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**BASELINE RISK ASSESSMENT OF
GROUNDWATER CONTAMINATION AT
THE URANIUM MILL TAILINGS SITE
NEAR SHIPROCK, NEW MEXICO**

Draft

September 1993

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Draft

September 1993

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

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Albuquerque, New Mexico**

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CITIZENS' SUMMARY

This *Baseline Risk Assessment of Groundwater Contamination at the Uranium Mill Tailings Site near Shiprock, New Mexico* evaluates potential impact to public health or the environment resulting from groundwater contamination at the former uranium mill processing site. The tailings and other contaminated material at this site were placed in a disposal cell on the site in 1986 by the U.S. Department of Energy's (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project. Currently, the UMTRA Project is evaluating groundwater contamination. This risk assessment is the first document specific to this site for the Groundwater Project.

There are no domestic or drinking water wells in the contaminated groundwater. This includes two distinct groundwater units: the contaminated groundwater in the San Juan River floodplain alluvium below the site and the contaminated groundwater in the terrace area where the disposal cell is located. Because no one is drinking the affected groundwater, there are currently no health or environmental risks directly associated with the contaminated groundwater. However, there is a potential for humans, domestic animals, and wildlife to be exposed to surface expressions of groundwater such as seeps and pools in the area of the San Juan River floodplain below the site. For these reasons, this risk assessment evaluates potential exposure to contaminated surface water and seeps as well as potential future use of contaminated groundwater.

This risk assessment follows the approach outlined by the U.S. Environmental Protection Agency (EPA). The first step is to evaluate groundwater data collected from monitor wells at the site. Evaluation of these data showed that the main contaminants in the floodplain groundwater are arsenic, magnesium, manganese, nitrate, sodium, sulfate, and uranium. The complete list of contaminants associated with the terrace groundwater could not be determined due to the lack of the background groundwater quality data. However, uranium, nitrate, and sulfate are evaluated since these chemicals are clearly associated with uranium processing and are highly elevated compared to regional waters. It also could not be determined if the groundwater occurring in the terrace is a usable water resource, since it appears to have originated largely from past milling operations.

The next step in the risk assessment is to estimate how much of these contaminants people would be exposed to if a drinking well were installed in the contaminated groundwater or if there were exposure to surface expressions of contaminated water. Potential exposures to surface water include incidental contact with contaminated water or sediments by children playing on the floodplain and consumption of meat and milk from domestic animals grazed and watered on the floodplain.

These estimated amounts of contaminants that people might take in are then compared to the toxic effects these contaminant levels might cause. Based on this analysis, there would be no adverse health effects associated with children playing on the floodplain and wading in surface water or incidentally ingesting surface water or sediments. Ingestion of beef and milk from domestic animals grazed and watered on the floodplain would not be associated with adverse health effects.

The most significant health hazard in the groundwater at the Shiprock mill site is nitrate. Nitrate is of primary concern for infants, since at these levels it interferes with their blood's ability to carry oxygen. This toxic effect occurs only in infants because their stomachs absorb food differently than adults' stomachs. The levels of nitrate in the floodplain and terrace groundwater near the site exceed levels that have been reported elsewhere to be lethal when ingested by infants. For this reason, the DOE is working with the Navajo Nation to restrict access to contaminated groundwater until the site can be further evaluated.

Other contaminants that occur at levels associated with adverse health effects in the floodplain groundwater are sulfate, magnesium, manganese, and sodium. The sulfate and magnesium concentrations in the floodplain and terrace groundwater near the site would cause severe diarrhea and dehydration if groundwater were ingested. The amounts of manganese potentially ingested would be expected to affect the nervous system with long-term exposure, and sodium levels would contribute to the development of hypertension.

The uranium and arsenic concentrations at the site may increase the risk of cancer. The additional cancer risk from ingesting contaminated groundwater for a lifetime would be 7 in 1000 for the terrace and 1 in 100 for the floodplain, on average. The additional lifetime cancer risk from exposure to arsenic in the floodplain groundwater would be 1 in 1000, on average.

This risk assessment also evaluated potential effects on livestock if the groundwater were used to water cattle or sheep and if cattle or sheep were to drink water from the seeps of contaminated groundwater while grazing on the floodplain. These animals are sensitive to nitrate. The nitrate concentrations near the mill site are high enough to cause these animals to die if they drank the contaminated groundwater. The nitrate levels in the wet area below seep 425 make this water unsuitable for drinking by animals grazing on the floodplain.

Potential impacts to the environment from contaminated groundwater are not likely. Based on water quality criteria available for a number of contaminants of concern, no ecological threat exists to plants that may have roots in contact with the most contaminated groundwater in the floodplain alluvial aquifer. However, potential use of this groundwater would not be suitable for crop irrigation due to boron, manganese, and selenium levels, and due to the salinity and sodium hazard to the plants. This groundwater would also not be suitable as a source of water for fish to live in.

The mill tailings disposal cell on the terrace at the Shiprock site is currently under long-term surveillance and monitoring. As part of the monitoring, wells are routinely sampled. The Shiprock site will be further evaluated under the UMTRA Groundwater Project. This will include evaluation of the terrace groundwater and its movement and further evaluation of the floodplain groundwater and surface water. Based on these results and this risk assessment, an approach will be developed to address water contamination. This approach will be presented in an environmental assessment that will include public and tribal government involvement. In the interim, access to contaminated groundwater will be controlled.

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LIST OF ACRONYMS AND ABBREVIATIONS

<u>Acronym</u>	<u>Definition</u>
ac	acre
ACL	alternate concentration limit
BCF	bioconcentration factor
BIA	Bureau of Indian Affairs
cm	centimeter
cm ²	square centimeter
cm ³	cubic centimeter
cm/s	centimeters per second
°C	degrees Celsius
°F	degrees Fahrenheit
DOE	U.S. Department of Energy
DW	dry weight
EPA	U.S. Environmental Protection Agency
ER-L	effects range-low
ft	foot
ft/day	feet per day
ft ³ /s	cubic feet per second
FWQC	Federal Water Quality Criteria
gal	gallon
gal/min	gallons per minute
GJPO	Grand Junction Projects Office
HEAST	Health Effects Assessment Summary Tables
ha	hectare
IHS	Indian Health Service
in	inch
IRIS	Integrated Risk Information System
kg	kilogram
km	kilometer
L	liter
L/kg	liter per kilogram
L/min	liters per minute
LOAEL	lowest-observed-adverse-effect level
LTSP	long-term surveillance plan
m	meter
m ³ /s	cubic meters per second
MCL	maximum concentration limit
μg/kg	micrograms per kilogram
μmhos/cm	microhmhos per centimeter
mg/g	milligrams per gram
mg/kg	milligrams per kilogram
mg/kg/day	milligrams per kilogram per day
mg/L	milligrams per liter
μg/L	micrograms per liter
mi	mile

LIST OF ACRONYMS AND ABBREVIATIONS (Concluded)

<u>Acronym</u>	<u>Definition</u>
mL	milliliter
mm	millimeter
MSL	mean sea level
NECA	Navajo Engineering and Construction Authority
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NTUA	Navajo Tribal Utility Authority
PEIS	programmatic environmental impact statement
pCi/L	picocuries per liter
RDA	recommended daily allowance
RfD	reference dose
SAR	sodium adsorption ratio
SQC	sediment quality criterion
TDS	total dissolved solids
UCL	upper confidence limit
UMTRA	Uranium Mill Tailings Remedial Action
UMTRCA	Uranium Mill Tailings Radiation Control Act

1.0 INTRODUCTION

The purpose of this baseline risk assessment is to determine whether the groundwater contamination at the Shiprock, New Mexico, uranium mill tailings site could adversely affect public health or the environment. The Shiprock site is one of 24 abandoned uranium mill tailings sites that are undergoing remediation in accordance with the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA) of 1978 (Public Law 95-604). The remediation is being performed under the oversight of the U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project.

Under the UMTRA Surface Project, the DOE is stabilizing tailings in disposal cells that minimize radon and other radiation emissions and minimize further contamination of groundwater. Remedial action at Shiprock was completed in 1986 with stabilization of the tailings at the former uranium mill site.

Under the UMTRA Groundwater Project, the DOE will evaluate groundwater contamination at the 24 sites, determine whether any action is necessary, and implement the remedial action. The DOE was authorized to perform groundwater restoration in the 1988 amendments to the UMTRCA.

The strategy to evaluate and methods to clean up contaminated groundwater at the UMTRA Project sites will be described in a programmatic environmental impact statement (PEIS) for groundwater contamination at UMTRA Project sites, currently in preparation. This baseline risk assessment has been prepared to evaluate potential health and environmental risks and provide information to assist in determining what remedial actions are necessary. Following the PEIS and risk assessment, a site-specific environmental assessment will be prepared to evaluate and select a remedial action for the Shiprock site.

This first assessment of groundwater-related risks is based on available groundwater data. When this data base is insufficient to adequately evaluate risks, critical data gaps will be identified. Because for most UMTRA Project sites this is the first look at risks from site-related contamination of groundwater, only the major exposure pathways are evaluated here. If future decisions or actions at this site cause the primary pathway to change, minor pathways will be evaluated in more detail at that time. This document will be updated following completion of the site characterization phase of the project.

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

- Data evaluation.
 - Combining data from various site investigations.
 - Comparing sample results with background and tailings source data.
 - Selecting chemical data for use in the risk assessment.

- Exposure assessment.
 - Characterizing the exposure setting.
 - 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.
 - Characterizing potential biota exposure pathways.
 - Identifying potential ecological receptors.
 - Evaluating environmental risk qualitatively.

This approach is used to estimate risks from potential current and future use of contaminated groundwater or surface water near the site.

2.0 SITE DESCRIPTION

The Shiprock site is located on the Navajo Indian reservation, 1 mile (mi) [1.6 kilometers (km)] south of the town of Shiprock, San Juan County, in northwestern New Mexico. It lies on the southwest bank of the San Juan River. Farmington, New Mexico, is approximately 30 mi (50 km) east of the Shiprock site (Figure 2.1).

2.1 SITE BACKGROUND

The former Navajo Mill at the Shiprock site was constructed and operated from 1954 to 1963 by Kerr-McGee Oil Industries, Inc., and from 1963 to 1968 by Vanadium Corporation of America and its successor, Foote Mineral Company. Before and during the milling operations, the site was leased from the Navajo Nation. When the lease expired in 1973, ownership of the site reverted to the Navajo Nation.

The mill reportedly processed approximately 1.5 million short tons (1.4 million metric tons) of ore along with smaller quantities of bulk precipitates from heap leach operations from the Monument Valley area and from purchased vanadium liquor (DOE, 1985). Ore processing consisted of crushing, leaching with sulfuric acid, washing, and extracting uranium and vanadium with organic solvents [di(2-ethylhexyl) phosphoric acid and tributyl phosphate in kerosene]. Tailings from the washing circuit and yellow cake filtrates were pumped to tailings disposal areas. Raffinate from the solvent extraction operation was allowed to evaporate in separate unlined lagoons to the west and southeast of the tailings piles (Figure 2.2).

Between 1984 and 1986, the tailings were stabilized permanently on the site by consolidating the tailings and associated contaminated soils into a recontoured pile. The final cover at Shiprock consists of a 7-foot (ft) [2-meter (m)] thick compacted clayey layer that serves as a barrier to radon emanation and water infiltration. The riprap cap is designed to provide erosion protection for the effective life of the cell (up to 1000 years) and to act as a barrier to the intrusion of plants and animals into the infiltration/radon barrier and the stabilized tailings (DOE, 1990). The disposal cell covers about 76 acres (ac) [31 hectares (ha)] (Thiers, 1986). A security fence encloses the embankment.

The Shiprock site has been stabilized and is in precicensing status. The site is expected to remain in this status until licensed by the NRC under the provisions of 10 CFR 40 for long-term surveillance and maintenance. The general license becomes effective for the Shiprock disposal site after the NRC concurs with the completion of the surface remedial action and approves the long-term surveillance plan (LTSP). Upon licensing, responsibility for conducting the long-term surveillance program for the disposal cell will be transferred from the DOE UMTRA Project Office in Albuquerque to the long-term surveillance and maintenance program at the DOE Grand Junction Projects Office (GJPO).

FIGURE 2.1
SHIPROCK SITE LOCATION MAP
SHIPROCK, NEW MEXICO, SITE

FIGURE 2.2
AERIAL VIEW OF SHIPROCK SITE IN OPERATING PERIOD
SHIPROCK, NEW MEXICO, SITE

2.2 CLIMATE

The climate of the Shiprock area is semi-arid, with a total annual precipitation of about 6 inches (in) [15 centimeters (cm)]. The wettest months are July through October, when approximately half the annual precipitation falls. The driest month is June. During the winter, snowfall is light, occurring only about 10 days per year and averaging 4 in (10 cm). Skies are normally clear; solar illumination and surface water evaporation are high. The average annual pan evaporation rate is 65 in (165 cm); 74 percent of the annual evaporation occurs from May through October.

Summer days are warm to hot, while summer evenings are cool. Winters are relatively mild. The annual average temperature is 52 degrees Fahrenheit (°F) [11 degrees Celsius (°C)]. Normal daily highs range from 42°F (5.5°C) in January to 94°F (34°C) in July, while normal daily lows range from 15°F (-9.4°C) in January to 58°F (14°C) in July. The extreme temperatures that can be expected in Shiprock range from -26°F (-32°C) in December to 107°F (42°C) in June through August (DOE, 1984). Prevailing light winds are from the south and east quadrants. Higher prevailing winds are usually associated with frontal systems and are from the north and west.

2.3 PHYSIOGRAPHIC SETTING

The disposal cell is on an elevated river terrace southwest of the San Juan River (Figure 2.3). The site is relatively level, with natural elevation ranging from a high of approximately 4980 ft (1520 m) above mean sea level (MSL) along the southwestern edge of the property to a low of about 4950 ft (1510 m) along the top of a 50-ft (15-m) high, northwest-southeast trending escarpment, separating the elevated terrace from the modern floodplain of the San Juan River. South of the site, the terrace continues gently upward for approximately 2500 ft (760 m) where it meets the weathered and colluvial covered bedrock uplands.

The modern floodplain of the San Juan River is located at the base of the escarpment north of the disposal cell. It begins approximately 1000 ft (300 m) upstream of the northeast corner of the site, widens to a maximum width of about 1500 ft (460 m), then pinches out against the bedrock escarpment at the bridge supporting Route 666 over the San Juan River, approximately 2200 ft (670 m) downstream of the northwestern corner of the site.

Two arroyos are located east and west of the tailings site. Bob Lee Wash is an arroyo bordering the western side of the site. Many Devils Wash parallels Bob Lee Wash approximately 2500 ft (760 m) southeast of the site.

2.4 HYDROGEOLOGY

2.4.1 Geology

There are four hydrogeologic units relevant to the Shiprock baseline risk assessment: the terrace alluvium, the floodplain alluvium, the Mancos Shale, and the deep, confined bedrock aquifers (Dakota Sandstone and Morrison Formation).

The disposal cell foundation rests on approximately 10 to 45 ft (3 to 13 m) of terrace alluvium. This alluvium consists of interbedded sands and silts with numerous lenses of gravel and cobbles.

The terrace deposit is underlain by Mancos Shale consisting of flat-lying beds of shales and sandy shales. In the Shiprock area, the Mancos Shale is approximately 1000 ft (300 m) thick (McLean and Johnson, 1987). The upper 10 to 30 ft (3 to 9 m) of the Mancos Shale is highly weathered, exhibiting fractures, fissility, and low strength. Below the highly weathered zone, the shale is more competent and relatively impermeable. Beneath the Mancos Shale are the Dakota Sandstone and the Morrison Formation.

North of the disposal cell is the floodplain of the San Juan River. The floodplain alluvium consists of unconsolidated, interbedded boulders, cobbles, gravels, sands, silts, and clays. The Mancos Shale underlies the floodplain alluvium at an average depth of approximately 15 ft (5 m). The alluvium within the floodplain is separated from the terrace deposit beneath the disposal cell by a 50-ft high vertical cliff. Mancos Shale is exposed in the lower two-thirds of the cliff.

2.4.2 Terrace hydrology

Groundwater below the disposal site on the terrace occurs in the alluvium; in the upper, weathered part of the Mancos Shale; and in the Dakota Sandstone and Morrison Formation. Some groundwater may also occur in fractures or sandstone tongues in the unweathered Mancos Shale, but the extent of these occurrences has not been determined. The groundwater in the alluvium and the upper part of the Mancos Shale is unconfined and constitutes the uppermost aquifer. Groundwater in the Dakota Sandstone and Morrison Formation is confined. A flowing well near the site is reportedly completed in the Morrison Formation from approximately 1500 to 1900 ft (450 to 570 m) below land surface (McLean and Johnson, 1987). The free-flowing condition demonstrates that the piezometric surface in the Morrison Formation is higher than the water table in the alluvium or the upper part of the Mancos. This higher head, combined with the low permeability of the unweathered Mancos Shale, will preclude movement of impacted water beneath the tailings pile into the deeper aquifers.

The terrace alluvium is recharged by precipitation, and in the vicinity of the site probably still contains water discharged from the former milling operations and tailings pile. The numerous lagoons active during the milling operations, as shown on Figure 2.2, reportedly lost as much as 4.9 million gallons (gal) [19 million

liters (L)) per month. This amount of water in such a dry climate could have formed a considerable groundwater mound perched on the Mancos Shale. Infiltrating seepage would move across the shale surface in all directions and infiltrate the upper, weathered zone of the shale. As the size of the mound diminished after the milling operations ceased, the movement of the perched groundwater would be controlled by the topography of the Mancos Shale.

Historically, the San Juan River meandered across the terrace surface. Monitor wells show that remnant erosional swales and low ridges exist on the top of the eroded surface of the Mancos Shale (Figure 2.4). These ridges and swales could cause the perched groundwater to flow to the west-northwest instead of directly toward the present San Juan River, as would normally be anticipated. To date, there are not enough data to develop an accurate flow map of the groundwater in the terrace alluvium.

Monitor wells installed to bracket the alluvial/bedrock interface and test pits that were excavated into the weathered shale indicate that only a relatively thin layer of water is perched on the bedrock. This perched groundwater represents a very limited resource for future development.

The groundwater in the alluvium also percolates down into the upper, fractured part of the Mancos Shale. Some of this water moves horizontally along bedding planes and can be seen seeping from bedding planes in the shale along the escarpment face immediately north of the disposal site. It is also anticipated that groundwater in the deeper Mancos may flow toward the San Juan River, the local base level in the region. This water would then move up out of the Mancos into the alluvium of the floodplain or directly into the river (Figure 2.5).

2.4.3 Floodplain hydrology

The floodplain alluvium contains a shallow groundwater system. It is recharged primarily by water from the San Juan River that enters the floodplain at its upstream end, approximately 1000 ft (300 m) east of the tailings pile. Additional recharge comes from the flowing well at Bob Lee Wash and from the seeps along the escarpment. The floodplain alluvium is also the local discharge zone for the underlying Mancos Shale. The groundwater in the floodplain then discharges back to the river along the downstream half of the floodplain. A water table map of the floodplain is shown in Figure 2.6.

Some of the groundwater flow in the floodplain enters a drainage ditch that separates the northwestern third of the floodplain (Figure 2.6). This ditch follows a preexisting natural meander channel in the floodplain (see Figure 2.2) but appears to have been artificially deepened. During periods of high water flow, the water table intersects this canal. Some of the groundwater flow is diverted into the ditch, which discharges to the river at the extreme northwestern end of the floodplain. During low water periods, however, the water table is beneath the bottom of the ditch, and this channel will not affect the movement of the groundwater.

FIGURE 2.5
CROSS SECTION A-A'
SHIPROCK, NEW MEXICO, SITE

The water table map (Figure 2.6) does not indicate the possible influence of the swale cutting across the floodplain, but it shows the groundwater mound resulting from the flow down Bob Lee Wash.

The groundwater within the floodplain on the north side of the San Juan River enters at the upstream end and discharges along the downstream portion. This floodplain also probably receives recharge from the upland areas farther to the north. The water table elevations on the opposite sides of the river will preclude the movement of groundwater from one side of the river to the other. All water will eventually discharge to the San Juan River or the channel fill. Therefore, tailings leachate within the floodplain alluvium will not migrate beneath the San Juan River.

The stratigraphy of the floodplain is extremely variable, both horizontally and vertically. The fluvial deposits are typical of floodplains in such environments; they vary from coarse gravel and cobbles found in high energy channels to fine, uniform sands indicative of point bars to silty sediments typical of quiet backwater areas. This heterogeneity means that there is no one value of hydraulic conductivity representative of the entire floodplain.

Visual inspection of the sediments and comparisons with typical ranges described in the literature (Hunt, 1984) suggest that the hydraulic conductivity could vary by two orders of magnitude, from less than 5 feet per day (ft/day) [0.001 centimeters per second (cm/s)] to more than 300 ft/day (0.1 cm/s). The sediments encountered during installation of monitor wells 734 and 736 (see Figure 3.1), where groundwater discharges from the floodplain into the San Juan River, were relatively uniform, medium-grained sands. Because the conductivity of these sediments will control the rate of discharge of impacted water to the San Juan River, an intermediate hydraulic conductivity typical of such sediments [approximately 30 ft/day (0.01 cm/s)] will be used in this study.

The hydraulic gradient also varies across the floodplain, ranging from 0.001 to 0.002. A hydraulic gradient of 0.002 was calculated for the area of discharge and is used in this study.

A calculation based on a hydraulic conductivity of 30 ft/day (0.01 cm/s), a hydraulic gradient of 0.002, and an assumed cross-sectional area of discharge along the southern bank of the San Juan River 8 ft (2.4 m) deep and 2500 ft (750 m) long results in an average groundwater discharge of 0.015 cubic feet per second (ft³/s) [30 liters per minute (L/min)].

2.5 SURFACE WATER

The San Juan River forms the eastern and northern boundaries of the floodplain north of and below the disposal cell. Surface runoff south and west of the site flows to the east and north into Bob Lee Wash. Bob Lee Wash also receives a constant discharge of approximately 60 gallons per minute (gal/min) [230 liters per minute (L/min)] from an artesian well located west of the wash (see well 648 in

Figure 3.1). Discharge from this well flows down Bob Lee Wash and discharges into a swampy area on the floodplain. This water eventually flows into a drainage swale that cuts across the floodplain and joins the San Juan River approximately 0.5 mi (0.8 km) northwest of the disposal cell. Surface runoff east of the tailings pile either flows into a borrow area east of the site, then down the escarpment onto the floodplain, or it reaches Many Devils Wash, which discharges into the San Juan River approximately 0.5 mi (0.8 km) east of the cell.

Three unnamed arroyos that cut the escarpment above the floodplain can be seen in the aerial photograph in Figure 2.2, which was taken prior to the site cleanup. These arroyos were filled during the surface restoration activities at the site. According to a 1977 letter report (Hans, 1977), seeps were identified and sampled in these arroyos.

In 1990, two seeps were identified originating from the escarpment of Mancos Shale that rises from the San Juan River floodplain (Figure 2.3). One seep is located on the escarpment about 15 ft (5 m) above the floodplain near the northwest corner of the disposal cell. This seep is about 30 ft (9 m) long and consists of a series of drips under an overhanging indurated sand lens within the Mancos Shale. The flow rate from the seepage face was approximately 0.5 gal/min (2 L/min) in January 1991. The second seep is located immediately south of where Bob Lee Wash enters the San Juan River floodplain. This seep is approximately 5 ft (2 m) above the floodplain and was flowing at a rate of about 1 gal/min (4 L/min) in January 1991. It is anticipated that the flow rate from these seeps will decrease during the dry months of the year.

2.6 LAND USE

The Shiprock site is on the Navajo Indian reservation adjacent to the unincorporated town of Shiprock. Figure 2.7 shows the land uses in the vicinity of the site.

According to unofficial estimates of Shiprock's population in 1980 (FBDU, 1981), there were 90 people living within 0.5 mi (0.8 km) of the site and 2200 within 1 mi (1.6 km). In 1983, the population of Shiprock was estimated at 8000.

A mix of residential and commercial development exists near the site. A U.S. Public Health Service building, Navajo Engineering and Construction Authority (NECA) facilities, Abandoned Mine Land Program office buildings, and fairgrounds are immediately west of the former mill site. West of U.S. Highway 666 is a residential area with a day care center and community development offices.

Approximately 80 people (including 30 children 15 years of age and younger) live in a residential area west of Bob Lee Wash (TAC, 1993). This community consists of approximately 15 households, with an average size of five persons per household. A hogan located within this community is used for ceremonial purposes only. Most of the residents have been living at the present location for at least 10 years. Many of them raise livestock such as horses, cows, sheep, donkeys, rabbits, ducks, and

REF: MODIFIED FROM FBDU, 1981.

FIGURE 2.7
LAND USE IN THE VICINITY OF
SHIPROCK, NEW MEXICO, SITE, 1993

chickens. These animals graze near the Bob Lee Wash and may also graze in the San Juan River floodplain. Only a few residents keep vegetable gardens.

Approximately 10 households are located southwest of the disposal site, with an average size of four persons per household. This community consists of approximately 40 persons, including 14 children. Most of these residents have been living at the present location for 18 years. Very few of them raise domestic animals such as chickens, and they do not have vegetable gardens.

Individual members from the Navajo Nation do not literally own land; rather, land use rights are maintained by several systems of land tenure. The Navajo system for land assignment consists of grazing permits. The grazing permit system was developed in the 1940s to assign land based on sheep units. A Navajo cannot establish a residence without a grazing permit, which is generally passed down or subdivided for family members. The Bureau of Indian Affairs (BIA) Land Operations oversees permit registration. As shown in Figure 2.7, a grazing permit has been issued for the floodplain beneath the site.

For several miles upstream and downstream from the site, the San Juan River floodplain is used for irrigated farming and low-density grazing. The irrigated farm land is used mainly to produce hay for feeding livestock. The floodplain also provides habitat for indigenous wildlife and migrating birds.

2.7 WATER USE

A field search and inquiries in 1993 did not identify any domestic or other wells within or near the residential area northwest of the site. Treated San Juan River water is provided to these residents by the Navajo Tribal Utility Authority (NTUA) (Charley, 1993).

Residences located west of Bob Lee Wash and southwest of the disposal site are also connected to the NTUA Shiprock municipal water supply system. Water from the artesian well (see monitor well 648 in Figure 3.1) is the primary source of water for livestock kept at residences located west of Bob Lee Wash, as well as for watering gardens. However, animals may also consume surface water from the floodplain and Bob Lee Wash.

The NECA facilities, Public Health Service building, and Abandoned Mine Land Program buildings, located on the site, are connected to the municipal water and sewer systems.

The community water system for Shiprock obtains water from the San Juan River at two locations. One intake is located in the San Juan River approximately 8 mi (13 km) upstream (east) of the town (DOE, 1984; NTUA, 1993). This intake is used as a source of water from April to October. It is capable of providing a year-round supply of water, though the NTUA does not have permission to withdraw year-round at this location. The primary source of water for NTUA in winter is located approximately 300 ft (90 m) upstream of the bridge supporting

Route 666 over the river. This latter intake, built around 1978, is on the north side of the San Juan River across the river from the site. Pumps for this intake are installed 15 ft (5 m) below the river surface. A pipeline connection with the Farmington community water system provides a backup system for Shiprock.

Surface water supplies from the San Juan River are treated at NTUA's water treatment plant, which has a design capacity of 3 million gal/day (11 million L/day). In 1991, NTUA's plant treated approximately 200 million gal (800 million L) of water, with an average of 10 to 27 million gal (38 to 102 million L) per month. Approximately 30 percent of this amount originated from the water intake near the former mill site. In 1992, 280 million gal (1 billion L) of water were treated at the plant (NTUA, 1993). The water treatment process consists of three stages: sedimentation (clarification and flocculation with aluminum sulfate and polymer), sand filtration, and treatment with chlorine.

The NTUA's water supply system provides potable water to Shiprock. It is also a water source for industrial and agricultural uses. In 1983, the NTUA provided service to more than 1300 residences and 100 commercial or institutional establishments.

3.0 EXTENT OF CONTAMINATION

Water quality data have been collected from 1984 to 1993 from most of the existing monitor wells and surface locations at Shiprock. At present, there are 8 monitor wells on the terrace, 51 monitor wells on the north and south floodplains, and 8 surface water locations that are currently monitored or have been sampled in the past. Water quality samples have also been collected from a well (648) west of Bob Lee Wash that is completed in the regional aquifer (Morrison Formation). Monitor wells and surface sampling locations are shown in Figure 3.1.

3.1 BACKGROUND WATER QUALITY SUMMARY

Background water quality is defined as the quality of water if uranium milling activities had not taken place. Because there are two distinct shallow hydrologic systems at the Shiprock site, the terrace and the alluvial floodplain, it is necessary to determine background water quality for each hydrologic system.

3.1.1 Terrace

There are currently no background water quality data for the terrace. Therefore, the impact from the uranium milling process can be described only in a general manner. Drilling would be needed to determine both background water quality and the areal and vertical extent of influence from milling activities.

3.1.2 Floodplain

Groundwater in the floodplain below the terrace has been degraded by milling activities. Therefore, data from this groundwater cannot be used to determine background. Instead, background water quality data were taken from wells completed in the alluvial system north of the San Juan River. Wells used in the qualitative and quantitative background analysis are listed in Table 3.1. The locations of background wells are shown in Figure 3.2.

Monitor wells 732 and 733 and domestic wells 634 and 635 were used in the quantitative statistical background analysis. Water quality data from the USGS locations (SJ3 and G10) were used to qualitatively examine and compare groundwater quality in the shallow alluvial system above and below Shiprock.

Background groundwater can be described as a sulfate-bicarbonate, calcium-sodium type with slightly basic pH and total dissolved solids (TDS) ranging from 800 to 5000 milligrams per liter (mg/L). Background groundwater from the six locations (Figure 3.2) has a similar chemical composition. Background groundwater quality data show some variability but fall into the same general chemical field. Variability in water chemistry can be explained by 1) distance of the well from the river (TDS decrease away from the river) and 2) depth of the well (TDS appear to decrease with depth due to the diminishing effect of evaporation with depth).

FIGURE 3.2
BACKGROUND WELLS USED FOR QUALITATIVE (SJ3,G10) AND QUANTITATIVE (732,733,634,635) ANALYSIS OF GROUNDWATER IN THE ALLUVIAL AQUIFER SHIPROCK, NEW MEXICO, SITE

Table 3.1 Wells used to determine background water quality

Well number	Location	Screened interval (depth)	
		Feet	Meters
Background wells installed 02/93			
732	Downstream	7-17	2.1-5.2
733	Across river from floodplain	6.5-11.5	2.0-3.5
Domestic wells across from the floodplain			
634	Across river from floodplain	?-24	?-7
635	Across river from floodplain	?-12	?-4
USGS sampling locations			
SJ3	Well, upstream by Hogback	5-10	1.5-3.0
G10	Seep, downstream by Cudei	NA	NA

USGS – U.S. Geological Survey.

The San Juan River influences groundwater quality adjacent to the river channel. The solute load of the river varies, depending upon which formation the river flows through, the amount of water each tributary contributes to flow, the evaporation rate, and the volume of flow in the river. Because of these effects, groundwater chemistry is not homogeneous in the floodplain adjacent to the river. Sulfate/chloride ratios in background wells in the alluvial aquifer range from 16 to 37, illustrating the variability in solute concentrations in this system. A statistical summary of water quality data for the four alluvial wells used to determine background is shown in Table 3.2.

3.2 MAGNITUDE OF SITE-RELATED CONTAMINATION

3.2.1 Terrace

Groundwater within the terrace is present in alluvial gravels and the upper weathered zone of the Mancos Shale Formation. During the operation of the mill, tailings leachate and tailings were discharged to tailings embankments adjacent to the mill site. Raffinate was discharged to evaporation ponds east and south of the tailings embankments.

Fluids from both raffinate and tailings containment structures infiltrated into the subsurface, contaminating groundwater on the terrace. As a result of milling activities, a groundwater mound probably developed beneath the site. Groundwater then flowed away from the site in all directions, imposing a temporary gradient on the terrace system. The mound has since subsided, but residual contamination remains.

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site

Constituent ^a	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median mg/L	Maximum
<u>Aluminum</u>						
Background	2	0.10	100	0.20	0.20	0.20
Plume ^c	23	0.05-0.20	30	<0.05	<0.10	0.77
<u>Ammonium</u>						
Background	4	0.1	50	<0.1	0.4	2.3
Plume ^d	8	0.1	100	380	516	542
<u>Antimony</u>						
Background	4	0.003	0	-	-	-
Plume ^c	29	0.001-0.1	62	<0.003	0.005-0.02	0.103
<u>Arsenic</u>						
Background	4	0.005-0.01	0	-	-	-
Plume ^c	33	0.005-0.10	42	<0.005	<0.028	0.05-0.10
<u>Barium</u>						
Background	2	0.10	50	<0.10	<0.10	0.10
Plume	5	0.01-0.10	40	<0.01	<0.05	0.03-0.10
<u>Beryllium</u>						
Background	0	-	-	-	-	-
Plume	4	0.01-0.03	0	-	-	-
<u>Boron</u>						
Background	2	0.1	0	-	-	-
Plume	5	0.05-0.20	100	0.60	0.80	0.92
<u>Bromide</u>						
Background	0	-	-	-	-	-
Plume	2	0.01-0.1	50	<0.1	<0.1	0.1
<u>Cadmium</u>						
Background	4	0.001	0	-	-	-
Plume ^c	33	0.001-0.005	36	<0.001	<0.001	0.018

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site (Continued)

Constituent ^a	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median mg/L	Maximum
<u>Calcium</u>						
Background	4	0.01-0.5	100	130	238	353
Plume	7	0.005-2	100	380	490	510
<u>Chloride</u>						
Background	4	1	100	10	57	181
Plume	7	0.5-1	100	385	605	700
<u>Chromium</u>						
Background	4	0.01	50	<0.01	0.02	0.05
Plume ^c	29	0.01-0.05	7	<0.01	<0.01	0.32
<u>Cobalt</u>						
Background	4	0.05	25	<0.05	<0.05	0.08
Plume ^c	25	0.03-0.10	12	<0.01	<0.05	0.08-0.10
<u>Copper</u>						
Background	4	0.02	25	<0.02	<0.02	0.02
Plume ^c	25	0.01-0.10	44	<0.01	<0.02	0.05-0.10
<u>Cyanide</u>						
Background	2	0.01	0	-	-	-
Plume	1	0.01	0	-	-	-
<u>Fluoride</u>						
Background	2	0.1	100	0.6	0.6	0.6
Plume	5	0.1	100	0.3	0.4	0.9

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site (Continued)

Constituent ^a	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
				mg/L		
<u>Iron</u>						
Background	4	0.03	75	<0.03	0.84	6.83
Plume	6	0.01-0.30	33	<0.01	<0.13	0.19-0.30
<u>Lead</u>						
Background	4	0.003-0.01	25	<0.003	<0.007	0.01
Plume ^c	29	0.003-0.05	24	<0.003	<0.02	0.03-0.05
<u>Magnesium</u>						
Background	2	0.001	100	31	91	150
Plume	7	0.001-10	100	1252	2030	2750
<u>Manganese</u>						
Background	4	0.01	100	0.02	1.19	3.99
Plume	7	0.006-0.05	100	5.67	8.08	9.75
<u>Mercury</u>						
Background	2	0.0003	0	-	-	-
Plume	5	0.0003	0	-	-	-
<u>Molybdenum</u>						
Background	4	0.01	50	<0.01	0.03	0.09
Plume ^c	33	0.004-0.05	36	<0.01	<0.01	0.36
<u>Nickel</u>						
Background	2	0.04	0	-	-	-
Plume ^c	27	0.01-0.2	48	<0.01	<0.04	0.40

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site (Continued)

Constituent ^a	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
				mg/L		
<u>Nitrate</u>						
Background	4	1	75	< 1	4.5	310
Plume	7	0.04-1	100	400	3300	5500
<u>Phosphate</u>						
Background	2	0.10	0	-	-	-
Plume ^c	28	0.10	71	< 0.10	0.10	1.53
<u>Potassium</u>						
Background	2	0.01	100	3.25	5.90	8.54
Plume	7	0.01-20	100	108	121	165
<u>Selenium</u>						
Background	4	0.005	25	< 0.005	< 0.005	0.007
Plume	7	0.001-0.05	100	0.070	0.122	0.599
<u>Silica</u>						
Background	2	2	100	11	12	13
Plume	5	0.1-2	100	10	13	15
<u>Silver</u>						
Background	2	0.01	0	-	-	-
Plume	17	0.01-0.05	12	< 0.01	< 0.01	0.03-0.05
<u>Sodium</u>						
Background	4	0.002-0.1	100	66	276	489
Plume	7	0.002-20	100	1650	3500	3810

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site (Continued)

Constituent ^a	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median mg/L	Maximum
<u>Strontium</u>						
Background	4	0.01-0.10	100	1.60	3.28	4.61
Plume	7	0.01-0.10	100	8.94	10.10	14.00
<u>Sulfate</u>						
Background	4	0.1-1	100	256	1465	2860
Plume	7	0.1-10	100	6230	13000	15600
<u>Sulfide</u>						
Background	2	0.1	0	-	-	-
Plume	2	0.1	0	-	-	-
<u>Thallium</u>						
Background	0	-	-	-	-	-
Plume	4	0.01-0.05	25	<0.01	0.035	0.02-0.05
<u>Tin</u>						
Background	2	0.005	0	-	-	-
Plume ^c	23	0.005-0.1	48	<0.005	<0.035	0.089-0.1
<u>Uranium</u>						
Background	4	0.001-0.003	100	0.010	0.016	0.023
Plume	7	0.001-0.003	100	1.64	2.80	4.07
<u>Vanadium</u>						
Background	4	0.01	50	<0.01	0.10	0.23
Plume ^c	26	0.001-0.05	54	<0.01	0.10	0.34
<u>Zinc</u>						
Background	4	0.005	50	<0.005	0.009-0.011	0.150
Plume ^c	29	0.001-0.1	90	<0.005	0.081-0.087	0.202

Table 3.2 Statistical summary of the San Juan River floodplain groundwater quality, 1987 - 1993 filtered samples, Shiprock, New Mexico, site (Concluded)

Constituent ^a	Number of samples	Observed concentrations ^b		
		Minimum	Median	Maximum
		pCi/L		
Radionuclides				
Lead-210				
Background	2	0.00	0.05	0.10
Plume ^c	6	0.10	1.15	1.30
Polonium-210				
Background	2	0.0	0.0	0.0
Plume ^c	13	0.0	0.6	3.3
Radium-226				
Background	2	0.0	0.1	0.2
Plume ^c	32	0.0	0.2	3.0
Thorium-230				
Background	2	0.20	0.30	0.40
Plume ^c	14	0.00	0.35	1.60

^aPlume data are from well 615 except as noted. Background data are from wells 634, 635, 732, and 733.

^bDue to nondetectable measurements, these statistics may not be available. If a range is reported, the statistic is known to lie somewhere within that range.

^cBecause of low detectability, the following plume wells' data were combined: 608, 609, 610, 611, 613, 614, and 615.

^dWell 608 was used to determine ammonium concentration.

mg/L - milligrams per liter.

pCi/L - picocuries per liter.

Water quality data from existing wells on the terrace demonstrate the impact on water quality from milling activities (Table 3.3). Groundwater from monitor wells 600 and 602, analyzed between 1988 and 1993, show high concentrations of uranium, nitrate, and sulfate. Groundwater samples from monitor wells 725, 726, 727, 728, and 731, constructed in the spring of 1993, were also high in nitrate, uranium, and sulfate. The latest drilling program better defined the extent of the plume to the east, south, and west of the disposal cell. However, the location of the edge of the plume is still unknown.

Groundwater flowing from the terrace to the alluvial floodplain probably contaminates groundwater in the alluvial system. Water quality data from seeps originating in the terrace show elevated nitrate, uranium, and sulfate.

3.2.2 Alluvial floodplain

The alluvial floodplain is characterized by elevated TDS and other constituents associated with the milling of uranium (e.g., sulfate, nitrate, and uranium). The sulfate, nitrate, and uranium plumes are delineated in Figures 3.3, 3.4, and 3.5. Groundwater sampled from the floodplain wells has TDS values ranging from 700 to 64,200 mg/L and a pH ranging from 6.5 to 8.0. The wide variability may be due to the presence of several plumes that contribute to the overall plume on the floodplain. These plumes may have formed from leachate and raffinate spills from the terrace onto the floodplain or from ponds, located on the floodplain, used during mill operations.

Monitor wells 608 through 615, excluding well 612 (Figure 3.1), are located in the plume and show elevated concentrations of contamination in the floodplain. Filtered water quality data collected between 1987 and 1993 indicate that monitor wells 608 and 615 have the highest concentrations of milling-related contaminants. Monitor wells located between these two wells show contamination to a lesser degree.

To characterize water quality in the floodplain and to determine constituents that are elevated above background, water quality data from the well showing the worst contamination between 1987 and 1993 were used. In most cases, data from well 615 were used because concentrations were slightly but consistently higher in well 615 than in well 608. However, in the case of ammonium, data from well 608 were used because ammonium appears in the highest concentrations in a limited area around well 608. Many constituents were detected only sporadically at levels near or below the laboratory detection limits in the floodplain. For constituents with very low detectability, it was necessary to pool data from wells 608 through 611 and 613 through 615 to assess possible contamination. Plume water quality data are summarized in Table 3.2.

Nineteen constituents were statistically elevated above background in the alluvial floodplain groundwater. These constituents are ammonium, antimony, arsenic,

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993

Constituent	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
					mg/L	
<u>Aluminum</u>						
600/602	10	0.05-0.87	0	-	-	-
725-731 ^c	0	-	-	-	-	-
<u>Ammonium</u>						
600/602	12	0.1	100	108	333	670
725-731	5	0.1	100	0.3	1.7	218
<u>Antimony</u>						
600/602	12	0.001-0.02	58	<0.007	0.018-0.024	0.106
725-731	5	0.003	0	-	-	-
<u>Arsenic</u>						
600/602	14	0.005-0.1	36	<0.005	<0.04	0.31
725-731	5	0.005	0	-	-	-
<u>Barium</u>						
600/602	12	0.002-0.1	33	<0.01	<0.08	0.016-0.1
725-731	0	-	-	-	-	-
<u>Beryllium</u>						
600/602	8	0.005-0.03	0	-	-	-
725-731	0	-	-	-	-	-
<u>Boron</u>						
600/602	8	0.05-0.2	100	0.7	0.9	1.1
725-731	0	-	-	-	-	-
<u>Bromide</u>						
600/602	2	0.1	0	-	-	-
725-731	0	-	-	-	-	-

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993 (Continued)

Constituent	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median mg/L	Maximum
<u>Cadmium</u> ^d						
600/602 filt	14	0.0001-0.005	50	≤0.0007	<0.002	0.013
600/602 unfilt	2	0.001	50	<0.001	<0.003	0.005
725-731 filt	5	0.001	20	<0.001	<0.001	0.001
725-731 unfilt	5	0.001	100	0.001	0.002	0.006
<u>Calcium</u>						
600/602	14	0.005-2	100	354	367	500
725-731	5	0.5	100	337	433	541
<u>Chloride</u>						
600/602	16	0.5-1	100	79	645	900
725-731	5	0.5	100	52	218	491
<u>Chromium</u>						
600/602	14	0.01-0.1	14	<0.01	<0.015	0.24
725-731	5	0.01	0	-	-	-
<u>Cobalt</u>						
600/602	12	0.005-0.1	17	<0.027	<0.05	0.12
725-731	5	0.05	0	-	-	-
<u>Copper</u>						
600/602	12	0.01-0.05	42	<0.01	<0.02	0.3-0.5
725-731	5	0.02	0	-	-	-
<u>Cyanide</u>						
600/602	2	0.01	0	-	-	-
725-731	0	-	-	-	-	-
<u>Fluoride</u>						
600/602	10	0.1	100	0.2	0.4	0.5
725-731	0	-	-	-	-	-

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993 (Continued)

Constituent	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
				mg/L		
<u>Iron^d</u>						
600/602 filt	12	0.018-0.1	50	<0.018	0.04-0.09	0.21
600/602 unfilt	0	-	-	-	-	-
725-731 filt	5	0.03	0	-	-	-
725-731 unfilt	5	0.03	80	<0.03	4.9	10.4
<u>Lead^d</u>						
600/602 filt	14	0.01-0.05	29	<0.01	<0.02	0.02-0.05
600/602 unfilt	2	0.003-0.03	50	<0.03	0.048-0.063	0.095
725-731 filt	5	0.003	0	-	-	-
725-731 unfilt	5	0.003	60	<0.003	0.003	0.009
<u>Magnesium</u>						
600/602	14	0.001-10	100	543	1339	2940
725-731	0	-	-	-	-	-
<u>Manganese</u>						
600/602	12	0.006-0.05	100	0.6	1.3	2.4
725-731	5	0.01	80	<0.01	0.4	1.5
<u>Mercury</u>						
600/602	12	0.0002	17	<0.0002	<0.0002	0.0005
725-731	0	-	-	-	-	-
<u>Molybdenum</u>						
600/602	14	0.01-0.1	36	<0.01	<0.028	0.45
725-731	5	0.01	40	<0.01	<0.01	0.01
<u>Nickel</u>						
600/602	10	0.04-0.2	30	<0.04	<0.06	0.1-0.2
725-731	0	-	-	-	-	-

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993 (Continued)

Constituent	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
				mg/L		
<u>Nitrate</u>						
600/602	14	0.044-1.0	100	36	236	945
725-731	5	1.0	100	26	738	2310
<u>Phosphate</u>						
600/602	12	0.1	58	<0.1	0.1	4.2
725-731	5	0.1	100	0.1	1.0	1.5
<u>Potassium</u>						
600/602	14	0.01-20	100	45	132	266
725-731	0	-	-	-	-	-
<u>Selenium</u>						
600/602	14	0.001-0.05	57	<0.005	0.04-0.06	0.80
725-731	5	0.005	60	<0.01	0.072	0.25
<u>Silica</u>						
600/602	8	0.1-2	100	8	11	13
725-731	0	-	-	-	-	-
<u>Silver</u>						
600/602	10	0.01-0.05	20	<0.01	<0.01	0.02-0.05
725-731	0	-	-	-	-	-
<u>Sodium</u>						
600/602	14	0.002-20	100	2120	3215	4090
725-731	5	0.1	100	534	1130	3030
<u>Strontium</u>						
600/602	14	0.01-0.1	100	6.2	9.2	15.6
725-731	5	0.01	100	4.9	7.4	14.7

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993 (Continued)

Constituent	Number of samples	Detection limit(s) (mg/L)	Percent above detection	Observed concentrations ^b		
				Minimum	Median	Maximum
				mg/L		
<u>Sulfate</u>						
600/602	16	0.1-10	100	9200	13,900	18,100
725-731	5	1.0	100	3870	4,780	18,100
<u>Sulfide</u>						
600/602	2	0.1	0	-	-	-
725-731	0	-	-	-	-	-
<u>Thallium</u>						
600/602	8	0.01-0.05	50	<0.01	<0.04	0.03-0.05
725-731	0	-	-	-	-	-
<u>Tin</u>						
600/602	8	0.005-0.1	25	<0.005	<0.026	0.1
725-731	0	-	-	-	-	-
<u>Uranium</u>						
600/602	14	0.001-0.01	100	0.81	1.18	1.57
725-731	5	0.001	100	0.022	0.48	0.50
<u>Vanadium^d</u>						
600/602 filt	11	0.001-0.05	55	<0.01	0.04-0.05	0.39
600/602 unfilt	0	-	-	-	-	-
725-731 filt	5	0.01	20	<0.01	<0.01	0.02
725-731 unfilt	5	0.01-0.03	80	<0.03	0.03-0.04	0.05
<u>Zinc</u>						
600/602	12	0.001-0.1	50	<0.005	<0.022	0.091-0.1
725-731	5	0.005-0.05	40	<0.005	<0.012	0.159

Table 3.3 Summary of groundwater quality^a in the terrace, Shiprock, New Mexico, site, 1988 - 1993 (Concluded)

Constituent	Number of samples	Observed concentrations ^b		
		Minimum	Median	Maximum
		pCi/L		
Radionuclides				
Lead-210 ^d				
600/602 filt	4	1.4	2.2	3.2
600/602 unfilt	2	1.3	4.7	8.0
725-731 filt	0	-	-	-
725-731 unfilt	5	0.0	0.0	0.9
Polonium-210 ^d				
600/602 filt	4	0.0	1.3	3.1
600/602 unfilt	2	0.1	0.3	0.4
725-731 filt	0	-	-	-
725-731 unfilt	5	0.0	0.2	1.3
Radium-226 ^d				
600/602 filt	14	0.2	1.5	5.2
600/602 unfilt	2	5.6	6.2	6.7
725-731 filt	0	-	-	-
725-731 unfilt	5	0.3	3.6	6.4
Thorium-230 ^d				
600/602 filt	6	0.0	0.1	0.3
600/602 unfilt	2	1.1	3.7	6.2
725-731 filt	0	-	-	-
725-731 unfilt	5	0.4	0.5	0.8

^aData are from filtered samples unless otherwise noted.

^bDue to nondetected measurements, these statistics may not be observable. If a range is given, the statistic is known to lie somewhere within the range.

^c725-731 refers to wells 725, 726, 727, 728, and 731. These wells were sampled only once, in 1993.

^dConcentrations from both filtered and unfiltered samples are presented for those constituents where filtering may be expected to remove a significant amount of the constituent from the water. However, when concentrations are low, natural variation and measurement error may be greater than the effects of filtering.

boron, cadmium, calcium, chloride, magnesium, manganese, nickel, nitrate, phosphate, potassium, selenium, sodium, strontium, sulfate, uranium, and zinc.

Time series plots of chemical constituents show slight trends, either increasing or decreasing, of solute concentrations in groundwater. In the most highly impacted area of the floodplain (the eastern portion), species such as nitrate and chloride appear to be increasing, sulfate and TDS are constant, and uranium shows a slight decrease. Observable concentrations of selenium and arsenic in groundwater are sporadic, most likely caused by analytical inconsistencies. Erratic behavior in the data probably indicates that numerous plumes are traveling through the alluvial system, affecting wells on an individual basis. This may be consistent with ponds and/or episodic discharges on different portions of the floodplain, which would have produced several plumes, and/or preferential flow paths through more permeable zones.

3.3 CONTAMINANTS OF CONCERN

The data presented in Sections 3.1 and 3.2 were used to compile a list of contaminants that are of potential concern for the assessment of human health or environmental risks at the Shiprock site. Since background water quality data are not available for the terrace, constituents that are highly elevated and clearly associated with the site are evaluated as contaminants of concern. This includes uranium, sulfate, and nitrate.

For the floodplain alluvium, a constituent was placed on the list of contaminants of potential concern if the constituent was detected at concentrations statistically above background monitor wells and if the site is a likely source of the contamination. Constituents exceeding background are shown in Table 3.4.

Several constituents that were above background were further screened because they are essential nutrients and the levels at which they are detected are within nutritional ranges. These constituents are calcium, chloride, phosphate, potassium, and zinc.

Ammonium, boron, and nickel were eliminated as contaminants of potential concern based on very low toxicity and a relatively high dietary intake. Antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, and uranium remain as contaminants of potential concern (Table 3.4).

3.4 CONTAMINANT FATE AND TRANSPORT

This section describes the fate and transport of contaminants in the floodplain.

Contaminant mobility, uptake, and toxicity depend on the species of elements that exist in an aqueous environment. The type and concentrations of species and complexes in natural waters depend on the concentration, availability of various anions and cations, and the pH and Eh conditions. Species of the contaminants of

Table 3.6 Statistical summary of the San Juan River water quality at the Shiprock UMTRA site, New Mexico^a, February 25, 1993

Constituent	Number of samples	Detection limit	Observed concentrations (mg/L)		
			Minimum	Median	Maximum
Arsenic ^b					
Upstream ^c	3	0.005	0.007	0.007	0.008
Downstream ^d	4	0.005	0.008	0.010	0.012
Cadmium ^e					
Upstream	3	0.001	-	-	-
Downstream	4	0.001	-	-	-
Calcium					
Upstream	3	0.5	73	73	77
Downstream	4	0.5	69	90	95
Chromium					
Upstream	3	0.01	0.03	0.03	0.03
Downstream	4	0.01	0.03	0.05	0.07
Iron					
Upstream	3	0.03	42	44	47
Downstream	4	0.03	40	54	69
Lead					
Upstream	3	0.003	0.024	0.025	0.026
Downstream	4	0.003	0.024	0.030	0.032
Manganese					
Upstream	3	0.01	1.03	1.12	1.23
Downstream	4	0.01	0.88	1.10	1.17
Molybdenum ^e					
Upstream	3	0.01	-	-	-
Downstream	4	0.01	-	-	-
Nitrate ^b					
Upstream	3	1.0	< 1.0	< 1.0	< 1.0
Downstream	4	1.0	1.1	1.5	1.7
Selenium ^e					
Upstream	3	0.05	-	-	-
Downstream	4	0.05	-	-	-
Sodium ^b					
Upstream	3	1.0	35	35	36
Downstream	4	1.0	36	56	58
Sulfate ^b					
Upstream	3	1.0	116	118	121
Downstream	4	1.0	121	190	205

Table 3.6 Statistical summary of the San Juan River water quality at the Shiprock UMTRA site, New Mexico^a, February 25, 1993 (Concluded)

Constituent	Number of samples	Detection limit	Observed concentrations (mg/L)		
			Minimum	Median	Maximum
Strontium					
Upstream	3	0.01	0.82	0.83	0.85
Downstream	4	0.01	0.80	1.12	1.18
Uranium					
Upstream	3	0.001	< 0.001	0.001	0.002
Downstream	4	0.001	0.001	0.001	0.001
Vanadium					
Upstream	3	0.01	0.09	0.09	0.09
Downstream	4	0.01	0.08	0.11	0.14
Radionuclides			(pCi/L)		
Lead-210					
Upstream	3		2.1	2.6	3.3
Downstream	4		1.0	1.8	2.6
Polonium-210					
Upstream	3		0.0	0.0	0.3
Downstream	4		0.0	0.3	1.1
Radium-226					
Upstream	3		1.3	2.5	2.9
Downstream	4		1.5	1.8	1.9
Thorium-230					
Upstream	3		0.1	0.2	0.3
Downstream	4		0.1	0.2	0.6

^aNonfiltered water samples.

^bStatistically elevated above background at the 0.05 level of significance.

^cUpstream locations: 546, 552, 554 (each location sampled one time).

^dDownstream locations: 548, 549, 550, 551 (each location sampled one time).

^eAnalyzed for, but not detected.

responsible for the flux of solute to the San Juan River. This takes place on the downstream half of the floodplain.

To estimate the impact of contaminated groundwater on water quality in the San Juan River, the contribution of contaminant flux from the alluvial aquifer to the San Juan River was calculated. The following instantaneous mixing equation was used to calculate the potential impact:

$$C_f = \frac{Q_{gw} C_{gw} + Q_r C_r}{Q_{gw} + Q_r}$$

where

C_f is the final concentration of a constituent in the final mixture,
 Q_{gw} is the flow of groundwater into the San Juan River (cubic feet per second),
 C_{gw} is the concentration of a constituent in groundwater (milligrams per liter),
 Q_r is the flow of the San Juan River (cubic feet per second), and
 C_r is the concentration of the same constituent in the San Juan River (milligrams per liter).

Initial concentrations taken from monitor well 736 (1993 water quality analysis) used in the calculation are uranium at 2.28 mg/L in groundwater and 0.0005 mg/L in the San Juan River; nitrate concentrations at 175 mg/L in groundwater and 0.05 mg/L in the San Juan River; and sulfate concentrations at 14,600 mg/L in groundwater and 187 mg/L in San Juan River water.

Using a groundwater flow of 0.2 ft³/s [0.006 cubic meters per second (m³/s)] and the lowest recorded 7-day flow of the San Juan River since the construction of the Navajo Dam [81 ft³/s (2.3 m³/s)], the following concentrations were calculated for the potentially impacted San Juan River: uranium at 0.006 mg/L, nitrate at 0.5 mg/L, and sulfate at 222 mg/L. These concentrations are within the range of ambient San Juan River water quality. From this calculation, it appears that the impact of contaminated groundwater from the floodplain on San Juan River water quality will be negligible.

4.0 EXPOSURE ASSESSMENT

This section discusses and quantifies the potential exposures that could be incurred by current or future residents, recreational users, and others who use groundwater or surface water contaminated by the Shiprock processing site. The methodology used here is consistent with the latest EPA guidance on exposure assessments (EPA, 1989a), which recommends an analysis based on the reasonable maximum exposure under both current and future land-use conditions. The reasonable maximum exposure is defined as the highest exposure that can be reasonably expected to occur at the site.

4.1 EXPOSURE PATHWAYS

Groundwater contaminated by uranium processing at the Shiprock site is not currently being used as a domestic or potable water supply. This includes the area that is contaminated in the San Juan River floodplain and contaminated groundwater in the terrace area, although the areal and vertical extent of the terrace contamination has not been defined. No points of groundwater withdrawal from the floodplain alluvium, alluvial terrace, or the Mancos Shale exist within the site vicinity.

However, the potential for exposure to surface expressions of groundwater exists on the San Juan River floodplain. A potential human secondary exposure pathway may be from consumption of meat or milk from livestock grazed and watered on the floodplain. Livestock may consume both the floodplain surface water contaminated by groundwater and pasture grasses that became contaminated through the uptake of contaminants from groundwater. Incidental access to the floodplain area by children or adults and subsequent direct contact with surface water bodies contaminated by groundwater may create another human exposure pathway.

There is a point of water withdrawal from the San Juan River near the site. This water intake is approximately 300 ft (90 m) upstream of the bridge supporting Route 666 over the river, on its north side. As discussed in Section 3.5, the water intake will not likely be affected by contaminants that may emanate from the site.

4.1.1 Groundwater exposure pathways for the floodplain

There is no current use of groundwater for household purposes in the site vicinity. Although it is unlikely that groundwater will be used in the future for household purposes because of the existing public water supply system, this risk assessment will assume hypothetical future use of groundwater resources, including the use of groundwater from the San Juan River floodplain for drinking water, cooking, and bathing. Other uses typical of the region that could indirectly lead to human exposure include irrigation and livestock watering with groundwater, as well as livestock grazed on the floodplain area, since grazing permits in the Shiprock region

include the portion of the floodplain located north and east of the former uranium processing site.

Figure 4.1 provides a conceptual model for potential floodplain groundwater exposure pathways.

4.1.2 Groundwater exposure pathways for the terrace

The only exposure pathway that is screened for the terrace groundwater is drinking water ingestion, since this is the pathway with the highest intake for nonvolatile compounds. Because data on the terrace background groundwater quality are lacking, the areal and vertical extent of the aquifer contamination is not known, and the yield of the aquifer seems to be very low, it is not known if groundwater is suitable for use as a drinking water resource. However, potential future beneficial use of the terrace groundwater is evaluated, since it cannot be ruled out at this time.

4.1.3 Surface water exposure pathways for the floodplain area

This exposure scenario is considered because of potential access to the floodplain area, especially by children playing in summer and by domestic animals that may graze on the floodplain. Three contaminants of concern detected in pools of surface water on the floodplain (refer to Section 7.0 for a detailed discussion), selenium, strontium, and uranium, were selected for a detailed evaluation of their potential to pose adverse health effects. Figure 4.2 provides a conceptual model for potential exposure pathways for the floodplain of the San Juan River below the Shiprock site.

4.2 POTENTIALLY EXPOSED POPULATION

The floodplain groundwater pathways evaluate domestic groundwater uses consistent with uses of a rural population of the region. The potentially exposed population includes Navajo infants (birth to 1 year old) and adults. These age groups were selected because toxicological responses are similar in these age groups, including recognition of sensitive subgroups for the contaminants of concern (infants), consistent intake to body weight ratios, and similar toxicokinetics.

For these same reasons, the terrace groundwater pathway of drinking water ingestion is also evaluated for infants and adults.

Direct exposure to contaminated surface water in the floodplain is evaluated for children since they are likely to play in the floodplain and would receive higher exposures than adults. Secondary exposures, such as meat or milk ingestion from livestock grazed and watered on the floodplain, could occur in both children and adults.

FIGURE 4.1
CONCEPTUAL MODEL FOR THE FLOODPLAIN ALLUVIUM GROUNDWATER USE PATHWAYS
SHIPROCK, NEW MEXICO, SITE

FIGURE 4.2
CONCEPTUAL MODEL FOR THE FLOODPLAIN ALLUVIUM SURFACE WATER USE PATHWAYS
SHIPROCK, NEW MEXICO, SITE

A sensitive subpopulation of Navajos are diabetics. The type II (adult onset) diabetes rate among the Navajo population is reported at 20 percent of the population over 20 years old and appears to be on an upward trend in regions of the Navajo Reservation (Hoy, 1993). Another sensitive subpopulation of residents living in the site's immediate vicinity might consist of residents with a history of occupational exposure primarily to uranium (uranium dust and radiation) at the former Shiprock uranium ore mill and/or during uranium mining, as revealed by a 1993 water use survey conducted among residents (TAC, 1993). Though this risk assessment does not quantitatively evaluate these subpopulations, it is recognized that diabetes could increase susceptibility to toxic exposures due to compromised renal function and increased drinking water ingestion rates, and previous occupational exposure could make that subpopulation more vulnerable to subsequent toxic exposures.

4.3 EXPOSURE CONCENTRATIONS

The exposure concentration of a contaminant in groundwater is that concentration contacted by an individual over the period of exposure being considered. In this evaluation, the contaminant concentrations are assumed to be in a steady state, even though actual exposure concentrations are expected to decrease with time because surface remediation has been completed at the site. Nonetheless, they are reasonable estimates for chronic exposure soon after remediation (chronic exposure for noncarcinogens is considered to be a period of at least 7 years).

Exposure point concentrations used for the floodplain groundwater-use scenario are the 95 percent upper confidence interval for the median contaminant concentrations, based on groundwater data collected from monitor well 615. This well has consistently shown the highest levels of contamination through the past 8 years of monitoring. Two exceptions to this rule occurred. First, ammonium levels were consistently higher in well 608 than in 615; therefore, ammonium median concentration was determined based on groundwater data collected from monitor well 608. Second, levels of antimony, arsenic, cadmium, lead-210, polonium-210, radium-226, and thorium-230 measured in seven monitor wells in the immediate vicinity of the disposal cell (608, 609, 610, 611, 613, 614, and 615) were combined to characterize these constituents' concentrations in the plume because of their low detectability. Exposure point concentrations for the floodplain alluvium are shown in Table 4.1.

Exposure point concentrations for nitrate and sulfate in the terrace groundwater are represented by the groundwater data from monitor well 727, which shows the highest levels of both contaminants (Table 4.2). Uranium concentration is represented by the maximum observed concentration in groundwater from monitor well 600.

Table 4.1 Exposure concentrations for the floodplain groundwater

Constituent	Upper 95 percent confidence interval estimate for the median (mg/L)
Antimony	0.037
Arsenic	0.03
Cadmium	0.005
Magnesium	2,540
Manganese	9.38
Nitrate	4,220
Selenium	0.26
Sodium	3,630
Strontium	12.2
Sulfate	15,000
Uranium	3.0

Table 4.2 Exposure concentrations for the terrace groundwater

Constituent	Observed concentration (mg/L)	Comments
Nitrate	2,310	Concentration in downgradient monitor well 727, measured in 1993.
Sulfate	18,100	Concentration in downgradient monitor well 727, measured in 1993.
Uranium	1.57	Maximum measured concentration in downgradient monitor well 600.

For the human exposure pathway associated with ingestion of meat and milk from livestock grazed and watered on the San Juan River floodplain, observed maximum values for surface water and sediment data were used. The median value of the contaminant concentrations in groundwater was used to estimate pasture grass root uptake (Table 4.3).

4.4 ESTIMATION OF INTAKE

Individuals within the population of future residents are anticipated to vary with respect to water consumption habits, stable body weight, and length of time they may reside in the potential contamination zone. Consequently, health risks associated with groundwater consumption will also vary among members of the population. Nevertheless, to describe the potential risks to the future population, daily water intake, body weight, and residency time were incorporated into this

Table 4.3 Exposure concentrations for the floodplain surface water and sediments

Constituent	Surface water concentration C _w (mg/L)	Groundwater concentration (root uptake) C _w (mg/L)	Sediments concentration (mg/kg)
Selenium	0.035	0.12	4.2
Strontium	7.34	9.79	1620
Uranium	1.1	1.30	44

assessment using the default values according to the standard EPA procedure (EPA, 1989a).

The exposure routes and potentially exposed populations that are quantitatively evaluated are summarized in Table 4.4. In addition to exposure routes and potentially exposed population, intakes are calculated separately for noncancer health effects (noncarcinogens) and cancer (carcinogens). Contaminants such as arsenic and uranium are associated with both noncancer effects and cancer, so they appear under both carcinogen and noncarcinogen estimations. Carcinogenic intakes for radionuclides are cumulative and are estimated for a lifetime exposure, whereas noncarcinogens intakes are estimated as a dose per day.

Table 4.4 Quantitatively evaluated exposure routes

Source	Exposure route	Potentially exposed population		
		Infant	Child	Adult
Floodplain groundwater	Drinking water ingestion	X		X
	Dermal contact (bathing)			X
	Groundwater-irrigated produce ingestion			X
Terrace groundwater	Drinking water ingestion	X		X
Floodplain surface water	Incidental water ingestion		X	
	Incidental sediment ingestion		X	
	Dermal contact (wading)		X	
	Meat ingestion			X
	Milk ingestion			X

4.4.1 Floodplain groundwater exposure routes

Drinking water ingestion

Drinking water ingestion is generally the most significant exposure route for groundwater contaminated with metals and other nonvolatile compounds. For this evaluation, drinking water consumption includes amounts of water consumed for drinking as well as amounts of water used for food preparation (e.g., reconstituted juices, soup, rice, beans). The assumptions and equations used to estimate intakes from drinking water ingestion are shown in Table 4.5.

Under the land tenure system of the Navajos, when a family has a grazing permit, the family usually resides on its land for its lifetime. Since there are no residence data for the Navajos in the vicinity of Shiprock, a chronic (i.e., at least 7-year) exposure duration is assumed for noncarcinogens and 70 years for carcinogens.

Dermal absorption

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

To evaluate this exposure route, a screening calculation was performed to determine whether the contribution from dermal absorption would be significant compared to the drinking water pathway for the contaminants of concern. Since chemical-specific absorption factors are not available for these contaminants, it was assumed that they are absorbed across the skin at the same rate as water. This assumption is likely to overestimate any potential contribution from dermal absorption.

The results of the screening are given in Table 4.5. Based on these results, the dermal absorption exposure route is eliminated from more detailed evaluation, since it contributed to less than 1 percent of the total dose from drinking water.

Ingestion of groundwater-irrigated produce

This exposure route was also evaluated for relative significance to the drinking water ingestion route. The results of the screening calculation are shown in Table 4.6. The assumptions for this evaluation are likely to overestimate the potential for exposure from this route, since it is assumed that all garden produce would be consumed from a garden. The results of this screening show that for all contaminants of concern except strontium, exposure through ingestion of garden vegetables and fruit irrigated with contaminated groundwater can be eliminated from more detailed evaluation, since this exposure route contributed to less than 3 percent of the total dose from drinking water. However, potential strontium

Table 4.5 Exposure dose calculations and equation definitions for groundwater ingestion and dermal contact, Shiprock, New Mexico, site

Equation definitions for exposure dose calculations	
Ingestion of groundwater	
Carcinogens and noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{C_w \times IR_w \times EF \times ED}{BW \times AT}$
Radionuclides (pCi)	$= C_w \times IR_w \times EF \times ED$
Dermal contact with groundwater	
Carcinogens and noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{(C_w \times SA \times Pc \times Cf) \times ET \times EF \times ED}{BW \times AT}$
Radionuclides (pCi)	$= C_w \times SA \times Pc \times Cf \times ET \times EF \times ED$
Where:	
Cw	Contaminant concentration in groundwater (upper 95 percent confidence interval estimate for the median concentration from wells 608, 609, 610, 611, 613, 614, or 615) (mg/L or pCi/L).
IRw	Ingestion rate for water (L/day) (2 L/day for an adult, 0.64 L/day for an infant).
EF	Exposure frequency (350 days/year).
ED	Exposure duration (7 years for an adult and 1 year for an infant for noncarcinogens; 70 years for carcinogens).
BW	Body weight (70 kg for an adult, 4 kg for an infant).
AT	Averaging time (350 days x ED for noncarcinogens, 350 days x 70 years for carcinogens).
SA	Skin surface area (19,400 cm ²).
Pc	Dermal permeability constant (0.001 cm/hour).
Cf	Conversion factor (0.001 L/cm ³).
ET	Exposure time (0.2 hour/day).
kg - kilogram.	
cm ² - square centimeter.	
cm ³ - cubic centimeter.	
mg/kg/day - milligrams per kilogram per day.	

Table 4.5 Exposure dose calculations and equation definitions for the floodplain groundwater ingestion and dermal contact, Shiprock, New Mexico, site (Concluded)

Contaminant of concern	Cw (mg/L)	Floodplain groundwater exposure doses (mg/kg/day)		Dermal Ingestion
		Ingestion	Dermal contact	
Noncancer effects				
Antimony	0.037	1E-03	2E-06	0.002
Arsenic	0.03	9E-04	2E-06	0.002
Cadmium	0.005	1E-04	3E-07	0.002
Magnesium	2540	7E + 02	1E-01	0.002
Manganese	9.38	3E-01	5E-04	0.002
Nitrate	4220	1E + 02 7E + 02 ^a	2E-01	0.002
Selenium	0.26	7E-03	1E-05	0.002
Sodium	3630	1E + 02	2E-01	0.002
Strontium	12.2	4E-01	7E-04	0.002
Sulfate	15000	4E + 02 2E + 03 ^a	8E-01	0.002
Uranium	3.0	9E-02	2E-04	0.002
Cancer effects				
Arsenic	0.03	6E-04	2E-06	0.002
Uranium ^b	2058 ^c	1E + 08 ^d	2E + 05 ^d	0.002

^aExposure dose calculated for an infant (aged 0 to 1 year).

^bUranium-234 and uranium-238 combined.

^cUnits are in picocuries per liter; 1 mg uranium is assumed to equal 686 pCi.

^dUnits are in picocuries per lifetime.

Table 4.6 Exposure dose calculations and equation definitions from groundwater-irrigated garden produce ingestion for future hypothetical adult scenarios, the San Juan floodplain, Shiprock, New Mexico, site

Equation definitions for exposure dose calculations	
Ingestion of garden produce irrigated with groundwater	
Carcinogens and noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{C_w \times K_d \times B_v \text{ or } B_r^a \times DF \times IR_p \times FI \times EF \times ED}{BW \times AT}$
Radionuclides (pCi)	$= C_w \times K_d \times B_v \text{ or } B_r^a \times DF \times IR_p \times FI \times EF \times ED$

Where:

- Cw Contaminant concentration in groundwater (upper 95 percent confidence level of the median of data from wells 608, 609, 610, 611, 613, 614, or 615) (mg/L or pCi/L).
- Kd Soil-water partition coefficient (L/kg).
- Bv Soil-to-plant concentration ratio for vegetative portions of plants (unitless).
- Br Soil-to-plant concentration ratio for reproductive portions (fruits, tubers) of plants (unitless).
- DF Dry weight fraction of plant (0.066; unitless).
- IRp Ingestion rate for garden produce (0.05 kg/day for vegetative parts; 0.03 kg/day for reproductive parts).
- FI Fraction of garden produce ingested from contaminated source (1.0; unitless).
- EF Exposure frequency (350 days/year).
- ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens).
- BW Body weight (70 kg for an adult).
- AT Averaging time (350 days x ED for noncarcinogens; 350 x 70 years for carcinogens).

^aExposure doses due to vegetative portions and reproductive portions of garden produce are calculated separately.

Table 4.6 Exposure dose calculations and equation definitions for the San Juan River floodplain groundwater-irrigated garden produce ingestion for future hypothetical adult scenarios, Shiprock, New Mexico, site (Concluded)

Contaminant of concern	Cw (mg/L)	Kd (L/kg)	Bv	Br	Garden produce ingestion exposure doses (mg/kg/day)		Vegetative + reproductive
					Vegetative parts	Reproductive parts	Ingestion
Noncancer effects							
Antimony	0.037	45	0.20	0.03	2E-05	1E-06	0.02
Arsenic	0.03	200	0.04	0.006	1E-05	1E-06	0.01
Cadmium	<0.005	6.5	0.55	0.15	8E-07	1E-07	0.007
Magnesium	2540	4.5	1.0	0.55	5E-01	2E-01	0.01
Manganese	9.38	65	0.25	0.05	7E-03	9E-04	0.03
Nitrate	4220	0.01	30	30	6E-02	4E-02	0.0008
Selenium	0.26	300	0.025	0.025	9E-05	6E-05	0.02
Sodium	3630	100	0.075	0.055	1E + 00	6E-01	0.02
Strontium	12.2	35	2.5	0.25	5E-02	3E-03	0.15
Sulfate	15,000	7.5	0.5	0.5	3E + 00	2E + 00	0.01
Uranium	3.0	450	0.0085	0.004	5E-04	2E-04	0.008
Cancer effects							
Arsenic	0.03	200	0.04	0.006	8E-06	7E-07	0.01
Uranium	2058 ^a	450	0.0085	0.004	6E + 05 ^b	2E + 05 ^b	0.01

^aPicocuries per liter.

^bPicocuries per lifetime.

exposure at these levels would not be expected to cause adverse health effects in humans.

Ingestion of meat or milk from groundwater-fed livestock

These pathways were eliminated from further consideration, because the concentrations of nitrate and sulfate are so high that livestock could not survive chronic ingestion of the water. If the livestock cannot consume the water, there is no potential for bioaccumulation or transfer of contaminants ingested to meat tissue or milk. Further evaluation of the direct toxicity to livestock is presented in Section 7.0. This pathway is evaluated in Section 4.4.3 for floodplain surface water, where nitrate concentrations are not so high.

4.4.2 Terrace groundwater exposure routes

Drinking water ingestion is generally the most significant exposure route for groundwater contaminated with metals and other nonvolatile chemical compounds. For this evaluation, drinking water consumption includes amounts of water consumed for drinking and amounts of water used for food preparation. Due to the lack of background groundwater quality information, contaminants of concern for the terrace groundwater could not be determined. However, three constituents clearly associated with uranium processing that are highly elevated compared to regional waters were selected for the pathway screening: nitrate, sulfate, and uranium. As can be seen in Tables 4.5 and 4.6, the relative pathway contribution for these contaminants is greater than 99 percent from drinking water ingestion. The exposure assessment of drinking water intake of terrace groundwater is shown in Table 4.7. This exposure pathway is further evaluated in Section 6.0.

4.4.3 Floodplain surface water exposure routes

The pathways that may occur on the floodplain include incidental ingestion of surface water and sediments while wading, dermal contact with surface water, and consumption of contaminated meat and milk from livestock grazed and watered on the floodplain.

For this evaluation, it is assumed that incidental ingestion of sediments and water from the surface water bodies identified on the floodplain area, as well as dermal contact with this water, would most likely occur during summer. Children aged 6 to 12 years are considered the subpopulation most likely to play on the floodplain.

Incidental ingestion of surface water

Incidental ingestion of surface water could occur while children play on the floodplain area. During this play, it is assumed that children may incidentally ingest

Table 4.7 Exposure dose calculations for hypothetical adult scenario of groundwater ingestion from the terrace, Shiprock, New Mexico, site^a

Constituent	Exposure point concentration (mg/L)	Well location	Estimated intake (mg/kg/day)
Noncarcinogens			
Nitrate	2,310	727	7E + 01 4E + 02 ^b
Sulfate	18,100	727	5E + 02 3E + 03 ^b
Uranium	1.57	600	0.05
Carcinogens			
Uranium ^c	1,077 ^d	600	5E + 07 ^e

^aEquation definitions used to estimate human intake of the constituent through ingestion of the terrace groundwater are the same as those presented in Table 4.5.

^bExposure dose calculated for an infant (aged 0 to 1 year).

^cUranium-234 and uranium-238 combined; 1 mg uranium is assumed to equal 686 pCi.

^dPicocuries per liter.

^ePicocuries per lifetime.

a few tablespoons of water. It is further assumed that children would visit the floodplain every day during the summer as well as visiting it on weekends throughout 3 other months of the year. The results for this exposure route are presented in Table 4.8.

Dermal contact with surface water

For the same scenario presented above, it is assumed that children's arms and legs will come into contact with surface water that has pooled in the floodplain. This leads to a potential exposure route from dermal absorption, although metals are generally not well absorbed across the skin. Because chemical-specific absorption factors are not available for these contaminants, it was assumed that they are absorbed across the skin at the same rate as water. This assumption is highly likely to overestimate any potential contribution from dermal absorption. The results for this pathway are presented in Table 4.8. This table shows that contributions from dermal absorption are approximately 1 percent of the incidental water ingestion pathway for the selected contaminants of concern.

Incidental ingestion of sediments

The third exposure route evaluated for children playing in the floodplain is incidental ingestion of sediments. The equation and assumptions for the exposure route are shown on Table 4.9. When this exposure route is compared to incidental ingestion of surface water, the relative contribution is 24 percent for selenium, 43 percent for strontium, and 8 percent for uranium. Therefore, incidental ingestion of surface water and sediments is further evaluated in Section 6.0 for potential exposures for children playing on the floodplain.

Ingestion of meat and milk from livestock grazed and watered on the floodplain

These pathways are considered because a grazing permit has been issued for the floodplain below the site. Moreover, many residents living near the site keep livestock that may consume surface water and pasture grasses from the floodplain.

For the purpose of this exposure pathway evaluation, concentrations of selected contaminants of concern in pasture grasses were determined assuming root intrusion into shallow groundwater and pasture grass uptake of these constituents. The method used to estimate plant tissue concentrations of the contaminants of concern is described in detail in Section 7.0. The median concentrations of groundwater from all wells installed within the entire floodplain are used for this evaluation. Site- and chemical-specific soil-water partition coefficient (K_d) values are calculated for each sampling location as the ratio of measured contaminant concentration in the sediments to the measured contaminant concentration in the

Table 4.8 Exposure dose calculations and equation definitions for hypothetical incidental surface water ingestion and dermal contact by children on the floodplain, Shiprock, New Mexico, site

Equation definitions for exposure dose calculations	
Incidental ingestion of surface water from the floodplain	
Noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{C_w \times IR_w \times EF \times ED}{BW \times AT}$
Incidental dermal contact with surface water	
Noncarcinogens:	
Chronic daily absorbed dose (mg/kg/day)	$= \frac{C_w \times S_a \times P_c \times C_f \times ET \times EF \times ED}{BW \times AT}$
Where:	
C _w	Contaminant concentration in surface water bodies (the highest determined concentration at the floodplain specified location) (mg/L).
IR _w	Ingestion rate for water (L/day) (0.05 L/day for children aged 6 to 12 years).
EF	Exposure frequency (days/year) (3 months per year during 7 days per week = 90 days, and 3 months per year during weekends = 24 days; total incidental exposure frequency equals 114 days per year).
ED	Exposure duration (7 years for children aged 6 to 12 years playing on the floodplain area).
BW	Body weight (38.3 kg for children aged 6 to 12 years; 90th percentile body weight for 9- to 10-year old male child).
AT	Averaging time (365 days x ED).
S _a	Skin surface area available to dermal exposure (497 cm ² for children aged 6 to 12 years; calculated using 50th percentile: hands - 57 cm ² ; arms - 130 cm ² ; legs - 310 cm ²) (EPA, 1989a).
P _c	Dermal permeability constant (0.001 cm/hr).
C _f	Conversion factor (0.001 L/cm ³).
ET	Exposure time (1 hour per day).

Table 4.8 Exposure dose calculations and equation definitions for hypothetical incidental surface water ingestion and dermal contact by children on the floodplain, Shiprock, New Mexico, site (Concluded)

Contaminant of concern	Cw (mg/L)	Surface water bodies location ID ^a	Exposure doses (mg/kg/day)		Dermal Ingestion
			Ingestion	Dermal contact	
Selenium	0.035	659	1E-05	1E-07	0.01
Strontium	7.34	659	3E-03	3E-05	0.01
Uranium	1.1	660	5E-04	5E-06	0.01

^aRefer to Section 7.0 for a description of surface water body locations.

Table 4.9 Exposure dose calculations and equation definitions for hypothetical incidental ingestion of sediments by children on the floodplain at the Shiprock, New Mexico, site

Equation definitions for exposure dose calculations

Incidental ingestion of sediments in the floodplain area

Noncarcinogens:

$$\text{Chronic daily intake (mg/kg/day)} = \frac{Cs \times Csf \times IRs \times FI \times EF \times ED}{BW \times AT}$$

Where:

Cs Contaminant concentration in sediments (the highest determined concentration in the floodplain soil at specified location) (mg/kg).

Csf Conversion factor (10^{-6} mg/kg) (EPA, 1989a).

IRs Ingestion rate for sediments (100 mg/day) (EPA, 1989a).

FI Fraction ingested from sediments (1.0; unitless).

EF Exposure frequency (days/year) (3 months per year during 7 days per week = 90 days, and 3 months per year during weekends = 24 days; total incidental exposure frequency equals 114 days per year).

ED Exposure duration (7 years for children aged 6 to 12 years playing in the floodplain area).

BW Body weight (38.3 kg for children aged 6 to 12 years; 90th percentile body weight for 6- to 12-year-old male child).

AT Averaging time (365 days x ED).

Contaminant of concern	Cs (mg/L)	Sampling location ID ^a	Contaminant intake from ingested sediments (mg/kg/day)	Sediment ingestion
				Water ingestion
Selenium	4.2	659	3E-06	0.2
Strontium	1620	659	1E-03	0.4
Uranium	44	659	4E-05	0.08

^aRefer to Section 7.0 for a description of sediment sampling locations.

mg/kg - milligrams per kilogram.

floodplain surface water bodies. These results are summarized in Tables 4.10 and 4.11.

It is assumed that livestock grazing on the floodplain will drink water and will ingest sediments contaminated at the levels measured at identified surface locations. The results of the potential human intakes of selected contaminants of concern due to ingestion of meat and milk from livestock grazed and watered on the floodplain are summarized in Table 4.12. This human exposure pathway is further evaluated in Section 6.0.

Table 4.10 Kd values for selected contaminants of concern based on sediment and surface water concentrations measured on the floodplain, Shiprock, New Mexico, site

Contaminant of concern	Wet area below					Average of chemical-specific Kd values (L/kg)
	Drainage canal location ID ^a		Stagnant pools location ID ^a		Seep 425 location ID ^a	
	655	656	660	661	659	
Selenium	1000	--	--	--	120	560
Strontium	81.8	117.8	97.9	32.2	220.7	110
Uranium	137.5	87.9	20	17.9	63.8	66

^aRefer to Section 7.0 for a detailed description of sampling locations.

Table 4.11 Estimated concentrations of selected contaminants of concern in pasture grasses on the San Juan River floodplain, Shiprock, New Mexico, site

Contaminant of concern	Cw (mg/L) ^a	Location ID	Kd (L/kg) ^b	Estimated soil concentration (Cs) ^c (mg/kg DW)	Bv	Estimated concentration in vegetative portions of plants (Cp) ^d (mg/kg DW)
Selenium	0.12	607-630	560	67	0.025	1.7
Strontium	9.79	607-630	110	1080	2.5	2690
Uranium	1.30	607-630	66	86	0.0085	0.73

^aCw values represent the median of groundwater concentrations from all wells in the San Juan River floodplain for 1986-1993.

^bKd values are based on calculated mean values for sediment/surface water concentrations measured at the locations shown in Table 4.10.

^cConcentration in soil (Cs) = Cw x Kd.

^dConcentration in plants (Cp) = Cs x Bv.

DW - dry weight.

Table 4.12 Exposure dose calculations and equation definitions for future hypothetical adult meat and milk consumption from cattle grazed and watered on the San Juan River floodplain, Shiprock, New Mexico, site

Equation definitions for exposure dose calculations	
Ingestion of contaminated meat from cattle grazed on pasture grasses and fed with the floodplain surface water bodies	
Noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{C_b \times IR_b \times FI \times EF \times ED}{BW \times AT}$
	$C_b = F_b[(C_p \times Q_p) + (Q_s \times C_s) + Q_w \times C_w]$
Radionuclides:	
Chronic daily intake (pCi/year)	$= C_b \times IR_b \times EF \times FI \times ED$
Ingestion of contaminated milk from cattle grazed on pasture grasses and fed with the floodplain surface water bodies	
Noncarcinogens:	
Chronic daily intake (mg/kg/day)	$= \frac{C_m \times IR_m \times FI \times EF \times ED}{BW \times AT}$
	$C_m = F_m[(C_p \times Q_p) + (Q_s \times C_s) + (Q_w \times C_w)]$
Radionuclides:	
Chronic daily intake (pCi/year)	$= C_m \times IR_m \times EF \times FI \times ED$
Where:	
<p> C_b Contaminant concentration in beef (calculated value; chemical-specific; mg/kg). IR_b Ingestion rate for beef (0.075 kg/day) (EPA, 1989). FI Fraction of diet (meat/milk) ingested (0.75, unitless; subsistence farm family). EF Exposure frequency (350 days/year). ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens). BW Body weight (70 kg for an adult). AT Averaging time (365 days x ED). F_b Forage-to-beef transfer coefficient (chemical-specific; unitless). C_p Contaminant concentration in pasture grasses (calculated value; chemical-specific; mg/kg). Q_p Quantity of pasture ingested daily by cattle (19 kg DW/day). Q_s Quantity of soil ingested daily by cattle (0.38 kg based on 2% of dry matter from feed ingestion rate). C_s Contaminant concentration in soil (the highest determined concentration in the floodplain specified location; mg/L). Q_w Quantity of water ingested daily by cattle (56 L/day). C_w Contaminant concentration in surface water bodies (the highest determined concentration at the floodplain specified location; mg/L). C_m Contaminant concentration in milk (calculated value; chemical-specific; mg/kg). IR_m Ingestion rate for milk (0.30 kg/day) (EPA, 1989). F_m Feed-to-milk transfer coefficient (chemical-specific; unitless). </p>	

Table 4.12 Exposure dose calculations and equation definitions for future hypothetical adult meat and milk consumption from cattle grazed in the San Juan River floodplain, Shiprock, New Mexico, site (Concluded)

Contaminant of concern	Contaminant concentration ^a		Transfer coefficients		Exposure dose (mg/kg/day)		Total intake (mg/kg/day)
	Surface water Cw (mg/L)	Sediment Cs (mg/kg)	Fb	Fm	Beef	Milk	
Noncarcinogens							
Selenium	0.035	4.2	0.015	0.004	4E-04	4E-4	8E-04
Strontium	7.34	1620	0.0003	0.0015	1E-02	2E-01	2E-01
Uranium	1.1	44	0.0002	0.0006	1E-05	2E-04	2E-4
Carcinogens							
Uranium ^b	755 ^c	501 ^d	0.0002	0.0006	20 ^e	280 ^e	300 ^e

^aAll values except uranium in surface water are from location 659. Uranium in surface water is from location 660. Refer to Section 7.0 for a detailed description of surface water and sediments sampling locations.

^bUranium-234 and uranium-238 combined.

^cPicocuries per liter.

^dPicocuries per kilogram.

^ePicocuries per lifetime.

4.5 EXPOSURE ASSESSMENT UNCERTAINTIES

A number of potential sources of error may arise in all phases of the exposure assessment. These uncertainties could lead to an underestimate or overestimate in the exposure analysis. Some of the more significant sources of uncertainty include the following:

- Uncertainties resulting from the lack of thorough environmental sampling data.
- Uncertainties arising from the assumption that the groundwater 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 at least 7 years).
- Uncertainties associated with the model used to estimate uptake of contaminants into plants and pasture grasses present in the floodplain area, as well as contaminant intake by grazing livestock. The plant uptake factors could vary substantially from the default estimates under the site conditions at Shiprock.
- Uncertainties associated with the relationship between an applied dose (used in this assessment) and absorbed dose or effective toxic dose for dermal absorption.
- Uncertainties associated with differing sensitivities of subpopulations such as diabetics or occupationally exposed subpopulations.

Regardless of these uncertainties, the intake estimates derived in this section likely represent reasonable, maximum exposures if groundwater or its surface expressions are used at the site.

5.0 TOXICITY ASSESSMENT

Several contaminants that could adversely affect human health and the environment have been detected in groundwater at the site. This section summarizes the toxicological effects of the chemical contaminants and the carcinogenic potentials of arsenic and the radionuclides.

5.1 CONTAMINANT TOXICITY SUMMARIES

The following summaries address the basic toxicokinetics and toxicity of the 11 contaminants of concern at Shiprock based on the preliminary screening discussed previously. The contaminants of concern are antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, and uranium. Wherever possible, data from human studies will be used. Only in cases where human data are unavailable will animal studies be reported. Although these contaminants have a wide range of toxic effects depending on the exposure levels, the following discussions will focus most heavily on toxic effects observed in the exposure range most relevant to contamination at the Shiprock site.

5.1.1 Antimony

Absorption

Absorption of antimony can occur both via inhalation and ingestion. At least 15 percent of a single oral dose of labeled antimony potassium tartrate given to mice is absorbed, i.e., recovered in urine and tissues. Absorption may, however, be higher since gastrointestinal excretion starts immediately after the metal is taken up from the gut (Friberg et al., 1986).

Tissue accumulation and clearance

Surface body scanning of people given intravenous injections of labeled antimony as sodium antimony dimercaptosuccinate revealed the highest amounts accumulating in the liver, thyroid, and heart. Forty-three days after the last injection, the liver still showed values of about one-sixth of the maximum, which was reached one day after the last injection.

Single intravenous or intramuscular injections to human volunteers produced higher 24-hour urinary excretion of pentavalent (80 percent) than of trivalent (25 percent) antimony compounds, a pattern similar to the one found in animals.

A study on the elimination of a single intravenously administered dose of labeled antimony potassium tartrate (trivalent) indicated that the urinary excretion was about four times higher than fecal excretion. The initial excretion of antimony in humans is rapid. When pentavalent antimony compounds are given intravenously

or intramuscularly, more than 90 percent of the dose will be found in the urine within 24 hours.

Environmental sources of antimony

Dietary intake of antimony ranged from 0.25 to 1.25 mg/day in a study of institutional diets for children in the United States (Friberg et al., 1986). Assuming an average body weight of 35 kg, this is equivalent to 0.007 to 0.04 mg/kg/day.

In freshwater fish, antimony concentrations have been reported to be on the order of 3 micrograms per kilogram ($\mu\text{g/kg}$) wet weight. Levels of 3 and 8 $\mu\text{g/kg}$ have been found in milk and potato powder. Antimony is sometimes present in the binding coat between enamel and metal, especially in older cooking utensils, and can be dissolved by acid food and drinks when the enamel coating is worn. In soil, antimony usually ranges from 0.1 to 10 mg/kg dry weight.

Toxicity of antimony

The EPA oral reference dose (RfD) for antimony is 0.0004 mg/kg/day. The RfD was based on the lifetime study in rats exposed to antimony trioxide in drinking water. The lowest dose producing adverse health effects (increased mortality of animals and alteration of blood chemistry) was 0.42 mg/kg/day.

After drinking lemonade containing 0.013 percent antimony, 70 people developed acute symptoms, including burning stomach pain, colic, nausea, and vomiting. It was estimated that a person consuming 300 milliliters (mL) of lemonade would have received a dose of approximately 0.5 mg/kg for a 70-kg adult.

One study indicated that female workers exposed in an antimony plant experienced a greater incidence of spontaneous abortions than did a control group of nonexposed working women. A high rate of premature deliveries among women who worked in antimony smelting and processing was also observed. The health effects from exposure to antimony as a function of dose are summarized in Figure 5.1.

5.1.2 Arsenic

Absorption

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

FIGURE 5.1
ANTIMONY TOXICITY RANGES

Tissue accumulation and clearance

After absorption by the gastrointestinal tract, arsenic is transported via the blood to most tissues. In man 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 the hair, skin, squamous epithelium of the upper gastrointestinal tract, epididymis, thyroid, lens, and skeleton (Lindgren et al., 1982). These organs are rich in proteins containing sulfhydryl groups. The chemical form of arsenic influences its retention time and target tissue site. Higher retention of arsenic in different organs occurs after exposure to trivalent arsenic than to pentavalent form. Trivalent arsenicals can combine chemically with sulfhydryl groups, which may explain its accumulation in keratin-rich tissues such as hair, skin, and the epithelium of the upper gastrointestinal tract (Webb, 1966; Casarett and Doull, 1991).

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

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

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

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

Environmental sources of arsenic

Arsenic is ubiquitous in nature in both inorganic and organic compounds. Water is the major means of transport of arsenic under natural conditions. In oxygenated

water, arsenic occurs in a pentavalent form; but under reducing conditions (e.g., in artesian well waters), the trivalent form predominates. Sedimentation of arsenic in association with iron and aluminum represents a considerable factor in environmental transport and deposition of this element (Marcus and Rispin, 1988).

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

Drinking water usually contains a few micrograms of arsenic, predominantly as inorganic salts in the trivalent and pentavalent states (WHO, 1981). However, concentrations of up to 1100 micrograms per liter ($\mu\text{g/L}$) in drinking water have been reported in Chile, Argentina, Taiwan, the United States, and the United Kingdom (WHO, 1981).

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

Toxicity of arsenic

Levels of exposure associated with acute arsenic toxicity vary with the valency form of the element. Trivalent arsenicals (arsenites) are generally more toxic than pentavalent (arsenates) (Morrison et al., 1989), and inorganic arsenic compounds are more toxic than organic (Shannon and Strayer, 1989). Based on geochemical models for the Shiprock site, arsenic exists primarily in the pentavalent form in groundwater (Table 3.4). For arsenic trioxide, the reported estimated lethal dose in humans ranges from 70 to 300 mg (EPA, 1984). Acute exposure to inorganic arsenic compounds may lead to a severe inflammation of the gastrointestinal tract after ingestion.

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

Chronic arsenic intoxications result from exposure to small doses of arsenic over a long period of time. These intoxications are frequently caused by arsenic content in drinking water and in food. In humans, chronic exposure to arsenic compounds may lead to skin lesions such as hyperpigmentations, hyperkeratoses, portal hypertension, and disturbances of the peripheral vascular and nervous systems (Finkel, 1983). 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).

Hyperpigmentation, hyperkeratoses, and skin cancer with prevalence of 7.1 percent, 18.4 percent, and 1.1 percent, respectively was reported in Taiwanese studies of more than 40,000 people exposed to arsenic in drinking water at concentrations ranging from 1.4 to 6.3 mg/L.

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, 1988a), based on the occurrence of increased lung cancer mortality in populations exposed primarily via inhalation and of increased skin cancer prevalence in populations exposed through consumption of drinking water containing high concentrations of arsenic. Currently, the EPA is reviewing recently published data concerning internal cancer risks from exposure to arsenic. The health effects from exposure to arsenic as a function of dose are summarized in Figure 5.2.

5.1.3 Cadmium

Absorption

In humans, approximately 5 percent of ingested cadmium in water is absorbed, but this figure can increase substantially following exposure to other metals (such as calcium or iron) or with increased protein intake (Friberg et al., 1986). The amount of cadmium absorbed from food sources is about half the amount absorbed from water. Absorption is also substantially increased in individuals with low iron stores (Flanagan et al., 1978). Once absorbed, cadmium is bound to protein, primarily metallothionein. The ability of many metals to increase the concentration of metallothionein is likely the basis for interactions in absorption, tissue concentrations, and toxicity of combined exposures to metals.

Tissue accumulation and clearance

Humans with low-level exposure to cadmium show approximately 50 percent of the body burden in the kidneys, 15 percent in the liver, and 20 percent in muscle (Kjellström, 1979). The kidney concentration will increase with continued exposure only to about age 50, but the concentration in muscle will increase throughout life. In cases of high exposure resulting in kidney damage, kidney concentrations can be quite low, but liver concentrations can be up to 100 times higher than normal. Only 0.01 to 0.02 percent of the total body burden of cadmium is excreted daily, resulting in continuously increasing body burdens with continuous exposure. The biological half-time of cadmium, or the time necessary to eliminate 50 percent of

FIGURE 5.2
ARSENIC TOXICITY RANGES

the cadmium in the body at a given time, is 10 to 30 years in humans (Nordberg et al., 1985).

Environmental sources of cadmium

Normal cadmium content of food and water in nonpolluted areas results in 0.01 to 0.06 mg/day intake of cadmium (0.0001 to 0.0009 mg/kg/day). Cadmium occurs naturally with zinc and lead; it is therefore often present as an impurity in products using these metals, such as solders and galvanized metals. These sources lead to contact with water supplies in water heaters and coolers, some pipes, and taps.

Toxicity of cadmium

Acute exposure to high concentrations of cadmium (15 mg/L in water) results in acute gastrointestinal effects, including abdominal cramps, diarrhea, and vomiting (0.07 mg/kg). These gastrointestinal effects have not been reported in any chronic environmental exposure.

The primary toxic effect of long-term exposure to cadmium is disturbance of reabsorption in the proximal tubules of the kidney. This effect is first observed by an increase of low molecular-weight proteins in the urine. This initial effect is observed following a daily intake of 0.0075 mg/kg/day. Progressive disruption of kidney function will lead to an increase in amino acids, glucose, phosphate, and protein in urine. Long-term exposures can also disturb calcium metabolism, leading to osteoporosis and osteomalacia. A combination of these two effects is referred to as *Itai-itai* disease and was seen in epidemic proportions in a cadmium-contaminated region in Japan in the 1950s (Friberg et al., 1986). These health effects are summarized in Figure 5.3 as a function of dose.

5.1.4 Magnesium

Absorption

Ingested magnesium is absorbed mainly in the small intestine. Absorption of dietary magnesium has been determined at the rate of 30 to 40 percent. Calcium and magnesium are competitive with respect to their absorptive sites, and excess calcium may partially inhibit the absorption of magnesium (Aikawa, 1963).

Tissue accumulation and clearance

Most magnesium present in the body is in the bones. Bone magnesium seems to be a reservoir to ensure that some will be on hand for vital reactions, regardless of dietary intake. Magnesium, a principal intracellular cation of soft tissue, is mainly involved as a cofactor in enzymatic reactions associated with the metabolism of

FIGURE 5.3
CADMIUM TOXICITY RANGES

carbohydrates and proteins. It is also involved in neurochemical transmission and neuromuscular excitability (AHFS Drug Information, 1991).

The major excretory pathway for absorbed magnesium is the kidney. In subjects on a normal diet, one-third or less of the ingested magnesium is excreted by the kidney. Approximately two-thirds of ingested magnesium is excreted in the feces.

Environmental sources of magnesium

Magnesium is readily available in ingested food, particularly from nuts, cereals, seafood, and meats. The average city water contains about 6.5 parts per million of magnesium, but amounts vary considerably, increasing with the hardness of the water.

The recommended daily allowance (RDA) for magnesium ranges from 4.5 to 6.7 mg/kg/day. In the United States, the average adult ingests between 240 and 480 mg of magnesium daily (from 3.4 to 6.9 mg/kg/day).

Toxicity of magnesium

Available human data on chronic magnesium toxicity following ingestion is limited. It has been reported that therapeutic doses of ingested magnesium as high as 6.9 mg/kg/day may have a laxative effect (AHFS Drug Information, 1991).

In patients with severe renal impairment, hypermagnesemia (characterized by hypotension, nausea, vomiting, electrocardiogram changes, respiratory or mental depression, and coma) has occurred after administration of antacids containing magnesium (AHFS Drug Information, 1991). However, the threshold dose level for these side effects has not been reported. The health effects from exposure to magnesium as a function of dose are summarized in Figure 5.4.

5.1.5 Manganese

Absorption

Following ingestion, manganese absorption is homeostatically controlled; the rate of absorption depends on both the amount ingested and tissue levels of manganese. For adult humans, approximately 3 to 4 percent of dietary manganese is absorbed (Saric, 1986). Manganese can be absorbed following exposure by inhalation, ingestion, and dermal contact. In humans, available data indicate that only 3 percent of an ingested dose of manganese chloride is absorbed (Mena et al., 1969). The rate of absorption is influenced by iron and other metals. In states of iron deficiency, manganese is actively absorbed from the intestine. Individuals with anemia can absorb more than twice the percentage of an ingested dose. However, in states of excess iron, absorption of manganese is by diffusion only (Saric, 1986).

FIGURE 5.4
MAGNESIUM TOXICITY RANGES

High levels of dietary calcium and phosphorus have been shown to increase the requirements for manganese in several species (Lönnerdal et al., 1987).

Tissue accumulation and clearance

Manganese is widely distributed throughout the body. Highest concentrations are found in the liver and kidney and, to a lesser extent, the hair. The biological half-life in humans is 2 to 5 weeks, depending on body stores. Manganese readily crosses the blood-brain barrier and is more slowly cleared from brain than from other tissues (Goyer, 1991). Normal concentrations in the brain are low, but the half-life in the brain is longer and the metal may accumulate in the brain with excessive absorption (National Research Council, 1973).

Absorbed manganese is rapidly eliminated from the blood and concentrates in mitochondria. Initial concentrations are greatest in the liver. Manganese penetrates the placental barrier in all species and is more uniformly distributed throughout the fetus than in adult tissues. It is secreted into milk.

Absorbed manganese is almost totally secreted in bile and reabsorbed from the intestine as necessary to maintain body levels. At excessive exposure levels, other gastrointestinal routes may participate. Excess manganese is eliminated in the feces; urinary excretion is negligible (Goyer, 1991; Saric, 1986).

Environmental sources of manganese

On the whole, food constitutes the major source of manganese intake for humans. The highest manganese concentrations are found in plants, especially wheat and rice. Drinking water generally contains less than 100 $\mu\text{g/L}$. Manganese levels in soil range from 1 to 7000 mg/kg, with an average of 600 to 900 mg/kg. Mining and natural geological background variation can contribute to this variability. Manganese bioaccumulates in marine mollusks up to 12,000-fold, and there is evidence for toxic effects in plants (phytotoxicity) and plant bioaccumulation. The Illinois Institute for Environmental Quality has recommended a criterion of 1 to 2 mg/kg for manganese in soil and 200 mg/kg in plants (Saric, 1986).

Variations in manganese intake can be explained to a large extent by differences in nutritional habits. In populations with cereals and rice as main food sources, the intake will be higher compared to areas where meat and dairy products make up a larger part of the diet. The average daily intake has been estimated to be between 2.0 to 8.8 mg/day (0.03 to 0.13 mg/kg/day) (EPA, 1993), but intakes as high as 12.4 mg (about 0.2 mg/kg/day) have been reported in countries with high cereal intake (Saric, 1986).

Drinking water generally results in an intake of less than 200 μg (0.003 mg/kg/day), although some mineral waters can increase this amount by more than three-fold (Saric, 1986). One study from Greece reported drinking water concentrations of

manganese in excess of 2 mg/L, which would result in daily intakes in the range of 0.06 to 0.07 mg/kg/day (EPA, 1993).

Toxicity of manganese

Manganese is an essential nutrient. Estimated safe and adequate daily dietary intakes for adults range from 0.03 to 0.07 mg/kg/day (Saric, 1986). The EPA no-observed-adverse-effect level (NOAEL) for drinking water is set at 0.005 mg/kg/day while the lowest-observed-adverse-effect level (LOAEL) for drinking water sources is 0.06 mg/kg/day. The EPA RfD for drinking water is 0.005 mg/kg/day. The RfD for food ingestion is 0.14 mg/kg/day. There is some indication that manganese in drinking water is potentially more bioavailable, i.e., more readily absorbed, than is manganese in dietary food sources. This would result in toxic effects with lower ingested doses of manganese in drinking water than in food (EPA, 1993).

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

Limited information is available on the effects of manganese ingestion. Because effects from drinking water seem to differ from those from food sources, only studies on water consumption will be considered here. A Japanese study of 25 people drinking well water with manganese concentrations of 14 mg/L (0.4 mg/kg/day estimated intake) reported symptoms of intoxication, including a mask-like face, muscle rigidity and tremors, and mental disturbances. There were two cases (8 percent) of death among intoxicated people. A Greek study of over 4000 individuals drinking water with manganese concentrations varying from 81 to 2300 μ g/L (estimated intake at 2 L/day for a 70-kg individual range from 0.002 to 0.07 mg/kg/day) showed neurological effects in individuals drinking from 0.05 to 0.07 mg/kg/day, but no effects in individuals drinking less than 0.005 mg/kg/day (Kondakis et al., 1989).

The chemical form of manganese has complex effects on its toxicity. Although more soluble forms are more readily absorbed from the gastrointestinal tract, they also appear to be more rapidly cleared. Exposure to insoluble forms results in lower manganese absorption, but higher chronic tissue levels and therefore greater toxicity (EPA, 1993). Only limited information is available on the effects of various forms of manganese.

Few data are available on manganese toxicity in infants, but it is likely that infants will be more susceptible to toxicity due to greater absorption and greater penetration into the central nervous system (EPA, 1993; Saric, 1986).

The toxicity of manganese is summarized in Figure 5.5.

5.1.6 Nitrate

Absorption

Ingested nitrate is converted in the gut to the toxic nitrite ion, which is readily absorbed. The conversion rate depends on both gut flora and pH, with a more rapid conversion in a more alkaline environment. Infants have a higher gut pH, which is more conducive to growth of bacteria. Therefore, the combination of an alkaline environment and increased bacterial conversion increases the production of nitrite from nitrate in infants, resulting in higher blood levels for a given dose of nitrate. In healthy adults, nitrates are rapidly absorbed from the upper intestine. This rapid absorption reduces the contact time with gut flora, thereby reducing the conversion to nitrite and the resultant toxicity.

Tissue accumulation and clearance

After absorption, the nitrite ion binds to hemoglobin in the blood and oxidizes it, thereby reducing the oxygen-carrying capacity of the blood and decreasing the rate of oxygen release. The oxidized hemoglobin is called methemoglobin and can be reduced back to normal hemoglobin enzymatically by methemoglobin reductase. Infants are more sensitive to these effects due to 1) the presence of fetal hemoglobin, which is more sensitive to oxidation by nitrite, and 2) lower activity of methemoglobin reductase, meaning the methemoglobin remains oxidized for a longer period. Certain individuals have a rare genetic deficiency in methemoglobin reductase and therefore exhibit higher levels of circulating methemoglobin as well. Although these individuals develop alternate metabolic pathways to maintain adequate levels of circulating hemoglobin in the normal state, exposure to high levels of nitrate can result in excessive levels of methemoglobin in these individuals.

In healthy adult humans, the half-time for methemoglobin reductase conversion of methemoglobin back to hemoglobin has been estimated to range from 6 to 24 hours for levels of methemoglobin in the 80 percent to 100 percent range (Bolyai et al., 1972).

Environmental sources of nitrate

Nitrates accumulate in soils from the application of fertilizers, human and animal waste, bacterial nitrogen fixation, mineral dissolution, and plant and animal tissue breakdown. These nitrates can filter through the soil into groundwater. Concentrations of nitrate in well water have been reported to exceed 440 mg/L, or 10 times the current regulatory levels (Lee, 1970).

Bioaccumulation of nitrates from soil and water to plants results in a wide range of nitrate concentrations in fresh fruits and vegetables, with levels as high as

FIGURE 5.5
MANGANESE TOXICITY RANGES

2000 mg/kg being reported in beets and 9000 mg/kg in radishes (Kamm et al., 1965; Smith, 1966). The accumulation of nitrates in plant material is increased by factors such as drought, high temperatures, cloudiness, and insect and herbicide damage to plants. Nitrates and nitrites are also used to preserve meats, especially corned or smoked products.

Toxicity of nitrate

The primary toxicity of nitrate is methemoglobinemia, which is a function of the balance between circulating levels of nitrite and methemoglobin reductase activity. A very high acute dose can produce the same toxicity as a lower dose that slowly increases the concentration of methemoglobin over time. Therefore, the acute and chronic toxicity of nitrate will be summarized together. To allow easier comparisons between ingested doses of nitrate and groundwater levels at Shiprock, dose ranges will be presented in terms of nitrate intake. The reader should be aware that nitrate exposure levels are frequently converted to the nitrogen concentration in the nitrate by dividing the nitrate number by 4.4. Therefore, 44 mg/L nitrate is equivalent to 10 mg/L nitrate-nitrogen.

Symptoms of methemoglobinemia are correlated with the percentage of methemoglobin in the blood. With less than 10 percent methemoglobin, individuals are asymptomatic; more than 25 percent methemoglobin produces weakness, rapid pulse, and tachypnea (rapid breathing); more than 50 to 60 percent methemoglobin can be fatal (EPA, 1991). All of these symptoms reflect the progressive decrease in the availability of oxygen. As explained above, infants are more sensitive to the production of methemoglobin and therefore are considered the most sensitive population. The route of exposure for infants is from the use of contaminated water for formula.

No symptoms of toxicity have been reported with nitrate intakes below 7 mg/kg/day. Mild symptoms such as weakness, rapid pulse, and rapid breathing occur with intakes from 7 to 30 mg/kg/day. The severity of these symptoms increases as increased nitrate intake results in greater levels of methemoglobin and therefore a reduced availability of oxygen. Cyanosis, or a blue appearance to the skin, will occur, followed by unconsciousness as the availability of oxygen is further reduced. The lowest reported fatal dose of chronic nitrate is 35 mg/kg/day for an infant and 116 mg/kg (acute intake) for an adult. A wide range of nitrate intake can produce similar symptoms among individuals because of net differences in gut pH, bacterial activity, and methemoglobin reductase activity. These health effects are summarized in Figure 5.6 as a function of dose.

Data on nitrate toxicity are primarily based on epidemiologic studies of human adults and infants who report to hospitals with symptoms of methemoglobinemia. Exposure doses have been back-calculated from sampling their drinking water in most cases. Therefore, these data do not represent well-controlled studies where dosage ranges can be readily defined. Many of the water sources in these clinical studies showed contamination with bacteria, leading to the possibility that this bacterial exposure was a necessary cofactor in the development of

FIGURE 5.6
NITRATE TOXICITY RANGES FOR INFANTS

methemoglobinemia. One laboratory study indicated that healthy infants could drink nitrate at 24 mg/kg/day in solutions free of bacteria and show no symptoms of methemoglobinemia (EPA, 1993).

Gastrointestinal distress has also been suggested as a cofactor in the development of methemoglobinemia. Anecdotally, infants with colic are more susceptible to nitrate-induced methemoglobinemia (EPA, 1993).

5.1.7 Selenium

Absorption

Although water-soluble forms of selenium such as selenite are approximately 90 percent absorbed in the gastrointestinal tract in rats, humans show lower percentages of absorption (40 to 80 percent; Bopp et al., 1982). Absorption by ruminants is only 30 to 35 percent, probably due to bacterial reduction in the rumen. Absorption of the less soluble elemental selenium or selenium sulfide is poor in rats (Medinsky et al., 1981; Cummins and Kimura, 1971).

Tissue accumulation and clearance

Human studies suggest similar distributions of selenium between humans and laboratory animals (Bopp et al., 1982). At low intake levels, selenium is retained and accumulates in the reproductive organs, brain, and thymus, with only transient accumulation in other organs. Selenite-derived selenium accumulates in the liver and kidneys more rapidly than selenium derived from selenate (Millar et al., 1973). There is some indication that organically bound forms of selenium exist in a separate, more bioavailable pool than either selenite and selenate.

Although urinary excretion is the primary route of elimination under normal dietary conditions (67 percent), in deficiency states fecal excretion is the major pathway. At toxic doses, the major route of excretion is through expired air as dimethylselenide (50 to 60 percent) (Friberg et al., 1986). Although these data were obtained in rats, available data suggest human excretion is similar (Bopp et al., 1982). The elimination of selenium in humans follows three phases with the following half-times: 1, 8 to 20, and 65 to 116 days.

Environmental sources of selenium

The main source of selenium for the general population is foods such as seafood, meat, and grains. Dietary intake of selenium in the United States ranges from 0.0007 to 0.0029 mg/kg/day. Selenium concentrations in groundwater and surface water range from 0.00006 to 0.400 mg/L, with highs of 6 mg/L being reported (Friberg et al., 1986). Concentrations in U.S. public water supplies rarely exceed 0.010 mg/L (EPA, 1980). High selenium concentrations occur in volcanic rock [0.120 milligrams per gram (mg/g)] and in sandstone uranium deposits (1.0 mg/g).

The soil content of selenium varies widely, as does the rate of accumulation by plants. Although grasses and grains do not accumulate selenium in concentrations greater than 50 mg/kg, some plants can accumulate as much as 10,000 mg/kg if grown in high-selenium regions. These high-accumulating plants are generally not used as food sources but can produce toxic effects if consumed by livestock.

Toxicity of selenium

Ingestion of 350 to 4300 mg of selenium by adults has produced vomiting, diarrhea, abdominal cramps, numbness in arms, and marked hair loss and irregular menstrual bleeding in women. Higher intakes can result in unsteady gait, cyanosis of mucous membranes, labored breathing, and sometimes death.

Selenium is an essential nutrient. The recommended daily allowance for adults is 0.0007 to 0.0029 mg/kg/day. Although some biochemical alterations, including prolonged prothrombin time and reduced blood glutathione concentrations, can be observed with intakes of selenium from 0.0107 to 0.0121, no clinical signs of selenosis are observed with these intakes. Mild toxicity, including hair loss or breakage, thickening and brittleness of nails, and a garlic odor in dermal excretions and breath, were reported in human populations with dietary intakes of selenium from 0.015 mg/kg/day. However, selenium intake as high as 0.013 mg/kg/day can produce symptoms of selenosis such as hair and nail loss in susceptible populations.

Persistent clinical symptoms of selenosis are seen due to the chronic dietary intake of selenium by human populations living in areas of China with high selenium concentrations in soil (from 7 mg/kg soil to 12 mg/kg soil). Estimated selenium dietary intake was reported to be 0.018 mg/kg/day and 0.021 mg/kg/day, respectively, for adult women and men (Yang et al., 1989a, 1989b). The average blood selenium concentration associated with this intake of selenium was 1.3 mg/L (ranging from 1.05 mg/L to 1.85 mg/L). Symptoms of chronic selenosis with hair and nail loss and below normal hemoglobin level were reported in the same study with selenium intake of 0.071 mg/kg/day. The serious outbreak of selenium poisoning, including the possible occurrence of neurotoxic effects such as peripheral anesthesia, acroparesthesia, and pain in extremities, was observed with selenium intake of 0.54 mg/kg/day in both women and men. It is important to note that protein intake by members of this population is unknown. These health effects are summarized in Figure 5.7 as a function of dose.

Symptoms of "alkali disease and blind staggers" have been seen in grazing livestock feeding over a long period of time upon selenium-accumulating plants in areas with high soil selenium content (Rosenfeld and Beath, 1964). These symptoms include neurological dysfunction such as impaired vision, ataxia, disorientation, and respiratory distress.

FIGURE 5.7
SELENIUM TOXICITY RANGES

5.1.8 Sodium

Absorption

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

Tissue accumulation and clearance

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

Excretion of sodium is mainly urinary, with appreciable amounts excreted in feces, sweat, and tears (Venugopal and Luckey, 1978). Mammalian renal excretion of sodium is a two-phase process involving glomerular filtration and reabsorption in proximal tubules; of about 600 grams of sodium involved in 24-hour glomerular filtration, approximately 99.5 percent is reabsorbed in human adults. A homeostatic mechanism for sodium functions at the renal excretory level.

Environmental sources of sodium

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

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

No RDA has been set for sodium. The NRC recommendations advise limiting daily sodium intake to less than 2400 mg (34 mg/kg/day). A healthy person requires about 115 mg sodium daily (1.6 mg/kg/day), yet average sodium dietary intake is estimated at 57 to 85 mg/kg/day. The American Heart Association recommends limiting sodium intake to 3000 mg daily.

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

Toxicity of sodium

Acute toxicity symptoms from sodium chloride in healthy adult males accompanied by visible edema may occur with an intake of 35 to 40 grams of salt per day (20 to 23 mg sodium/kg/day since sodium is 39 percent of the weight of sodium chloride) (Meneely and Battarbee, 1976). The mean lethal dose of sodium for humans was reported to be 3230 mg/kg (Venugopal and Luckey, 1978).

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

Research shows that there appear to be critical levels of sodium ingestion that cause blood pressure to rise with age and some people to develop hypertension. It has been reported that with sodium intake below 227 mg/day (3 mg/kg/day for a 70-kg adult), hypertension was absent (Freis, 1976). In the range of 227 to 1591 mg/day (3 to 23 mg/kg/day for a 70-kg adult), a few cases of hypertension may appear, while in the range of 1590 to 8000 mg/day (23 to 114 mg/kg/day for a 70-kg adult), approximately 15 percent of adults would exhibit hypertension. When sodium intake rises above 8000 mg/day, hypertension may be found in about 30 percent of the population. The potential sodium health effects as a function of dose are summarized in Figure 5.8.

5.1.9 Strontium

The isotopes of strontium present at UMTRA Project sites are all natural, stable isotopes. The radioactive element strontium-90 is not naturally occurring and is produced only as a product of fission reactions. Therefore, no radiation exposures are associated with the presence of strontium at UMTRA Project sites.

Absorption

In humans, 14 to 50 percent of an orally administered dose of strontium is absorbed, with peak blood levels occurring within 4 hours. Absorption is proportional to dose, although large doses may overwhelm homeostatic mechanisms. Strontium is absorbed by passive diffusion from the intestinal lumen (Comar and Wasserman, 1964). Because of their chemical resemblance, strontium can effectively displace calcium. In cases of dietary calcium deficiency, strontium

FIGURE 5.8
SODIUM TOXICITY RANGES

is absorbed to a higher degree. The bioavailability of ingested strontium is estimated to be 20 percent but depends on age, species, form of strontium, and dietary levels of phosphorus, vitamin D, and calcium.

Tissue accumulation and clearance

Because of its strong similarity to calcium, 99 percent of the body burden of strontium is found in bone. The average adult body burden of strontium is estimated to be 320 mg (Snyder et al., 1975). Clearance of absorbed strontium is primarily through urine and feces. In humans, 12 to 13 percent of an intravenous dose is eliminated in the feces. Urinary excretion accounts for elimination of nearly 60 percent of an intravenous dose and 4 to 18 percent of an oral dose (EPA, 1990). Although strontium is filtered by the kidney at a rate 3.5 times greater than calcium, calcium is reabsorbed more efficiently than strontium, resulting in a more rapid clearance of strontium.

Environmental sources of strontium

Normal dietary intake of strontium in adult humans ranges from 0.013 to 0.021 mg/kg/day. Stable strontium has been reported in drinking water supplies in Wisconsin and Ohio at concentrations ranging from 0.02 to 34.5 mg/L (Curzon, 1985). Strontium has been used medicinally since 1884, although its medicinal use has steadily declined.

Toxicity of strontium

No data are available on the acute toxicity of stable strontium in humans. The range of lethal doses for orally administered strontium varied across species from a lethal dose of 1826 mg/kg for 50 percent of experimental mice to a lethal dose of 7500 mg/kg in rabbits (EPA, 1990). Death resulted from respiratory failure. Intravenous administration decreased the toxic dose by as much as an order of magnitude (148 mg/kg in mice).

No good data are available for estimating toxic effects related to long-term intake of excess stable strontium. Strontium was administered in the treatment of osteoporosis at a dose of 24 mg/kg/day for as long as 3 years (McCaslin and Janes, 1959). Although no side effects were observed, the bone loss occurring in these patients makes it difficult to compare their response to what might be observed in a healthy population.

In rat studies, the toxicity of strontium is related to its displacement of calcium in bone; this toxicity differs with the developmental stage of the animals. The lowest intake level observed to produce toxicity in young rats was 380 mg/kg/day of strontium carbonate. This dose inhibited calcification of the epiphyseal plate after 3 weeks of exposure. In adult rats, this dose had no effect, but a much larger epiphyseal plate was observed following intake of 750 or 1500 mg/kg/day in the

adult animals (Storey, 1961). Intake of 190 mg/kg/day resulted in no observed toxicity in the young rats. In weanling rats, drinking 633 mg/kg/day of strontium chloride in water resulted in slower mineralization of the bone, slower calcification, and defective long bone growth (Marie et al., 1985). No toxicity was observed in the weanling animals at 525 mg/kg/day intake of strontium chloride. These health effects are summarized in Figure 5.9 as a function of dose.

5.1.10 Sulfate

Absorption

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

Tissue accumulation and retention

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

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

Environmental sources of sulfate

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

FIGURE 5.9
STRONTIUM TOXICITY RANGES

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 in diarrhea; the severity of the diarrhea is dose-dependent. Chronic ingestion of sulfate can result in persistent diarrhea, leading to ionic imbalances and dehydration similar to that seen with extremely high acute doses. In the case of drinking water contaminated with sulfate, the taste of the water may make it unpalatable and reduce consumption. This lower water intake could compound the dehydration effects of the diarrhea. Extreme dehydration can lead to death. As with nitrate toxicity, infants seem to be the most susceptible population for sulfate-induced diarrhea. These health effects are summarized in Figure 5.10 as a function of dose.

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

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

5.1.11 Uranium

Naturally occurring uranium, present at UMTRA Project sites, consists of three radioactive isotopes: uranium-234, uranium-235, and uranium-238. More than 99 percent of natural uranium occurs in the form of uranium-238 (Cothorn and Lappenbusch, 1983). Uranium-238 undergoes radioactive decay by emitting alpha particles to form uranium-234, thorium-230, radium-226, radon 222, polonium-210, and other radioisotopes. The radioactive decay chain of uranium-238 and uranium-234 is summarized in Figure 5.11. As all uranium isotopes in nature are radioactive, the hazards of a high uranium intake are its chemical toxicity and potential radiological damage. This section focuses on the chemical toxicity of natural uranium. Carcinogenic potential associated with exposure to radioactive isotopes of natural uranium is discussed in Section 5.3.

FIGURE 5.10
SULFATE TOXICITY RANGES FOR INFANTS

FIGURE 5.11
HALF-LIVES AND EMISSIONS FROM DECAY CHAIN OF URANIUM-238

Absorption

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

Tissue accumulation and clearance

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

In body fluids, uranium tends to be converted into water-soluble hexavalent uranium (Berlin and Rudell, 1986). Approximately 60 percent of the uranium in plasma complexes with low-molecular-weight anions (e.g., bicarbonates, citrates), while the remaining 40 percent binds to the plasma protein transferrin (Stevens et al., 1980). Following oral exposure in humans, more than 90 percent of uranium that has not been absorbed from the gastrointestinal tract is excreted in the feces. Of the small percent that is absorbed (typically less than 5 percent), approximately 60 percent is excreted in the urine within 24 hours and 98 percent is excreted within 7 days, based on animal studies (Ballou et al., 1986; Leach et al., 1984; Sullivan et al., 1986). A small portion of the absorbed uranium is retained for a longer period.

Environmental sources of uranium

Uranium is a ubiquitous element, present in the earth's crust at approximately 4 parts per million. Uranium concentrations in groundwater and surface water averaged 1 pCi/L and 3 pCi/L, respectively (NCRP, 1984). It is absorbed from the soil into plant tissues to an extent that depends on the plant species and the depth of its root system (Berlin and Rudell, 1986). Plant concentrations of uranium averaged 0.075 $\mu\text{g/kg}$ of fresh plant material (Tracy et al., 1983).

The main source of uranium for the general population is food products such as potatoes, bakery products, meat, and fresh fish, which may contain uranium concentrations between 10 and 100 $\mu\text{g/kg}$ (Prister, 1969). The total dietary intake of uranium from the consumption of average foods is approximately 1 $\mu\text{g/day}$. Cereals and vegetables, particularly root crops, are likely to contribute most to the daily intake of uranium (Berlin and Rudell, 1986).

Toxicity of uranium

Exposure of the general public to natural uranium is unlikely to pose an immediate lethal threat to humans. No human deaths have been reported that are definitely attributable to the uranium exposure. The lethal doses in animals range from 8 to 242 mg/kg, depending on the solubility of the uranium compound tested (higher solubility compounds have greater toxicity) and animal species. High doses of uranium cause complete kidney and respiratory failure.

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

The lowest dose of uranyl nitrate that caused moderate renal damage was given to rabbits in diet at 2.8 mg/kg/day (Maynard and Hodge, 1949). The health effects for uranium are summarized in Figure 5.12 as a function of dose.

5.2 CONTAMINANT INTERACTIONS

Of primary concern among the contaminants in groundwater at Shiprock is the potential for nitrate-sulfate interactions. As discussed above, no clear data are available to assess this interaction, but the epidemiological evidence suggests that in infants, gastrointestinal upset, such as that caused by sulfate, could increase the sensitivity of infants to methemoglobinemia produced by nitrates. However, it is also possible that sulfate-induced diarrhea could decrease the nitrate-reducing bacteria in the intestine, thereby causing a decreased nitrate toxicity.

Interactions between nitrate, sulfate, and hemoglobin also occur. However, no studies have been found that address the potential interactions in a combined exposure to high concentrations of both nitrate and sulfate. Nitrate has been used in the treatment of hydrogen sulfide poisoning. The hydrosulfide anion binds to methemoglobin to form sulfmethemoglobin, effectively removing circulating hydrosulfide. Additional data are needed to assess the likelihood of hydrosulfide formation with oral sulfate exposure or the subsequent formation or stability of sulfmethemoglobin.

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, the absorption of cadmium can be considerably increased under conditions of low intake of calcium, iron, or protein (Nordberg et al., 1985). Low body iron stores can increase cadmium uptake fourfold (Flanagan et al., 1978). Because cadmium binds strongly to metallothionein and its toxicity depends on

FIGURE 5.12
URANIUM TOXICITY RANGES

this binding, metals that increase metallothionein concentrations will increase cadmium binding and potentially its toxicity. Metallothionein can be induced by exposure to a wide range of other metals, including copper and zinc. However, in the continued presence of these other metals, there may be competition for metallothionein binding sites.

Strontium toxicity is strongly influenced by calcium intake. Toxicity is enhanced in low calcium conditions and decreased in the presence of high calcium.

Selenium interacts with a wide range of metals, including arsenic, bismuth, cadmium, cobalt, copper, lead, mercury, platinum, silver, tellurium, and thallium (Friberg et al., 1986). Selenium forms insoluble complexes with silver, copper, cadmium, and mercury. Selenium deficiency may develop in the presence of these other metals, as is seen with cobalt and copper. The formation of these complexes can reduce the toxicity of both selenium and the other metal (Casarett and Doull, 1991). Most of these interactions have been observed in laboratory animals or in livestock. The mechanisms are not completely understood in many cases. Often the selenium-metal complex binds in a stable complex to a larger protein than the metal alone, and a redistribution of this complex away from target tissues occurs.

No information on interactions of uranium with other metals has been found. However, the common target organ suggests interaction with cadmium and magnesium in the production of kidney toxicity.

5.3 CONTAMINANT RISK FACTORS

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

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

Table 5.1 Toxicity values: potential noncarcinogenic effects

Chemical	Chronic oral RfD (mg/kg/day)	Confidence level	Critical effect/organ	RfD basis/RfD source	Uncertainty factor
Antimony	0.0004	Low	Increased mortality, altered blood chemistries	Water/HEAST	1000
Arsenic inorganic	0.0003	High	Keratoses, hyperpigmentation	Water/HEAST	3
Cadmium	0.0005	High	Kidney	Water/HEAST	10
Magnesium	NA	High	Diarrhea, central nervous system depression, methemoglobinemia	Water/Diet/NA	1
Manganese	0.005	Medium to Low	Central nervous system	Water/IRIS	1
Nitrate	1.6	High	Methemoglobinemia, hematologic	Water/IRIS	1
Sodium	NA	High	Hypertension	Water/Diet/NA	1
Selenium	0.005	High	Hair, nail, and skin	Diet/IRIS	3
Strontium	0.6	Medium	Bone	Water/IRIS	300
Sulfate	NA	High	Diarrhea	Water/NA	1
Uranium	0.003	Medium	Kidney, body weight	Water/IRIS	1000

Table 5.2 Toxicity values: carcinogenic effects

Parameter	Oral slope factor (pCi) ⁻¹ , (mg/kg/day) ⁻¹	Weight of evidence classification	Type of cancer	Slope factor basis/slope factor source
Arsenic inorganic	1.8E + 0	A	Skin	Water/HEAST
Lead-210	5.1E-10	A	Bone	Water/HEAST
Polonium-210	2.6E-10	A	Liver, kidney, spleen	Water/HEAST
Radium-226	1.2E-10	A	Bone	Water/HEAST
Thorium-230	2.4E-11	A	Bone	Water/HEAST
Uranium-238	1.3E-10	A	Note ^a	Water/HEAST
Uranium-234	1.4E-10	A	Note ^a	Water/HEAST

^aNo human or animal studies have shown a definite association between oral exposure to uranium and development of cancer.

6.0 RISK EVALUATION

To evaluate health risks to an individual or population, the results of the exposure assessment are combined with the results of the toxicity assessment. As discussed in Section 5.0, potential adverse health effects are a function of how much of the contaminant an individual takes into his or her body. At lower levels, many of the contaminants associated with the mill tailings are beneficial to health, since they are essential nutrients. At higher levels, these same elements can cause adverse health effects or, at very high levels, death. In this section, the estimated reasonable maximum intake, if the floodplain or terrace groundwater were used, is correlated to potential health effects. Additionally, the expected intake for children potentially exposed to selected contaminants of concern while playing on the floodplain and for ingestion of meat and milk from livestock raised on the floodplain are correlated to potential health effects.

6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

6.1.1 Floodplain groundwater potential use

The results from the exposure assessment for the floodplain groundwater exposure pathways showing either the adult intake doses or the toxicologically most sensitive group are used to evaluate potential health effects for noncarcinogens. For antimony, arsenic, cadmium, magnesium, manganese, selenium, sodium, strontium, and uranium, the reasonable maximum intake-per-body-weight is estimated for the adult population. For sulfate and nitrate, the infant exposures are used to evaluate health risks, since this is the toxicologically sensitive population.

The most significant health risk associated with the floodplain groundwater contamination at the Shiprock site is nitrate. As can be seen in Figure 6.1, if groundwater were used for drinking water, the potential exposure is more than 10 times the potentially lethal level for infants.

The levels of nitrate associated with lethal cases of methemoglobinemia vary considerably. One of the more significant factors appears to be that infants prone to gastrointestinal distress seem to be more sensitive. For this reason, it is possible that the gastrointestinal effects associated with sulfate exposures could increase the toxicity of nitrate. However, it is also possible that sulfate-induced diarrhea could decrease the intestinal content of nitrate-reducing bacteria, making nitrate less toxic. Likewise, these high sulfate concentrations may cause the water to be unpalatable to infants, thus reducing their exposure. Figure 6.2 shows that the exposure for sulfate is well above the range where very severe diarrhea or even death due to dehydration would be expected.

Arsenic exposure levels detected in the floodplain groundwater at the Shiprock site are below the levels known to produce pathologic lesions of the skin (Figure 6.3), but are slightly above the EPA oral RfD.

FIGURE 6.1
HEALTH EFFECTS OF POTENTIAL NITRATE EXPOSURE FOR INFANTS
SHIPROCK , NEW MEXICO, SITE

FIGURE 6.2
HEALTH EFFECTS OF POTENTIAL SULFATE EXPOSURE FOR INFANTS
SHIPROCK, NEW MEXICO

FIGURE 6.3
HEALTH EFFECTS OF POTENTIAL ARSENIC EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

Cadmium and uranium exposures should be considered additive, since they affect reabsorption in the proximal tubule of the kidney (Figures 6.4 and 6.5). This causes protein and other nutritive compounds to be spilled in the urine rather than retained by the body. The levels of cadmium alone detected at the Shiprock site are below the threshold level for these effects; nonetheless, the presence of both metals indicates a potential for additive effects.

The estimated uranium intake is less than one-tenth the level of any observed health effects in humans or animal studies. However, the estimated exposure is substantially above (30 times) the EPA acceptable intake level (RfD of 0.003 mg/kg/day). This apparent discrepancy occurs largely because there is little toxicity information for uranium in humans. Uranium has not been demonstrated to serve a beneficial purpose in biological systems; therefore, unlike nutrient metals, it is difficult to define a threshold of toxicity. Because of these uncertainties, the EPA has set a very low acceptable intake level.

Magnesium exposures at these levels would be associated with strong cathartic effects (Figure 6.6). There are not sufficient toxicity data to know whether these high levels are also associated with other, more severe effects.

Exposure levels for antimony and strontium detected at the site would be within ranges for no observed adverse health effect in humans. However, the estimated exposure level for antimony is above EPA oral RfD values (Figure 6.7), whereas the estimated exposure level for strontium is below the EPA oral RfD value (Figure 6.8).

For selenium (Figure 6.9), the estimated exposure level is above the EPA oral RfD and above the upper level of the range for no observed adverse effects in humans. However, it falls into the exposure range between unknown toxicity in humans and the exposure level demonstrated to be associated with mild toxicity, manifested in the brittleness of nails or hair loss.

Exposure levels for manganese are determined to be above the EPA acceptable oral intake levels (RfD). This intake is approximately half the levels reported to cause neurotoxic effects in humans (Figure 6.10).

The sodium exposure level is about three times the upper limit recommended by the NRC and would cause hypertension and renal impairment (Figure 6.11). It is not known if more severe effects would be associated with this high dose.

6.1.2 Terrace groundwater potential use

The exposure assessment for the terrace is limited to groundwater use as drinking water and considers three contaminants (nitrate, sulfate, and uranium) that are most likely to be associated with the former uranium processing site. Exposures to nitrate, sulfate, and uranium are evaluated for their potential to pose adverse health effects in humans exposed through ingestion of contaminated groundwater. The results of this evaluation are shown in Figures 6.1, 6.2, and 6.5.

FIGURE 6.4
HEALTH EFFECTS OF POTENTIAL CADMIUM EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.5
HEALTH EFFECTS OF POTENTIAL URANIUM EXPOSURE
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.6
HEALTH EFFECTS OF POTENTIAL MAGNESIUM EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.7
HEALTH EFFECTS OF POTENTIAL ANTIMONY EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.8
HEALTH EFFECTS OF POTENTIAL STRONTIUM EXPOSURE
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.9
HEALTH EFFECTS OF POTENTIAL SELENIUM EXPOSURE
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.10
HEALTH EFFECTS OF POTENTIAL MANGANESE EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.11
HEALTH EFFECTS OF POTENTIAL SODIUM EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

As can be seen in Figures 6.1 and 6.2, if the terrace groundwater were used for drinking water, the expected exposure would be 10 times the threshold of the potentially lethal level for infants due to nitrate and sulfate exposures. Uranium exposures do not appear to be associated with adverse health effects in potentially exposed humans; however, as discussed above, this level of exposure in humans has not been adequately studied. Due to these uncertainties in the uranium toxicology data base, these exposure levels are more than 10 times the EPA acceptable intake level (oral RfD).

6.1.3 Floodplain surface water potential uses

Potential health effects are evaluated for children aged 6 to 12 years who might be exposed to contaminants while playing on the floodplain. Uranium, strontium, and selenium were detected in surface water bodies and sediments at several locations within the floodplain. As can be seen in Figures 6.5, 6.8, and 6.9, no adverse health effects would be expected from incidental exposures (such as incidental ingestion of sediments and surface water or dermal contact with surface water) estimated for this subpopulation.

Likewise, chronic exposures following consumption of meat and milk from domestic animals grazed and watered on the floodplain area would not be anticipated to pose adverse health effects from exposure to uranium, strontium, and selenium (Figures 6.5, 6.8, and 6.9).

6.2 POTENTIAL CARCINOGENIC HEALTH EFFECTS

If groundwater in the floodplain were used as drinking water, skin cancer could develop as a result of chronic exposure to detected arsenic levels (Figure 6.12). This evaluation is based on the EPA oral slope factor of $1.8 \text{ (mg/kg/day)}^{-1}$ for skin cancer development. The potential lifetime excess cancer risk from exposure to arsenic at these levels is estimated at 1 in 1000.

All uranium isotopes are radioactive and, as such, are considered potential carcinogens. The uranium intake and the potential lifetime carcinogenic risk associated with ingestion of groundwater from the floodplain is shown in Figure 6.13. This estimate is based on the cancer slope factor developed by the EPA; however, natural uranium has not been demonstrated to cause cancer in humans or animals following ingestion exposures.

Uranium is the only radionuclide consistently measured above background in the floodplain plume. However, since uranium decays to radioactive progeny, the entire uranium decay series was evaluated for carcinogenic risk. The estimated carcinogenic risks from these radionuclides are shown in Table 6.1.

FIGURE 6.12
INCREASED CANCER RISK FROM POTENTIAL ARSENIC EXPOSURE FOR ADULTS
SHIPROCK, NEW MEXICO, SITE

FIGURE 6.13
INCREASED LIFETIME CANCER RISK FROM POTENTIAL URANIUM EXPOSURE
SHIPROCK, NEW MEXICO, SITE

Table 6.1 Carcinogenic risk for the floodplain groundwater consumption pathway at the Shiprock, New Mexico, site

Radionuclides	Exposure point concentration (pCi/L)	Well location	Intake (pCi/lifetime)	Ingestion slope factor (pCi) ⁻¹	Lifetime risk
Radionuclides					
Lead-210	1.3	608-615	66,430	5.1E-10	3E-05
Polonium-210	0.9	608-615	45,990	2.6E-10	1E-05
Radium-226	0.2	608-615	10,220	1.2E-10	1E-06
Thorium-230	0.6	608-615	30,660	2.4E-10	7E-06
Total					5E-05

1 mg/L uranium = 686 pCi/L.

Ingestion rate: 2 L.

Exposure frequency: 350 days/year.

Exposure duration: 70 years.

Carcinogenic risks due to ingestion of the terrace groundwater contaminated with uranium is determined to be 7 in 1000 excess cancer lifetime risk (Figure 6.13). Radiological risks due to ingestion of meat and milk from domestic animals grazed and watered on the floodplain area are determined to be at 4 in 1 million excess cancer risk per lifetime.

7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION

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

Ecological assessment can be distinguished from human health assessments in that the nature of ecological relationships influences the impacts of constituents. Environmental toxicology, or ecotoxicology, combines the sciences of ecology and toxicology to study the ecological effects of environmental contaminants. Contaminants are defined as environmental constituents that occur at high enough concentrations to cause deleterious biological effects (Moriarty, 1988). Toxicology has focused largely on studying the effects of single compounds on individual organisms. In ecotoxicology, this must be extended to include effects of multiple constituents on the ecosystem. An ecosystem is composed of both abiotic and biological components. The abiotic component is called the habitat. Biological components are organized into species, populations, and communities. A population is composed of individuals of a species that occur within a defined area, and a community is a collection of all populations (plant, animal, bacteria, and fungi) that live in a defined area and interact with one another. In practice, it is not always easy to set the boundaries for populations and communities. The community plus its habitat is an ecosystem (Moriarty, 1988).

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

It is possible that effects on individual organisms or even populations may not affect the ecosystem at all. If a prey species is affected, predators may be able to shift to feeding on other species; losses of predators may be compensated for by other predators or by immigration of another predator population. The challenge is recognizing when an adverse effect has occurred or is occurring. Unless there is a mass killing within a population or community, effects may go unnoticed. Sublethal effects such as behavioral changes, reduced reproductive success, enzyme level changes, effects on microorganisms, etc., can have repercussions at the population or community level of organization due to an effect on reproductive success in one species that may influence other dependent species. It is often difficult to identify and measure the sublethal effects, and establishing a causal relationship to a specific environmental stressor, such as a specific contaminant, is only

rarely accomplished. Evidence of sublethal effects and gross impacts were not observed during the field survey.

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

7.1 EXPOSURE CHARACTERIZATION

This section identifies the ecological resources present at the site that are likely to be exposed to site-related contaminants and identifies the possible and probable exposure pathways. For risk to exist, a receptor must be exposed to contaminants. Exposure can occur only if there is both a source of contamination and a mechanism of transport to a receptor population or individual.

The tailings piles and associated contaminated soils have been stabilized permanently in a disposal cell on the site. Thus, direct exposure pathways such as incidental ingestion of soil, dermal contact with soil, and inhalation of air containing particulates do not represent an ecological concern. However, other direct exposure pathways such as ingestion of surface water potentially affected by contaminated groundwater in the floodplain and bioconcentration of contaminants in surface water by aquatic organisms and indirect exposure pathways such as consumption of previously exposed organisms (known as bioaccumulation) are possible at the site.

The net accumulation by organisms of a constituent directly from the surrounding environment is known as bioconcentration. Net accumulation as a result of all routes of exposure, including the diet, is known as bioaccumulation. Generally, bioconcentration is measured for uptake of chemicals from water by aquatic organisms. Bioconcentration factors (BCF) for soils have been too variable and dependent on site conditions to make identification of generic soil BCFs possible. Fish BCFs in the scientific literature for the contaminants of concern detected in site surface waters range from 1 liter per kilogram (L/kg) for arsenic to 64 L/kg for cadmium (EPA, 1992). No fish BCFs were found in the available literature for the remainder of the contaminants of concern. Identification of significant fish BCF values have ranged from 1000 L/kg down to 300 L/kg (Kenaga, 1980; EPA, 1989a). None of the contaminants of concern detected in site surface waters would be identified as significant.

The nearest perennial surface water body to the stabilized tailings pile is the wetlands area located on the floodplain at the bottom of Bob Lee Wash. This wetlands area receives a constant supply of water from an uncapped artesian well on the terrace that flows down Bob Lee Wash into the wetlands (Figure 3.6). Surface water bodies and associated wetlands occur in Bob Lee Wash and on the

floodplain. Water from the artesian well on the terrace flows down Bob Lee Wash, into an approximately 2-ac (0.8-ha) wetland, then through a ditch and artificially dug drainage canal to the San Juan River approximately 0.5 mi (0.8 km) northwest of the disposal cell. The San Juan River forms the eastern and northern boundaries of the floodplain (Figure 3.6). A small area of ponded water (approximately 10 by 20 ft (3 by 6 m)) also exists at the base of the escarpment approximately 250 ft (76 m) east of Bob Lee Wash. This wet area is fed by a seep issuing from a fracture in the bedrock of the escarpment. Several other small areas of standing water were observed in depressions in the floodplain. The water in these depressions was stagnant and may be associated with shallow groundwater levels.

All of these water bodies are potential exposure points for resident aquatic life and for terrestrial wildlife (including domestic animals) to come in contact with surface water and/or sediments. These exposure pathways were evaluated in this risk assessment.

A potential current pathway could involve plant uptake of contaminants in groundwater. Due to the shallow depth to groundwater, plants can access contaminated groundwater. Plant uptake was evaluated in this risk assessment assuming that the plant roots accessed groundwater containing the 95 percent upper confidence limit (UCL) concentrations for the contaminants of concern.

For the purposes of this baseline risk assessment, a future hypothetical exposure pathway was also evaluated. Several residences are located west of Bob Lee Wash, and livestock from these residences could forage and drink water on the floodplain. In the future, a well could be placed in the floodplain plume, and contaminated groundwater could be used as a water supply for a livestock watering pond (which could also be stocked with fish) or for irrigation of agricultural crops.

7.2 ECOLOGICAL RECEPTORS

A qualitative survey of the terrestrial flora and fauna existing in the alluvial floodplain and Bob Lee Wash and the aquatic flora and fauna observed in the surface water bodies on the alluvial floodplain and in Bob Lee Wash was conducted on May 19 and 20, 1993.

7.2.1 Terrestrial flora

Dense salt cedar habitat type

Dense salt cedar habitat type occurs in bands along the San Juan River. The salt cedar growth is very dense along the river and is typically 15 to 20 ft (4.5 to 6 m) tall. There is little ground cover except for a layer of leaves from this species. Within this habitat are scattered Russian olive and patches of *Phragmites* (common reed). In small open areas, grass forms the dominant ground cover, with *Bromus* sp. being the most common species observed. Salt grass is also common.

Dense stands of salt cedar were also observed along the drainage canal that traverses the floodplain. Along the upstream end of the canal, the salt cedars have been burned over in many areas. Most of this vegetation has recovered from this burn and has grown to 10 to 15 ft (3 to 4.6 m) high. Most areas within the canal contain a fairly dense growth of cattails.

Open salt cedar habitat type

Open salt cedar habitat type occurs in the interior of the floodplain back from the river. The salt cedars are 10 to 20 ft (3 to 6 m) tall and are scattered approximately 10 to 20 ft (3 to 6 m) apart or more. In many areas, salt grass is the dominant ground cover. Within this type are scattered Russian olives, Fremont cottonwoods, black greasewoods, and a few four-winged salt bushes. A few wild privets were also noted.

Disturbed ground habitat type

A large area of disturbed ground that had much less vegetation than the other areas was observed. There was much bare rock (river cobbles), and most of the top soil has been removed. There were very widely scattered salt cedars and Russian olives in this area. Weedy annuals such as tumble mustard, Russian thistle, and summer cypress were observed here.

Many of the low-lying areas within the floodplain have soft saturated soil. Standing water was observed in two locations (locations 660 and 661, Figure 3.6). The areas of saturated soil and standing water did not have a growth of wetland species. Instead, species such as salt cedar and salt grass were the common species in these areas. It is believed that these areas were wet because the river was very high, resulting in a high water table.

Wetland habitat type

A band of wetlands is present where the water from the artesian well (648) enters Bob Lee Wash. This band is 15 to 40 ft (4.6 to 12 m) wide and approximately 600 ft (183 m) long. It consists of a mixture of plant community types. The area is dominated by cattails, salt grass, mixed salt grass and salt cedars, grass, and wild parsnip. The major grass species in this area is manna grass, although there were other herbaceous grasses and sedges. The wetland downstream of the Bob Lee Wash wetlands covers approximately 2 ac (0.8 ha) and contains a mixture of cattails, softstem bulrushes, salt grass, and open water.

7.2.2 Terrestrial fauna

Detailed studies regarding wildlife that occur in the Shiprock site area were not conducted. Several bullfrogs were seen throughout Bob Lee Wash and the

associated wetland areas. A few garter snakes were also observed near the wetland areas. The only other reptiles observed were several whiptails. Some bird species were seen in the floodplain during the field survey, including mourning dove, brown-headed cowbird, red-winged blackbird, meadow lark, Gambel's quail, yellow-billed magpie, yellow warbler, common raven, spotted sandpiper, house finch, mallard, blue-winged teal, black-headed grosbeak, ring-necked pheasant, American gold finch, song sparrow, and killdeer.

An adult coyote was seen on the hillside leading up to the disposal cell. Tracks of numerous small mammals were also seen in the plant communities and disturbed ground on the floodplain. Insects observed included numerous grasshoppers, crickets, and ant mounds.

No domestic animals were observed during the field survey. However, livestock from the residences located west of Bob Lee Wash may forage or drink water on the floodplain.

7.2.3 Aquatic organisms

A brief qualitative survey of the aquatic organisms in Bob Lee Wash and associated wetlands, the drainage canal, the standing stagnant water, and the pooled water near seep 425 was conducted during the field survey. A fine-mesh dip-net was used to collect fish and benthic macroinvertebrates at these locations.

No observations were made in the San Juan River due to the turbidity, extremely high water level, and rapid velocity of the river.

Other than some mosquito larvae, no aquatic organisms were observed in the two small areas of standing stagnant water (locations 660 and 661). This is not unexpected, considering the ephemeral nature of this standing water.

Visual observations and dip-net collections were made in the wetlands at the mouth of Bob Lee Wash and in the drainage ditch downstream of the wetlands area (locations 658 and 657). The water was clear in the wetlands and ditch. The water depth in the wetlands ranged from less than 1 in (2.5 cm) along the shoreline to approximately 18 in (45 cm) in the deeper areas. The water depth in the drainage ditch ranged from approximately 1 to 12 in (2.5 to 30 cm). There was no perceptible water flow in the wetlands area; the water in the ditch was flowing slowly. The substrate in the wetlands was composed of fine-grained brown sand for the first 0.08 to 0.2 in [2 to 5 millimeters (mm)], then a black gelatinous muck below the surficial sand. The bottom sediments in the drainage ditch were composed of fine- and coarser-grained sands. No floating algae were observed; however, filamentous algae covered a majority of the bottom. Submergent macrophytes were also interspersed with the filamentous algae.

A diverse assemblage of aquatic organisms was seen in the wetlands area and the downstream ditch. Water boatmen (*Corixidae*), backswimmers (*Notonectidae*), water striders (*Gerridae*), mosquito larvae, and unidentified cladocerans were

observed. A large number of damselfly (*Zygoptera*) nymphs were seen, along with a lesser number of dragonfly (*Anisoptera*) nymphs. A few lace-wing larvae (*Chrysopidae*) were collected. Snails were seen grazing on periphyton, and diving water beetles were common. Chironomid larvae and unidentified nematode worms were found in the bottom sediments. An abundance of fish was observed. They appeared to be of the same type, topminnow, and were identified as *Gambusia* sp. Young-of-the-year fish were seen among the reeds and cattails in the shallow water areas. A few adult bullfrogs were seen in the wetlands and ditch. Black tadpoles, ranging in size from approximately 0.2 to 0.4 in (5 to 10 mm), were observed in high numbers in the shallow water areas of the ditch downstream of the wetlands. No small tadpoles were observed in the wetlands; a few large brown tadpoles [approximately 2.5 to 3 in (63 to 75 mm) long] were observed.

The wetlands area in Bob Lee Wash above the floodplain contained a few areas of ponded water where the flow was slowed down, but for most of the wash the water flowed rapidly over cobbles and rocks until reaching the wetland at the mouth of the wash. In general, the same types of aquatic organisms observed in the wetlands were seen in the wash, with one exception. Several case-making caddis fly larvae (*Trichoptera*) were observed on the undersides of rocks in the flowing portions of the wash. Caddis fly larvae may be collected from virtually all ponds, lakes, and streams, although they are more common in cold, unpolluted water (Bland and Jaques, 1978). No fish were observed. This is not surprising, as it is not likely that fish from the wetlands could migrate upstream against the rapid flow of water.

The water in the drainage canal was turbid, which restricted visual observations and limited the effectiveness of dip-net collecting. Most of the water in the canal had probably backed up from the San Juan River. The bottom sediments were composed of very fine brownish-black silts and mud 8 to 12 in (20 to 30 cm) deep. No emergent or submergent macrophytes were observed, nor were any fish seen or collected. Water striders were seen, and some damselfly nymphs and water beetles were collected.

7.3 CONTAMINANTS OF ECOLOGICAL CONCERN

The list of groundwater contaminant levels that exceed background (refer to Table 3.3) was used as the list of contaminants of concern in groundwater for ecological receptors. Additionally, lead-210, polonium-210, radium-226, and thorium-230 are evaluated.

The list of contaminants of concern in the San Juan River water was developed from the list of contaminants detected above background in groundwater (refer to Table 3.3). This list of contaminants was then compared with the surface water data. If a contaminant was not detected in the river water (e.g., nickel) or the median concentration adjacent to and downstream of the site was less than upstream of the site, it was excluded as a contaminant of concern for ecological receptors.

The same screening process used to select the contaminants of concern for the San Juan River water was also used for the river sediment data.

In summary, after all the groundwater constituents exceeding background were examined, the contaminants of concern in the San Juan River are antimony, arsenic, magnesium, sodium, strontium, sulfate, and thorium-230. For the San Juan River sediments, the contaminants of concern remaining after screening are arsenic, manganese, radium-226, strontium, and uranium.

No surface water or sediment samples had been collected from the floodplain water bodies prior to the field survey. Thus, the contaminants of concern for these water bodies were those contaminants that were analyzed for in the samples.

7.4 POTENTIAL IMPACTS TO WILDLIFE AND PLANTS

7.4.1 Terrestrial risk

Terrestrial vegetation was evaluated as a potential exposure pathway. Terrestrial vegetation can be directly exposed to contaminants in groundwater through uptake by the roots. Contaminants may accumulate in various plant parts and exert a wide range of influences, depending on the contaminant. Plant uptake rates vary greatly among species and are affected by factors such as soil characteristics (pH, moisture, redox potential, organic matter, etc.), plant sensitivity, input-output balance, cumulative effects, etc. Foraging wildlife can be exposed indirectly to contaminants in groundwater by ingesting plants that may have bioaccumulated certain contaminants. Based on the shallow depth to contaminated groundwater at the site, it is possible that some plants could have rooting zones in soils that intercept contaminated groundwater.

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

Concentrations of the contaminants of concern in plant tissue were estimated using soil-to-plant BCFs. No soil data are available for the site, nor are water-to-plant BCFs. However, because plants require nutrients to be in an aqueous form for root uptake, it was considered appropriate to use the soil-to-plant BCFs for estimating potential plant uptake at the site. Soil concentrations in the saturated zone were estimated by multiplying the groundwater concentration by the soil-water distribution coefficient, K_d . The methodology and parameters used to estimate root uptake and plant tissue concentrations for the contaminants of concern are presented in Table 7.1. This methodology is described in the literature (ORNL, 1984).

The estimated tissue concentrations for the contaminants of concern in the vegetative portions (e.g., stems, leaves) and in the nonvegetative portions (e.g., fruits, tubers) were compared to approximate concentrations in mature leaf tissue

Table 7.1 Comparison of estimated plant concentrations to phytotoxic concentrations, Shiprock, New Mexico, site

Contaminant of concern	Ground-water UCL (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factors		Estimated concentration in vegetative growth (mg/kg DW) ^a	Estimated concentration in fruits/tubers (mg/kg DW) ^b	Approximate concentration in mature leaf tissue that is toxic (mg/kg DW) ^c
				Bv	Br			
Nonradionuclides								
Antimony	0.037	45	1.7	0.20	0.03	0.33	0.05	150
Arsenic	0.030	200	6.0	0.04	0.006	0.24	0.036	5-20
Boron	0.8	3.0	2.4	4.0	2.0	9.6	4.8	50-200
Calcium	490	4.0	2,000	3.5	0.35	6,900	690	NA
Cadmium	0.0049	6.5	0.031	0.55	0.15	0.018	0.0048	5-30
Chloride	605	0.25	150	70	70	11,000	11,000	NA
Magnesium	2540	4.5	11,000	1	0.55	11,000	6,300	NA
Manganese	9.38	65	610	0.25	0.05	150	30	400-1000
Nickel	0.04	150	6.0	0.06	0.06	0.36	0.36	10-100
Nitrate ^d	4220	0.1	420	30	30	13,000	13,000	NA
Potassium	120	5.5	670	1.0	0.55	670	370	NA
Strontium	12.2	35	430	2.5	0.25	1,100	110	NA
Sulfate ^e	15,000	7.5	110,000	0.5	0.5	56,000	56,000	NA
Uranium	3.0	450	1,400	0.0085	0.004	11	5.4	NA
Sodium	3630	100	360,000	0.075	0.055	27,000	20,000	NA
Selenium	0.255	300	77	0.025	0.025	1.9	1.9	5-30
Zinc	0.087	40	3.5	1.5	0.9	5.2	3.1	100-400

Table 7.1 Comparison of estimated plant concentrations to phytotoxic concentrations, Shiprock, New Mexico, site (Concluded)

Contaminant of concern	Ground-water UCL (mg/L)	Kd (L/kg)	Estimated soil concentration (mg/kg DW)	Soil-to-plant concentration factors		Estimated concentration in vegetative growth (mg/kg DW) ^a	Estimated concentration in fruits/tubers (mg/kg DW) ^b	Approximate concentration in mature leaf tissue that is toxic (mg/kg DW) ^c
				Bv	Br			
Radionuclides								
Lead-210	2.9E-13	900	2.6E-10	0.045	0.009	1.2E-11	2.3E-12	NA
Polonium-210	1.7E-11	500	8.5E-9	0.0025	0.0004	2.1E-11	3.4E-12	NA
Radium-226	2.0E-10	450	9.0E-8	0.015	0.0015	1.4E-9	1.4E-10	NA
Thorium-230	3.0E-8	150,000	4.5E-3	0.00085	0.000085	3.8E-6	3.8E-7	NA

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

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

^cConcentrations are not presented for very sensitive or for highly tolerant plant species (Kabata-Pendias and Pendias, 1992).

^dBv and Br factors available for elemental nitrogen only. It is expected that the Bv and Br factors for nitrate would be higher than for nitrogen.

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

Kd Soil-water distribution coefficient.

Bv Soil-to-plant elemental transfer factor for vegetative portions of food crops and feed plants (ORNL, 1984).

Br Soil-to-plant elemental transfer factor for nonvegetative portions (e.g., fruits, tubers) of food crops and feed plants (ORNL, 1984).

DW Dry weight.

mg/kg Milligrams per kilogram.

mg/L Milligrams per liter.

L/kg Liters per kilogram.

UCL The upper 95 percent one-tailed confidence interval of the median.

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

that have been reported to be toxic to plants (phytotoxic) Table 7.1. The phytotoxic concentrations reported in Table 7.1 are not representative of very sensitive or highly tolerant plant species. The estimated tissue concentrations for the contaminants of concern in the plants on the floodplain do not exceed the available phytotoxicity data. However, no comparison data were available for calcium, chloride, lead-210, magnesium, nitrate, potassium, polonium-210, radium-226, strontium, sodium, sulfate, thorium-230, and uranium. Thus, it is not possible to evaluate whether the estimated tissue concentrations could result in adverse effects to plants.

Bioaccumulation in terrestrial organisms as a function of contaminants of concern in ingested plants or animals (e.g., birds eating aquatic macroinvertebrates) is a potential exposure pathway at the site. Birds and other vertebrates consuming these plants and animals can bioaccumulate some of the contaminants of concern from this diet if the amount ingested exceeds the amount eliminated. This is often a function of the areal extent of contamination versus the areal extent of the animals' feeding range. In the case of small contaminated areas, the amount of food in the diet usually exceeds the impacted food and bioaccumulation is not a concern. Therefore, exposure via the diet for all trophic level species is possible in certain areas (e.g., wetland areas), but the potential for bioaccumulation is not always a concern.

Biomagnification is a more severe situation in which the concentration of a constituent increases in higher levels of the food chain because the contaminant concentrations are accumulated through each successive trophic level. Biomagnification effects are of particular concern for top predators, especially carnivorous birds and mammals. Only a limited number of constituents have the potential for magnifying in the food chain. Most constituents are metabolized in organisms and eliminated at each level of the food chain.

Based on available information, the potential for the detected contaminants of concern to represent a health hazard via food chain transfer is probably low. However, no sampling of plant or animal tissue has been conducted to date as part of the site characterization, and it is not possible to make a definitive statement concerning the bioaccumulation or biomagnification of site-related contaminants of concern.

To evaluate the future hypothetical impact that use of contaminated groundwater in a livestock pond might have on wildlife (i.e., animals drinking from the pond or fish stocked in the pond), the 95 percent UCL groundwater concentrations for the contaminants of concern were compared to available comparison water quality criteria (Table 7.2). No available Federal or state criteria or standards are established for the protection of terrestrial wildlife via water exposure. Therefore, it is difficult to evaluate the potential hazards to terrestrial receptors without additional information. Available surface water quality criteria for the protection of freshwater aquatic life, however, do exist and include the Federal Water Quality Criteria (FWQC) (EPA, 1986).

Table 7.2 Comparison of contaminants of concern in groundwater with available water quality criteria, Shiprock, New Mexico, site

Contaminant of concern	Groundwater UCL	FWQC	Water concentration protective of livestock ^a	Concentration in irrigation water protective of plants ^a
Ammonium	516	NA	NA	NA
Antimony	0.037	0.030 (Sn III)	NA	NA
Arsenic	0.030	0.19 (As III)	0.20	0.10
Boron	0.8	NA	5.0	0.75
Calcium	490	NA	NA	NA
Cadmium	0.0049	0.048 ^b	0.050	0.010
Chloride	605	230	NA	NA
Lead-210	0.9 pCi/L	NA	NA	NA
Magnesium	2540	NA	NA	NA
Manganese	9.38	1.5 ^c	NA	0.20
Nickel	0.04	8.9 ^b	NA	0.20
Nitrate	4220	90 ^d	100	NA
Polonium-210	1.3 pCi/L	NA	NA	NA
Potassium	120	NA	NA	NA
Phosphate	0.1	NA	NA	NA
Radium-226	0.2 pCi/L	NA	NA	NA
Selenium	0.255	0.035	0.050	0.020
Sodium	3630	NA	NA	NA
Strontium	12.2	NA	NA	NA
Sulfate	15,000	NA	1000	NA
Thorium-230	0.6 pCi/L	NA	NA	NA
Uranium	3.0	NA	NA	NA
Zinc	0.087	6.0 ^b	25	2.0

^aFrom EPA (1972), unless specified otherwise.

^bWater hardness-related chronic FWQC (EPA, 1986). Criterion presented was calculated using the average hardness (11,710 mg/L) determined from concentrations of calcium and magnesium in alluvial aquifer wells.

^cNo FWQC available. Value presented is the lower end of the tolerance range for freshwater aquatic life (EPA, 1986).

^dConcentration at or below which no adverse effects are expected for warm water fish (EPA, 1986).

All concentrations reported in milligrams per liter, unless specified otherwise.

UCL The upper 95 percent one-tailed confidence interval of the median.

FWQC Federal Water Quality Criteria for the protection of freshwater aquatic life via chronic exposure (EPA 1986), unless specified otherwise.

NA Not available.

The 95 percent UCL groundwater concentrations for antimony, chloride, manganese, nitrate, and selenium exceeded the comparison water quality criteria (Table 7.2), while the groundwater concentrations for arsenic, cadmium, nickel, and zinc were below the comparison aquatic criteria. The 95 percent UCL for antimony, chloride, and selenium exceed the chronic FWQC, indicating that this water would be unacceptable for aquatic organisms. There are no comparison water quality criteria available for ammonium, boron, calcium, lead-210, magnesium, potassium, phosphate, radium-226, sodium, strontium, sulfate, thorium-230, and uranium.

Another future hypothetical use of the groundwater in the area is for irrigating agricultural crops. Table 7.2 shows the approximate concentrations in water used for irrigation that should be protective of plants (EPA, 1972). Seven of the contaminants of concern, arsenic, boron, cadmium, manganese, nickel, selenium, and zinc, have comparison criteria. The 95 percent UCL groundwater concentrations for boron, manganese, and selenium exceed the comparison criteria, while the concentrations of arsenic, cadmium, nickel, and zinc are below the comparison criteria. No comparison criteria are available for the remaining contaminants of concern.

Other basic criteria required to evaluate the suitability of water quality for irrigation, in addition to the concentrations of metals that may be toxic to plants, are the total soluble salt content and the sodium hazard. Excess salt in water increases the osmotic pressure of the soil solution. This increase can elicit a physiological drought condition in the plants. The total soluble salt content of water can be measured by the specific conductance (electrical conductivity). The 95 percent UCL for specific conductance from a representative plume well (well 615) is 16,600 micromhos per centimeter ($\mu\text{mhos/cm}$). This value is well above the upper end of the acceptable range for salt-tolerant plants, 7,500 $\mu\text{mhos/cm}$ (Follett and Soltanpour, 1985). Sodium concentrations can contribute to the total salinity of an irrigation water and may be directly toxic to sensitive crops. However, the primary concern with elevated sodium concentrations is the adverse effects on the physical characteristics of soils (e.g., increase in colloiddally absorbed sodium, resulting in hard compact soil). The sodium hazard of a water is expressed as the sodium adsorption ratio (SAR), which is calculated as the proportion of sodium to calcium plus magnesium in the water. Using the 95 percent UCL groundwater concentrations for sodium, calcium, and magnesium, a SAR of 14.6 was calculated. This SAR is above the upper SAR limit of 10 (Follett and Soltanpour, 1985). Water with a SAR value of greater than 10 should not be used as the sole source of irrigation water for long periods of time.

Based on the available information, use of the alluvial groundwater (containing the upper 95 percent UCL concentrations) as irrigation water would result in deleterious effects to plants, primarily due to elevated concentrations of boron, manganese, and selenium and to the salinity and sodium hazard.

7.4.2 Aquatic risk

Bob Lee Wash and associated wetlands

Surface water (unfiltered) and surficial sediment [0 to 4 in (0 to 10 cm)] were collected from Bob Lee Wash above the floodplain (location 662) and from the wetlands areas on the floodplain (locations 658 and 657) (Figure 3.6) on May 19 and 20, 1993. The surface water samples were analyzed for manganese, nitrate, selenium, strontium, and uranium. The sediment samples were analyzed for manganese, selenium, strontium, and uranium.

The surface water concentrations for the analyzed constituents were very similar among the sampling locations and compared with concentrations detected in the artesian well groundwater (Table 7.3). This similarity is expected, considering the constant input of artesian well water to these areas.

The detected concentrations relative to available comparison water quality criteria show that none of the constituents were detected at a concentration above the comparison criteria. Selenium was not detected in the surface water collected from Bob Lee Wash and the wetlands.

There are no sediment quality criteria or background sediment quality data for the floodplain with which to compare the detected concentrations. In general, the concentrations were similar among locations, although the highest concentrations of strontium and uranium were detected in the large wetlands at the mouth of Bob Lee Wash (Table 7.4). No information was found in the available literature on the potential for these concentrations to represent a hazard to ecological receptors, but additional research is recommended.

Based on this qualitative evaluation and the diversity of organisms observed in Bob Lee Wash and the wetlands, it does not appear that the concentrations detected in surface water and sediments are having an adverse impact on the ecosystem. However, additional field work and quantitative analyses would be required to fully evaluate the biological integrity of this ecosystem.

Drainage canal

Surface water (unfiltered) and surficial sediment samples were also collected during the field survey from two locations (locations 655 and 656) in the drainage canal that traverses the floodplain. The results are presented in Tables 7.3 (surface water) and 7.4 (sediment). The detected surface water concentrations for each contaminant of concern were generally similar, although selenium was reported at the detection limit at location 655. The detected concentrations for all of the contaminants of concern were below available comparison water quality criteria.

The sediment concentration of selenium was highest in the canal at the location near the confluence with the San Juan River (location 655), while the concentrations for the other contaminants of concern were highest at the upstream

Table 7.3 Comparison of contaminants of concern in site surface water bodies with available water quality criteria, Shiprock, New Mexico, site

Contaminant of concern	<u>Bob Lee Wash and associated wetlands</u>			<u>Drainage canal</u>		<u>Stagnant pools</u>		<u>Wet area below seep 425</u>	FWQC
	Location ID			Location ID		Location ID		Location ID	
	657	658	662	655	656	660	661	659	
Manganese	0.03	0.03	<0.01	0.40	0.27	0.54	0.15	0.01	1.5 ^a
Nitrate	1.2	1.0	1.3	8.7	8.6	1.8	1.6	236	90 ^b
Selenium	<0.005	<0.005	<0.005	0.005	<0.005	<0.005	<0.005	0.035	0.035
Strontium	12.0	12.0	10.3	1.92	1.57	7.13	3.98	7.34	NA
Uranium	<0.001	<0.001	0.001	0.024	0.066	1.1	0.33	0.69	NA

^aNo FWQC available. Value presented represents lower end of tolerance range for freshwater aquatic life (EPA, 1986).

^bNo FWQC available. Value presented is the concentration at or below which no adverse effects are expected for warmwater fish (EPA, 1986).

All concentrations reported in milligrams per liter in total (unfiltered) samples

NA - Not available.

FWQC - Federal Water Quality Criteria for the protection of freshwater aquatic life via chronic exposure (EPA, 1986), unless specified otherwise.

Table 7.4 Contaminants of concern detected in sediment from site surface water bodies, Shiprock, New Mexico, site

Contaminant of concern	<u>Bob Lee Wash and associated wetlands</u>			<u>Drainage canal</u>		<u>Stagnant pools</u>		<u>Wet area below seep 425</u>
	Location ID			Location ID		Location ID		Location ID
	657	658	662	655	656	660	661	659
Manganese	162	179	533	360	1050	654	312	184
Selenium	1.2	<1.7	<0.9	5.0	<1.0	<0.8	<3.0	4.2
Strontium	188	448	232	157	185	698	128	1620
Uranium	2.6	4.7	3.8	3.3	5.8	22	5.9	44

All concentrations reported in milligrams per kilogram.

location (656) in the drainage canal. As discussed earlier, there are no sediment quality criteria or background data with which to compare the canal data. Observations of aquatic life in the canal were hampered by the turbid water and lack of substrate (e.g., aquatic macrophytes, rock/cobble, etc.).

Stagnant pools

Two small pools of standing stagnant water on the floodplain were sampled during the field survey (locations 660 and 661) (Figure 3.6). The unfiltered surface water concentrations of manganese and uranium detected at location 660 were the highest detected in any of the floodplain surface water bodies. This stagnant pool of water was located midway between monitor well 616 and the San Juan River. No selenium was detected (Table 7.3). Similar to the situation in Bob Lee Wash and the drainage canal, none of the contaminants of concern were detected at concentrations above the comparison water quality criteria (Table 7.3). However, the maximum uranium concentration of 1.1 mg/L detected at location 660 is more than one order of magnitude higher than concentrations detected in the San Juan River, the drainage canal, and Bob Lee Wash and the associated wetlands. It is not known whether this concentration would represent a hazard to aquatic life. As mentioned earlier, these small pools of standing water are ephemeral and were probably associated with the high water table that occurred during the field survey.

The concentrations detected in sediments from these pools were similar to concentrations detected at locations in the drainage canal, with the exception of the elevated concentrations of strontium (698 mg/kg) and uranium (22 mg/kg) at location 660.

Wet area near seep 425

Unfiltered surface water and sediment samples were collected from the area of ponded water at the base of the escarpment directly below seep 425. The primary source of water for this area of ponded water is probably discharge from the seep. A review of the surface water data seems to support this. The highest concentrations for nitrate and selenium anywhere on the floodplain were detected in this water (Table 7.3). The reported concentration of nitrate exceeds the comparison water quality criteria, and the concentration of selenium equals the FWQC. Concentrations this high could adversely affect aquatic life.

The sediment concentrations of strontium (1620 mg/kg) and uranium (44 mg/kg) were the highest detected in any of the floodplain surface water bodies. Again, this is probably a reflection of the seep water input. It is not known whether the fish observed in this ponded water are year-round residents or whether they migrated from the Bob Lee Wash wetlands area during high water, then were trapped as the water receded.

San Juan River water

Another potential exposure point is the San Juan River in the vicinity of the site along the alluvial floodplain. As described in Section 2.4, the plume of contaminated alluvial groundwater is believed to be discharging to the river. Also, contaminants in a dissolved state or associated with particulates may have been transported from the site to the floodplain and to the river via over land surface runoff. Another potential release mechanism involves expression of contaminated groundwater to a floodplain surface water body that intersects the water table (e.g., the drainage canal). Thus, a comparison of the historical surface water data collected from the San Juan River at locations upstream (locations 546 and 554 pooled together) and at locations adjacent to and downstream (locations 548 through 553 and 555 pooled together) of the site was conducted. Unfiltered data were collected from the river in 1993, and filtered data were collected in 1987 through 1989.

A review of the surface water data indicates that thorium-230 was not detected upstream of the site; however, the median concentration detected adjacent to and downstream of the site was less than the detection limit. The median concentrations for antimony, arsenic, magnesium, sodium, strontium, and sulfate at the locations adjacent to and downstream of the site were higher than at the upstream locations. However, the median concentrations for each contaminant of concern were within the same order of magnitude, and the differences between the upstream and downstream concentrations ranged from approximately 9 to 39 percent. Although these data suggest a relationship with the site, they do not provide compelling evidence that site-related contamination has affected the water quality of the San Juan River. For purposes of this risk assessment, the median concentrations for the contaminants of concern were compared to available comparison water quality criteria. These comparison criteria are FWQC for the protection of freshwater aquatic life via chronic exposure (EPA, 1986; 1988b).

A comparison of the historical surface water quality data with available comparison water quality criteria indicates that none of the concentrations exceed their respective comparison criteria upstream or downstream of the site (Table 7.5). Water quality criteria for aquatic life are not available for magnesium, sodium, strontium, sulfate, and thorium-230. When it is considered that there is essentially no difference between the upstream and downstream median concentrations for these metals, it is unlikely that these concentrations represent an ecological concern or that they can be attributed to impacted groundwater from the floodplain. However, continued monitoring of the river is recommended.

San Juan River sediments

There are no established tribal, state, or Federal sediment quality criteria (SQC) for the protection of aquatic life, although interim SQCs have been presented by the EPA for several nonpolar hydrophobic organic compounds (EPA, 1988c). However, none of the contaminants of concern at this site are organic compounds.

Table 7.5 Comparison of contaminants of concern in San Juan River water with available water quality criteria, Shiprock, New Mexico, site

Contaminant of concern	Median concentration in San Juan River		FWQC
	Adjacent and downstream of site ^a	Upstream of site ^b	
Antimony*	0.0059	0.0049	0.030 ^c
Arsenic	0.01	0.0075	0.19 ^d
Magnesium*	11	10	NA
Sodium	56	35	NA
Strontium	1.1	0.84	NA
Sulfate	192	117	NA
Thorium-230	0.3 pCi/L	<0.4 pCi/L	NA

^aSampling locations 548, 549, 550, 551, 552, 553, and 555.

^bSampling locations 548 and 554.

^cDraft chronic FWQC for antimony (III) (EPA, 1988b).

^dChronic FWQC for arsenic (III) (EPA, 1986).

All concentrations reported in milligrams per liter, unless specified otherwise, from unfiltered samples. No unfiltered data available for contaminants of concern noted with "*." Therefore, filtered data are reported. Unfiltered data are from April 1993. Filtered data are from the period of March 19, 1987, through April 22, 1989.

NA - Not available.

FWQC Federal Water Quality Criteria for the protection of freshwater aquatic life via chronic exposure (EPA, 1986), unless specified otherwise.

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

National Oceanic and Atmospheric Administration (NOAA) effects-based sediment quality values are available for evaluating the potential for constituents in sediment to cause adverse biological effects. "Effects range-low" (ER-L) values are concentrations equivalent to the lower 10th percentile of available data screened by the NOAA and indicate the low end of the range of concentrations in specific sediments at which adverse biological effects were observed or predicted in sensitive species and/or life stages.

NOAA sediment quality values are available for only one of the detected contaminants of concern, arsenic (Table 7.6). The mean arsenic concentrations upstream of the site of 1.9 mg/kg and downstream of the site (2.5 mg/kg) are well below the NOAA ER-L value of 33 mg/kg. This suggests that the potential for arsenic to represent a hazard to aquatic life is low. Because there are no sediment quality values for the other contaminants of concern, it is not possible to evaluate whether the detected sediment concentrations could adversely affect biota.

A comparison between the mean surface water concentrations for those contaminants that exceeded background groundwater quality (refer to Table 3.3) upstream versus downstream of the site indicates that the concentrations were slightly higher downstream for arsenic, manganese, radium-226, strontium, and uranium. Conversely, the concentrations of polonium-210 and thorium-230 were slightly higher upstream of the site. Antimony, magnesium, nitrate, sodium, and sulfate were not analyzed for in the river sediment samples, and the remaining groundwater contaminants (cadmium, lead-210, and selenium) were not detected (Table 7.6).

7.5 POTENTIAL IMPACTS TO LIVESTOCK

Impacted groundwater under the floodplain may discharge to the drainage canal or may appear as surface water bodies during certain times of the year (e.g., periods of high water in the San Juan River). The potential exists for livestock from the residences west of Bob Lee Wash to search the floodplain for food and water. Ingestion by livestock of vegetation that may have bioconcentrated contaminants from alluvial groundwater is a potential pathway. However, without additional data (e.g., actual plant tissue concentrations), it is difficult to evaluate this exposure pathway.

To evaluate the potential impact to livestock that might drink out of the floodplain surface water bodies, the detected concentrations were compared to approximate

Table 7.6 Comparison of contaminants of concern in San Juan River sediment with available sediment quality screening levels, Shiprock, New Mexico, site

Contaminant of concern	Mean concentration in San Juan River		NOAA values
	Adjacent to and downstream of site ^a	Upstream of site ^b	ER-L
Arsenic	2.5	1.9	33
Manganese	309	263	NA
Radium-226	1.3 pCi/g	1.2 pCi/g	NA
Strontium	66	58	NA
Uranium	4.5	4.2	NA

^aSampling locations 548, 549, 550, 551, 552, 553, and 555.

^bSampling locations 546 and 554.

All concentrations reported in milligrams per kilogram, unless specified otherwise.

NA - Not available.

ER-L - Effects range-low (NOAA, 1990).

pCi/g - Picocuries per gram.

drinking water concentrations considered to be protective of livestock (Table 7.7). A review of available comparison criteria suggests that livestock could use the floodplain surface water as a source of drinking water, with the exception of the wet area below seep 425. The detected nitrate concentration (236 mg/L) in this area is more than twice the comparison criterion (100 mg/L) (Table 7.7). There are no comparison criteria available for manganese, strontium, or uranium.

Based on past and current agricultural activities in the area surrounding the site, the possibility exists that groundwater could be used in the future to provide water for a livestock watering pond. In an attempt to evaluate the potential impact to livestock in this future hypothetical scenario, the 95 percent UCL groundwater concentrations for the contaminants of concern were compared to approximate drinking water concentrations considered to be protective of livestock (EPA, 1972) (Table 7.2). The comparison water quality criteria for nitrate, selenium, and sulfate are exceeded by the 95 percent UCL groundwater concentrations, while the 95 percent UCL concentrations for the other contaminants of concern are below the comparison criteria. The groundwater concentration of nitrate (4220 mg/L) is more than one order of magnitude above the comparison criterion (100 mg/L). If this groundwater were used as the sole source of drinking water for livestock, it would ultimately result in death to ruminants (e.g., cattle) from methemoglobinemia (Deeb and Sloan, 1975; NAS, 1972). No comparison water quality criteria have been reported for the remaining contaminants of concern. However, the available information suggests that the use of groundwater as a source of drinking water for livestock would be unacceptable due to nitrate, selenium, and sulfate.

7.6 SUMMARY

Insufficient water quality criteria and sediment quality criteria were available to allow a comprehensive evaluation of the impact of surface water, sediments, and contaminated groundwater on ecological receptors.

Surface water data from the San Juan River indicate the presence of slightly higher concentrations for several of the contaminants of concern at locations adjacent to and downstream of the site. However, there is no clear trend suggesting that site-related constituents have adversely affected the water quality of the river. There are not enough data to adequately evaluate whether these concentrations represent a hazard to aquatic life.

For many of the contaminants of concern, the sediment concentrations detected in the river at locations adjacent to and downstream of the site were higher than at the upstream locations. A qualitative evaluation of the sediment data suggests that past or current conditions at the site may act as a release source for the detected contaminants of concern. Concentrations of arsenic detected in the river sediments (both upstream and downstream of the site) are below the available sediment quality values. Sediment quality values or criteria are not available for the majority of the contaminants of concern; thus, the potential for these concentrations to represent an ecological concern cannot be evaluated with current data.

Table 7.7 Comparison of contaminants of concern in site surface water bodies with water concentrations protective of livestock, Shiprock, New Mexico, site

Contaminant of concern	<u>Bob Lee Wash and associated wetlands</u>			<u>Drainage canal</u>		<u>Stagnant pools</u>		<u>Wet area below seep 425</u>	Water concentration protective of livestock ^a
	Location ID			Location ID		Location ID		Location ID	
	657	658	662	655	656	660	661	659	
Manganese	0.03	0.03	<0.01	0.40	0.27	0.54	0.15	0.01	NA
Nitrate	1.2	1.0	1.3	8.7	8.6	1.8	1.6	236	100
Selenium	<0.005	<0.005	<0.005	0.005	<0.005	<0.005	<0.005	0.035	0.050
Strontium	12	12	10.3	1.92	1.57	7.13	3.98	7.34	NA
Uranium	<0.001	<0.001	0.001	0.024	0.066	1.1	0.33	0.69	NA

^aFrom EPA (1972).

All concentrations reported in milligrams per liter.

Selected contaminants of concern were analyzed for in the unfiltered surface water and sediment samples collected from Bob Lee Wash and from the floodplain surface water bodies. The concentrations of nitrate and selenium detected in water from the wet area below seep 425 may be of concern to aquatic life if chronic exposure occurs. None of the contaminants of concern analyzed for in the sediments have comparison sediment quality criteria. This makes it difficult, without additional data, to evaluate the potential for adverse effects to biota.

Potential current exposure to livestock using the floodplain for forage and drinking water was evaluated. A comparison of available livestock drinking water quality values with concentrations detected in the floodplain surface water bodies suggests that livestock could use these water bodies as the sole drinking water source, with the exception of the wet area below seep 425. The nitrate detected in this water exceeded the comparison criterion. However, these data represent only a "snapshot" in time and may not be representative of long-term conditions. Additional monitoring could provide information on possible temporal and seasonal variations in water quality, as well as sediment quality.

Based on available data and criteria, no ecological threat exists to plants that may have roots in contact with the most contaminated groundwater in the floodplain alluvial aquifer. However, this groundwater would be unsuitable for hypothetical future use as irrigation water for crops due to boron, manganese, and selenium and to the salinity and sodium hazard. Water from the most contaminated wells in this aquifer would not be suitable as a source of water for fish to live in or as a sole source of drinking water for livestock.

The potential for the contaminants of concern detected in media at the site to represent a food chain hazard (via bioaccumulation and biomagnification) is considered low, based on available data. However, no plant or animal tissue analysis has been conducted to date.

8.0 INTERPRETATION AND RECOMMENDATIONS

8.1 RISK SUMMARY

The UMTRA Project is required by the UMTRCA to protect public health and the environment from radiological and nonradiological hazards associated with the uranium mill sites. This baseline risk assessment was conducted on the Shiprock site to evaluate the presence of these hazards. Because groundwater is currently not used by area residents, human health is not at risk from direct groundwater uses. However, health risks would be associated with potential future use of contaminated groundwater in the floodplain and terrace.

The primary risks associated with future use of contaminated groundwater at the Shiprock processing site result from ingestion of nitrates and sulfate by humans and other animals. Using groundwater from the plume for drinking purposes would result in nitrate intakes in the potentially lethal range for infants following short-term exposure. These nitrate levels are also unacceptable for domestic livestock and wildlife. Sulfate levels in plume wells are associated with severe diarrhea and possible death due to dehydration. Lifetime excess cancer risks associated with uranium (1×10^{-2}) and arsenic (1×10^{-3}) are also at unacceptable levels.

Although the potential for current exposure to surface expressions of groundwater exists in the area of the San Juan River floodplain near the Shiprock site, these exposures are not anticipated to pose an immediate threat to public health. Additionally, adverse health effects would not be expected following ingestion of meat and milk from animals grazed and watered on the San Juan River floodplain. However, this evaluation is based on very limited data. Additional data are required to completely evaluate the significance of these exposure pathways.

Human adverse health effects would not be expected if meat and milk from animals grazed and watered on the San Juan River floodplain were ingested. However, at least one surface water location had nitrate at a level that exceeds livestock drinking water criteria.

8.2 GROUNDWATER CRITERIA

In 1983, the EPA established health and environmental protection standards for the UMTRA Project, and in 1987 the EPA proposed revised groundwater standards. The UMTRA Project is required to adhere to the 1987 proposed groundwater standards until final standards are published. The groundwater standards consist of 1) groundwater protection standards to evaluate disposal cell performance, and 2) groundwater cleanup standards for existing contamination at processing sites. These standards are summarized in Table 8.1 for contaminants that have a proposed maximum concentration limit (MCL). Because an MCL is not established for every contaminant, the proposed standard requires meeting background levels for those contaminants that do not have an MCL.

Table 8.1 Concentration limits of constituents

Constituent	UMTRCA MCL 40 CFR 192.02 (mg/L)	Health advisories 10-kg child, 10-day (mg/L)	Health advisories 70-kg adult lifetime (mg/L)
Chemicals (inorganic)			
Antimony	-	0.015 ^a	0.003 ^a
Arsenic	0.05	-	-
Barium	1.0	-	2
Boron	-	0.9	0.6 ^a
Cadmium	0.01 ^a	0.04	0.005 ^a
Chromium	0.05 ^a	1.0	0.1 ^a
Copper	-	-	-
Fluoride	-	-	-
Lead	0.05	-	0.015 ^b
Manganese	-	-	-
Mercury	0.002	-	-
Molybdenum	0.1	0.08 ^c	0.04 ^c
Nickel	-	1.0	0.1
Nitrate	44 ^{a,d}	44 ^e	-
Selenium	0.01 ^a	-	-
Silver	0.05	0.2	0.1
Strontium	-	25.0	17 ^a
Sulfate	-	-	-
Thallium	-	0.007	0.0004
Vanadium	-	0.08 ^{a,c}	0.02 ^{a,c}
Zinc	-	6.0	2
Radionuclides			
Radium-226/-228	5 pCi/L	-	-
Uranium (U-234/-238)	30 pCi/L ^a (0.044 mg/L)	-	-

^aExceeded in plume wells 608, 609, 610, 611, 613, 614, or 615.

^bAction level.

^cExceeded in background.

^dEqual 10 mg/L nitrate as nitrogen.

^eUnder review.

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

The following contaminant concentrations in plume wells have consistently exceeded the MCL and health advisory levels: nitrate, selenium, cadmium, and uranium. Antimony concentrations exceeded the 10-day advisory for a 10-kg child and the lifetime health advisory for a 70-kg adult. Boron, chromium, molybdenum, and vanadium marginally exceeded the health advisories; however, the concentrations of these metals are not statistically above background. Strontium concentrations exceeded the lifetime health advisory for a 70-kg adult, but they were not elevated above the 10-day health advisory for a 10-kg child.

8.3 RISK MITIGATION MEASURES

Because there is a potential for serious health effects following short-term use of nitrate-contaminated groundwater in the floodplain alluvium and the terrace, this section presents possible ways to restrict access to groundwater so as to mitigate risks.

Institutional controls are defined in the proposed groundwater standards for UMTRA as mechanisms that can be effectively used to protect human health and the environment by controlling access to contaminated groundwater. Although the proposed standards refer to institutional controls for long periods of time (e.g., up to 100 years during natural flushing), this concept can also be applied to short-term or interim restriction of access to groundwater. Because it will take years to characterize groundwater at the Shiprock site, and because not all 24 UMTRA sites can be evaluated simultaneously, interim institutional controls are needed before remedial action decisions are made for individual sites.

The Division of Natural Resources manages water resources within the Navajo Nation. Both the Division of Natural Resources of the Navajo Nation and the Navajo Nation Environmental Protection Agency are responsible for water supply, standards, and discharge. The Division of Natural Resources is responsible for administering the water permit system and developing water-code compliance regulations for the Navajo Nation.

Currently, two permitting categories exist for water access: permits for well drilling and permits for water use. Both permits are necessary for drilling new wells. Permits usually undergo an administrative and a technical review. Agency implementation of institutional controls would be most effective at the technical review level. Tribal agencies and local authorities must agree to the technical criteria for implementing institutional control of groundwater use.

After obtaining signature approval from the director of Water Resource Management, a permit is sent to the Navajo Department of Justice to determine any jurisdictional issues or problems with water rights. The permit is then reviewed

by the Executive Director of the Division of Natural Resources to determine the potential for regional, political, or social concerns or impacts.

Establishing interim institutional controls for the protection of human health and the environment will require a consensus among Navajo Nation governmental agencies and local Navajo governing authorities and chapter houses. It would be most effective if local authorities would be responsible for monitoring new wells to ensure that they have been approved. The governing authorities will also need to be informed of monitoring results and the anticipated duration of contamination problems. Chapter houses would be the most effective organizations for educating local residents about potential risks and the necessity of access restrictions.

Although the presence of a public water supply system largely reduces the likelihood of a well being installed at this site, interim institutional controls are needed to restrict access to contaminated water, since short-term exposures could be lethal.

8.4 FUTURE SITE ACTIVITIES

The terrace at the Shiprock site is currently under long-term surveillance and monitoring of the mill tailings disposal cell.

For the groundwater phase, monitoring the floodplain groundwater and potential surface expression points will continue until the detailed characterization of the site groundwater is completed.

The San Juan River water will also continue to be monitored to assess the potential impact of contaminated groundwater from the floodplain on the river water quality.

To resolve outstanding questions with respect to the terrace area at the Shiprock site, better hydrologic and groundwater chemistry characteristics are required (new wells, pump tests, etc).

Additional sampling will be conducted to better evaluate the potential for ecological impact. These sampling efforts would further evaluate the potential for surface water, sediment, and biota contamination. If these results indicate a significant potential for adverse effects, additional studies might include toxicity tests, quantitative investigations, or biomonitoring.

8.5 RECOMMENDATIONS

In general, the groundwater standards of MCLs or background concentrations are sufficient to protect human health and the environment. However, in some cases, a risk assessment may identify site-specific factors that suggest that these standards may be either too restrictive or not restrictive enough. In the case of standards that are too restrictive, there may be no potential for exposure, and a less restrictive alternate concentration limit (ACL) may be sought. In other cases,

the standards may not be sufficiently protective (for example, if there are many contaminants near the MCL with additive or synergistic adverse health effects).

For Shiprock, there is no permanent physical barrier to prevent access to contaminated groundwater at the former processing site. Therefore, ACLs could not be justified for those constituents with MCLs. However, for those constituents that exceed background and do not have MCLs, this assessment suggests that background levels are more restrictive than necessary. This includes contaminants that were screened on the basis of their concentrations falling within nutritional levels (e.g., zinc) and other contaminants, such as strontium, that were demonstrated to be at concentrations well below adverse health effects levels. ACLs should be sought for these contaminants.

The nitrate MCL of 44 mg/L (10 mg/L as nitrogen) may not be sufficiently protective for infants where sulfate concentrations are also high. Further study of this potential synergism is needed before remedial action decisions are made for this site.

The levels of nitrate present a serious health risk if the floodplain groundwater at this site is used. The levels in monitor wells 908 and 915 substantially exceed levels at which fatal cases of methemoglobinemia have been reported elsewhere in infants. This situation requires that institutional controls be implemented as soon as possible, regardless of which remedial action strategy is pursued.

It is recommended that an additional round of surface water and sediment samples be collected from the San Juan River and the floodplain surface water bodies during a low flow period in the San Juan River. These data would provide information on the contribution of groundwater to these water bodies.

It is also recommended that samples of plant and animal tissue be collected from the floodplain and from a background location. These data would be useful for assessing bioaccumulation and the potential for food chain transfer.

The terrace groundwater hydrology, background groundwater chemistry, and the areal and vertical extent of contamination require more characterization. Existing data are not sufficient for identifying mill-related potential contaminants and receptors or for precisely evaluating potential risks associated with the future uses of the terrace groundwater. However, the yield of the aquifer seems to be very low, and it is questionable whether groundwater can be used as a drinking water resource. Other future beneficial uses of the terrace groundwater (for example, for livestock watering or crop irrigation) should be evaluated.

9.0 LIST OF CONTRIBUTORS

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