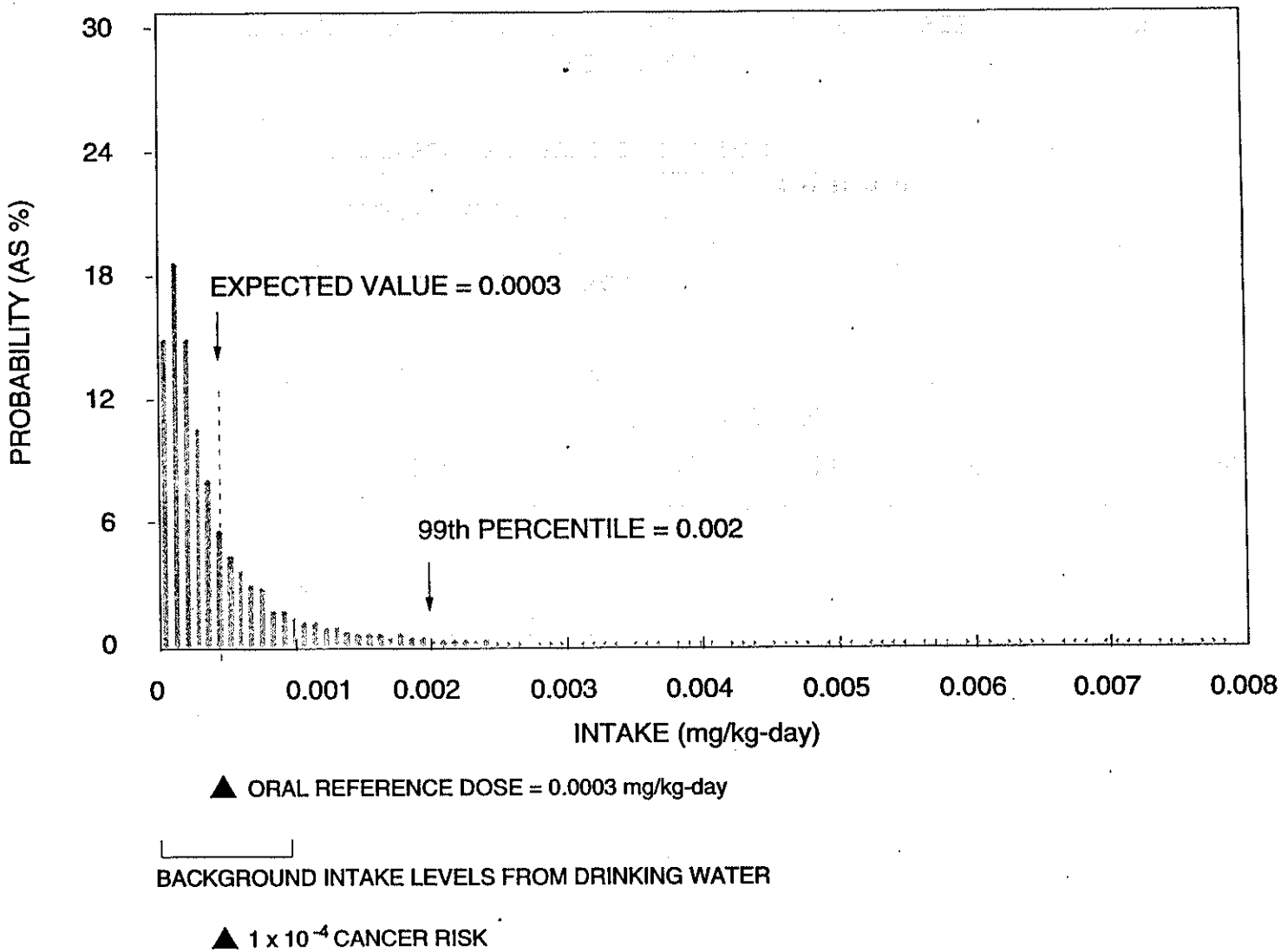


FIGURE 6.1
HEALTH EFFECTS OF POTENTIAL ARSENIC EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE

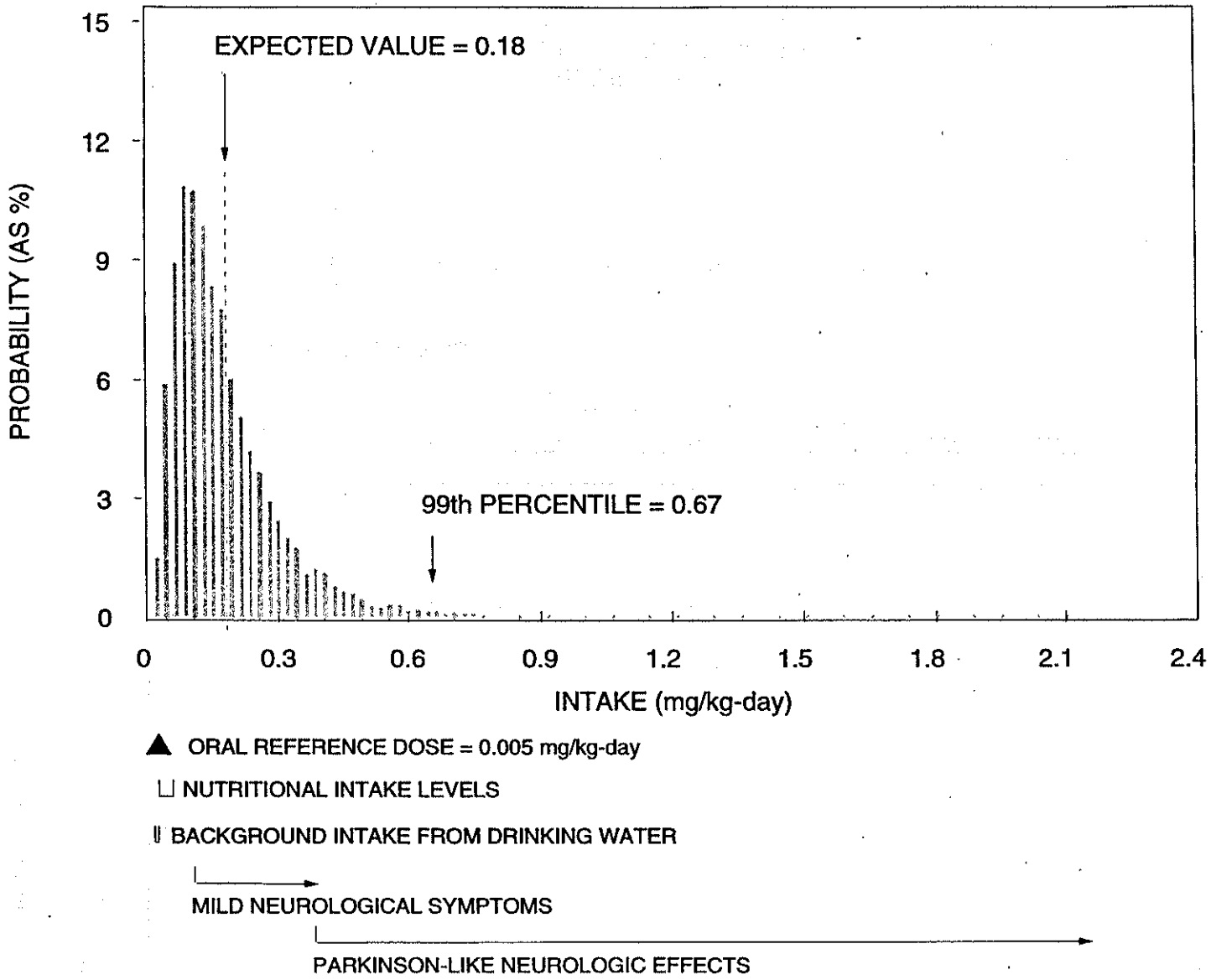


FIGURE 6.2
HEALTH EFFECTS OF POTENTIAL MANGANESE EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE

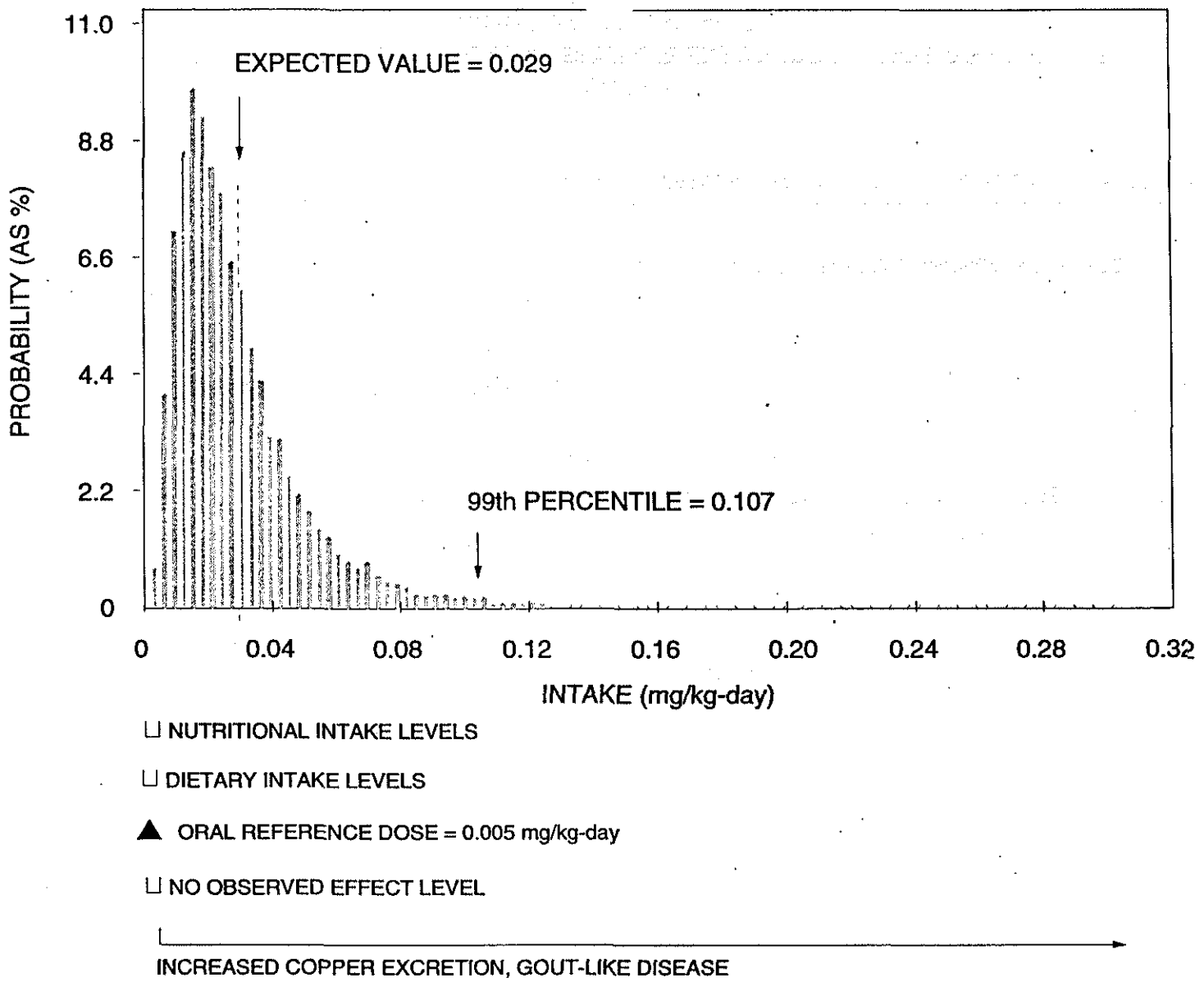
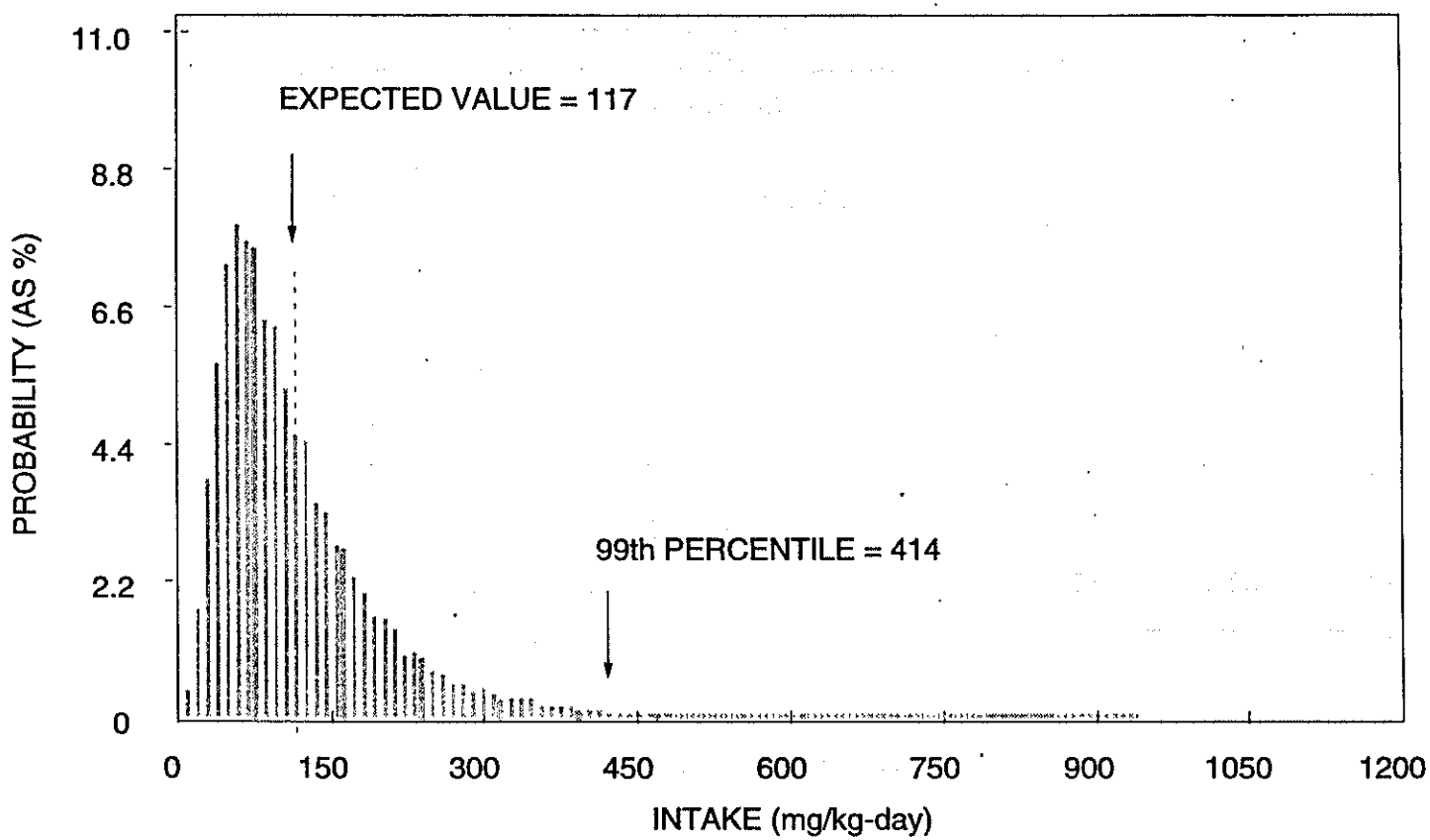


FIGURE 6.3
HEALTH EFFECTS OF POTENTIAL MOLYBDENUM EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE



BACKGROUND INTAKE LEVELS

NO OBSERVED EFFECTS

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

SEVERE TOXICITY - PERSISTENT INFANT DIARRHEA LEADING TO DEHYDRATION ····► DEATH

FIGURE 6.4
HEALTH EFFECTS OF POTENTIAL SULFATE EXPOSURE RANGES FOR INFANTS
RIVERTON, WYOMING, SITE

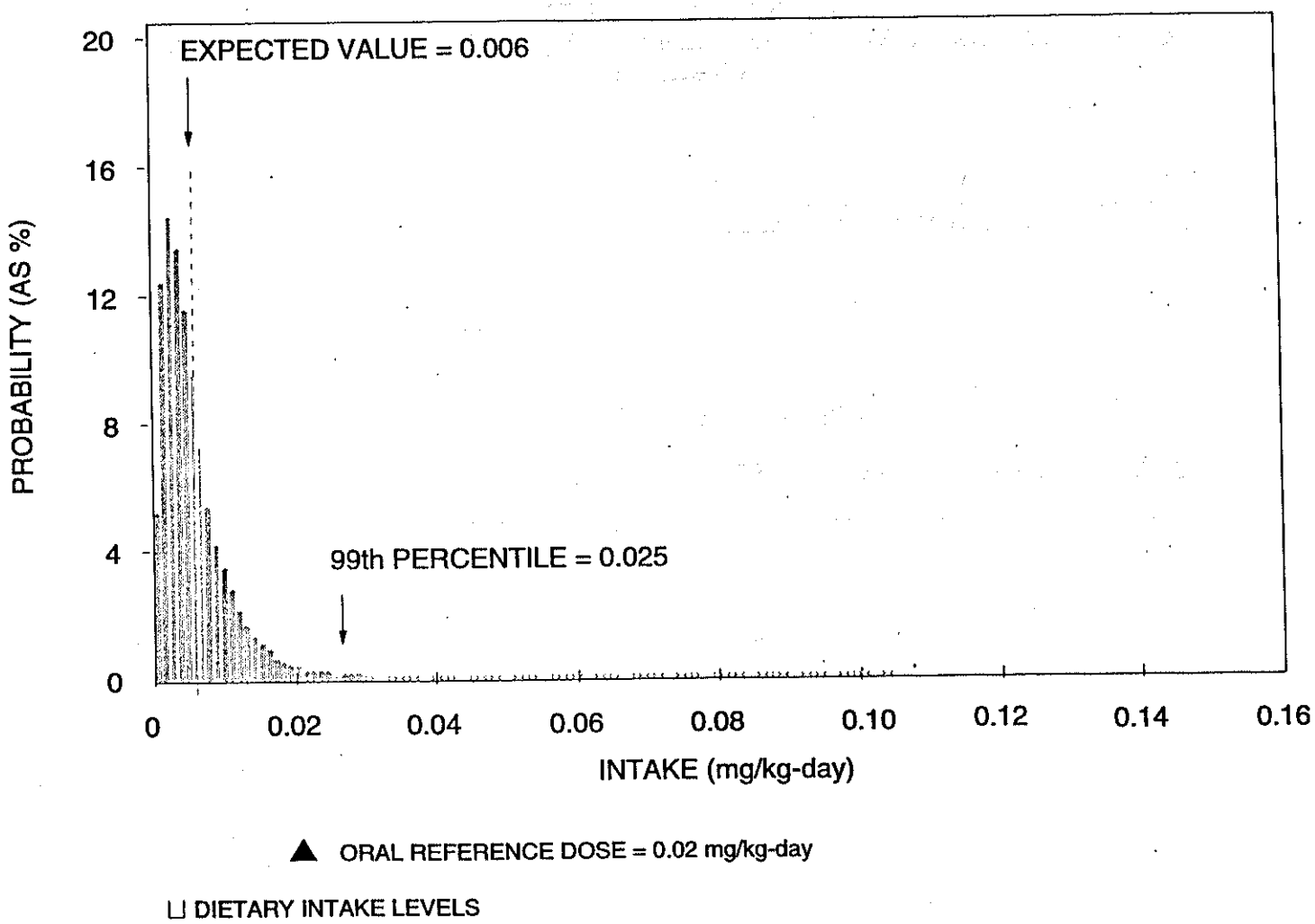


FIGURE 6.5
HEALTH EFFECTS OF POTENTIAL NICKEL EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE

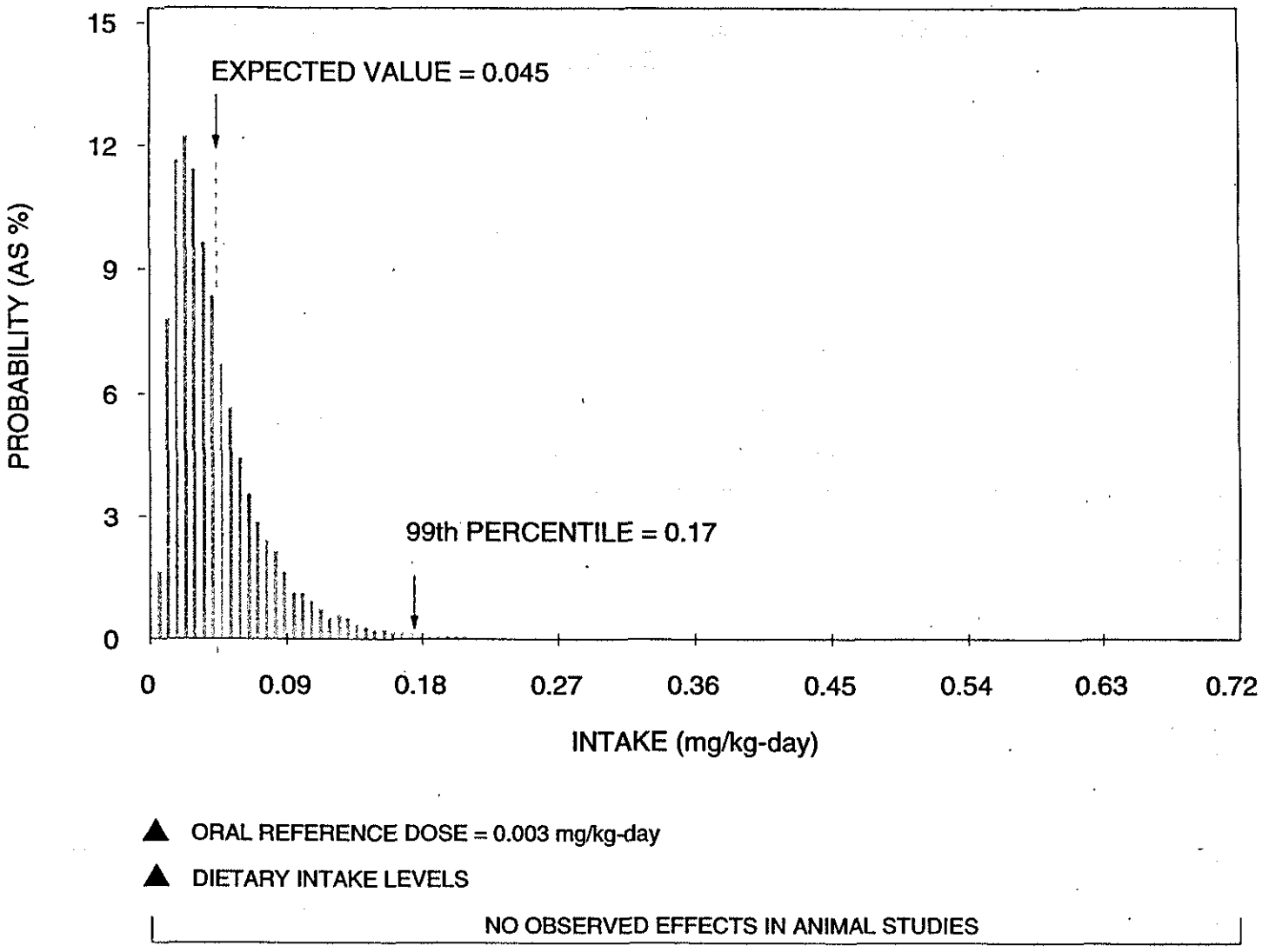


FIGURE 6.6
HEALTH EFFECTS OF POTENTIAL URANIUM EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE

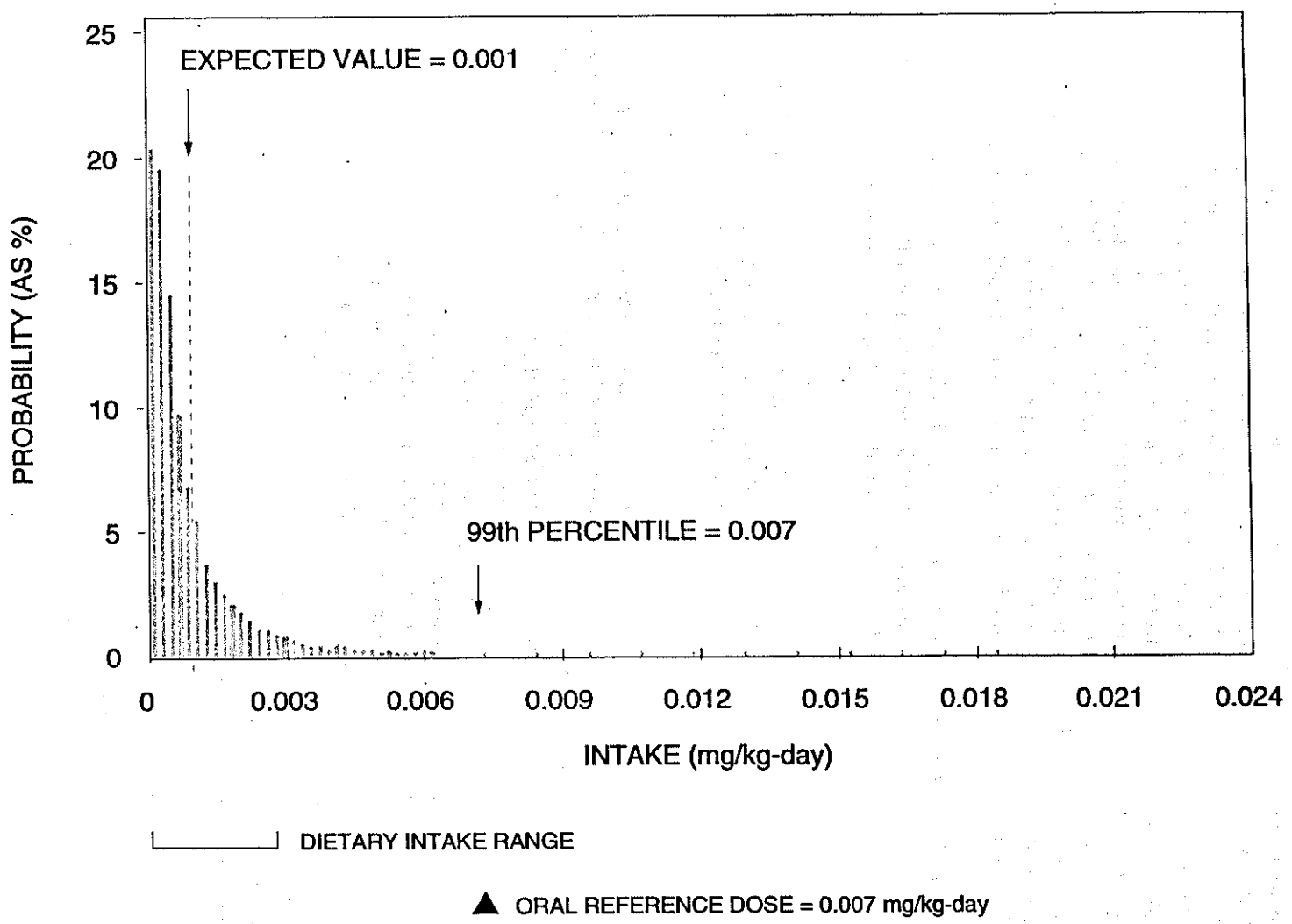


FIGURE 6.7
HEALTH EFFECTS OF POTENTIAL VANADIUM EXPOSURE RANGES FOR CHILDREN
RIVERTON, WYOMING, SITE

The exposure distribution for nickel is almost completely below the acceptable intake level (RfD of 0.02 mg/kg-day) (Figure 6.5).

The entire exposure distribution for uranium falls within the no-observed-effect level and is less than one-tenth the level of any observed health effects in human or animal studies (Figure 6.6). However, 99 percent of the exposure distribution falls above the EPA acceptable intake level (RfD of 0.003 mg/kg-day). This apparent discrepancy occurs largely because the toxicological database is incomplete and the EPA has considered this uncertainty in determining its acceptable intake level. Uranium has not been demonstrated to serve a beneficial purpose in biological systems; therefore, unlike nutrient metals, a toxicity threshold is difficult to define. Though these low intake levels are not associated with adverse effects in humans or test animals, it is important that most of the exposure distribution falls above this RfD criterion because of the low level of confidence in the toxicological data.

For vanadium, 99 percent of the exposure distribution is below the acceptable intake level; most of the distribution occurs within dietary ranges (Figure 6.7). If the potential ground water exposure is added to dietary intake, the combined dose is still below acceptable values.

Based on the assumptions of this risk assessment, the levels of arsenic, uranium, and vanadium detected in the shallow well southeast of the site would not be associated with adverse health effects if ground water were used for irrigation and watering livestock.

The exposure distributions in Figures 6.1 through 6.7 are based only on the ground water ingestion pathway for residential drinking water. In Section 4.2, the following additional exposure pathways were screened and found to be negligible compared to the drinking water ingestion pathway: dermal absorption; ingestion of ground water-irrigated produce; ingestion of meat from ground water-fed livestock; and ingestion of milk from ground water-fed livestock. To verify that these additional pathways would not contribute substantial risks, the results of the screening calculations for each contaminant of potential concern were used to adjust the calculated risks for ground water ingestion by an amount appropriate to the incremental risks associated with those additional exposure pathways. Noncarcinogenic risks (and carcinogenic risks) did not increase notably when additional exposure pathways were considered; therefore, the drinking water ingestion pathway is the dominant ground water exposure pathway, as indicated by the screening calculations in Section 4.2.

Semiconfined aquifer

Levels of manganese, molybdenum, sulfate, and uranium are also elevated in the semiconfined aquifer. Chronically ingested at these levels, manganese would cause mild neurologic effects; molybdenum would cause increased copper excretion, potentially leading to gout; and sulfate could cause severe diarrhea. Uranium is associated with carcinogenic effects at these levels.

6.2 POTENTIAL CARCINOGENIC HEALTH EFFECTS

All uranium isotopes are radioactive and, as such, are considered carcinogens. Table 6.1 shows the uranium intake from the most contaminated well in the surficial aquifer and the potential lifetime carcinogenic risk associated with exposure to this drinking water. This estimate is based on the cancer SF developed by the EPA; however, natural uranium has not been demonstrated to cause cancer in humans or animals following ingestion.

Table 6.1 Calculation of excess lifetime cancer risks from ingesting ground water, Riverton, Wyoming, site

Radionuclides	CW[a] (pCi/L)	Exposure dose[b] (pCi/lifetime)	Oral slope factor[c] $(pCi)^{-1}$	Cancer risk[d]
Lead-210	4.0	9E+04	1.01E-09	9E-05
Polonium-210	2.4	5.0E+04	3.26E-10	2E-05
Thorium-230	6.8	1.4E+05	3.75E-11	5E-06
Uranium-234	4E+02[e]	8.8E+06	4.44E-11	4E-04
Uranium-238	4E+02[e]	8.8E+06	6.20E-11	5E-04

[a]The exposure point concentration is represented by the maximum concentration detected in plume well 707 for lead-210, polonium-210, and thorium-230, and by the mean concentration detected in plume well 707 for uranium-234 and -238.
[b]Calculated using equations given in Table 4.1 for exposure dose resulting from ingesting ground water, and the following assumptions: the ingestion rate = 2 L water per day; the exposure frequency = 365 days per year; the exposure duration = 30 years.
[c]The basis for these oral slope factors is presented in Table 5.1.
[d]Cancer risk = exposure dose x slope factor.
[e]The mean of uranium concentration in ground water from plume well 707 is 1.2 mg/L; 1 mg uranium is assumed to equal 686 pCi; the secular equilibrium between uranium-234 and uranium-238 is assumed to estimate activity of different radioisotopes of uranium.

The estimated lifetime excess cancer risk from ingesting uranium in ground water (9×10^{-4}) exceeds the National Contingency Plan guidance for maximum increased lifetime cancer risk of 1×10^{-4}. This calculation is based on a cumulative, 30-year exposure. As discussed previously, this exposure duration is probably appropriate, but uranium concentrations in ground water are expected to decline because the tailings have been removed. Therefore, this calculation probably overestimates risk.

Uranium decays to other radioactive elements that are also considered carcinogenic. Table 6.1 shows cancer risks from other progeny detected above background levels. Uranium is the only radionuclide that has a potential risk of greater than 1×10^{-4}.

Arsenic is not radioactive but is carcinogenic due to its chemical properties. The carcinogenic potential of arsenic is currently under evaluation by the EPA. However, based on the cancer potency previously assigned to arsenic, the expected

concentration value of 0.014 mg/L in ground water would be associated with a lifetime excess cancer risk of 2×10^{-4}, which is above the National Contingency Plan guidance value of 1×10^{-4}.

6.3 LIMITATIONS OF THIS RISK EVALUATION

The following potential limitations should be noted in interpreting this risk evaluation.

- This risk evaluates only risks related to ground water contaminated with inorganics. This document does not address potential contamination from any of the few organic constituents used in uranium processing.
- Populations that might have increased sensitivity, such as the elderly or individuals with preexisting diseases, are not specifically addressed on the toxicity ranges presented on the graphs.
- For reasons not yet determined, some individuals may be more sensitive to the toxic effects of certain constituents.
- Data available to interpret potential health effects may not always be 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 has been evaluated hydrologically and geochemically, the monitoring locations sampled may not be in the most contaminated portion of the plume. Additionally, concentrations may increase or decrease substantially as the plume moves.
- Only the drinking water exposure pathway was considered in depth, although other pathways were screened to determine their contributions, which are small compared to the drinking water ingestion pathway.

The evaluation presented here considers these limitations and compensates where possible by presenting toxicity ranges and probabilistic exposure assessments rather than point estimates, incorporating as much variability as could be reasonably defined. The impact of these potential limitations is discussed more fully in Section 8.2.

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7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION

The objective of the environmental portion of this risk assessment is to determine if contaminants detected at the site have the potential to adversely affect the existing biological community at or surrounding the site. Currently, the EPA has not issued guidelines for quantifying potential impacts to ecological receptors but has developed a qualitative approach generally used for ecological evaluation (EPA, 1989b). With the qualitative approach, the EPA recommends comparing ambient environmental media concentrations to water quality, sediment quality, or other relevant criteria to determine whether any concentrations the ecological receptors are expected to encounter exceed these criteria.

The effects of contaminants on ecological receptors are of concern; however, it is difficult to predict whether observed effects on specific populations will damage the ecosystem. Since populations are dynamic, information concerning the normal range of variability within a population is important. Although difficult to detect, sublethal effects may be very important to overall ecosystem health (e.g., 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 site. An exposure pathway generally consists of four elements: a source and mechanism of contaminant release, a retention or transport medium, a point of receptor contact with the contaminated medium, and an exposure route. If any of these elements are missing, exposure cannot occur.

Originally, the source of contamination at the site was the uranium mill and 70-ac (30-ha) tailings pile. However, removal of the tailings pile and associated contaminated soil was completed in 1990 with permanent stabilization in the off-site disposal cell (DOE, 1991). Thus, exposure to tailings or contaminated near-surface soils do not represent an ecological concern at this site.

Currently, the primary source of contamination at the site is the contaminant plume in ground water. Ecological receptors could be exposed to ground water at the site. The shallow depth to ground water may allow plant roots to access soil saturated with contaminated ground water, resulting in plant uptake of contaminants. Plants would also be exposed to contaminants in ground water if ground water were used to irrigate fields or gardens. In addition, livestock watered with ground water drawn from wells are potentially exposed to contaminants. These potential exposure pathways will be evaluated in this assessment.

Surface water in the site vicinity could be affected if contaminated ground water from the site discharges into area surface water. In addition, it is possible that contaminants migrated to surface water bodies through surface runoff before the tailings pile was removed. Surface water bodies in the site vicinity that could be affected by site contaminants include the Little Wind River and other water bodies

(wetland areas, streams, and drainage ditches) south of the site (Figure 2.2). All these surface water bodies are potential exposure points for resident aquatic life and terrestrial wildlife that may come in contact with surface water or sediment. In addition, terrestrial wildlife may ingest plants or animals that have bioaccumulated contaminants from surface water or ground water. All these potential exposure pathways will be addressed in this risk assessment.

7.2 ECOLOGICAL RECEPTORS

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

7.2.1 Terrestrial flora

The Riverton tailings site is in an arid, high-desert environment. Vegetation includes shrubs, grasses, and forbs typical of the low desert. The riparian zone along the Wind and Little Wind Rivers contains dense populations of cottonwood, Russian olive, and willow trees. The site, including a large empty field north of the former tailings pile location, has been severely disturbed. As a result, approximately 30 percent of the plant species at the site are of the primary succession type that invade disturbed areas, such as Russian knapweed and Swainson's pea. Typical native species include wheatgrass, sand dropseed, big sagebrush, and rabbitbrush (DOE, 1987). Appendix D of the 1987 EA conducted for this site (DOE, 1987) lists plant species found at or in the site vicinity.

7.2.2 Terrestrial fauna

No comprehensive wildlife surveys or inventories have been conducted for the area around the Riverton site. The marshy areas near the site contain pheasant, chukar, hawk, owl, blackbird, small game, and ground-dwelling rodent populations. Mule deer and white-tailed deer are confined mainly to the riparian zone along the Wind River and brushy drainages approximately 1 mi (1.6 km) from the site. A large variety of waterfowl, including many riparian species, are found along the rivers and adjacent marshy areas. Canada geese, snow geese, and many duck and shorebird species are common in these areas (DOE, 1987).

During a field survey conducted 8 and 9 June 1993, terrestrial mammals observed in the site vicinity included prairie dogs and a red fox. Signs of beaver, deer, and possibly muskrat were observed in the wetland and riparian areas. Bird species observed in the wetland areas south of the site include numerous red-winged and yellow-headed blackbirds, swallows, eastern kingbirds, a northern oriole, a western tanager, a common yellowthroat, a cormorant, killdeer, American robins, mallard ducks, and herons.

Reptiles found in the area include the short-horned lizard and the fence lizard. The garter snake is likely to occur in the marshy areas adjacent to the site. Depending on seasonal weather conditions and runoff, the marshes contain water intermittently

and could provide breeding habitats for amphibians such as the leopard frog and spadefoot toad (DOE, 1987).

Appendix D of the 1987 EA (DOE, 1987) lists animal species found at or in the vicinity of the Riverton site.

7.2.3 Aquatic life

A brief qualitative survey of the aquatic organisms observed in the Little Wind River, the wetland areas, the drainage ditch east of the site, and the unnamed tributary to the Little Wind River was conducted during the June 1993 field survey. A fine-mesh dip net was used to collect fish and benthic macroinvertebrates at these locations.

Sampling location 742 is on the west bank of the river, between sampling locations 794 and 796 (Figure 2.2). The water at sampling location 742 moved more slowly than at the other two sampling locations because the main channel and water flow bypassed the side channel from which the sample was taken. The river's velocity at sampling locations 794 and 796, coupled with its high turbidity at all three locations, limited visual observation to the immediate shoreline area. At sampling locations 794 and 796, the substrate along the shore-water interface was primarily large pebbles and cobbles, with small, isolated areas of sand. The substrate at sampling location 742 was composed of fine-grained sands and silt; no rocks or cobbles were seen. A few small sucker fish, ranging in size from approximately 2 to 3 inches (5.1 to 7.6 cm) were caught with a dip net at sampling locations 794 and 796. Several damselfly and dragonfly nymphs were also collected. The Little Wind River reportedly contains rainbow, brown, and brook trout; three species of suckers (longnose, white, and mountain); carp; flathead chub; and the longnose dace (DOE, 1987).

A wetland area is approximately 0.25 mi (0.40 km) southeast (downstream) of Mission Road and just north of the unnamed tributary levee (Figure 2.2). This wetland area is approximately 100 ft (30 m) long and 40 ft (12 m) wide, with water depths ranging from less than 1 inch (2.5 cm) to approximately 20 inches (50 cm). Sampling location 745 is at the approximate center. This wetland contained an abundance of filamentous algae and submergent macrophytes. The bottom sediments were covered with algal mats or macrophytes in most areas. Many adult and young-of-the-year minnows were observed. A few large (approximately 2-inch [50-millimeter]) tadpoles were seen, as well as numerous water scavenger beetles. Damselfly and dragonfly nymphs were observed, as well as unidentified amphipods and cladocerans.

A wetland area approximately 1300 ft (396 m) east of the site starts just south of Goes In Lodge Road and trends southeast to the Little Wind River (Figure 2.2). Sampling location 746 is in this wetland area, approximately 25 ft (7.6 m) north of Highway 137. No areas of open water were observed in the portion of the wetland that could be seen from the highway. The wetland contained an extremely dense

cattail stand. The only aquatic organisms observed were chironomid worms in the sediment.

Observations were also made at the drainage ditch that runs east from the site. This drainage ditch parallels an irrigation canal. Sampling location 744 was 60 ft (18 m) east of the fence that constitutes the eastern boundary of the site (Figure 2.2). There was no perceptible water flow in this ditch. The water depth ranged from approximately 8 to 12 inches (20 to 30 cm). No fish were observed. Water striders, unidentified water fleas, large numbers of mosquito larvae, and small isolated patches of green algae were seen.

The water level was very high in the Little Wind River, which flowed rapidly in the vicinity of sampling locations 794 (upstream location) and 796 (downstream location).

Sampling location 743 is in a drainage area approximately 1500 ft (457 m) east of St. Stephens Mission and approximately 400 ft (122 m) north of the Little Wind River. This area is pasture. Because of the high river water, it was flooded during the field survey. Two carp approximately 18 inches (46 cm) long were observed in this flooded area. These fish probably migrated there from the river. No other aquatic life was observed.

The unnamed tributary is approximately 1250 ft (380 m) southwest of the site and flows northwest to southeast, eventually joining the Little Wind River approximately 750 ft (230 m) east of St. Stephens Mission (Figure 2.2). Observations along the reach of this unnamed tributary from 170 ft (52 m) upstream of Mission Road (sampling location 741) downstream to Highway 137 showed the water was clear, ranging from approximately 4 inches (10 cm) deep at sampling location 741 to approximately 24 inches (61 cm) deep where the tributary flows under Highway 137. The substrate was composed primarily of fine-grained sand interspersed with small rocks in certain areas. The following organisms were observed: water striders, snails, backswimmers, damselfly and dragonfly nymphs, and unidentified minnows. Floating mats of filamentous algae also were observed in this tributary.

7.3 CONTAMINANTS OF ECOLOGICAL CONCERN

The list of ground water contaminants that exceed background levels (Table 3.4) was used as the list of contaminants of potential concern in ground water for ecological receptors (e.g., plants).

Before the June 1993 sampling activities, sediment and surface water samples had not been collected from the site vicinity in more than 3 years. Therefore, the constituents analyzed in these water bodies were evaluated in this assessment.

7.4 POTENTIAL IMPACTS TO TERRESTRIAL AND AQUATIC LIFE

7.4.1 Risk to terrestrial life

Plants

The contaminant plume in ground water currently represents the primary source of contamination at the site. Because the depth to ground water is from 6 to 10 ft (2 to 3 m), plant roots could extend into the ground water table, resulting in plant uptake of contaminants. In addition, plants may be exposed to contaminants if ground water is used to irrigate fields or gardens. Plants exposed to ground water may accumulate contaminants in various plant parts, which may result in adverse effects.

One way to evaluate plant exposure to chemicals in ground water is to estimate the plant tissue concentrations resulting from contaminant uptake from ground water and compare these values to toxic levels in mature leaf tissue. Ground water data from monitor well 707 were used to evaluate this pathway, because contaminant concentrations in this well were the highest values observed at the site for most contaminants of potential concern.

Constituent partitioning is assumed to occur between ground water and the soil in which the plants are growing. Therefore, a soil concentration was estimated for each contaminant of concern from the mean ground water concentrations by multiplying the water concentrations by a chemical-specific soil-to-water partition coefficient (Kd). Soil concentrations were estimated because only soil-to-plant uptake factors are available. The estimated soil concentrations were then multiplied by soil-to-plant uptake factors (Baes et al., 1984), resulting in an estimated plant tissue concentration. Baes et al. (1984) present two types of uptake factors: those used to estimate contaminant concentrations in the vegetative portions of plants (e.g., stems, leaves) and those used to estimate contaminant concentrations in the reproductive portions of plants (e.g., fruits, tubers). This assessment estimates concentrations for both the vegetative and reproductive portions of plants. Estimated plant tissue concentrations for the contaminants of potential concern were compared with toxic concentrations in mature leaf tissue (Kabata-Pendias and Pendias, 1992). Table 7.1 presents the results of this comparison.

Where toxicity data were available, all estimated plant tissue concentrations fell below levels shown to be toxic in mature leaf tissue (Table 7.1). However, toxicity data were not available for the following 12 of the 21 contaminants of potential concern: aluminum, bromide, calcium, chloride, iron, magnesium, potassium, silica, sodium, sulfate, strontium, and uranium.

Potential effects on plants exposed to ground water were also evaluated by comparing mean ground water concentrations to concentrations in continuously applied irrigation water reported to be protective of plants. The water

Table 7.1 Comparison of estimated plant concentrations to phytotoxic concentrations, Riverton, Wyoming, site

Contaminant of potential concern	Mean concentration in ground water[a] (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factors Bv	Br	Estimated concentration in vegetative growth (mg/kg DW)	Estimated concentration in fruits/tubers (mg/kg DW)	Approximate toxic concentration in mature leaf tissue (mg/gram DW)[b]
Aluminum	0.14	1500	2.1E+02	0.004	0.00065	0.8	0.14	NA
Arsenic	0.011	200	2.1E+00	0.04	0.006	0.09	0.013	5-20
Boron	0.20	3	6.0E-01	4	2	2.4	1.2	50-200
Bromide	0.22	7.5	1.6E+00	1.5	1.5	2.4	2.4	NA
Calcium	40	4	1.6E+03	3.5	0.35	560	56	NA
Chloride	120	0.25	3.0E+01	70	70	2100	2100	NA
Fluoride	0.80	150	1.2E+02	0.06	0.006	7.2	0.72	50-500
Iron	0.35	25	8.8E+00	0.004	0.001	0.035	0.0088	NA
Magnesium	200	4.5	9.1E+02	1	0.55	910	500	NA
Manganese	4.9	65	3.2E+02	0.25	0.05	80	16	200[c]-1000
Molybdenum	0.79	20	1.6E+01	0.25	0.06	4.0	0.96	10-50
Nickel	0.16	150	2.4E+01	0.06	0.06	1.4	1.4	10-100
Potassium	14.3	5.5	7.9E+01	1	0.55	79	43	NA
Selenium	0.022	300	6.6E+00	0.025	0.025	0.17	0.17	5-30
Silica	29	30	8.6E+02	0.35	0.07	300	60	NA
Sodium	900	100	9.0E+04	0.075	0.055	6800	5000	NA
Sulfate	3100	7.5	2.3E+04	0.5	0.5	12,000	12,000	NA
Strontium	2.5	35	8.8E+01	2.5	0.25	220	22	NA
Uranium	1.2	450	5.4E+02	0.0085	0.004	4.6	2.2	NA
Vanadium	0.030	1000	3.0E+01	0.0055	0.003	0.17	0.09	5-10
Zinc	0.0075	40	3.00E-01	1.5	0.9	0.45	0.3	100-400

[a]Data from monitor well 707.
[b]Concentrations are not presented for very sensitive or for highly tolerant plant species (Kabata Pendias and Pendias, 1992).
[c]The value of 200 mg/kg is a recommended criterion of the Illinois Institute of Environmental Quality (Saric, 1986).

Bv - soil-to-plant uptake factor for vegetative portions (e.g., leaves, stems) of plants (Baes et al., 1984).
Br - soil-to-plant uptake factor for reproductive portion (e.g., fruits, tubers) of plants (Baes et al., 1984).
Kd - soil-to-water partition coefficient (Baes et al., 1984).
DW - dry weight.
NA - not available.

concentrations protective of plants were developed for agricultural purposes (WDEQ, 1990; EPA, 1972). Table 7.2 presents the results of this comparison, showing that manganese and molybdenum concentrations could build up to toxic levels in the soil following long-term use. In addition, plants could be affected by the sulfate in ground water. Because comparison values were not available for 7 contaminants of potential concern (bromide, calcium, magnesium, potassium, silica, sodium, and strontium), the plant toxicity of these contaminants cannot be evaluated.

Another way to evaluate the suitability of the ground water for plants is to evaluate the water's total soluble salt content. Excess salts in water increase the osmotic pressure of a soil solution, creating a physiological drought condition in plants growing in the soil. The total soluble salt content of water can be measured by its specific conductance (electrical conductivity). The mean specific conductance from monitor well 707 is 3730 micromhos per centimeter (micromhos/cm). This value falls above the upper end of the acceptable range (1500 to 3000 micromhos/cm) for water that may have adverse effects on many crops; using this water would require careful management (Follett and Soltanpour, 1985).

Wildlife

Wildlife may be exposed to contaminants from the site by ingesting surface water that has received site contamination in runoff or ground water discharge. However, state and federal criteria or standards have not been established for the protection of terrestrial wildlife from water exposure. Thus, this potential exposure pathway was not evaluated.

Wildlife also may be exposed to contaminants by ingesting plants or other animals that have bioaccumulated site contaminants (e.g., birds eating aquatic macroinvertebrates). Standards or guidelines are not available to evaluate this potential exposure pathway.

Biomagnification refers to the tendency of some chemicals to accumulate in higher concentrations at higher levels of the food chain through diet. Top predators, especially carnivorous birds and mammals, are of particular concern in determining the effects of biomagnification.

7.4.2 Risk to aquatic life

Little Wind River

Aquatic life in the Little Wind River may be exposed to contaminants associated with the site. It is possible that contaminants from the ground water plume at the site are discharging to the Little Wind River or that surface water runoff washed contaminants into the river before the tailings were removed.

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

Contaminant of potential concern	Mean concentration in ground water[a] (mg/L)	Water concentration protective of livestock[b] (mg/L)	Water concentration protective of plants[b] (mg/L)
Aluminum	0.14	5.0	5.0[c]
Arsenic	0.011	0.2	0.10
Boron	0.20	5.0	0.75
Bromide	0.22	NA	NA
Calcium	410	NA	NA
Chloride	120	2000[c]	100[c]
Fluoride	0.80	2.0	1.0
Iron	0.35	NA	5.0
Magnesium	200	NA	NA
Manganese	4.9	NA	0.20
Molybdenum	0.79	10[d]	0.01
Nickel	0.16	NA	0.20
Potassium	14	NA	NA
Selenium	0.022	0.05	0.02
Silica	29	NA	NA
Sodium	900	NA	NA
Sulfate	3100	1000[e], 3000[c]	200[c]
Strontium	2.5	NA	NA
Uranium	1.2	5.0[c]	5.0[c]
Vanadium	0.030	0.1	0.10
Zinc	0.0075	25	2.0

[a]Data from monitor well 707.
[b]EPA (1972), unless specified otherwise.
[c]WDEQ (1990).
[d]No state or federal guidelines available. Value shown is the U.S. Fish and Wildlife Service safe level for cattle (Eisler, 1989).
[e]National Research Council (1971).

NA - not available.

Little Wind River water

The three sampling locations on the Little Wind River are 794 (background), 742, and 796 (see Figure 2.2). These stations were last sampled in June 1993. Surface water was analyzed for calcium, iron, magnesium, manganese, molybdenum, lead, and uranium. The samples were not filtered and thus represent total metals in the surface waters. These data were compared with available chronic federal water quality criteria (FWQC) for the protection of aquatic life (EPA, 1986). The ambient water quality criteria for the state of Wyoming do not differ from the FWQC for the constituents evaluated in surface water at the site (WDEQ, 1990).

Surface water data from the Little Wind River collected in the June 1993 sampling event are shown in Table 7.3. Because molybdenum was not detected in surface water, a comparison for molybdenum was not necessary. Of the remaining constituents, FWQC were available only for iron and lead. Iron concentrations at all three river sampling locations exceeded the criterion of 1.0 mg/L, including the background location. The highest iron concentration was found at sampling location 796, which exceeded the criterion by a factor of 3. However, it should be noted that iron concentrations in upstream and downstream sampling locations were not notably different. The criterion for lead, 0.0058 mg/L, is hardness-dependent and was calculated using an average hardness for the Little Wind River of 161 mg/L as $CaCO_3$. Lead concentrations measured in the Little Wind River did not exceed this criterion. In addition, the downstream lead concentrations were the same as the concentration measured at the background location.

Table 7.3 Occurrence of constituents in the Little Wind River surface water, June 1993 sampling event, Riverton, Wyoming, site

Contaminant of potential concern	Location ID 794 (background)	Location ID 742	Location ID 796
Calcium	41	41	42
Iron	2.0	1.4	3.1
Magnesium	14	14	14
Manganese	0.09	0.12	0.14
Molybdenum	ND	ND	ND
Lead	0.004	0.004	0.004
Uranium	0.002	0.025	0.002

All concentrations reported in milligrams per liter.
ND - not detected.

Uranium concentrations at sampling location 742 exceeded background by more than 1 order of magnitude and may thus represent an ecological concern. This

sampling location is closest to the highest chemical levels detected in ground water (monitor well 707). Farther downstream at sampling location 796, uranium concentrations decline to background levels.

FWQC were not available for calcium. However, calcium probably is not of ecological concern in surface water at this site because the observed calcium concentration is low.

Historical data collected from August 1987 through March 1990 are available for sampling locations 794 and 796. Samples from these locations were analyzed for 45 metals and radionuclides. Most of the samples were filtered. However, one sample taken at sampling location 796 in March 1990 was unfiltered. A comparison of the data at sampling location 796 (filtered) with the data from sampling location 794 (background) shows that most concentrations at sampling location 796 fall below background levels. Five constituents essentially were at background levels, but six slightly exceeded the background levels: calcium, chloride, lead-210, mercury, strontium, and sulfate. Three constituents (radium-226, radium-228, and uranium) in the earlier sampling rounds exceeded background levels but in the later rounds either fell below background levels or only slightly exceeded background. Historical data show a total uranium concentration of 0.0088 mg/L at sampling location 796 in March 1990. In June 1993, the level of total uranium measured at this same location was 0.002 mg/L, showing a slight decline in concentration.

Little Wind River sediment

No tribal, state, or federal sediment quality criteria (SQC) have been established for the protection of aquatic life. Although the EPA has developed interim SQC for several nonpolar hydrophobic organic compounds (EPA, 1988), no such SQC have been developed for the contaminants of potential concern at this site.

In the absence of SQC, effects-based sediment guidelines available from the National Oceanic and Atmospheric Administration (NOAA) (NOAA, 1990) were compared to the chemical concentrations detected in sediments of the Little Wind River. The NOAA sediment-effects values were developed to determine which chemical concentrations are likely to result in adverse effects, based on established data. The effects range-low (ER-L) values are concentrations equivalent to the lower 10th percentile of available data screened by NOAA and indicate the low end of the concentration range in which adverse effects were observed or predicted.

The sediment data collected at the three sampling locations on the Little Wind River in June 1993 are shown in Table 7.4. Of the five constituents analyzed, a NOAA value was available only for lead. The levels of lead detected in Little Wind River sediment fall below the ER-L value of 35 mg/kg. Guidelines were not available with which to assess the potential ecological effects of iron, manganese, molybdenum, and uranium in sediment at the site. With the exception of molybdenum,

Table 7.4 Occurrence of constituents in the Little Wind River sediment, June 1993 sampling event, Riverton, Wyoming, site

Constituent	Location ID 794 (background)	Location ID 742	Location ID 796
Iron	5900	21,000	8500
Manganese	170	620	210
Molybdenum	ND	ND	4.0
Lead	4.5	14	3.9
Uranium	2.0	10	2.3

All concentrations reported in milligrams per kilogram.
ND - not detected.

concentrations of these constituents did exceed background levels, with the highest concentrations measured at sampling location 742. Molybdenum was not detected at the background location (794) or at location 742.

Other water bodies

Several other surface water bodies in the site vicinity may be potential exposure points for aquatic life. It is possible that contaminants from the ground water plume at the site discharge to these areas or that contaminants washed into these areas through surface water runoff before the tailings were removed.

Surface water

During the June 1993 sampling event, five areas were sampled and surface water was analyzed for calcium, iron, magnesium, manganese, molybdenum, lead, and uranium. The samples were not filtered and thus represent total metals in the surface waters.

Table 7.5 presents the surface water data from the five sample locations. These data were compared with available chronic FWQC for the protection of aquatic life. Because lead was not detected in surface water, comparisons were not applicable. Neither were comparisons applicable for molybdenum, which was detected only at the background location. For the remaining chemicals, an FWQC was available only for iron. Iron concentrations did not exceed the FWQC of 1.0 mg/L at any sampling locations.

Table 7.5 Constituents in surface water from water bodies other than Little Wind River in the site vicinity, June 1993 sampling event, Riverton, Wyoming, site

Constituent	Location ID 741[a] (background)	Location ID 743	Location ID 744	Location ID 745	Location ID 746
Calcium	46-70	160	58	81	110
Iron	0.43-0.46	0.24	0.13	0.36	0.09
Magnesium	13-20	38	14	22	39
Manganese	0.05-0.05	0.12	0.02	0.05	0.06
Molybdenum	ND-0.02	0.01	ND	ND	ND
Lead	ND	ND	ND	ND	ND
Uranium	0.006-0.006	0.013	0.016	0.008	0.008

[a]Range represents values from duplicate samples.

ND - not detected.

FWQC were not available for calcium, magnesium, manganese, and uranium. Sampling location 743 had the highest calcium concentration, exceeding the maximum background level by a factor of 2.3. Where measured concentrations are similar to background concentrations, however, these concentrations probably do not represent an ecological concern, nor can they be attributed to contaminants from the site. Many concentrations measured for these constituents were similar to background levels. The detected concentrations exceeded maximum background concentrations at most by a factor of 2 to 2.5. Based on the limited data available, the significance of such an increase is uncertain.

Sediment

Table 7.6 shows the sediment data collected in June 1993 at the five sampling locations. Of the five constituents analyzed in the sediments, a NOAA value was available only for lead. Lead levels detected at sampling location 746 exceeded the NOAA ER-L of 35 mg/kg. This sampling station is only a few feet from Highway 137. Therefore, although this lead concentration may result in ecological effects, the source of this lead probably is related to automobile exhaust fumes from the highway rather than from activities at the Riverton site. Sediment guidelines were not available to assess the potential ecological effects of iron, manganese, molybdenum, and uranium in sediment at the site. However, because manganese and molybdenum concentrations did not exceed maximum concentrations measured at the background location, they probably are not of concern. Iron and uranium exceeded maximum background concentrations by a factor of 1.5 at most. The significance of this increase above background is not known.

Table 7.6 Occurrence of constituents in sediments from water bodies in the site vicinity, June 1993 sampling event, Riverton, Wyoming, site

Constituent	Location ID 741[a]	Location ID 743	Location ID 744	Location ID 745	Location ID 746
Iron	3000-11,000	18,000	12,000	3200	11,000
Lead	3.6-11	21.9	6.0	4.8	68
Manganese	180-460	240	190	54	310
Molybdenum	ND-11	2.0	ND	1.0	9.0
Uranium	1.9-7.6	5.0	5.5	1.7	11

[a]Range represents duplicate sample taken at this location.

All concentrations reported in milligrams per kilogram.
ND - not detected.

7.5 POTENTIAL IMPACTS TO LIVESTOCK

Both cattle and horses are raised in the site vicinity. Currently, some residents water their livestock with ground water. To evaluate the potential impact to livestock that may be exposed to ground water contaminants, Table 7.2 compares the concentrations detected in monitor well 707 to drinking water concentrations considered protective of livestock (WDEQ, 1990; EPA, 1972; Eisler, 1989). Data from monitor well 707 were used to evaluate livestock exposure because contaminant concentrations in this well were the highest values observed at the site for most contaminants.

Comparison values for livestock were available for 11 of the 21 contaminants of potential concern in ground water. Results of the comparison (Table 7.2) show that the mean sulfate concentration in ground water exceeds by a factor of about 3 the water concentrations the EPA considers protective of livestock while the state of Wyoming criterion is exceeded slightly. Nevertheless, watering livestock with ground water may cause adverse effects.

Information was not sufficient to evaluate the following 12 contaminants: bromide, calcium, chloride, iron, magnesium, manganese, nickel, potassium, silica, sodium, strontium, and uranium. Therefore, the potential effects on livestock from the intake of these chemicals could not be evaluated with available information.

7.6 LIMITATIONS OF THE ECOLOGICAL RISK ASSESSMENT

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

- 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 mixed contaminants.

7.7 SUMMARY

Comparing mean ground water concentrations with water concentrations protective of plants indicates that manganese, molybdenum, and sulfate are at concentrations that could adversely affect plants. Thus, plants whose roots may contact ground water-saturated soil or plants irrigated with ground water could be affected by these contaminants of potential concern in ground water.

Essentially, no data were available to evaluate the effects of exposure on terrestrial wildlife species.

The surface water concentrations for iron detected in the Little Wind River exceed available FWQC. However, for most constituents analyzed in surface water, the downstream location differed little from concentrations at the background location. The downstream location is approximately 9000 ft (2740 m) downriver from the background location. The uranium concentration measured at sampling location 742 is approximately 1 order of magnitude higher than the background concentration and could represent an ecological concern. This sampling location is closest to the highest contaminant levels detected in ground water. Farther downstream at sampling location 796, uranium concentrations decline to background levels.

An evaluation of sediment data from the Little Wind River showed that downstream concentrations exceeded background concentrations, with the highest concentrations observed at sampling location 742. This is the same location at which higher uranium values were observed in surface water. Sediment quality values were available only for lead. The sediment lead concentrations did not exceed the sediment quality values.

In other water bodies in the site vicinity, the surface water concentrations did not exceed FWQC. Since FWQC are not available for several constituents, however, it is not certain whether any of these concentrations threaten aquatic organisms. At most, the concentrations exceeded maximum background concentrations by a factor of 2 to 2.5. No clear trend is associated with surface water concentrations in these areas. The locations of highest values varied for the different constituents.

A comparison of sediment quality values with sediment data from these areas shows that the lead concentration at sampling location 746 exceeds the NOAA ER-L. Because this sampling location is only a few feet from Highway 137, these concentrations probably are due primarily to automobile exhaust fumes rather than site-related activities. Sediment quality values were not available for any other constituents detected in sediments. Manganese and molybdenum concentrations did not exceed background concentrations, and concentrations of the other constituents exceeded maximum background concentrations by a factor of 1.5 at most. No clear trend is associated with sediment concentrations in these water bodies. The locations of highest values varied for the different constituents.

Potential exposure to livestock that ingest ground water was evaluated by comparing mean ground water concentrations to water concentrations determined

to be protective of livestock. Based on the sulfate concentrations in ground water, the results showed that adverse effects could occur because the mean sulfate ground water concentration was 3 times greater than the guidance value.

Water and sediment quality criteria and/or guideline values were insufficient to comprehensively evaluate the impact of ground water, surface water, and sediments on ecological receptors. Therefore, this assessment of ecological risk could evaluate only some of the constituents detected at the site.

8.0 INTEPRETATION AND RECOMMENDATIONS

8.1 RISK SUMMARY

The UMTRCA requires the UMTRA Project to protect public health and the environment from radiological and nonradiological hazards associated with the former uranium mill sites. This baseline risk assessment of the Riverton site was conducted to evaluate the presence of these hazards. This risk assessment is conservative in that it evaluates residential exposure associated with the highest level of exposure by drinking water from the most contaminated well near the site. Because contaminated ground water is not currently used for drinking water by area residents, human health is not at risk. This situation will continue if land and water use at or near the site remain the same. Changes of land use may or may not create future risks. As new land uses are determined, they should be evaluated to identify potential health and environmental risks from the contaminated ground water. Although one private irrigation well showed very low levels of contamination, these levels are not associated with adverse health effects if water from this well were used to irrigate crops or water livestock.

Future drinking water use of contaminated ground water could, however, be associated with adverse health effects. However, the likelihood of such use is considered low because the natural quality of the shallow aquifer water is poor and it is not used for drinking in the site area. Note also that in the future residential scenario, only the people who drill a well in the most contaminated portion of the aquifer (a small fraction of contamination) could experience the health problems discussed below. Drinking water from a future well drilled farther downgradient from the site could result in risks lower than estimated here. Furthermore, the ground water contaminant concentrations will decline over time.

Using the surficial aquifer as a source of drinking water in the future could cause an unacceptable risk of 9×10^{-4} for uranium and 2×10^{-4} for arsenic, exceeding the EPA National Contingency Plan guideline of a maximum of 1×10^{-4} lifetime excess cancer risk. Potential noncarcinogenic health effects from exposure to molybdenum, manganese, and sulfate could be of concern. These exposures could result in adverse health effects such as severe diarrhea (sulfate), neurologic changes (manganese), and biochemical imbalances (molybdenum, manganese, and sulfate).

Using contaminated ground water from the surficial aquifer to irrigate crops or gardens or to water livestock is not expected to threaten human health. Adverse human health effects would not be expected to follow ingestion of milk or 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.

Available data indicate recreational use of the Little Wind River in the site vicinity (which could result in ingestion of fish and incidental ingestion of surface water and sediments) is not expected to result in adverse human health effects.

The contaminated ground water near the processing site would not be acceptable as a source of continuous irrigation water for agricultural crops. In addition, the adverse effects of sulfate may make the ground water unsuitable as a sole source of drinking water for livestock.

Based on data from monitor wells near the river and surface water and sediment quality data, it appears that contaminated ground water has reached the Little Wind River. However, water and sediment quality criteria and/or guidelines were insufficient to comprehensively evaluate the impact of ground water, surface water, and sediments on ecological receptors. Therefore, this assessment of ecological risk could evaluate only some constituents detected at the site.

8.2 LIMITATIONS OF THIS RISK ASSESSMENT

The following limitations to this health risk evaluation should be noted:

- This document evaluates risks associated only with exposure to inorganic contaminants of ground water at the UMTRA Project site near Riverton. 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-micrometer) water samples. The effect of filtration differs for different elements: e.g., filtering uranium and sulfate results in only slight differences, whereas filtering manganese can considerably underestimate levels present. Constituents in suspension can be lost with filtration but still produce toxic effects if ingested and broken down in the acidic environment of the stomach.

- The toxicity of contaminants varies from individual to individual. This assessment emphasizes that variability by using the probability distributions for potential exposure and by presenting exposure ranges that can produce toxic effects. Additionally, data used to determine toxicity were obtained by testing laboratory animals at exposure doses different from those expected at the site. The relationship between dose and response is not always linear, and humans and animals do not always exhibit similar toxic effects.

- To assess toxicity, standardized reference values (developed by various agencies) and literature values were used to determine plant uptake, tissue concentrations in livestock, and toxic effects in humans. These reference values are limited: exposure to multiple chemicals occurs simultaneously; toxicity, uptake, and bioconcentration data are not available for all constituents; and data used to determine toxicity are generally based on exposure to only one chemical. The interactive effects of multiple constituents and the impact of these interactions on expected toxicity generally cannot be assessed from existing data.

- Although considerable effort has been directed at determining plume movement and locating monitor wells to identify maximum contamination, variability in

physical systems and models used to determine contaminant plume migration could still result in well placements that do not measure the highest contaminant concentrations or determine the fullest extent of plume impact.

- Variability can be introduced through sampling and analytical processes. However, the data at UMTRA Project sites have been collected over many years and subjected to rigorous quality assurance procedures. The use of multiple samples introduces high confidence in the reliability and validity of the collected data.

- The drinking water pathway is considered the major determinant of exposure in this assessment. Although other pathways were screened and determined not to contribute substantially to the total exposure, the additivity of exposure from these pathways should be considered. The potential risk contributions from the additional ground water exposure pathways (dermal absorption, ingestion of irrigated produce, and ingestion of meat and milk from livestock that drink the ground water) were reviewed to verify that those pathways are insignificant relative to the drinking water pathway.

Nonetheless, by presenting ranges of toxic effects, probable exposure distributions, summaries of available data on health effects and interactions, and outlines of potential limitations, this document realistically interprets the potential health risks associated with use of contaminated ground water at this site. Based on available data, this assessment is designed to accurately describe contamination, risk, and uncertainties.

8.3 GROUND WATER CRITERIA

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

In general, the EPA ground water standards are sufficient to protect humans and the environment. However, some risk assessments may identify site-specific factors that suggest these standards are not appropriate. When standards are too restrictive, as for example, when there may be no potential for exposure, a less restrictive ACL may be sought. In other cases, the standards may not be sufficiently protective (as for example, if many contaminants near the MCL have additive or synergistic adverse health effects).

Table 8.1 Concentration limits of constituents

Constituent	EPA MCL for UMTRA (mg/L)	EPA health advisories 10-kg child, 10-day[a] (mg/L)	EPA health advisories 70-kg adult lifetime[a] (mg/L)
Chemicals (inorganic)			
Antimony	NA	0.015[b]	0.003[c]
Arsenic	0.05	NA	NA
Barium	1.0	NA	2
Boron	NA	0.9	0.6
Cadmium	0.01	0.04	0.005
Chromium	0.05[c]	1.0	0.1[b]
Lead	0.05	NA	0.015[b,d]
Mercury	0.002[c]	NA	NA
Molybdenum	0.1[b]	0.08[b]	0.04[b]
Nickel	NA	1.0	0.1[b]
Nitrate (as NO_3^-)	44[e]	44[f]	NA
Selenium	0.01[b]	NA	NA
Silver	0.05	0.2	0.1
Strontium	NA	25.0	17
Thallium	NA	0.007	0.0004
Zinc	NA	6.0	2
Radionuclides			
Radium-226/-228	5 pCi/L	NA	NA
Uranium (U-234/-238)	30 pCi/L[b] (0.044 mg/L)	0.03 mg/L[g,h]	0.1 mg/L[g]

[a]EPA (1995).
[b]Exceeded in plume wells.
[c]Exceeded in background and plume wells in surficial aquifer.
[d]Action level for lead.
[e]Equal 10 mg/L nitrate as nitrogen.
[f]Under review.
[g]Proposed values, under review; expected revision 1995.
[h]Longer-term health advisory.
NA - not available.

While these ground water protection and cleanup standards apply specifically to the UMTRA Project, the EPA has also published drinking water health advisory levels (EPA, 1995) for both long-term and short-term exposures (Table 8.1). Molybdenum, nickel, and uranium have consistently exceeded these levels at this site.

8.4 RISK MITIGATION MEASURES

Using the contaminated ground water from the shallow aquifers could cause health problems. Possible ways to restrict ground water access and thus mitigate the risk of adverse health effects are presented below.

The UMTRA standards require permanent government entities to implement institutional controls (controlling access to contaminated ground water). The Wind Rivers Tribes' Environmental Quality Commission has negotiated with the state of Wyoming to define its role in administering ground water activities on the reservation (Stockdale, 1993). The negotiations, held during the summer of 1993, and the development of tribal water quality standards, may help determine appropriate site-specific institutional controls.

Currently, the state administers a permitting process for the installation of new wells on tribal lands. Water quality testing is voluntary for owners of new wells; some analytical results are attached to well completion reports and sent to the state. There is no minimum acreage restriction for new wells.

Some wells installed on tribal lands were not permitted by the state. The Wind River Environmental Quality Commission tabulates all wells on tribal lands. Tribal engineers conduct technical review of well installations to lessen the potential for aquifer cross contamination and to confirm that wells have been constructed and installed correctly. Until the Wind River Tribes develop their own water quality standards and regulations, the tribal water quality requirements must comply with EPA standards (Hart, 1993).

The Wyoming Engineer's Office of Regulations and Instructions, Part II Ground Water, Section 16, provides for the designation of control areas by the Wyoming Board of Control. Control areas can be imposed if ground water levels are declining, if ground water use is approaching current recharge rate, or if "other conditions exist or . . . arise that require regulation for the protection of the public interest." To establish a new control area, the state engineer reports to the state Board of Control, proposing the control area and describing why a control area should be established. After a public hearing, the state Board of Control evaluates the testimony and either designates a control area or rejects the proposal.

Section 16 provides for the election of a Control Area Advisory Board composed of five control area residents to advise and assist the state engineer in formulating policies concerning ground water development in the control area. The advisory board provides a means by which regulations for the use of ground water can be developed and implemented, should they be needed. When a control area is established, a Control Area Advisory Board is elected and the appropriate water

division superintendent adjudicates all ground water rights within the designated control area. The state engineer may then establish necessary controls and regulations with the advice of the Control Area Advisory Board and the Division Advisory Committee on Ground Water.

Establishing interim institutional controls to ensure the protection of human health and the environment will require a consensus among the Arapaho and Shoshone Tribes and other governmental agencies. Local authorities will need to monitor new wells and ensure they have been approved. Governing authorities will need to be informed of monitoring results and the expected duration of contamination problems. The DOE, the state of Wyoming, and environmental officials from both tribes should work together to educate local residents about potential risks and the need for access restrictions.

8.5 RECOMMENDATIONS

The levels of arsenic, manganese, molybdenum, sulfate, and uranium in the surficial aquifer between the former processing site and the Little Wind River could be associated with adverse health effects if the ground water is used for drinking in the future; therefore, ground water from the contaminated portion of the aquifer should not be used until the water quality improves.

Monitoring ground water from the unconfined surficial aquifer, the semiconfined aquifer, and potential surface expression points should continue until detailed characterization of the site ground water is complete. Monitoring the Little Wind River, including sampling during a low-flow period, may be desirable to assess the potential impact of contaminated floodplain ground water on river water quality. Additional sampling will help to better evaluate the potential for ecological impact. Such additional characterization will be discussed in future Riverton ground water documents and work plans.

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