1 of 2

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REV. 1

**BASELINE RISK ASSESSMENT OF
GROUND WATER CONTAMINATION
AT THE MONUMENT VALLEY
URANIUM MILL TAILINGS SITE,
CANE VALLEY, ARIZONA**

August 1994

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AT THE MONUMENT VALLEY URANIUM MILL TAILINGS SITE
CANE VALLEY, ARIZONA**

August 1994

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

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CITIZENS' SUMMARY

This baseline risk assessment evaluates the potential impact to public health or the environment from ground water contamination at the former uranium mill processing site in Cane Valley near Monument Valley, Arizona. The U.S. Department of Energy (DOE) Uranium Mill Tailings Remedial Action (UMTRA) Project has relocated and stabilized this site's tailings and other contaminated material in a disposal cell at Mexican Hat, Utah. The second phase of the UMTRA Project is to evaluate ground water contamination. This risk assessment is the first document specific to this site for the Ground Water Project that evaluates potential health and environmental risks. It will help determine the approach required to address contaminated ground water at the site.

Currently, no domestic drinking water wells exist in the contaminated ground water plume. Because the affected water is not used for drinking, no current health or environmental risks are associated with the currently defined contaminated ground water plume. However, because ground water contamination extends nearly 1 mile from the former processing site and ground water in that vicinity is used for drinking, this risk assessment evaluates potential future use of the contaminated ground water. Recent data indicated elevated levels of uranium, sulfate, and other plume contaminants in water slightly below ground surface in Cane Valley Wash. Levels are elevated downgradient from the frog ponds and the site and, to a lesser extent, in water from a hand-dug domestic well crossgradient from the site. However, the pattern of elevated levels of constituents is not entirely consistent with the contamination in site ground water. For example, nitrate, a major plume constituent, is not detectable in these samples. Because the precise source of this contamination has not yet been determined, risks associated with using this water are evaluated separately from those associated with the site ground water plume. A plan to determine the source of elevated constituents in Cane Valley Wash is being developed. Surveys indicated the residents are not using the hand-dug well as a drinking water source due to the unpalatable taste.

This risk assessment follows the basic approach outlined by the U.S. Environmental Protection Agency. The first step is to evaluate ground water data collected from monitor wells at the site over the past 5 years. These data show manganese, nitrate, sulfate, strontium, vanadium, and uranium as the main contaminants associated with the site.

The next step in the risk assessment is to estimate how much of these contaminants people would be exposed to if a drinking water well were installed in the contaminated ground water. Because the contaminant concentrations vary each time a well is sampled and because people vary in the amount they weigh and drink, this risk assessment uses probability distributions to reflect these differences when estimating how much of each contaminant would likely be ingested if a well were at the site. The probability distributions determine the likelihood of an event. For example, based on population survey data, probability distributions can reflect the percentage of people who drink a half gallon of water daily and the percentage who drink only 1 cup of water daily.

The estimated amounts of contaminants that could be ingested in drinking water are then compared to the potential toxic effects of these contaminant levels. Nitrate is the most

Livestock drink water from the frog ponds east of the site. These ponds are fed from a portion of the unconsolidated aquifer that is not affected by the contaminant plume. An assessment of the water quality in these ponds indicates no adverse effects to livestock or vegetation would occur from exposure to this water.

The potential exists for future surface expression as the contaminant plume moves to the northeast. The elevated levels of contaminants associated with the plume in samples of near-surface water in Cane Valley Wash suggest the plume already may have intersected the wash. However, deeper monitor wells close to Cane Valley Wash currently do not show contamination, and the absence of nitrate and other geochemical markers of the plume means additional information will be necessary to determine whether the plume is the source of this contamination. Plans for additional sampling are being developed to determine the source of these elevated levels of contaminants.

Monitor wells were sampled routinely as part of the Monument Valley remedial action (the relocation of tailings to Mexican Hat). The Monument Valley site will be further evaluated during the UMTRA Project's ground water activities. This evaluation is under way and includes further characterization of the ground water and its movement. Based on these results and this risk assessment, an approach will be proposed for managing the contaminated ground water. The overall ground water decision-making framework will be presented in a programmatic environmental impact statement, and site-specific remedial action decisions will be presented in an environmental assessment. Both of these evaluations will include public and tribal involvement. In the interim, control of access to contaminated ground water is being pursued.

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LIST OF ACRONYMS AND ABBREVIATIONS

<u>Acronym</u>	<u>Definition</u>
ac	acre
ACL	alternate concentration limit
BIA	Bureau of Indian Affairs
cm	centimeter
cm ²	square centimeter
DHHS	Department of Health and Human Services
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
ft	foot
FWQC	Federal Water Quality Criteria
FWS	Fish and Wildlife Service
g/L	grams per liter
ha	hectare
HEAST	Health Effects Assessment Summary Tables
IRIS	Integrated Risk Information System
kg	kilogram
km	kilometer
km ²	square kilometers
L/cm ³	liters per cubic centimeter
L/kg	liters per kilogram
LOAEL	lowest-observed-adverse-effect level
μg	microgram
μg/kg	micrograms per kilogram
μg/L	micrograms per liter
μmhos/cm	micromhos per centimeter
m	meter
MCL	maximum concentration limit
mg/kg	milligrams per kilogram
mg/kg-day	milligrams per kilogram per day
mg/L	milligrams per liter
mi	mile
mV	millivolt
ng	nanogram
ng/L	nanogram per liter
NOAEL	no-observed-adverse-effect level
pCi	picocurie
pCi/L	picocuries per liter
PEIS	programmatic environmental impact statement
RfD	reference dose
SAR	sodium adsorption ratio
SF	slope factor
TDS	total dissolved solids
UCL	upper confidence limit

LIST OF ACRONYMS AND ABBREVIATIONS (Concluded)

<u>Acronym</u>	<u>Definition</u>
UMTRA	Uranium Mill Tailings Remedial Action
UMTRCA	Uranium Mill Tailings Radiation Control Act
USGS	U.S. Geological Service

1.0 INTRODUCTION

The purpose of this baseline risk assessment is to determine whether ground water contamination at the Monument Valley Uranium Mill Tailings Remedial Action (UMTRA) Project site in Cane Valley, Arizona, can adversely affect public health or the environment. The Monument Valley site is one of 24 abandoned uranium mill tailings sites undergoing remediation in accordance with the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA) of 1978 (42 USC §7901 *et seq.*) under the oversight of the U.S. Department of Energy (DOE).

The first phase of the UMTRA Project at the Monument Valley site was to relocate and stabilize the tailings in a disposal cell that minimizes 1) radon and other radiation emissions and 2) further contamination of ground water. The tailings and other contaminated materials were transported 17 miles (mi) (27 kilometers [km]) north to the Mexican Hat, Utah, disposal cell for stabilization.

Next, ground water contamination will be evaluated to determine if any remedial action is required and, if so, to implement such action. The DOE was authorized to perform ground water restoration in the 1988 amendments to UMTRCA.

A programmatic environmental impact statement (PEIS) for ground water contamination at the UMTRA Project sites (DOE, 1994) describes the evaluation strategy and remediation methodology for contaminated ground water. This baseline risk assessment is the first site-specific document prepared to evaluate potential health and environmental risks and to provide information to help determine the remedial action, if any, required. Based on the PEIS, additional ground water characterization, and this risk assessment, a site-specific environmental assessment will be prepared to evaluate and select remedial action for the Monument Valley site.

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

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

This baseline risk assessment follows the basic framework outlined by the U.S. Environmental Protection Agency (EPA) for evaluating hazardous waste sites (EPA, 1989). This assessment includes the following steps:

- Data evaluation.
 - Combining existing 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 exposure settings.
 - Identifying exposure pathways.
 - Quantifying the exposure.
- Toxicity assessment.
 - Identifying toxicity values.
 - Evaluating noncarcinogenic effects.
 - Evaluating carcinogenic effects from radionuclides and chemical carcinogens.
- Public health risk characterization.
 - Comparing toxicity ranges with predicted exposure ranges.
 - Combining risks across exposure pathways and multiple contaminants.
 - Characterizing uncertainties.
- Environmental risk.
 - Characterizing potential biota exposure pathways.
 - Identifying the potential ecological receptors.
 - Evaluating the environmental risk qualitatively.

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

2.0 SITE DESCRIPTION

The Monument Valley site is on the Navajo Reservation in northern Arizona, 13 mi (22 km) east of the Monument Valley Navajo Tribal Park in a remote area near Cane Valley (Figure 2.1). The site is on Bureau of Indian Affairs (BIA) Navajo Service Road 6440, 17 road mi (27 km) south of the Mexican Hat disposal site. Two uranium mill tailings piles were at the base of Yazzie Mesa at approximately 4900 feet (ft) (1500 meters [m]) above mean sea level (Figure 2.2). Several year-round residences are within 0.5 mi (0.8 km) of the site. At least three points of surface-water expression (seeps) have been identified in the valley.

2.1 SITE BACKGROUND

The mill at Monument Valley operated from the summer of 1955 until 1968. From 1955 until 1964, the site consisted of an upgrader plant that produced a slime concentrate, which was trucked to mills in Durango, Colorado, and Shiprock, New Mexico, leaving behind sand tailings that formed the majority of the lower tailings pile. In 1964, the upgrader was replaced by a concentrator and batch leach facility. Sand tailings were leached in a batch process with sulfuric acid. Low-grade ores were heap-leached with sulfuric acid in the region of the upper tailings pile. The acid-leach solutions from both operations were adjusted to pH 4 with ammonia and then to pH 7 with lime.

The tailings piles have been moved to the Mexican Hat disposal cell as part of the surface remediation at the site. The upper tailings pile was in an ephemeral drainage channel flowing into Cane Valley; the pile covered approximately 11 acres (ac) (4 hectare [ha]) and had an average thickness of 11 ft (3 m). The lower pile covered 17 ac (7 ha) and had an average thickness of 24 ft (7 m). The total tailings volume in the two piles was approximately 842,600 cubic yards) (644,300 cubic meters). As a result of milling operations, an additional 55 ac (22 ha) at the site showed surface contamination. These areas included an evaporation pond, an ore storage/mill yard, a batch leach yard, and areas contaminated by windblown tailings.

2.2 CLIMATE

The Monument Valley tailings area is arid, with less than 6 inches (15 centimeters [cm]) of precipitation relatively evenly distributed throughout the year (although somewhat greater during the July through October thunderstorm period characteristic of the Colorado plateau). Snowfall statistics are not available for Monument Valley, but Mexican Hat, 17 road mi (27 km) north, had an annual average of 3.3 inches (8.4 cm) for the 20-year period ending in 1980 (Stevens et al., 1983).

Temperatures can vary considerably within 24 hours and across seasons. Winters are cold, with temperatures below freezing from November through March. Summers are hot, with highs from the 90s to the low 100s degrees Fahrenheit (approximately 32 to 40 degrees Celsius).

FIGURE 2.1
LOCATION OF THE MONUMENT VALLEY UMTRA PROJECT SITE
IN CANE VALLEY, ARIZONA

2.3 HYDROGEOLOGY

2.3.1 Physiographic setting

The tailings site is on the west side of Cane Valley, a northward draining strike valley eroded into the Chinle Formation. The valley floor elevation is approximately 4800 ft (1500 m) along Cane Valley Wash east of the tailings pile (Figure 2.3). The valley is bordered on the east by Comb Ridge (Figure 2.2), a 600-ft (200-m)-high escarpment of Navajo, Kayenta, and Wingate Sandstones. On the west side of the valley (where the tailings were located), the bedrock dips east at approximately 5 degrees, rising up to Yazzie Mesa at an elevation of more than 5300 ft (1600 m) (Figures 2.4 and 2.5). The valley between these bedrock outcrops is filled with windblown (eolian) sand and some transported gravel and rock fragments.

2.3.2 Stratigraphy

The major hydrostratigraphic formations of concern underlying the Monument Valley UMTRA site are, in descending order, the unconsolidated eolian and alluvial deposits (hereafter referred to as the unconsolidated aquifer), the Shinarump Member of the Chinle Formation (called the Shinarump Conglomerate), the Moenkopi Formation, and the De Chelly Sandstone of the Cutler Formation (Witkind and Thaden, 1963). Figure 2.5 shows the Shinarump Conglomerate, the Moenkopi Formation, and the De Chelly Sandstone in an outcrop on the eastern flank of Yazzie Mesa west of the tailings pile. Figure 2.6 depicts a cross section beneath the site at the location shown in Figure 2.3.

The dune sands are well-sorted, windblown deposits of fine- to medium-grained sands. The unconsolidated aquifer also includes sands and gravels and transported fragments of bedrock. This unconsolidated aquifer is more than 80 ft (24 m) thick in the center of Cane Valley, feathering out on the dipping bedrock along the western side of the valley. The western side of the lower tailings pile rests on the bedrock while the eastern side is on more than 50 ft (15 m) of the unconsolidated aquifer, as shown on the cross section in Figure 2.6.

The Shinarump Conglomerate is a geologic member of the Chinle Formation consisting of lenticular, cross-bedded formations of sandstone and conglomerate with occasional thin mudstone layers. It is 20 to 85 ft (6 to 26 m) thick in the site vicinity. The Shinarump Conglomerate directly underlies the upper tailings pile and occurs below the unconsolidated aquifer under the lower tailings pile.

The Moenkopi Formation, a shaley siltstone and sandstone, underlies the Shinarump Conglomerate. It is 50 to 60 ft (15 to 20 m) thick beneath most of site, but thins to the west to only about 20 ft (6 m) beneath the upper pile. The Moenkopi Formation acts as an aquitard between the Shinarump Conglomerate and the underlying De Chelly Sandstone Member of the Cutler Formation.

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FIGURE 2.5
OUTCROP WEST OF TAILINGS PILES SHOWING SHINARUMP CONGLOMERATE,
MOENKOPF FORMATION, AND DE CHELLY SANDSTONE

FIGURE 2.4
VIEW TO NORTHWEST SHOWING LOWER TAILINGS PILE,
LINED EVAPORATION POND, BUFF-COLORED SHINARUMP CONGLOMERATE,
AND PINK DE CHELLY SANDSTONE (TELEPHOTO SHOT WITH 150-mm LENS)

LEGEND

- SANDS, GRAVEL, AND BEDROCK FRAGMENTS
- SANDSTONE AND CONGLOMERATE WITH OCCASIONAL THIN MUDSTONE LAYERS
- SHALEY SILTSTONE AND SANDSTONE
- FINE-GRAINED SANDSTONE

NOTE: SEE FIGURE 2.3 FOR CROSS SECTION LOCATION.

FIGURE 2.6
GEOLOGIC CROSS SECTION X - X'
MONUMENT VALLEY UMTRA PROJECT SITE

The De Chelly Sandstone is a massive, cross-bedded, fine-grained sandstone, approximately 500 ft (150 m) thick beneath the site. It is the lowest aquifer beneath the Monument Valley tailings site and is underlain by the low permeability Organ Rock Shale.

2.3.3 Ground water presence and movement

Unconsolidated aquifer

Ground water within the unconsolidated aquifer is unconfined. Depths to ground water range from the land surface near Cane Valley Wash to slightly more than 10 ft (3 m) below the lower tailings pile. Precipitation and upgradient underflow recharge this unconfined ground water. Some leakage may occur upwards from the deeper confined portions of the Shinarump Conglomerate and De Chelly Sandstone through the confining beds. Discharges from flowing artesian monitor wells 611, 613, and 615 and from former production well 625 completed in the De Chelly Sandstone also recharge the unconsolidated aquifer. Figure 2.3 shows the locations of the wells discussed in this report.

East of the tailings area, on the opposite side of Cane Valley Wash (Figure 2.3), a spring-fed pond may recharge the unconsolidated aquifer in the lower part of the valley.

Figure 2.7 is a water table contour map for the unconsolidated aquifer. Ground water in the area of the site flows north-northeast under an average hydraulic gradient of 0.01. This water moves down the valley as underflow, with some discharging to Cane Valley Wash, the local discharge area.

The piezometric head in one of the monitor wells (654) screened in the unconsolidated aquifer is approximately 1 ft (0.3 m) above land surface even though the ground around the well is dry. This elevated water level indicates that the well screen at 34 to 58 ft (10 to 18 m) below land surface encountered water with a higher piezometric head than the water table. In an unconfined aquifer, this occurrence means the ground water flow lines curve upward. At this location, the ground water rises to join the surface flow in Cane Valley Wash, the local discharge area.

The same ground water flow pattern is anticipated on the east side of Cane Valley Wash and is probably the source of the spring mentioned above. These opposing flow regimes will prevent water on one side of the valley from crossing to the opposite side. The water elevation from well 654 is not included in the water table contour map because it is not thought to be representative of the shallow water table flow regime.

The hydraulic conductivity computed from aquifer pumping tests in the unconsolidated aquifer was 19 ft per day (6 m per day). The linear ground water velocity in the unconsolidated aquifer was calculated using a hydraulic gradient of 0.01 and an assumed effective porosity of 0.25. Ground water

FIGURE 2.7
WATER TABLE CONTOURS IN UNCONSOLIDATED AQUIFER
MONUMENT VALLEY UMTRA PROJECT SITE

velocity in the unconsolidated aquifer is approximately 300 ft per year (90 m per year). Appendix F of the Monument Valley and Mexican Hat remedial action plan (DOE, 1993) includes all calculations.

Shinarump Conglomerate

Ground water within the Shinarump Conglomerate occurs under both unconfined and confined conditions. Water entering the Shinarump through outcrops along the west side of the valley will be unconfined and move downdip to the east. This water may become confined moving beneath the sandstone layers. The infiltrating water joins other ground water moving from farther south or that may have leaked from the De Chelly Sandstone below the Moenkopi Formation.

Water level measurements at the site indicate the ground water moves north under approximately the same average hydraulic gradient (0.01) as in the unconsolidated aquifer. Figure 2.8 is a potentiometric surface map for the Shinarump Conglomerate.

Hydraulic conductivities computed from four slug tests in the Shinarump Conglomerate varied from 0.4 to 8 ft per day (0.1 to 2 m per day). The ground water velocities were calculated using a hydraulic gradient of 0.01 and an assumed effective porosity of 0.1. Linear velocities for ground water flow range from 15 to 300 ft per year (5 to 90 m per year).

De Chelly Sandstone

Ground water within the De Chelly Sandstone is generally confined by the overlying Moenkopi Formation. This confined ground water flows northward from recharge areas to the south and west. The ground water in the De Chelly Sandstone also flows under an average hydraulic gradient of approximately 0.01 to the north-northeast. Figure 2.9 is a potentiometric surface map for the De Chelly Sandstone.

The hydraulic conductivity computed from an aquifer pumping test in the De Chelly Sandstone was 6 ft per day (2 m per day). Using a hydraulic gradient of 0.01 and an assumed effective porosity of 0.1, the average linear ground water velocity is 200 ft per year (60 m per year) (DOE, 1993).

Vertical gradients

An upward, vertical hydraulic gradient exists between the De Chelly Sandstone and the overlying aquifers. In some cases, there is also evidence of an upward gradient between the Shinarump and the overlying unconsolidated aquifer (Figures 2.7, 2.8, and 2.9).

FIGURE 2.8
POTENTIOMETRIC SURFACE FOR THE SHINARUMP CONGLOMERATE
MONUMENT VALLEY UMTA PROJECT SITE

FIGURE 2.9
POTENTIOMETRIC SURFACE FOR THE DE CHELLY SANDSTONE
MONUMENT VALLEY UMTA PROJECT SITE

2.4 SURFACE WATER REGIME

Surface water features in the Monument Valley site vicinity include Cane Valley Wash and several small ephemeral drainage channels (arroyos), as shown in Figure 2.10. Cane Valley Wash has a drainage area approximately 90 square miles (230 square kilometers) (57,000 ac [23,000 ha]) south of the tailings site. East and South Arroyos drain about 710 ac (290 ha) south of the tailings site. These arroyos join approximately 300 ft (90 m) east of the lower tailings pile and continue northward into the wash. West Arroyo drains a 1120-ac (453-ha) watershed on Yazzie Mesa southwest of the tailings pile. The upper tailings pile and heap-leach pad area were in this arroyo.

Discharge from a spring on the east side of Cane Valley Wash supplements surface water flow in the wash, as shown in Figure 2.11. Spring water flows out of an excavation in the unconsolidated aquifer into a rectangular-shaped excavation and then into Cane Valley Wash. From there, the water flows into a series of excavated ponds and depressions on the floodplain of Cane Valley Wash, known locally as the "frog ponds." Figure 2.12 shows these ponds and the lush vegetation indicating high water table conditions in the bottom of Cane Valley downstream of the spring. The U.S. Geological Survey (USGS) topographical maps (Figure 2.3) note another series of springs feeding into Cane Valley Wash, approximately 2 mi (3.2 km) north of the site near sampling location 620. In wet years, the meadow surrounding Cane Valley Wash continues for several miles, providing pasture for cattle and sheep and a watering and nesting area for waterfowl and other wildlife.

Of particular importance to this baseline risk assessment is that the spring east of the site is located on the opposite side of Cane Valley Wash from the tailings pile at a higher elevation than the bottom of the wash, as shown on the topographic map (Figure 2.3) and the photograph (Figure 2.11). As discussed earlier, this spring is probably fed by ground water moving up out of the unconsolidated aquifer from the east and south. Therefore, ground water contaminants associated with the former milling operations should not affect the quality of the spring water. Also important is the probability that the spring contributes significantly more flow to surface water in Cane Valley Wash than does ground water. Therefore, the uncontaminated spring outflow should dilute any contaminated ground water that may reach the surface stream from the west, or tailings side, of the valley.

2.5 LAND USE

Individual members from the Navajo Nation do not literally own land; several systems of land tenure maintain land use rights. The Navajo system for land assignment consists of grazing permits. The grazing permit system developed in the 1940s assigns land based on sheep units. A Navajo cannot establish a residence without a grazing permit. These permits are usually passed down or subdivided among family members. The BIA oversees permit registration.

FIGURE 2.10
SURFACE WATER FEATURES
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 2.11
VIEW TO SOUTHEAST SHOWING LOCATION OF SPRING AT ARROW BY LIGHT GREEN TREE. NOTE ELEVATION OF SPRING ABOVE BOTTOM OF CANE VALLEY

FIGURE 2.12
VIEW TO NORTH DOWN CANE VALLEY SHOWING EXCAVATED PONDS FED BY SPRING WATER. NOTE LUSH VEGETATION INDICATING HIGH WATER TABLE CONDITIONS

Figure 2.2 shows the location of area residences. One occasional and six year-round residences are within a 1-mi (1.6-km) radius of the site. Two of the year-round residences lie due south of the tailings pile, and four residences are along Cane Valley Wash, east and slightly south of the pile. The remaining full-time residence is adjacent to the site at the northeast corner. Three additional residences are located along BIA service road 6440 within 2 mi (3.2 km) north of the site.

Most residents of the valley maintain small herds of cattle, goats, and/or sheep. These animals range freely throughout the valley, but the primary grazing area is a meadow in Cane Valley. The meadow is supported by the frog-pond drainage running north through the valley for several miles (see Figure 2.12). Several residents also maintain vegetable gardens consisting primarily of corn, squash, and melons.

2.6 WATER USE

Because of the limited and highly variable surface water supply in the area, ground water is an important resource. The Indian Health Service (IHS) funds and arranges installation of wells and water systems on the Navajo Reservation.

Homes in the Monument Valley vicinity do not have plumbing, water, or sewage systems. Domestic water use encompasses drinking, bathing, and watering vegetable gardens and domestic pets. Livestock primarily drink surface water, as described in later sections.

In addition to DOE-placed monitor wells, five points of ground water withdrawal are known to exist within a 2.5-mi (4.0-km) radius of the pile (Figure 2.2):

- An IHS hand-pump well at each of the two residences due south of the pile.
- One hand-dug, hand-pump well southeast of the site along Cane Valley Wash.
- A windmill-pump well at a residence 2 mi (3.2 km) north of the site.
- An IHS well in Cane Valley Wash, approximately 2.5 mi (4.0 km) north and slightly east of the site.

Several families in the valley use the hand-pump wells at the residences south of the pile as a domestic water supply. These wells (616 and 617) have been monitored regularly as part of the UMTRA Project ground water monitoring effort at the site. Their exact completion depth is unknown. Water quality and geochemistry in wells 616 and 617 cannot be distinguished from upgradient water samples from either the unconsolidated or De Chelly aquifers.

The hand-pump well southeast of the site in Cane Valley Wash is not in use due to the bitter quality of the water. It is being monitored as well 640. Surveys of

area residents conducted in 1992 and 1993 indicated that residents in this well's vicinity carry their water from either well 616 or 617 or use water flowing from artesian monitor well 613, completed in the De Chelly Formation.

The residents at the northeast corner of the site use water from a flowing artesian well (625) that was a former production well for the milling operation. This well also is completed in the De Chelly Formation and is monitored regularly by the UMTRA Project as well 625. No contamination has been observed in this well.

Residents living north of the site along BIA road 6440 obtain their water from the windmill-pump well which also appears to be completed in the De Chelly Formation. This well also is used to fill a stock pond located on the same property.

Livestock drink water pooling around artesian monitor wells 613 and 625 and from the frog ponds and the ephemeral stream in Cane Valley Wash. USGS maps reveal at least one additional spring approximately 2 mi (3.2 km) north of the site, east of road 6440 near surface sampling location 620, which also drains to Cane Valley Wash. During periods of drought, residents dig out springs in the valley to water their stock. This has occurred both in the vicinity of the frog ponds and in the spring region north of the site around sampling location 620. While only these two instances have been verified, it is likely that this ground water access also occurs in other locations.

3.0 EXTENT OF CONTAMINATION

Figure 2.3 shows the locations of monitor wells for the unconsolidated aquifer, Shinarump Conglomerate, and De Chelly Sandstone. Table 3.1 gives the screened intervals within each of these wells and the number of samples obtained since 1988. In this risk assessment, only samples since 1988 were used to assess water quality. This decision is based on 1) sufficient number of samples for that period in wells used for evaluation and 2) improved analytical detection limits and quality control procedures for that period. Table 3.1 indicates, for each aquifer, which wells were used in this document to define background water quality and which wells were used to evaluate potential health and ecological risks. These latter wells were selected because they consistently showed higher levels of key contaminants from 1988 through 1992 than the other monitor wells screening the same aquifer. Ground water has not been screened for organic contaminants. However, a solvent extraction process to recover uranium from the pregnant solution was used at the site. Additional sampling and analyses would be needed to completely rule out the possibility of organic contamination.

3.1 BACKGROUND WATER QUALITY DATA SUMMARY

Background water quality is defined as the quality of water that would be present had uranium processing activities not occurred. Background ground water quality in the unconsolidated aquifer was evaluated by analyzing water samples collected from upgradient monitor wells 602 and 603 and from domestic wells 616 and 617 (Figures 2.2 and 2.3). This water is a sodium bicarbonate type with total dissolved solids (TDS) of 406 ± 43 milligrams per liter (mg/L) and pH of 7.73 ± 0.23 . Background ground water quality in the Shinarump Conglomerate was evaluated by analyzing water samples collected from upgradient wells 601 and 615 (Figures 2.3 and 2.8). This water is similar in composition to that of the unconsolidated aquifer and is also a sodium bicarbonate type with TDS of 382 ± 54 mg/L and pH of 7.77 ± 0.63 . Background ground water quality in the De Chelly Sandstone was evaluated by analyzing water samples collected from upgradient wells 612 and 613 (Figures 2.3 and 2.9). This water is a mixed sodium-calcium to sodium-magnesium bicarbonate type with TDS ranging from 118 to 321 mg/L and pH ranging from 7.37 to 8.46. All the background waters show oxidizing conditions (Eh greater than 400 millivolts [mV]).

3.2 MAGNITUDE OF SITE-RELATED GROUND WATER CONTAMINATION

The main contaminants (in terms of mass) resulting from milling operations at the site are sulfate and nitrate. Under the geochemical conditions found in the shallow ground water, sulfate travels essentially at the same rate as ground water. Therefore, sulfate can be used to define the extent of the contaminant plume originating from the site.

Table 3.1 Ground water monitor well network at tailings sites

Well no.	Screened interval (depth below surface)				No. samples since 1988
	(ft)		(m)		
Unconsolidated aquifer					
Upgradient					
602 ^a	33	53	10	16	7
616 ^a	--b	--b	--b	--b	5
617 ^a	--b	--b	--b	--b	4
Crossgradient					
603 ^a	33	53	10	16	6
604	13	28	4	8	6
605	14	29	4	9	8
654	57	77	17	23	9
Downgradient					
606 ^c	32	42	10	13	9
650	78	98	24	30	2
651	20	80	6	24	9
652	34	54	10	16	5
653	56	76	17	23	9
655 ^c	38	58	12	18	10
656	38	58	12	18	3
662	35.5	67.5	11	20	5
669	34	54	10	16	2
Shinarump aquifer					
Upgradient					
601 ^a	12	22	4	7	4
610	63	83	19	25	--d
658	135	155	41	47	--d
Crossgradient					
615 ^a	68	88	21	27	2
Downgradient					
607	12.5	22.5	4	7	--d
609	7	12	2	4	--d
614 ^c	48	68	15	21	8
659 ^c	87	107	27	33	9
660	133	153	41	47	6

Table 3.1 Ground water monitor well network at tailings sites (Concluded)

Well no.	Screened interval (depth below surface)				No. samples since 1988
	(ft)		(m)		
De Chelly aquifer					
Upgradient					
612 ^a	175	195	53	59	1
613 ^a	138	158	42	48	8
661	190	210	58	64	-- ^e
Crossgradient					
611 ^a	163	183	50	56	1
625 ^a	-- ^b				3
Downgradient					
608	98	118	30	36	-- ^e
618	-- ^b	--	--	--	-- ^e
619 ^c	-- ^b	--	--	--	-- ^e
657 ^d	121	136	37	41	6
663 ^d	175	215	53	66	9
664	211	231	64	70	-- ^e
668	180	200	55	61	1

^aWells used to establish background water quality for purposes of risk assessment.

^bWells are either domestic or production wells for which log data are not available. Based on water quality analysis, wells are assumed to be completed in the given aquifers.

^cWell 619 is an open borehole with no casing and therefore is likely to draw water from multiple aquifers.

^dWells with highest contaminant concentrations used in modeling potential exposures for purposes of risk assessment.

^eNo data available for these wells after 1988 due to changes made to sampling frequency and/or inadequate ground water flow.

Plume definition

The general shape of the plume has been defined by contouring sulfate concentrations in ground water downgradient from the site. Figure 3.1 is the resulting isopleth map. Only a single plume is depicted emanating from the tailings because there are not enough sampling points to distinguish contributions from the upper and lower tailings piles. The second, smaller plume depicted around well 605 is not assumed to be site-related.

The precise periphery of the plume is difficult to define solely on the basis of sulfate concentrations. This is evident in well 605, which has high sulfate, suggesting sulfate is in the plume, but does not have elevated concentrations of other site-related contaminants (e.g., nitrate). A better way to define the plume periphery is through chloride versus sulfate plots. Both of these analytes are conservative anions in the peripheral wells and, by definition, travel at essentially the same rate in ground water. Figure 3.2 shows that nearly all background ground waters have chloride:sulfate ($\text{Cl}:\text{SO}_4$) ratios greater than 1:10. Wells within the plume have $\text{Cl}:\text{SO}_4$ ratios inversely proportional to their degree of contamination. For example, in the center of the plume, well 655 has a $\text{Cl}:\text{SO}_4$ ratio near 1:100 (Figure 3.3). Therefore, the 1:10 ratio provides a reasonable discriminator between the plume and background. The peripheral wells in the unconsolidated aquifer (656, 662, and 669) have $\text{Cl}:\text{SO}_4$ ratios less than 1:10, suggesting they are within the plume (Figure 3.3). Wells 606 and 653 are intermediate in contaminant levels, falling midway between the central and peripheral wells in their $\text{Cl}:\text{SO}_4$ ratios. Although well 605 has relatively high sulfate concentrations, the $\text{Cl}:\text{SO}_4$ ratio is in the background values range (Figure 3.4), suggesting the high sulfate in this well is not due to contamination from the site. It may reflect dissolution of evaporite (e.g., playa lake) deposits in the unconsolidated aquifer. Downgradient unconsolidated aquifer wells 650, 651, 652, and 654 generally have $\text{Cl}:\text{SO}_4$ ratios greater than 1:10 and are not impacted by the plume (Figure 3.4).

Based on $\text{Cl}:\text{SO}_4$ ratios, ground waters from Shinarump wells 614 and 659 and deeper well 657 may also contain a small plume component, as shown in Figure 3.5. Well 657 was originally drilled as a De Chelly monitor well. However, the driller's log indicates that the sandpack brackets the unconsolidated aquifer and De Chelly Sandstones in a region where the Shinarump Conglomerate and Moenkopi Formation are absent. Therefore, it is difficult to determine how representative this well is of the De Chelly Sandstone.

Figures 3.6 to 3.9 present the observed concentrations of nitrate, uranium, strontium, and sulfate for background and plume wells in the unconsolidated aquifer and for corresponding wells in the deeper zones. These constituents associated with the plume are slightly but consistently elevated above background in wells 614 and 657 (Figures 3.6 to 3.9). Well 659 does not show elevated concentrations of these constituents. $\text{Cl}:\text{SO}_4$ ratios less than 1:10, combined with concentrations of the four main plume contaminants over background levels, reinforce the conclusion that some site-related contamination

FIGURE 3.2
CHLORIDE:SULFATE RATIOS IN BACKGROUND WELLS IN THE UNCONSOLIDATED AQUIFER
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 3.4
CHLORIDE:SULFATE RATIOS IN THE UNCONSOLIDATED AQUIFER
COMPARISON AMONG WELL 605, BACKGROUND WELLS, AND CONTAMINATED WELL 653
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 3.5
CHLORIDE:SULFATE RATIOS FOR SHINARUMP AND DE CHELLY WELLS:
BACKGROUND (601, 613, 615) AND DOWNGRAIDENT (614, 657, 659, 663)
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 3.6

DATA COLLECTED: 1988 - 1993

BOX PLOTS ILLUSTRATING NITRATE CONCENTRATIONS IN DOWNGRADIENT AND BACKGROUND WELLS WITH INCREASING DEPTH OF AQUIFER, MONUMENT VALLEY UMTRA PROJECT SITE

INCREASING DEPTH OF AQUIFER

UNCONSOLIDATED AQUIFER

SHINARUMP AQUIFER

DE CHELLY AQUIFER

DATA COLLECTED: 1988 - 1993

FIGURE 3.7
BOX PLOTS ILLUSTRATING URANIUM CONCENTRATIONS IN DOWNGRADIENT AND
BACKGROUND WELLS WITH INCREASING DEPTH OF AQUIFER
MONUMENT VALLEY UMTRA PROJECT SITE

INCREASING DEPTH OF AQUIFER

UNCONSOLIDATED AQUIFER

SHINARUMP AQUIFER

DE CHELLY AQUIFER

DATA COLLECTED: 1988 - 1993

FIGURE 3.8
BOX PLOTS ILLUSTRATING STRONTIUM CONCENTRATIONS IN DOWNGRAIDENT AND
BACKGROUND WELLS WITH INCREASING DEPTH OF AQUIFER
MONUMENT VALLEY UMTRA PROJECT SITE

DATA COLLECTED: 1988 - 1993

FIGURE 3.9

BOX PLOTS ILLUSTRATING SULFATE CONCENTRATIONS IN DOWNGRAIDENT AND BACKGROUND WELLS WITH INCREASING DEPTH OF AQUIFER, MONUMENT VALLEY UMTRA PROJECT SITE

may be present in these wells (657 and 614). A puzzling aspect of the water quality in these wells is that uranium concentrations actually increase with well depth from 655 to 614 to 657 (Figure 3.7), while other contaminants decrease with depth and show only minor elevations from background. Because there is no geochemical explanation for preferential migration of uranium from the tailings piles through ground water, these elevated levels may reflect the contribution from a nearby uranium ore body in addition to site-related effects, as evidenced by the presence of nitrate.

Figure 3.10 is an isopleth map of nitrate concentrations in unconsolidated aquifer ground water. The figure shows elevated nitrate concentrations at locations up to 3000 ft (900 m) downgradient of the tailings pile. Within the plume, nitrate concentrations range up to 1600 mg/L as nitrate.

Well 619 is an anomalous well that will be discussed and evaluated separately. Although this well is listed as a De Chelly well, field analysis has shown that this former production well is actually an open, uncased borehole. Therefore, samples from this well are likely to reflect water quality from shallower sources in addition to the De Chelly. Furthermore, well 619 was pumped during the tailings removal to provide water for dust control. This heavy pumping may have drawn contaminants from shallower zones into the well vicinity. In fact, moderate increases in nitrate, strontium, and sulfate similar to those seen in the deeper wells 614 and 657 (Figures 3.6, 3.8, and 3.9) were observed in well 619 in June and December 1993. However, levels of uranium fourfold greater than those seen in the contaminant plume and twofold greater than those observed in well 657 (Figure 3.7) were also observed in well 619 in 1993. Levels of uranium observed in 1985 were slightly below those seen in the plume wells and may reflect a contribution of the more contaminated water in the unconsolidated aquifer to the samples. With cessation of pumping, water quality in this well is expected to stabilize at levels different from those observed following heavy usage. Therefore, water quality in this well will be evaluated independently from plume wells where relatively stable patterns of contamination have been observed over a number of years.

3.3 CONTAMINANTS OF CONCERN

Table 3.2 summarizes the data presented in Sections 3.1 and 3.2 for the unconsolidated aquifer. These data were used to compile a list of contaminants of potential concern for the assessment of human health or environmental risks at the Monument Valley site. Table 3.3 presents corresponding data for the Shinarump Conglomerate. In general, a contaminant was placed on the list of contaminants of potential concern (Table 3.4) if it was detected, on average, in excess of background in monitor wells and if the site is a likely source for the contaminant. Although some constituents not identified as exceeding background have either median or maximum values in downgradient wells that appear to exceed background (e.g., arsenic, bromide, and selenium), the difference between background and downgradient data was not large enough or consistent enough to reach statistical significance.

DATA COLLECTED: 11/92

FIGURE 3.10
NITRATE PLUME IN UNCONSOLIDATED GROUND WATER
MONUMENT VALLEY UMTA PROJECT SITE

Table 3.2 Statistical summary of ground water quality in the unconsolidated aquifer at Monument Valley, 1988-1993

Constituent	Number of samples	Detection limit(s)	% above detection	Observed ^a		
				Minimum	Median (mg/L)	Maximum
Aluminum						
Background	13	0.05-0.2	0	-	-	-
Plume	7	0.05-0.2	0	-	-	-
Antimony						
Background	13	0.003-0.06	0	-	-	-
Plume	7	0.003-0.06	0	-	-	-
Ammonium ^b						
Background	12	0.1	25	<0.1	<0.1	0.6
Plume (606)	8	0.1-10	100	200	351	381
Arsenic						
Background	17	0.001-0.01	0	-	-	-
Plume	9	0.001-0.05	11	<0.001	<0.01	0.014-0.05
Barium						
Background	15	0.001-0.2	60	<0.038	0.04-0.1	0.12-0.2
Plume	8	0.001-0.2	38	<0.018	<0.10	0.03-0.2
Beryllium						
Background	9	0.005-0.01	0	-	-	-
Plume	7	0.005-0.01	0	-	-	-
Boron						
Background	5	0.05-0.1	0	-	-	-
Plume	5	0.05-0.1	0	-	-	-
Bromide						
Background	3	0.1	67	<0.1	0.1	0.2
Plume	2	0.1	50	<0.1	0.25-0.30	0.5
Cadmium						
Background	17	0.0001-0.005	12	<0.0001	<0.001	0.005
Plume	9	0.0001-0.005	11	<0.0001	<0.001	0.03

Table 3.2 Statistical summary of ground water quality in the unconsolidated aquifer at Monument Valley, 1988-1993 (Continued)

Constituent	Number of samples	Detection limit(s)	% above detection	Observed ^a		
				Minimum	Median (mg/L)	Maximum
Calcium						
Background	16	0.01-5	100	18	27	37
Plume	9	0.01-5	100	372	411	521
Chloride						
Background	16	0.02-1	100	10	14	17
Plume	9	0.02-1	100	29	33	38
Chromium						
Background	15	0.005-0.01	0	-	-	-
Plume (606)	8	0.005-0.01	0	-	-	-
Cobalt						
Background	5	0.03-0.05	0	-	-	-
Plume	5	0.03-0.05	0	-	-	-
Copper						
Background	5	0.01-0.02	0	-	-	-
Plume	5	0.01-0.02	20	<0.01	<0.02	0.02
Cyanide						
Background	2	0.01	0	-	-	-
Plume	2	0.01	0	-	-	-
Fluoride						
Background	5	0.1	100	0.1	0.2	0.3
Plume	5	0.1	100	0.1	0.1	0.2
Iron						
Background	14	0.03-0.1	0	-	-	-
Plume	8	0.03-0.1	38	<0.03	<0.055	0.09-0.1
Lead						
Background	7	0.001-0.01	0	-	-	-
Plume	6	0.001-0.01	17	<0.001	<0.01	0.01

Table 3.2 Statistical summary of ground water quality in the unconsolidated aquifer at Monument Valley, 1988-1993 (Continued)

Constituent	Number of samples	Detection limit(s)	% above detection	Observed ^a		
				Minimum	Median (mg/L)	Maximum
Magnesium^b						
Background	16	0.001-5	100	13	18	34
Plume	9	0.001-5	100	288	313	350
Manganese^b						
Background	16	0.001-0.01	13	<0.001	<0.01	0.01
Plume	9	0.001-0.01	100	0.04	0.07	0.12
Mercury						
Background	2	0.0002	50	<0.0002	<0.0002	0.0002
Plume	3	0.0002	0	-	-	-
Molybdenum						
Background	17	0.004-0.2	24	<0.004	<0.01	0.02-0.2
Plume	9	0.004-0.2	22	<0.004	<0.01	0.09-0.2
Nickel						
Background	5	0.04	0	-	-	-
Plume	5	0.04	0	-	-	-
Nitrate^b						
Background	18	0.04-1	100	1.0	5.4	44
Plume (606)	10	0.04-150	100	974	1140	1600
Phosphate						
Background	2	0.1	100	0.6	0.6	0.6
Plume	3	0.1	100	0.3	0.4	0.6
Potassium^b						
Background	16	0.01-0.6	100	0.55	1.9	2.6
Plume	9	0.01-5	100	25	30	43
Selenium						
Background	17	0.001-0.005	35	<0.001	<0.005	0.009
Plume	9	0.005-0.015	22	<0.005	<0.005	0.02

Table 3.2 Statistical summary of ground water quality in the unconsolidated aquifer at Monument Valley, 1988-1993 (Continued)

Constituent	Number of samples	Detection limit(s)	% above detection	Observed ^a		
				Minimum	Median (mg/L)	Maximum
Silica^b						
Background	5	0.1-2	100	12	14	26
Plume	5	0.1-2	100	21	21	23
Silver						
Background	5	0.01	0	-	-	-
Plume	5	0.01	0	-	-	-
Sodium^b						
Background	16	0.002-5	100	57	99	112
Plume	9	0.002-5	100	144	165	190
Strontium^b						
Background	13	0.0004-0.2	100	0.22	0.29	0.50
Plume	6	0.0004-0.2	100	2.5	3.0	3.2
Sulfate^b						
Background	18	0.06-20	100	69	120	164
Plume	10	0.1-10	100	2000	2660	3540
Sulfide						
Background	5	0.1-1	0	-	-	-
Plume	4	0.1-1	25	<0.1	<0.1	5.6
Thallium						
Background	5	0.01	0	-	-	-
Plume	5	0.1-1	0	-	-	-
Tin						
Background	5	0.005-0.01	0	-	-	-
Plume	6	0.005-0.03	16	<0.005	<0.008	0.063
Uranium^b						
Background	16	0.001-0.003	88	<0.001	0.004	0.005
Plume	9	0.0003-0.005	100	0.019	0.028	0.031

Table 3.2 Statistical summary of ground water quality in the unconsolidated aquifer at Monument Valley, 1988-1993 (Concluded)

Constituent	Number of samples	Detection limit(s)	% above detection	Observed ^a		
				Minimum	Median (mg/L)	Maximum
Vanadium^b						
Background	15	0.001-0.05	7	<0.001	<0.01	0.02-0.05
Plume	8	0.01-0.05	50	<0.008	0.004-0.015	0.08
Zinc^b						
Background	15	0.001-0.02	13	<0.001	<0.005	0.014-0.02
Plume	8	0.001-0.02	38	<0.005	<0.005	0.05

Radionuclide	Number of samples	Detection limit(s)	% above detection	Minimum	Observed ^a Median	Maximum
					(pCi/L)	
Lead-210						
Background	2	-	-	0.0	0.9	1.7
Plume (606 + 655)	2	-	-	0.3	1.1	1.9
Polonium-210						
Background	Not available	-	-	-	-	-
Plume		-	-	-	-	-
Radium-226						
Background	15	-	-	0.0	0.5	5.7
Plume	8	-	-	0.0	0.1	0.7
Thorium						
Background	3	-	-	0.4	0.8	1.0
Plume	4	-	-	0.0	0.3	0.7

Note: Plume values are from well 655 unless otherwise noted. Background is based on wells 602, 603, 616, and 617.

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

^bConstituents elevated above background in plume.

pCi/L = picocuries per liter.

**Table 3.3 Statistical summary of ground water quality in the Shinarump Conglomerate
at Monument Valley, 1988-1993**

Constituent	Well	Number of samples	Detection limit(s)	% above detection	Observed ^a		
					Minimum	Median (mg/L)	Maximum
Ammonium							
Background		3	0.1	33	<0.1	<0.1	0.2
Downgradient	614	5	0.1	0	-	-	-
	659	7	0.1	29	<0.1	<0.1	0.4
Calcium ^b							
Background		4	0.01-5	100	18	28	30
Downgradient	614	7	0.01-5	100	80	86	90
	659	8	0.01-5	100	18	22	25
Chloride ^b							
Background		4	0.016-1	100	10	14	15
Downgradient	614	7	0.016-1	100	16	19	22
	659	8	0.016-1	100	6	9	11
Magnesium ^b							
Background		4	0.001-5	100	16	18	18
Downgradient	614	7	0.001-5	100	51	57	63
	659	8	0.001-5	100	21	27	29
Manganese ^b							
Background		6	0.001-0.01	33	<0.001	<0.01	0.003-0.01
Downgradient	614	6	0.01	0	-	-	-
	659	8	0.001-0.01	100	0.1	0.11	0.12
Nitrate ^b							
Background		6	0.044-1	100	2.7	4.3	5.4
Downgradient	614	7	0.075-1	100	22	25	34
	659	9	0.044-1	100	0.1	3.3	7.3
Phosphate ^b							
Background		1	0.1	0	-	-	-
Downgradient	614	3	0.1	100	0.4	0.5	0.8
	659	3	0.1	100	0.1	1.2	1.4

BASELINE RISK ASSESSMENT OF GROUND WATER CONTAMINATION AT THE
MONUMENT VALLEY URANIUM MILL TAILINGS SITE, CANE VALLEY, ARIZONA

EXTENT OF CONTAMINATION

**Table 3.3 Statistical summary of ground water quality in the Shinarump Conglomerate
at Monument Valley, 1988-1993 (Continued)**

Constituent	Well	Number of samples	Detection limit(s)	% above detection	Observed ^a		
					Minimum	Median	Maximum
					(mg/L)		
Potassium ^b							
Background		4	0.010-0.575	100	1.6	1.9	3.3
Downgradient	614	7	0.010-0.575	100	1.4	1.6	2.1
	659	8	0.010-0.575	100	2.6	3.1	3.9
Silica							
Background		2	0.1-2	100	9	12	14
Downgradient	614	4	2	100	13	14	14
	659	5	0.1-2	100	14	14	14
Sodium ^b							
Background		4	0.002-5	100	92	98	99
Downgradient	614	7	0.002-5	100	27	30	32
	659	8	0.002-5	100	69	75	78
Strontium							
Background		5	0.0004-0.2	100	0.28	0.30	0.64
Downgradient	614	3	0.1-0.2	100	0.60	0.67	0.68
	659	6	0.0004-0.2	100	0.20	0.26	0.28
Sulfate ^b							
Background		6	0.059-10	100	60	92	131
Downgradient	614	7	0.1-1	100	240	255	312
	659	9	0.059-10	100	84	110	146
Uranium ^b							
Background		5	0.001-0.003	80	≤0.0007	0.001	0.006
Downgradient	614	6	0.003-0.003	100	0.027	0.031	0.036
	659	8	0.003-0.003	88	<0.0003	0.001	0.003
Vanadium							
Background		6	0.001-0.05	0	-	-	-
Downgradient	614	5	0.01-0.05	20	<0.01	<0.01	0.02-0.05
	659	8	0.001-0.05	0	-	-	-

Table 3.3 Statistical summary of ground water quality in the Shinarump Conglomerate at Monument Valley, 1988-1993 (Concluded)

Radionuclide	Well	Number of samples	Detection limit(s)	% above detection	Observed ^a		
					Minimum	Median (pCi/L)	Maximum
Lead-210							
Background		2	-	-	0.4	0.7	1.0
Downgradient	614	1	-	-	-	1.8	-
	659						
Polonium-210							
Background		Not	-	-	-	-	-
Downgradient	614	available	-	-	-	-	-
	659						
Radium-226							
Background		6	-	-	0.0	0.2	2.4
Downgradient	614	5	-	-	0.5	0.5	3.5
	659	8	-	-	2.9	5.0	8.1

Note: Plume values are from well 655 unless otherwise noted. Background is based on wells 602, 603, 616, and 617.

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

^bConstituents are elevated above background in 614 or 659.

pCi/L = picocuries per liter.

Table 3.4 Contaminants of potential concern for human health evaluation

Contaminant levels exceed background	Contaminant levels in nutritional range	Contaminants of low toxicity and/or high dietary range	Contaminants of potential concern (human health)
Ammonium		Ammonium	
Calcium		Calcium	
Chloride	Chloride		
Iron	Iron		
Magnesium		Magnesium	
Manganese			Manganese
Nitrate			Nitrate
Potassium	Potassium		
Silica		Silica	
Sodium		Sodium	
Strontium			Strontium
Sulfate			Sulfate
Vanadium			Vanadium
Uranium			Uranium
Zinc	Zinc		

The uranium concentration in well 619 for June and December 1993 was 0.13 mg/L. As discussed at the end of Section 3.2, the health risks associated with this uranium concentration will be considered separately from plume concentrations of uranium and monitoring will continue to determine the stability of these values over time.

The constituents identified in column 1 of Table 3.4 were screened for their impact on human health using the criteria discussed below to develop a final list of contaminants of concern for human health. Because ecological impacts differ from effects on human health, the complete list of contaminants will be considered for ecological risk assessment in Section 7.0. Several constituents detected above background were deleted from the final list of contaminants of potential concern for human health because they are essential nutrients whose detected levels are within nutritional ranges. These chemicals include chloride, iron, potassium, and zinc.

Final screening of the remaining contaminants was based on very low toxicity and relatively high normal dietary intake compared to the values detected. This criterion ruled out calcium, magnesium, silica, and sodium.

Although ammonium is not considered a dietary component, it is produced in the human body at levels exceeding 4000 mg per day (Summerskill and Wolpert, 1970), roughly an order of magnitude more than would result from ingestion of the most ammonium-contaminated water at the site (381 mg/L). Although these levels are substantially higher than background, they will not likely be associated with adverse health effects. However, detected concentrations will likely affect the taste and odor of the water.

Screening based on the criteria described above eliminated all of the contaminants from consideration except manganese, nitrate, strontium, sulfate, vanadium, and uranium. These constituents form the basis of this risk assessment for Monument Valley.

Of the final contaminants of potential concern, strontium, sulfate, and uranium were also elevated in the hand-dug domestic well (640). Nitrate and vanadium were not above background in this well nor were the following plume indicators: ammonium, manganese, potassium, and zinc. Constituents appearing in column 1 of Table 3.4 that were elevated in well 640 to levels greater than those observed in plume wells include chloride, silica, and sodium. Levels of calcium and magnesium were elevated above background in well 640, but were 10 and 30 percent, respectively, of the concentrations observed in plume wells. These constituents would therefore be screened from further consideration by the same criteria discussed above.

3.4 CONTAMINANT FATE AND TRANSPORT

Hazardous constituents in the plume waters will be subject to dilution, radioactive decay, and various chemical reactions including oxidation/reduction reactions, precipitation and coprecipitation reactions, adsorption onto aquifer mineral surfaces, and possibly transformation by organisms. Dilution, precipitation reactions, and sorption (e.g., ion exchange) reactions will control concentrations of the dominantly cationic metals such as strontium and the major cations calcium, magnesium, potassium, and sodium. These processes also will control manganese and zinc concentrations. Only dilution will affect chloride concentrations.

Dilution and adsorption will mainly decrease uranium and vanadium concentrations. Ion exchange reactions on clays and oxidation to nitrate perhaps mediated by bacterial action will decrease ammonium concentrations. These reactions may produce nitrogen, which would be released to the atmosphere.

Sulfate concentrations will be subject to dilution, precipitation/dissolution reactions, adsorption reactions, and possibly oxidation/reduction reactions. The precipitation/dissolution reactions will occur in the portion of the plume closest to the tailings piles. Precipitation reactions are currently active because the shallow ground water in this zone is oversaturated with gypsum. Now that the tailings piles have been removed, the ground water sulfate concentrations in this

zone will decrease, allowing gypsum to redissolve. This dissolution will buffer the sulfate concentrations at fairly high levels until the gypsum is exhausted. At this point, dilution with background waters will substantially lower sulfate concentrations in this zone. Adsorption reactions are not likely to affect sulfate concentrations in the shallow ground water significantly due to the relatively high concentrations involved. Given the high redox potential of the shallow ground water, reduction of sulfate to sulfide also is unlikely.

The chemical species present in natural systems are a function of pH, Eh, and the concentrations of various anions and cations. Speciation determines the mobility of the chemicals and may also influence their toxicity. Using the geochemical model PHREEQE, the predominant species of the contaminants of concern for human health were predicted (Parkhurst et al., 1980). Table 3.5 lists the dominant solution species for the hazardous contaminants of concern.

Table 3.5 Monument Valley speciation summary

Contaminant of potential concern	Identity of dominant species in ground water	Common name	Molar %
Manganese	Mn^{2+}	Manganese	64
	$MnSO_4 \text{ Aq}$	Manganese sulfate	31
Nitrate	NO_3^-	Nitrate	100
Strontium	Sr^{2+}	Strontium	100
Sulfur	SO_4^{2-}	Sulfate	100
Uranium	$UO_2(CO_3)_3^{4-}$	Uranyl tricarbonat	70
	$UO_2(CO_3)_3^{2-}$		30
Vanadium	$HV_2O_7^{3-}$	Pyrovanadic acid	83
	$H_2VO_4^-$		15

Note: Assume Eh = 400 mV, pH = 7.0.

3.5 SURFACE WATERING MONITORING

Figure 2.2 shows the locations of three surface water sampling points. Two sampling points are in the frog ponds in Cane Valley Wash (locations 621 and 622) and one (location 620) is at a spring downgradient of the site. Filtered samples were collected from locations 621 and 622 on seven separate occasions from 1985 through 1992. In November 1992, unfiltered samples were collected from locations 621 and 622. Filtered samples were collected from a spring at location 620 in 1985 and 1986. No samples have been collected from location 620 since 1986. A comparison of the data from location 620 with the data from locations 621 and 622 indicates similar water quality. For this reason, and because no data have been available in more than 7 years, location 620 will not be considered further in this risk assessment.

As mentioned above, the water quality has been monitored for several years at locations 621 and 622. These locations are crossgradient to the site (see Figure 2.2) and based on available data, which is discussed in Section 7.0, no evidence indicates that site activities have affected water quality.

3.5.1 Near-surface water in Cane Valley Wash

In 1993, three additional locations in Cane Valley Wash were sampled. These samples were taken from hand-dug holes within inches of ground surface in the dry stream bed of Cane Valley Wash at the locations (623, 624, and 627) shown on Figure 2.2. These locations were evaluated for the constituents known to be geochemical or toxicological markers of the contaminant plume. Concentrations of these constituents were compared with concentrations found in the region of the frog ponds (locations 621 and 622). Of the plume-related contaminants listed in column 1 of Table 3.4, only zinc was elevated at location 623. Downstream at location 624, plume indicators chloride, magnesium, manganese, and potassium were observed at levels greater than the 98-percent confidence interval for samples from locations 621 and 622. Strontium and uranium exceeded levels observed at locations 621 and 622. Location 627 was the furthest downstream and showed the greatest number of elevated constituents and the greatest magnitude of elevation. Ammonium and arsenic were below the maximum values observed at locations 621 and 622, but exceeded the 98-percent confidence interval for those locations. Chloride, magnesium, manganese, potassium, silica, sodium, sulfate, uranium, and zinc were observed at levels greater than the maximum observed values at locations 621 and 622. The TDS concentration at location 627 was elevated to 26,600 mg/L. However, the major plume indicator, nitrate, was not observed in any of the sampling locations in Cane Valley Wash. In addition, plume indicators calcium, strontium, and vanadium were not found to exceed background (concentrations found in 621 and 622) at location 627. For each contaminant listed in column 1 of Table 3.4, Table 3.6 lists the greater of the two concentrations of elevated constituents observed at location 627 and the relative relationship of these concentrations to the maximum concentrations found in the plume wells.

Table 3.6 shows that some unanswered questions remain regarding the source of elevated constituents found at locations 624 and 627. Although the pattern of elevated constituents closely resembles that seen in the plume, the major plume constituent nitrate is missing. In addition, concentrations of many plume constituents are elevated more than twofold over the maximum value observed in plume wells. Uranium is sevenfold greater than plume concentrations, and chloride and sodium show 24- and 50-fold increases, respectively. These results suggest several alternative theories as to the source of contamination.

- The elevations represent the intersection of the plume with Cane Valley Wash, and the increased concentrations represent concentrations present only in the upper few inches of the unconsolidated aquifer. These concentrations would be diluted by the lower concentrations found with

Table 3.6 Near-surface ground water maximum contaminant concentrations in Cane Valley Wash relative to plume maximums

Constituent	Concentration at 627 ^a (mg/L)	Plume maximum (mg/L)
Ammonium	1.6	381
Calcium	46.7	521
Chloride	938	38
Iron	NA	0.1
Magnesium	82	350
Manganese	0.62	0.12
Nitrate	ND	1600
Potassium	73	43
Silica	133	23
Sodium	9850	190
Strontium	0.43 ^b	3.2
Sulfate	5990	3540
Vanadium	0.31 ^b	0.08
Uranium	0.217	0.031
Zinc	0.081	0.05

^aJuly and December 1993 sampling.

^bAlthough concentration elevated above plume values, it does not exceed 98-percent confidence interval for concentrations at locations 621 and 622 (frog ponds).

NA—Not analyzed.

ND—Not detected.

increasing depth during sampling of monitor wells screened at deeper intervals relative to the water table. This possibility can be evaluated by installing shallow well points between the wash location and the known plume. If contamination is plume-related, elevated concentrations should be observed in these shallow wells on the site side of the wash, but not in wells drilled on the opposite side of Cane Valley Wash. This interpretation assumes the hydrogeologic gradients are as predicted, with ground water on both sides of the wash moving toward the wash. Installation of such wells is to be covered in future field investigation work plans.

- Tailings or associated contaminated refuse from the site was dumped in the wash and contaminants have migrated through the soil. Soil surface monitoring during surface remediation should have detected such a source term, however. Records of the precise location of monitoring during surface remediation can be checked and additional monitoring performed, if needed.
- The elevations are the result of an unknown source being carried into the wash via an underground stream tapped in these wash samples. This can be determined by sampling shallow subsurface water, if present, at a location upstream in Cane Valley Wash. These samples should be drawn above the frog ponds and further upstream south of the site for full evaluation of this possibility. Because the frog ponds are fed from the De Chelly aquifer, this clean water probably will dilute any contamination in the immediate vicinity. Future field investigation plans will cover such sampling.
- Increased concentrations could also result from evapotranspiration as ground water approaches the wash. Selective plant uptake of water could concentrate those constituents not taken up by the plants in the remaining water as it approaches the wash. The same could be true for movement from another source. Data on this point should result from data collection to evaluate the first two alternatives.

Section 7.5 evaluates the toxicity of these constituents to the potential ecological receptors.

Precipitation and snowmelt may have carried both dissolved and suspended constituents along surface drainages to Cane Valley Wash. Metal constituents transported as dissolved species would have become diluted after discharging to these water bodies. Alternatively, dissolved species could have precipitated, becoming adsorbed to sediments or absorbed in biota with varying biochemical and geochemical conditions. Constituents transported from the disposal site that were sorbed onto soil particles would have been deposited as sediments. Variations in geochemical conditions or biological action could release constituents adsorbed onto sediments into surface waters. Thus, deposited sediments could act as a source of site-related surface water contamination.

4.0 EXPOSURE ASSESSMENT

4.1 POTENTIALLY EXPOSED POPULATION

Exposure can occur only if there is both a source of contamination and a mechanism of transport to a receptor population or individual. Ground water contaminated by uranium processing at the Monument Valley site is not currently used. If the contaminant plume continues to move north-northeast, surface expression would be possible during wet seasons when the water table is elevated in the unconsolidated aquifer. If this expression is occurring or occurs in the future in Cane Valley, livestock could drink the water. Using the livestock for food would create an exposure pathway to humans. Plants with roots in the unconsolidated aquifer or irrigated with water from this zone could take up and concentrate contaminants, forming a pathway to humans through plant consumption. Finally, a domestic well could be drilled or hand-dug in the unconsolidated aquifer, creating an exposure pathway through drinking and bathing water.

Because there are no known current human receptors of contaminated ground water, a future ground water use scenario is assumed. This scenario evaluates domestic ground water use consistent with current use by the regional rural population. The potentially exposed population includes Navajos of the following age groups: infants (birth to 1 year old), children (1 to 10 years old), and adults (11 to 64 years old). These age groups were selected for the following reasons:

- Survey data for population variables such as age, weight, and daily water intake are available for these age groups.
- Toxicological variables are similar within these age groups, including responsiveness of sensitive subgroups (infants and children) to the contaminants of concern, toxicant intake to body weight ratios, and toxicokinetics.

The incidence of type II (adult onset) diabetes in Navajos is approximately 20 percent in members of the population more than 20 years old and appears to be increasing (Hoy, 1993). Compromised renal function and increased drinking water ingestion rates in this diabetic subpopulation could increase susceptibility to toxicants. Though insufficient data exist to allow this risk assessment to quantitatively evaluate risk to diabetics, it is recognized that this group could represent a sensitive subpopulation.

4.2 EXPOSURE PATHWAYS

Ground water in the region is primarily used for household purposes such as drinking, cooking, and bathing. Other uses typical of the region that could indirectly lead to human exposure include irrigation and livestock watering.

Figure 4.1 provides a conceptual model for potential ground water exposure pathways from these uses.

There is a history of hand-dug wells in this region; therefore, the potential exists for residents to access water near the surface as it nears Cane Valley Wash. The characteristics of this water (nearly 10 grams per liter [g/L] sodium; nearly 6 g/L sulfate; and more than 25 g/L TDS) are likely to make this a less desirable source of drinking water than the deeper wells currently in use. Cattle using this water as a drinking water source and plants irrigated with this water would probably not survive due to the high TDS and sulfate concentrations. However, plants with roots in this water may survive and provide an exposure point for cattle or other livestock. The source of near-surface contamination, the reliability of the data over time, and the potential concentrations that might result from surface expression of this water are under investigation but are unknown at this time. Therefore, these data will not be evaluated in the human pathway analysis in this document. Rather, this document will primarily examine potential human health risks posed by the contaminant plume as identified and tracked over the past 5 years. As more data on the near-surface water become available, the need for further evaluation of risks posed by that source will be reevaluated. Because the potential for ecological impacts from the near-surface water appears to have a greater probability of occurrence than human exposure, the ecological impacts will be addressed in Section 7.0.

The analysis of human health risks will consider use of ground water from the most contaminated wells in the contaminant plume north/northeast of the former mill site. Elevations in sulfate and uranium observed in the hand-dug domestic well (640) fall within the range of concentrations observed in the contaminant plume. Therefore, potential human health risks from exposure to sulfate and uranium through ingestion of water from well 640 will be determined through this analysis as well.

4.2.1 Drinking water ingestion

Drinking water ingestion is generally the most significant exposure pathway for ground water contaminated with metals and other nonvolatile compounds. For this evaluation, drinking water consumption includes water used for drinking and water used for food preparation (e.g., reconstituted juice, soup, rice, and beans). To compare relative pathway significance, Table 4.1 shows a screening level assessment of drinking water intake. These calculations are based on estimates of the average concentrations of contaminants within the plume (i.e., the 95-percent upper confidence limit [UCL] for the average concentration, using data from the most contaminated plume wells [well 606 for nitrate and well 655 for all other contaminants]).

FIGURE 4.1
CONCEPTUAL MODEL FOR POTENTIAL GROUND WATER EXPOSURE PATHWAYS

Table 4.1 Exposure dose calculations and equation definitions for ground water usage in future hypothetical adult scenarios, Monument Valley UMTRA Project site

Contaminant of potential concern	Cw (mg/L)	Ground water exposure doses (mg/kg-day)		Dermal absorption ^a water ingestion
		Ingestion	Dermal absorption	
Noncarcinogenic effects				
Manganese	0.086	3E-03	5E-06	0.002
Nitrate	1200	3E + 01 (18E + 01) ^b	7E-02	0.002
Strontium	3.1	9E-02	2E-04	0.002
Sulfate	2920	8E + 01 (45E + 01) ^b	2E-01	0.002
Uranium	0.029	8E-04	2E-06	0.002
Vanadium	0.02	6E-04	1E-06	0.002
Carcinogenic effects				
Uranium	20 ^c	1E + 06 ^d	2E + 03 ^d	0.002

Equation Definitions for Exposure Dose Calculations

Ingestion of ground water

Chemicals

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

Radionuclides

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

Dermal contact with ground water - carcinogens and noncarcinogens

Chemicals

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

Radionuclides

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

Table 4.1 Exposure dose calculations and equation definitions for ground water usage in future hypothetical adult scenarios, Monument Valley UMTRA Project site (Concluded)

Where:

- Cw = Contaminant concentration in ground water (upper 95% confidence level of the mean of data from plume wells).
IRw = Ingestion rate for water (L per day) (2 L per day for an adult; 0.64 L per day for an infant).
EF = Exposure frequency (365 days per year).
ED = Exposure duration (7 years for an adult, 1 year for an infant for noncarcinogenic effects; 70 years for carcinogenic effects).
BW = Body weight (70 kilograms [kg] for an adult; 4 kg for an infant).
AT = Averaging time (365 days x ED for noncarcinogens; 365 days x 70 years for carcinogens).
SA = Skin surface area (19,400 square centimeters [cm²]).
Pc = Dermal permeability constant (0.001 cm per hour).
Cf = Conversion factor (0.001 liters per cubic centimeter [L/cm³]).
ET = Exposure time (0.2 hour per day).

^aRatio of the dermal absorption exposure dose to the ground water ingestion exposure dose.

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

^cpCi/L = Picocuries per liter.

^dpCi per lifetime.

mg/kg-day = Milligrams per kilogram per day.

4.2.2 Dermal absorption

Dermal absorption is the process by which chemicals coming into contact with the skin are absorbed into the blood vessels near the skin surface. While some compounds are absorbed easily, metals do not possess chemical properties conducive to skin absorption.

To evaluate this exposure route, a screening calculation was conducted to determine the significance of a dermal absorption pathway compared to a drinking water pathway for the contaminants of concern. Because chemical-specific absorption factors are not available for these contaminants, they were assumed to be absorbed across the skin at the same rate as water. This assumption will probably overestimate any potential contribution from dermal absorption.

Table 4.1 gives the results of the screening. Based on these results, dermal absorption was eliminated from more detailed evaluation because it contributed less than 1 percent of the total intake from drinking water.

4.2.3 Ingestion of ground water-irrigated produce

This exposure route also was evaluated for its relative significance to the drinking water ingestion route. Table 4.2 gives the results of the screening calculation. The assumptions for this evaluation probably will overestimate the potential for exposure from this route because this garden is assumed to be the source of all garden produce in the diet. The results of this screening show that for the contaminants of potential concern at this site, ingesting garden vegetables and fruit irrigated with contaminated ground water would lead to potential exposures of 2 percent or less of that associated with drinking water ingestion (except for strontium where up to 15 percent of the ground water ingestion dose could be obtained from ingesting irrigated plant materials). This pathway was eliminated from further quantitative evaluation, although Section 6.1 qualitatively discusses the potential increment to drinking water exposures.

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

These pathways were eliminated from further consideration because the nitrate and sulfate concentrations are so high that livestock would not likely survive chronic ingestion of the water. If the livestock cannot consume the water, there is no potential for bioaccumulation or transfer of ingested contaminants to meat tissue or milk. In addition, infants aged 0 to 3 months (the population most sensitive to nitrate toxicity) do not consume meat or dairy products. Section 7.0 presents further evaluation of the direct toxicity to livestock.

The results of the pathway screening analyses indicate drinking water ingestion is the dominant pathway; Section 4.4 further evaluates this pathway probabilistically. Potential risks from non-drinking water pathways are discussed in Section 6.0.

Table 4.2 Exposure dose calculations and equation definitions for ground water-irrigated produce ingestion in future hypothetical adult scenarios, Monument Valley UMTRA Project site

Contaminant potential of concern	Cw (mg/L)	Kd (L/kg)	Bv	Br	Garden produce ingestion exposure doses (mg/kg-day)		Total produce ingestion ^a water ingestion
					Vegetative parts	Reproductive parts	
Noncarcinogenic effects							
Manganese	0.086	65	0.25	0.05	7E-05	8E-06	0.02
Nitrate	1200	0.01	30	30	2E-02	1E-02	0.001
Strontium	3.1	35	2.5	0.25	1E-02	8E-04	0.12
Sulfate	2920	7.5	0.5	0.5	5E-01	3E-01	0.01
Uranium	0.029	450	0.0085	0.004	5E-06	2E-06	0.008
Vanadium	0.02	1000	0.0055	0.003	5E-06	2E-06	0.01
Carcinogenic effects							
Uranium ^b	20 ^c	450	0.0085	0.004	4.36E + 03 ^d	2E + 02	0.05

Equation Definitions for Exposure Dose Calculations
Ingestion of garden produce irrigated with ground water

Chemicals

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

Radionuclides

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

Where:

- Cw = Contaminant concentration in ground water (upper 95% confidence level of the mean of data from plume wells).
- Kd = Soil-water partition coefficient (L/kg) (Baes et al., 1984).
- 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) (Baes et al., 1984).
- EF = Exposure frequency (365 days per year).
- ED = Exposure duration (7 years for an adult; 1 year for an infant).
- BW = Body weight (70 kilograms [kg] for an adult; 4 kg for an infant).
- AT = Averaging time (365 days x ED for noncarcinogens; 365 days x 70 years for carcinogens).
- * = Exposure doses due to vegetative portions and reproductive portions of garden produce are calculated separately.

Table 4.2 Exposure dose calculations and equation definitions for ground water-irrigated produce ingestion in future hypothetical adult scenarios, Monument Valley UMTRA Project site (Concluded)

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

IRp = Ingestion rate for garden produce (0.05 kg per day for vegetative parts; 0.03 kg per day for reproductive parts).

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

^aRatio of the total garden ingestion exposure dose to the ground water ingestion exposure dose.

^bUranium-234 and -238.

^cpCi/L = Picocuries per liter.

^dpCi per lifetime.

L/kg = Liters per kilogram.

mg/kg-day = Milligrams per kilogram per day.

4.3 EXPOSURE CONCENTRATIONS

Exposure concentration of a contaminant in ground water is defined as the average 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 although actual contaminant concentrations (and therefore exposures) are expected to decrease with time now that the tailings have been removed. Nonetheless, these estimates are reasonable for chronic exposure soon after surface remediation. (Chronic exposure for noncarcinogens is considered to be exposure for any period longer than 7 years.)

Exposure concentrations are evaluated as a probability of occurrence based on ground water data collected from monitor well 655 for strontium, sulfate, uranium, and vanadium. Since 1988, this well has consistently shown the highest concentrations of these constituents. Because nitrate levels in well 606 were greater than in well 655, well 606 was used to model nitrate exposures.

The probability distribution selected for each contaminant reflected the same mean, median, standard deviation, and shape as were observed in historical water quality data. The upper tail of the distributions was truncated at the 99th percentile. For every contaminant, this highest concentration in the distribution was higher than the maximum observed concentration in the historical water quality data. The software package @RISK (Palisade Corp, Newfield, New York) generated the contaminants of concern probability curves, the results of which are shown in Figures 4.2 through 4.7.

4.4 ESTIMATION OF INTAKE

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

The potential toxicity of noncarcinogenic contaminants in drinking water depends primarily on long-term (i.e., at least 7 years) average daily consumption

FIGURE 4.2
SIMULATED DISTRIBUTION OF NITRATE BASED ON DATA FROM WELL 606
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 4.3
SIMULATED DISTRIBUTION OF SULFATE BASED ON DATA FROM WELL 655
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 4.4
SIMULATED DISTRIBUTION OF STRONTIUM BASED ON DATA FROM WELL 655
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 4.5
SIMULATED DISTRIBUTION OF URANIUM BASED ON DATA FROM WELL 655
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 4.6
SIMULATED DISTRIBUTION OF VANADIUM BASED ON DATA FROM WELL 655
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 4.7
SIMULATED DISTRIBUTION OF MANGANESE BASED ON DATA FROM WELL 655
MONUMENT VALLEY UMTRA PROJECT SITE

of the contaminant per kilogram of body weight. For noncarcinogens, chronic daily intake in mg/kg-day is calculated as follows:

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

where:

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

kg = kilogram.

Potential carcinogenicity of radionuclides is thought to increase with *total* intake over time, instead of with *average daily* intake as for noncarcinogens. Also, body weight is relatively insignificant in determining risk from exposure. Only uranium ingestion will be presented here because it is the only carcinogen elevated above background at the Monument Valley site. Intake of a radioactive carcinogenic substance is therefore quantified as total exposure to radioactivity throughout the residency period of an individual:

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

Where:

pCi = picocuries.

pCi/L = picocuries per liter.

Average daily intake (L/day)

Lognormal probability distributions were used to describe the variation in average daily tap water intake among members of the population (Roseberry and Burmaster, 1992). These distributions were developed from data collected during the 1977-78 National Food Consumption Survey conducted by the U.S. Department of Agriculture. During the survey, total tap water consumption during a 3-day period was recorded for 26,081 survey participants nationwide (Figure 4.8).

Exposure frequency (days)

Individuals generally are not present at their homes and drinking water from the same source for 365 days per year. Therefore, calculation of intake assumes only 350 days of exposure per year, allowing for 14 days per year of drinking water intake from a different source. Because exposure is expressed and compared in terms of average daily intake, 365 days per year is retained in the averaging-time term in the denominator.

FIGURE 4.8
PROBABILITY DISTRIBUTIONS FOR TAP WATER INGESTION RATES BY AGE GROUP

Body weight (kg)

The National Health and Nutrition Survey collected extensive national data on weights of males and females, by age, between 1976 and 1980. These data were used to develop lognormal probability distributions for body weight by age, separately by gender. The distributions for males and females were then combined using census data on the national ratio of males to females within each age group (Figure 4.9).

Exposure duration (years)

Distributions of exposure duration were developed using data collected by the U.S. Department of Commerce, the Bureau of the Census, and the U.S. Department of Housing and Urban Development in 1985 and 1987 (Israeli and Nelson, 1992). For noncarcinogenic effects, the exposure duration term in the numerator and denominator of the drinking water intake equations cancel out, assuming all exposures are chronic; i.e., greater than 7 years. Therefore, deviations from the standard residence time assumptions do not affect the results. For carcinogens, however (because risk is cumulative throughout a lifetime), deviations from the hypothesized distribution could significantly affect the risk estimate. Given the grazing permit system used by the Navajo Nation for land allocation, residents frequently can be in the same region for an entire lifetime. No available data adequately model this longer residence time. Therefore, a fixed lifetime exposure duration of 70 years was used to model lifetime cancer risks.

Using exposure concentration distributions discussed in Section 4.3 and the intake parameter distributions described in this section, total intake distributions derived for the three age groups were generated for sulfate. Figure 4.10 presents the results, illustrating the effect of different age group characterizations on daily intake of noncarcinogens. From this figure, it can be seen that intake is greatest in the 1- to 10-year age group. Therefore, this group is used in risk evaluation unless one of the other age groups demonstrates increased sensitivity to a particular constituent. Of the contaminants at Monument Valley, a more sensitive age group is known only for nitrate and sulfate. Consequently, the nitrate and sulfate exposure risk determinations are based on infant exposure. Figures 4.11 through 4.15 present simulated intake distributions for appropriate age groups for nitrate and the remaining contaminants.

4.5 EXPOSURE ASSESSMENT UNCERTAINTIES

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

- Uncertainties resulting from the lack of thorough environmental sampling (ground water and surface water) data, which could lead to an underestimation or overestimation in the exposure analysis.

FIGURE 4.9
BODY WEIGHT PROBABILITY DISTRIBUTIONS BY AGE GROUP

NOTE: HIGHEST POTENTIAL EXPOSURE OCCURS IN CHILDREN AGE 1-10 (AVERAGE POTENTIAL EXPOSURE OF 99 mg SULFATE/kg BODY WEIGHT/DAY).

FIGURE 4.10

COMPARISON OF DISTRIBUTIONS OF POTENTIAL SULFATE EXPOSURES AS A RESULT OF DRINKING CONTAMINATED GROUND WATER FOR THE THREE AGE GROUPS EXAMINED MONUMENT VALLEY UMTRA PROJECT SITE

NOTE: AVERAGE POTENTIAL EXPOSURE IS 45 mg NITRATE/kg BODY WEIGHT/DAY. INFANTS' EXPOSURES HAVE BEEN MODELED FOR NITRATE BECAUSE INFANTS ARE MORE SENSITIVE TO NITRATE TOXICITY THAN THE OTHER TWO AGE GROUPS.

FIGURE 4.11
DISTRIBUTION OF POTENTIAL NITRATE EXPOSURES FOR INFANTS CONSUMING
CONTAMINATED GROUND WATER
MONUMENT VALLEY UMTRA PROJECT SITE

NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.11 mg STRONTIUM/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.12
DISTRIBUTION OF POTENTIAL STRONTIUM EXPOSURES FOR CHILDREN
AS A RESULT OF CONTAMINATED GROUND WATER INGESTION
MONUMENT VALLEY UMTRA PROJECT SITE

NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.0009 µg URANIUM/
kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE
BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT
IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN
IDENTIFIED.

FIGURE 4.13
DISTRIBUTION OF POTENTIAL URANIUM EXPOSURES FOR CHILDREN
AS A RESULT OF CONTAMINATED GROUND WATER INGESTION
MONUMENT VALLEY UMTRA PROJECT SITE

NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.55 µg VANADIUM/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.14
DISTRIBUTION OF POTENTIAL VANADIUM EXPOSURES FOR CHILDREN
AS A RESULT OF CONTAMINATED GROUND WATER INGESTION
MONUMENT VALLEY UMTRA PROJECT SITE

NOTE: AVERAGE POTENTIAL EXPOSURE IS 0.002 mg MANGANESE/kg BODY WEIGHT/DAY. EXPOSURES FOR CHILDREN HAVE BEEN MODELED BECAUSE THEIR INTAKE/kg BODY WEIGHT IS GREATEST AND NO SENSITIVE POPULATION HAS BEEN IDENTIFIED.

FIGURE 4.15
DISTRIBUTION OF POTENTIAL MANGANESE EXPOSURES FOR CHILDREN
AS A RESULT OF CONTAMINATED GROUND WATER INGESTION
MONUMENT VALLEY UMTRA PROJECT SITE

- Uncertainties arising from the assumption that the ground water contaminant source term at the site has reached a steady state and that contaminant concentrations at the exposure point will remain constant for chronic periods of exposure (generally greater than 7 years). Because the source of contamination at Monument Valley has been removed, the assumption of a constant source will likely lead to an overestimation of risk.
- Uncertainties associated with the model used to estimate uptake of contaminants into plants for the irrigated garden produce pathway. Under the site conditions at Monument Valley, plant uptake factors could vary substantially from the default literature estimates. As with environmental sampling, the net effect on risk estimates of this uncertainty cannot be predicted.
- Uncertainties associated with the relationship of an applied, absorbed, or effective toxic dose. (An applied dose was used here.)
- Uncertainties associated with differing sensitivities of subpopulations such as diabetics and the elderly.

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

5.0 TOXICITY ASSESSMENT

Six contaminants that could adversely affect human health have been detected in ground water at the Monument Valley site: nitrate, strontium, sulfate, manganese, uranium, and vanadium. This section summarizes the toxicological effects of the chemical contaminants and the carcinogenic potentials of the radionuclides. The following source materials were used in developing these toxicological profiles: when available, EPA's Integrated Risk Information System (IRIS); the Agency for Toxic Substances Disease Registry Toxicological Profiles published by the Department of Health and Human Services (DHHS); and the *Handbook on the Toxicology of Metals* (Friberg et al., 1986). When these review documents were not available, peer-reviewed scientific literature was used as cited. By basing toxicity information on the standardized review documents cited above, the evaluation of risks at UMTRA Project sites should be consistent with evaluations at sites regulated under different legislation.

The toxicity profiles presented here will focus on drinking water source material in humans whenever available and will include animal information only when human data are not obtainable. Widely spaced dotted lines will represent animal information on the toxicity range graphs. When uncertainty exists about the beginning or end of a range of exposures that produces specific toxic effects, closely spaced dots will be used at the applicable end of the line denoting range.

5.1 CONTAMINANT TOXICITY SUMMARIES

The basic toxicokinetics and toxicity of each of the contaminants of concern at the Monument Valley site are summarized below. Wherever possible, data from human studies will be addressed. 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 exposure level, the following discussions will focus on toxic effects observed in the exposure range most relevant to contamination in the Monument Valley site area.

5.1.1 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). Iron and other metals influence the rate of absorption. In states of iron deficiency, manganese is actively absorbed from the intestine. Individuals with anemia can absorb more than twice the percentage of an ingested dose. However, in states of excess iron, manganese absorption is by diffusion only (Saric, 1986). High levels of dietary calcium and

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

Tissue accumulation and clearance

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

Absorbed manganese is rapidly cleared 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

Food constitutes the major source of manganese intake for humans. The highest manganese concentrations are in plants, especially wheat and rice. Drinking water generally contains less than 0.1 mg/L. Manganese levels in soil range from 1 to 7000 milligrams per kilogram (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 recommends criteria of 1 to 2 mg/kg for manganese in soil and 200 mg/kg in plants (Saric, 1986).

Differences in eating habits can explain variations in manganese intake. The intake will be higher in populations with cereals and rice as main food sources than in populations where meat and dairy products form a larger part of the diet. The average daily intake is estimated to be between 2.0 and 8.8 mg per day (0.03 and 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 0.2 mg (0.003 mg/kg-day), although some mineral waters can increase this amount more than threefold (Saric, 1986). One study from Greece reported drinking water

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

Manganese toxicity

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 0.005 mg/kg-day while the lowest-observed-adverse-effect level (LOAEL) is 0.06 mg/kg-day (EPA, 1993). The EPA reference dose (RfD) for drinking water is 0.005 mg/kg-day. The RfD for food ingestion is 0.14 mg/kg-day. Manganese in drinking water appears to be potentially more bioavailable, i.e., more readily absorbed, than manganese in dietary food sources. This would result in toxic effects with lower ingested doses of manganese in drinking water than in food (EPA, 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 with symptoms 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 (Goyer, 1991), suggesting that manganese produces functional deficit in the central nervous system.

Limited information is available on the effects of manganese ingestion. Because effects from drinking water seem to differ from those from food sources, only water consumption studies will be considered here. A Japanese study of 25 people drinking well water with manganese concentrations of 14 mg/L (0.4 mg/kg-day estimated intake) reported symptoms of intoxication, including a mask-like face, muscle rigidity and tremors, and mental disturbances. Two cases (8 percent) of death were reported among intoxicated people. A Greek study of more than 4000 individuals drinking water with manganese concentrations varying from 0.081 to 2.3 mg/L (estimated intake at 2 L per day for a 70-kg individual range from 0.002 to 0.07 mg/kg-day) showed varying degrees of neurological effects in individuals drinking from 0.007 to 0.07 mg/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). Limited information is available on the effects of various forms of manganese.

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

Figure 5.1 summarizes the health effects of manganese as a function of dose.

5.1.2 **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 occurring in a higher pH environment. Infants have a higher gut pH, which is more conducive to bacteria growth. Therefore, the combination of a higher pH and an increased bacterial conversion exacerbates the production of nitrite from nitrate in infants, resulting in higher blood nitrite 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, lowering 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 because of 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. 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 adults, the half-time for methemoglobin reductase conversion of methemoglobin back to hemoglobin is estimated to range from 6 to 24 hours for theoretical levels of methemoglobin in the 80- to 100-percent range (Bolyai et al., 1972).

Environmental sources of nitrate

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

FIGURE 5.1
MANGANESE TOXICITY RANGES

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 2000 mg/kg 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 drought, high temperatures, cloudiness, and insect and herbicide damage to plants. Nitrates and nitrites also are used to preserve meats, especially corned or smoked products.

Nitrate toxicity

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 are summarized together. For easier comparison between ingested doses of nitrate and ground water levels at Monument Valley, dose ranges are 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 can be correlated with the percentage of methemoglobin in the blood as follows: less than 10 percent, individuals are asymptomatic; more than 25 percent produces weakness, rapid pulse, and tachypnea (rapid breathing); more than 50 to 60 percent can be fatal (EPA, 1993). These symptoms reflect a progressive decrease in oxygen availability. As explained above, infants are more sensitive to methemoglobin production and therefore are considered the most sensitive population. The route of exposure for infants is from contaminated water in formula.

No symptoms of toxicity have been reported with nitrate intakes below 7 mg/kg-day in infants. With intakes from 7 to 30 mg/kg-day, mild symptoms such as weakness, rapid pulse, and rapid breathing occur. The severity of these symptoms increases as increased nitrate intake results in greater levels of methemoglobin and therefore reduced oxygen availability. Cyanosis, or a blue appearance to the skin, occurs, followed by unconsciousness as oxygen availability is further reduced. The lowest reported fatal dose of chronic nitrate is 35 mg/kg-day for an infant and 116 mg/kg 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. Figure 5.2 summarizes the health effects of nitrates in infants 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. In most cases, exposure doses were back-calculated from sampling their drinking water. Therefore, these data do not represent well-controlled

FIGURE 5.2
NITRATE TOXICITY RANGES FOR INFANTS

studies with readily defined dosage ranges. Many of the water sources in these clinical studies showed contamination with bacteria, leading to the possibility that this bacterial exposure is a necessary cofactor in the development of 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.

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.

5.1.3 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 is absorbed to a higher degree. The bioavailability of ingested strontium is estimated to be 20 percent. This figure varies, depending 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 in bone. The average adult body burden (the amount found in normal, healthy adults) of strontium is estimated to be 320 mg (Snyder et al., 1975). Absorbed strontium is cleared from the body primarily through urine and feces. In humans, 12 to 13 percent of an intravenous dose is eliminated in the feces; urinary excretion eliminates nearly 60 percent of an intravenous dose and 4 to 18 percent of an oral dose (EPA, 1990). Strontium is filtered by the kidneys at a rate 3.5 times greater than calcium (which is reabsorbed more efficiently than strontium), resulting in a more rapid clearance.

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.

Strontium toxicity

No data are available on the acute toxicity of stable strontium in humans. In laboratory studies, 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 (LD₅₀) to an LD₅₀ 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 (EPA, 1993). Although no side effects were observed, the resultant bone loss renders questionable the extrapolation of these data to a healthy population.

In rat studies, strontium toxicity is related to its calcium displacement in bone; this toxicity differs with the developmental stage of the animal. The lowest intake level producing 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. Figure 5.3 summarizes these health effects as a function of dose.

5.1.4 Sulfate

Absorption

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

Tissue accumulation and retention

Ingestion of high levels of sulfate results in transient increases in both blood and urine concentrations. For sulfate doses of approximately 75 mg/kg, an estimated 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 also is excreted in feces in its inorganic

2

of

2

FIGURE 5.3
STRONTIUM TOXICITY RANGES

form. To date, no available data indicate sulfate is accumulated, even with chronic ingestion of above-normal levels. However, no extremely high chronic doses 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 sulfate concentration of 99 mg/L. The EPA estimates a normal sulfate intake range of 0.00023 to 0.0064 mg/kg-day from air and up to 2.9 mg/kg-day from drinking water in the concentration range in the western United States (EPA, 1992a). No estimates are available on sulfate intake from food sources.

Sulfate toxicity

As with nitrate toxicity, the acute and chronic effects of sulfate toxicity differ more in severity than in symptoms or mechanisms. Therefore, this discussion combines acute and chronic toxicity. As mentioned above, no available data indicate 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 primarily manifests itself in diarrhea. Serious gastroenteritis has been reported in some infants and adults drinking water containing 400 to 1000 mg/L sulfate (EPA, 1992a). 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. However, this is not necessarily the case. In regions such as Saskatchewan, with high sulfate concentrations in the drinking water, residents adapt to the taste and find the water palatable (EPA, 1992a). When drinking water is contaminated with sulfate, the taste of the water may make it unpalatable and reduce consumption. Also, data indicate diabetic and elderly populations with compromised kidney function may be more sensitive to the effects of sulfates than are healthy adults (EPA, 1992a). This lower water intake could compound the dehydration effects of the diarrhea which, in extreme cases, can lead to death. As with nitrate toxicity, infants may be the most susceptible population for sulfate-induced diarrhea. Figure 5.4 summarizes these health effects as a function of dose.

FIGURE 5.4
SULFATE TOXICITY RANGES

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

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 were back-calculated from sampling their drinking water. Therefore, these data do not represent well-controlled studies with readily defined dosage ranges.

5.1.5 Uranium

The 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 with shorter half-lives. Figure 5.5 summarizes the radioactive decay chain of uranium-238 and uranium-234. As all uranium isotopes in nature are radioactive, the hazards of a high uranium intake are from both its chemical toxicity and its potential radiological damage. This section focuses on the chemical toxicity of natural uranium. Section 5.3 discusses the carcinogenic potential associated with exposure to radioactive isotopes of natural uranium.

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 were found in bone, muscle, lung, liver, and kidney (Fisenne et al., 1988). Uranium retention in bone consists of a short retention half time of 20 days followed by a long retention half time of 5000 days for its residue (Tracy et al., 1992).

In body fluids, uranium tends to convert into water-soluble hexavalent uranium (Berlin and Rudell, 1986). Approximately 60 percent of the uranium in plasma complexes with low-molecular-weight anions (e.g., bicarbonates, citrates), while

FIGURE 5.5
HALF-LIVES AND RADIATION EMISSION PATTERNS
RESULTING FROM DECAY OF URANIUM-238

the remaining 40 percent binds to the plasma protein transferrin (Stevens et al., 1980). Following oral exposure in humans, more than 90 percent of uranium is excreted in the feces and not absorbed in the gastrointestinal tract. Of the small percentage 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. These data are based on animal studies by Ballou et al. (1986), Leach et al. (1984), and Sullivan et al. (1986). A small portion of the absorbed uranium is retained for a longer period.

Environmental sources of uranium

Uranium is an ubiquitous element, present in the earth's crust at approximately 4 parts per million. Uranium concentrations in samples of ground water and surface water worldwide 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 micrograms per kilogram ($\mu\text{g/kg}$) fresh plant material (Tracy et al., 1983).

The main dietary source of natural uranium for the general population is from bakery goods, potatoes, meat, and fresh fish, which may contain uranium concentrations between 10 and 100 $\mu\text{g/kg}$ (Prister, 1969). The total dietary uranium intake from the consumption of average foods is approximately 1 μg per day; additionally, approximately 20 to 50 percent of this total can come from drinking water. Cereals and vegetables, particularly root crops, probably contribute most to the daily uranium intake (Berlin and Rudell, 1986).

Uranium toxicity

Exposure of the general public to natural uranium is unlikely to pose an immediate lethal threat to humans. No human deaths reported have been definitely attributable to uranium ingestion; therefore, no lethal dose has been determined for humans. Lethal doses of uranium ($\text{LD}_{50,23}$) have been reported to be as low as 14 mg/kg-day following 23-day oral exposures, depending on both the solubility of the uranium compound (higher solubility compounds have greater toxicity), route of exposure, and the animal species tested. High uranium doses 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 from studies on experimental animals indicate that the kidney is the critical organ for chronic uranium toxicity, specifically the proximal tubule (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 to cause moderate renal damage was given to rabbits in their diet at 2.8 mg/kg-day (Maynard and Hodge, 1949). Figure 5.6 summarizes the health effects of uranium as a function of dose.

5.1.6 Vanadium

Absorption

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

Tissue accumulation and clearance

Vanadium is found in all body tissues in concentrations ranging from 0.08 μg per gram wet weight in spleen tissue to 0.14 μg per gram in brain and heart tissue and 0.33 μg per gram in aorta tissue (Yakawa and Suzuki-Yasumoto, 1980). Vanadium concentrations in human blood serum are reported to be 0.016 to 0.939 nanograms per milliliter (ng/mL). Different authors report vanadium concentrations in hair ranging from 20 to 60 ng per gram, with higher values found in manic-depressive patients than in normal control groups (57 versus 29 ng per gram, respectively).

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

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

FIGURE 5.6
URANIUM TOXICITY RANGES

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

Environmental sources of vanadium

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

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

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

Although vanadium is considered an essential element for chickens and rats, human dietary requirements are uncertain. For animals, the daily requirement is about 10 to 25 μg per day (Pennington and Jones, 1987).

Vanadium toxicity

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

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

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

Tank and Storvick (1960) report a relationship between the concentration of vanadium in drinking water and the incidence of dental caries in children. Dental caries incidence in children aged 7 to 11 years was reduced three times (compared to controls) by applying ammonium vanadate in glycerol to the teeth (Belehova, 1969). Other studies (Hadjimarkos, 1966; 1968) did not find this relationship.

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

Although animal studies have reported impaired conditioned reflexes following doses of vanadium from 0.05 (6 months) to 0.5 (21 days) mg/kg-day, effects on the nervous system have not been reported following repeated oral vanadium administration in humans. Workers exposed by inhalation to fairly high concentrations of vanadium compounds (milligrams per cubic meter) report nonspecific symptoms, including headache, weakness, vomiting, nausea, and ringing of the ears (WHO, 1988). Available data on vanadium toxicity are insufficient to evaluate effects on cholesterol levels, iron metabolism, blood cell production, and mutagenesis. However, due to poor absorption from the gut, the metal is not considered very toxic following oral administration (WHO, 1988).

Figure 5.7 summarizes the health effects of vanadium as a function of dose.

5.2 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 response variability from sensitive individuals. These values are updated quarterly and published in the Health Effects Assessment Summary Tables (HEAST) (EPA, 1992b). Following more extensive review, they also are provided through the EPA's IRIS data base. Table 5.1 summarizes the most recent oral RfDs for the noncarcinogenic contaminants of concern.

FIGURE 5.7
VANADIUM TOXICITY RANGES

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
Nitrate	7.0 ^a	High	Methemoglobinemia, hematologic	Water/IRIS	1
Strontium	0.6	Medium	Bone	Water/IRIS	300
Sulfate	NA	High	Diarrhea	Water/NA	1
Uranium (soluble salts)	0.003	Medium	Kidney, decreased body weight	Water/IRIS	1000
Vanadium	0.007	Low	Hair	Water/HEAST	100
Manganese	0.005	Medium to low	Central nervous system	Water/IRIS	1

^aValue presented as nitrate; nitrate-nitrogen RfD is 1.6 mg/kg-day.

NA - Not available.

The EPA currently classifies all radionuclides as Group A, or known human carcinogens, based on their property of emitting ionizing radiation and on the evidence provided by epidemiological studies of radiation-induced cancer in humans. At sufficiently high doses, ionizing radiation acts as a complete carcinogen (both initiator and promoter), capable of increasing the probability of cancer development. However, the actual risk is difficult to estimate, particularly for the low doses and dose rates encountered in the environment. Most reliable data were obtained under conditions of high doses delivered acutely. It is not clear whether cancer risks at lower doses are dose-proportional (i.e., the linear dose-response hypothesis) or whether the risk is greatly reduced at low doses and rates (the threshold hypothesis). A conservative assumption is that no threshold dose exists below which there is no additional cancer risk. Risk factors are published in HEAST and IRIS for correlating intake of carcinogens over a lifetime with the increased excess cancer risk from that exposure. Table 5.2 gives the most recent cancer slope factors for uranium-234 and -238.

Table 5.2 Toxicity values: carcinogenic effects

Parameter	Oral SF (pCi ⁻¹)	Weight of evidence classification	Type of cancer	SF basis/SF source
Uranium-238	1.6×10^{-11}	A	a	Water/HEAST
Uranium-234	1.6×10^{-11}	A	a	Water/HEAST

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

SF - Slope factor.

A - EPA classification for known human carcinogens.

6.0 RISK EVALUATION

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

6.1 POTENTIAL NONCARCINOGENIC HEALTH EFFECTS

The results from the exposure assessment showing either the highest intake-to-body-weight ratios (or highest doses) or the toxicologically most sensitive group are used to evaluate potential health effects for noncarcinogens. For manganese, strontium, uranium, and vanadium, the highest intake-per-body-weight group is children 1 to 10 years old. For nitrate and sulfate, infant exposures are used to evaluate health risks because this is the toxicologically sensitive population.

As discussed in Sections 3.2 and 3.3, limited recent data show elevated concentrations of sulfate in domestic well 640 and elevated uranium in wells 640 and 619. These concentrations exceed those typically found in the plume. Because only limited data were available from wells 640 and 619, distributions of expected intake could not be derived. Therefore, expected intake values for sulfate ingestion from well 640 and uranium ingestion from wells 640 and 619 have been calculated using the appropriate simulation distribution means for daily water ingestion rates and average body weight and the maximum contaminant concentration observed in the June or December 1993 samples. These values are indicated on the simulated distributions (Figures 6.1, 6.2, and 6.3). If these wells were used as a drinking water source, the exposure distribution might be expected to shift, relative to the distribution presented, in the same direction as the expected intake value for these wells. That is, because the expected value for well 640 is lower than the expected value of the simulated distribution, use of well 640 as the sole drinking water source would be expected to shift the exposure range to the left (lower exposures) as well.

The most significant potential health risk associated with drinking contaminated ground water at the Monument Valley mill site is from nitrate. As seen in Figure 6.4, if contaminated ground water were used for drinking water, more than 50 percent of the expected exposures would be above the potentially lethal level for infants. Some degree of methemoglobinemia is expected with any infant consumption of ground water, with more than 70 percent of the predicted exposure range falling above the severe toxicity level.

NOTE: LAXATIVE EFFECTS MANIFESTED AS DIARRHEA IN INFANTS COULD OCCUR EVEN WITHIN THE LOWER RANGE OF PREDICTED EXPOSURES, WITH SEVERE DEHYDRATION IN INFANTS OCCURRING AT HIGHER PREDICTED EXPOSURES.

FIGURE 6.1
HEALTH EFFECTS OF PREDICTED SULFATE EXPOSURE RANGES FOR INFANTS
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 6.2
HEALTH EFFECTS OF PREDICTED URANIUM EXPOSURE RANGES FOR CHILDREN
MONUMENT VALLEY UMTRA PROJECT SITE

NOTE 1: THE AVERAGE PREDICTED EXPOSURE IS RELATED TO AN INCREASED CANCER RATE OF 0.0009%.

NOTE 2: EXPOSURE IS ASSUMED TO BE HALF U-234 AND HALF U-238.

FIGURE 6.3
INCREASED LIFETIME CANCER RISK ASSOCIATED WITH PREDICTED RADIATION EXPOSURES
RESULTING FROM INGESTION OF URANIUM IN GROUND WATER
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 6.4
HEALTH EFFECTS OF PREDICTED NITRATE EXPOSURE RANGES FOR INFANTS
MONUMENT VALLEY UMTRA PROJECT SITE

The levels of nitrate associated with lethal cases of methemoglobinemia vary considerably. A major cofactor in nitrate toxicity may be the presence of bacterial contamination of the ground water, thus increasing reduction to nitrite in the gut. This factor has not been evaluated in Monument Valley ground water. A second significant factor is that infants prone to gastrointestinal distress seem more sensitive to the toxic effects of nitrates. For this reason, the gastrointestinal effects associated with sulfate exposures could increase nitrate toxicity. However, sulfate-induced diarrhea also 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 6.1 shows that much of the exposure distribution for sulfate is above the range where mild diarrhea is expected and nearly 50 percent of the exposures are above the range of severe diarrhea. The toxicity of nitrate and sulfate requires direct action in the gut. Because the predicted nitrate and sulfate toxicity from drinking water is so severe and because this is the primary ingestion pathway for infants, the contribution from other pathways would not alter the interpretation of health risks. Therefore, it will not be evaluated at this time. Children drinking water from domestic well 640 would have an average sulfate intake of 24 mg/kg-day. No nitrate exposure would occur from this source, so only mild, if any, diarrhea is expected from this exposure.

It is also important to note that the exposure distribution for infants is based on tap water intake rates across a population that includes breast-fed and canned formula-fed infants. Those infants consuming powdered formula reconstituted with well water would be in the upper percentiles of this exposure distribution and could be at high risk for severe diarrhea and methemoglobinemia. Furthermore, these effects are expected after very short-term exposures.

The predicted exposure to manganese from ingestion of contaminated ground water would exceed the EPA RfD (Figure 6.5) for drinking water in 11 percent of the cases. These predicted exposures could produce neurological symptoms similar to those seen in the early stages of Parkinson's disease, but these symptoms are slowly reversible after manganese exposure ceases (Section 5.1.1). Because the background intake of manganese is so close to the RfD, this additional source should be viewed as additive, shifting the entire distribution upward by the amount of background intake. This would result in 50 percent of expected exposures at or greater than the RfD. The RfD for manganese is derived from a study of human drinking water ingestion and therefore would be expected to more closely reflect actual toxicity than values extrapolated from data on other species or other exposure routes. There has been a suggestion that infants may be more sensitive to manganese toxicity than data shown here indicate, but data to evaluate that possibility are not available. Because drinking water ingestion is the most serious exposure route, the 2-percent additional contribution from other pathways would not alter the interpretation, nor would exposure from other pathways be likely to produce toxicity if the drinking water pathway were eliminated.

FIGURE 6.7
HEALTH EFFECTS OF PREDICTED VANADIUM EXPOSURE RANGES FOR CHILDREN
MONUMENT VALLEY UMTRA PROJECT SITE

For the remaining contaminants of concern at Monument Valley, very few if any adverse effects are anticipated from chronic ground water ingestion. For strontium (Figure 6.6) and vanadium (Figure 6.7), more than 99 percent of the exposure distribution falls below the EPA-derived oral RfD, and the remaining portion of these distributions is well below any toxic effects observed in humans. For uranium (Figure 6.2), more than 95 percent of the predicted exposure range is below the EPA oral RfD, and the entire distribution is well below any toxic effects in humans. Using the average distribution values for water ingestion and adult body weight and the maximum observed sample values, expected uranium exposures from ingestion of water from wells 619 and 640 can be calculated. If well 619 were used as a water source, the expected ingestion would fall in the upper half of the simulated distribution, but would be below the EPA oral RfD. (Note that the predicted distribution of potential intakes from this source would be shifted to the right, or in the direction of increased toxicity.) Exposure to uranium if well 640 were used as a drinking water source would result in an expected intake of 0.0003 mg/kg-day, shifting the predicted exposure to the left, or in the direction of lower toxicity. Because of the very low concentrations and therefore low predicted toxicity for these metals from drinking water, the minor exposure pathways presented in Section 4.0 is not expected to contribute significantly to toxicity at this site. For example, in the unconsolidated aquifer, screening calculations indicated that ingestion of produce contributed 15 percent of the strontium exposure obtained from water ingestion. Because the toxicity of strontium even from water ingestion was so low, the additional contribution from produce ingestion would not shift the distribution into a range expecting adverse health effects. Therefore, no further evaluation of this secondary pathway was considered.

As stated previously, diabetics and the elderly may be more sensitive to uranium toxicity due to their already impaired kidney function. Therefore, toxic effects may appear in these sensitive individuals at lower levels than those presented in Section 5.0.

Although some of these constituents have elevated levels in the deeper aquifers, no risk simulation has been conducted based on data from these aquifers. The concentrations of these constituents in the Shinarump were 2 percent of the unconsolidated aquifer value for nitrate, 10 percent for sulfate, and 20 percent for strontium. Therefore, the health effects from ingestion of the Shinarump concentrations of these constituents are expected to be minimal.

Although uranium concentrations from wells 614 in the Shinarump and 657 in the De Chelly exceeded concentrations observed in the unconsolidated aquifer, risk calculations have not been conducted based on these wells. Because the source of these elevated uranium levels is undetermined, further evaluation of this issue will be considered in the ground water phase of the UMTRA Project.

NOTE: NO OBSERVED ADVERSE HEALTH EFFECTS HAVE BEEN ASSOCIATED WITH THE PREDICTED RANGE OF EXPOSURES.

FIGURE 6.6
HEALTH EFFECTS OF PREDICTED STRONTIUM EXPOSURE RANGES FOR CHILDREN
MONUMENT VALLEY UMTRA PROJECT SITE

FIGURE 6.7
HEALTH EFFECTS OF PREDICTED VANADIUM EXPOSURE RANGES FOR CHILDREN
MONUMENT VALLEY UMTRA PROJECT SITE

6.2 POTENTIAL CARCINOGENIC HEALTH EFFECTS

All uranium isotopes are radioactive and, as such, are considered potential carcinogens. Figure 6.3 shows the exposure distribution for uranium intake from the most contaminated well in the unconsolidated aquifer and the potential lifetime carcinogenic risk associated with these drinking water exposures. These estimates are 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. Approximately 80 percent of the exposure distribution for ground water ingestion of uranium falls below the National Contingency Plan guidance for maximum increased lifetime cancer risk of 1×10^{-4} . The expected exposure value (549,000 pCi per lifetime) results in an excess lifetime cancer risk of 9×10^{-6} . The distribution presented here is considered conservative because it is based on cumulative 70-year exposure durations. As discussed previously, this exposure duration is probably appropriate, but ground water uranium concentrations resulting from processing at this site are expected to decline now that the tailings have been removed. Therefore, this distribution would overestimate risk.

If wells 619 or 640 were used as a drinking water source, the increase in lifetime cancer risk would be 4×10^{-5} or 7×10^{-6} , respectively. Both of these values would fall below the National Contingency Plan guidance value.

Uranium is the only radionuclide statistically above background in the plume. No other progeny have been statistically above background levels. Therefore, no evaluation of uranium decay products was needed.

6.3 LIMITATIONS OF RISK EVALUATION

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

- The risk assessment evaluates only risks related to inorganic ground water contamination. Potential contamination with any of the few organic constituents used in uranium processing has not been addressed.
- Subpopulations that might have increased sensitivity are not specifically addressed on the graphs.
- Some individuals may be more sensitive to the toxic effects of certain constituents for undetermined reasons.
- Data available to interpret potential adverse health effects are not always sufficient to allow accurate determination of all health effects (i.e., lack of testing in humans or testing of dose ranges other than those expected at this site).

- Although plume movement is evaluated hydrologically and geochemically, the monitoring locations sampled may not be the most contaminated portion of the plume.
- Only the drinking water exposure pathway has been considered in depth, although other pathways have been screened to determine their contribution.

The evaluation presented here has considered these limitations and compensated wherever possible by presenting toxicity ranges rather than point estimates to incorporate as much variability as could be reasonably defined. Section 8.2 discusses the impact of these potential limitations more fully.

7.0 LIVESTOCK AND ENVIRONMENTAL EVALUATION

The objective of this section is to assess the potential for site-related contaminants to adversely affect 1) the existing biological communities and 2) livestock grazing and other agricultural practices in the site area. Currently, the EPA has no guidance for quantifying potential ecological impacts of the release of hazardous constituents; the EPA has, however, developed a qualitative approach to be used in ecological risk evaluations (EPA, 1989). As part of this qualitative approach, the EPA recommends conducting ecological assessments in a phased approach because it ensures the most effective use of resources while ensuring performance of all necessary work (EPA, 1992d). This approach consists of four phases, with the analysis becoming more complex starting with the identification of potentially exposed habitats (phase 1), the collection of chemistry data (phase 2), the collection of biological samples (phase 3), and the performance of toxicity testing (phase 4). If the initial inspection of the habitats and the analysis of media samples indicate that there is no, or very low, potential for an ecological risk, the assessment will likely be complete. If the early phases of the assessment indicate that the contaminants may be adversely affecting ecological receptors, a higher level of analysis may be warranted.

The ecological risk assessment at the Monument Valley site consists of the first two phases of EPA's approach, because there is potential that habitats may have been impacted (phase 1) and water chemistry data collected for many years are available (phase 2). Water quality data from several years of sampling in the frog pond region (locations 621 and 622), plus limited recent near-surface water quality and sediment data from Cane Valley Wash (locations 623, 624, and 627), were used to prepare this assessment. This approach provides a screening level assessment of the risks associated with potential exposure to contaminated media at the site.

It is often difficult to determine if contaminants have affected the biological component of an ecosystem and to predict whether observed effects will result in damage to the ecosystem. However, even though the determination of effects of a contaminant on the ecosystem may be difficult, the sampling of environmental media such as surface water can be used to assess the possibility of an ecological risk occurring. For such an ecological risk to occur, a source of contamination must exist along with a pathway for the contamination to reach the biological communities. The following sections identify the areas of contamination and the potential pathways by which this contamination may be entering the aquatic and terrestrial biological communities at the Monument Valley site; the potential ecological receptors at the site; the contaminants of potential concern; and the potential hazards the contaminants of concern may pose to the ecological resources, livestock, and agricultural crops.

7.1 EXPOSURE CHARACTERIZATION

This section identifies the ecological resources at the site likely to be exposed to site-related contaminants and identifies the possible and probable exposure pathways. As noted in Section 4.0, exposure can occur only with both a

of contamination and a mechanism of transport to a receptor population or individual.

Contaminated media at the site included the two piles of tailings and associated contaminated soil and ground water. However, because the tailings and contaminated soil were stabilized in a disposal cell 17 mi (27 km) away, direct-exposure pathways (such as incidental ingestion of soil, dermal contact with soil, and inhalation of air containing particulates) will not represent a future ecological concern and will not be evaluated in this baseline risk assessment. Indirect exposure pathways (such as bioaccumulation or consumption of previously exposed organisms) are possible at the site.

Running south to north, the perennial surface water body nearest the tailings site is Cane Valley Wash, approximately 1500 ft east of the former lower tailings pile. A few areas of ponded water exist approximately 1500 to 2000 ft east-northeast of the lower tailings pile in the Cane Valley Wash floodplain (Figure 2.12). These ponds appear to have been excavated into Cane Valley Wash. These areas of ponded water are historically referred to as ephemeral, and the local residents call them the "frog ponds." As discussed previously, the plume of impacted ground water in the unconsolidated aquifer is traveling north-northeast up to 300 ft/year (100 m/year). Not enough data are available to determine the exact location of the plume or to predict when and where the plume could reach a surface expression point in Cane Valley Wash.

However, as discussed in Section 3.5.1, samples of near-surface water in Cane Valley Wash at sampling locations 624 and 627 show elevated levels of several plume constituents in addition to elevated TDS. Whether these elevated levels reflect intersection of the plume with the wash has not yet been determined. Investigations are under way to determine the source of this contamination. Potential ecological impacts of these constituents will be evaluated here.

Based on the frog ponds' location and the ground water flow direction, it is unlikely that site-related ground water has or will discharge to the frog ponds (Figure 2.7). The ponds are crossgradient to the site and are fed by a spring and seeps. The source of this spring and seep water is probably the unconsolidated aquifer from the east and south; therefore, former site activities probably do not affect this water source. The water quality has been monitored for several years in two ponds (sampling locations 621 and 622) and will be discussed in Section 7.2. Based on water quality data, no evidence suggests site activities have contaminated these ponds.

A potential current pathway involves plant uptake of contaminants in ground water. Because of the shallow depth to ground water (approximately 10 ft [3 m] or less below land surface), some plants could access contaminated ground water and near-surface water now or in the future as the plume moves downgradient. Plant uptake is evaluated in this risk assessment, assuming the plant roots accessed ground water containing the 95-percent UCL of the median concentrations for the contaminants elevated above background (column 1 of

Table 3.4). For near-surface water in Cane Valley Wash, maximum observed concentrations will be evaluated due to the limited data set. Because ecological effects differ from human health effects, the complete list of contaminants will be contaminants of potential concern used in evaluating ecological impacts in this section.

Because the local population uses the Cane Valley Wash floodplain to graze livestock (sheep, cattle, horses, and goats), this risk assessment evaluates potential impacts to these receptors. These include ingestion of plants with roots in contaminated ground water and ingestion of contaminated ground water by wildlife and livestock if a well were dug in the future. A well might also be used to irrigate agricultural crops and/or create a pond, which could be stocked with fish.

7.2 ECOLOGICAL RECEPTORS

A description of the ecological receptors at or in the vicinity of the Monument Valley UMTRA site is based on information compiled in previous DOE reports (DOE, 1989; 1993). No recent ecological field surveys have been conducted.

The dominant plant species observed in the desert shrub habitat adjacent to the tailings piles are greasewood, shadscale, broom snakeweed, and soapweed. These species (plus black brush, cliffrose, small Utah juniper, and singleleaf ash) occupy the rocky terrain south and west of the site. Rabbitbrush, vanclevea, and Russian thistle (EES, 1986) dominate the blow sand areas just west of the former tailings pile.

The former tailings site and adjacent areas (Baucom, 1985) do not include wetlands or riparian habitat (as defined by the U.S. Fish and Wildlife Service [FWS]) (Cowardin et al., 1979) .

A total of 19 species of amphibians and reptiles may occur in the site area. Observed species include the side-blotched, western whiptail, leopard, desert spiny, and sagebrush lizards. An estimated 42 species of nesting birds may occur near the site. Seventeen species were observed near the site, the black-throated sparrow and rock wren being the most common. Numerous ducks were observed in the frog ponds' vicinity. Twenty-six species of mammals may reside near the Monument Valley site; the black-tailed jackrabbit, desert cottontail, and white-tailed antelope squirrel were observed (Burt, 1985; FBD, 1983). Big game species are not known to occur at or near the site.

A list of threatened and endangered species and other species of concern that may occur in the tailings site area was developed through consultation with the FWS and the Navajo Nation (Baucom, 1985; Diswood, 1985; House, 1985; Ruesink, 1985). The consultation process resulted in the identification of 13 species (nine species of wildlife and four species of plant life). It was determined that no threatened and endangered wildlife species would likely occur near the site.

Suitable habitat for three of the four threatened and endangered plant species (Cutler milkweed, Monument Valley milkvetch, and *Eremocrinum alvomarginatum*) occur in the site area. However, no species were observed at or near the Monument Valley site in 1986 (EES, 1986).

As described previously, the water quality of the frog ponds (locations 621 and 622) is considered representative of background conditions because their water source is ground and surface water unaffected by the tailings site. However, for this baseline risk assessment, the detected concentrations for the contaminants of potential concern are compared with available water quality values, which include the federal water quality criteria (FWQC) (Table 7.1). The FWQC are designed to protect freshwater aquatic life from chronic exposure (EPA, 1986). If no FWQC were available for a particular contaminant, other available comparison values were reported.

A comparison of the historical surface water data from the frog ponds with available water quality values indicates that no concentrations exceeded their respective comparison criteria at both locations.

No comparison water quality values were available for many of the contaminants of potential concern. However, this water represents background water quality and not mill- or tailings-related contamination. Therefore, it is unlikely the detected concentrations represent an ecological concern.

7.3 POTENTIAL IMPACT ON WILDLIFE AND PLANTS

A potential exposure pathway involves terrestrial vegetation. Terrestrial vegetation can be directly exposed to contaminants in ground water and near-surface water 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 (e.g., pH, moisture, redox potential, organic matter, plant sensitivity, input-output balance, and cumulative effects). Foraging wildlife can be exposed to contaminants in ground water by ingesting plants that have bioconcentrated certain contaminants.

Although visual observation of vegetation (June 1994) indicated no gross evidence (chlorosis disease) of impact to plants, uptake of contaminants in ground water may be occurring at the Monument Valley UMTRA site. However, based on the shallow depth to contaminated ground water at the site (approximately 10 ft below land surface, or less for near-surface water), the rooting zones of some plants possibly could intercept contaminated ground water and near-surface water. The rooting depths for some of the types of plants occurring in this area (i.e., desert shrub community) reportedly reach maximum depths of approximately 30 ft below land surface in the Los Alamos, New Mexico, vicinity (Foxy and Tierney, 1986). Although no site-specific data exist for rooting depths, plant uptake of contaminants in ground water and near-surface water may represent a complete exposure pathway at the site.

Table 7.1 Comparison of contaminants of potential concern in frog pond water with available water quality values

Contaminant of potential concern	Median concentration in frog pond water (mg/L)		Aquatic life water quality value ^a
	Location 621	Location 622	
Ammonium	0.1	0.1	NA
Calcium	41	38	NA
Chloride	11	27	230 ^b
Iron	0.07	0.05	1.0
Magnesium	31	66	NA
Manganese	0.05	0.03	1.5 ^c
Nitrate	1.0	3.7	90 ^d
Phosphate	0.1	0.2	NA
Potassium	4.4	11	NA
Silica	8.2	15	NA
Sodium	47	390	NA
Strontium	0.42	0.40	NA
Sulfate	54	400	NA
Uranium	0.018	0.0052	NA
Vanadium	0.015	0.018	NA
Zinc	0.005	0.034	0.25 ^e

^aFWQC in mg/L for the protection of freshwater aquatic life from chronic exposure (EPA, 1986), unless specified otherwise.

^bFrom EPA (1992c).

^cValue presented represents 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).

^eWater hardness-related chronic FWQC (EPA, 1986).

NA - Not available.

Concentrations of the contaminants of potential concern in plant tissue are estimated using soil-to-plant bioconcentration factors. No soil data are available for this site. Table 7.2 presents the methodology and parameters used to estimate soil concentrations, root uptake, and plant tissue concentrations for the contaminants of potential concern. This methodology is described in detail elsewhere (Baes et al., 1984) and therefore will not be discussed here.

The estimated plant tissue concentrations for contaminants of potential concern in the vegetative portions (e.g., stems, leaves) and in the nonvegetative portions (e.g., fruits, tubers) of plants at harvestable maturity were compared with approximate concentrations (in mature leaf tissue) that are reportedly toxic to plants (phytotoxic) (Table 7.2). As shown in Table 7.2, few available data relate tissue concentrations to phytotoxicity. Note that the reported phytotoxic concentrations are not representative of very sensitive or highly tolerant plant species. The estimated contaminant of potential concern tissue concentrations do not exceed the available phytotoxicity data. However, no comparison data were available for calcium, chloride, iron, magnesium, nitrate, potassium, strontium, sodium, sulfate, and uranium. Therefore, it is not possible to determine if the estimated tissue concentrations could result in adverse effects to plants without further study.

Bioaccumulation in terrestrial organisms as a function of contaminants of potential concern in ingested plants is a potential exposure pathway at the site. Birds and other vertebrates consuming these plants can bioaccumulate some contaminants of potential concern in their 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. When the contaminated areas are small, the amount of food in the animals' diet usually exceeds the amount of contaminated food and bioaccumulation is not a concern. Therefore, exposure through diet for all trophic level species is possible in certain areas, but the potential for bioaccumulation is not always of concern.

Biomagnification is a more severe situation in which the concentration of a constituent increases in higher levels of the food chain because contaminant concentrations are accumulated through each successive trophic level. Of particular concern for biomagnification effects are the top predators, especially the carnivorous birds and mammals. Only a limited number of constituents can magnify in the food chain because most constituents are metabolized and eliminated at each level of the food chain. Based on available information, the potential for the contaminants of potential concern to represent a concern through 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 definitively determine the bioaccumulation or biomagnification of site-related contaminants of potential concern.

To evaluate the future hypothetical impact on wildlife of using contaminated ground water or near-surface water in a livestock pond (i.e., to animals drinking

Table 7.2 Comparison of estimated plant concentrations to phytotoxic concentrations

Constituent	Ground water UCL; near- surface maximum ^a	Kd	Estimated soil concentration from ground water; from near- surface water (mg/kg DW)	Soil to plant concentration factors		Estimated concentration in vegetative growth from ground water; from near- surface water (mg/kg DW) ^b	Estimated concentration in fruits/tubers from ground water; from near- surface water (mg/kg DW) ^c	Approximate concentration in mature leaf tissue that is toxic (mg/kg DW) ^d
				Bv	Br			
Calcium	450; 47	4	1800; 190	3.5	0.35	6300; 670	630; 67	NA
Chloride	35; 940	0.25	8.8; 240	70	70	620; 17,000	620; 17,000	NA
Iron	0.07; NA	25	1.8; NA	0.004	0.001	0.007; NA	0.00175; NA	NA
Magnesium	320; 82	4.5	1440; 370	1	0.55	1440; 370	790; 200	NA
Manganese	0.17; 0.62	65	11; 40	0.25	0.05	2.8; 10	0.55; 2	400-1000
Nitrate ^e	1200; ND	0.1	120; NA	30	30	3600; NA	3600; NA	NA
Potassium	34; 73	5.5	190; 400	1	0.55	190; 400	105; 220	NA
Sodium	76; 9850	100	7600; 985,000	0.075	0.055	570; 74,000	420; 54,000	NA
Strontium	3.1; 0.43	35	110; 15	2.5	0.25	280; 38	28; 3.8	NA
Sulfate ^f	2900; 5990	7.5	22,000; 45,000	0.5	0.5	11,000; 23,000	11,000; 23,000	NA
Uranium	0.029; 0.22	450	13; 99	0.0085	0.004	0.11; 0.84	0.052; 0.40	NA
Vanadium	0.02; 0.31	1000	20; 310	0.0055	0.003	0.11; 1.7	0.06; 0.93	5-10
Zinc	0.017; 0.081	40	0.68; 3.2	1.5	0.9	1.0; 4.8	0.61; 2.9	100-400

^aMaximum observed concentration, location 627.

^bEstimated concentration in vegetative portions; calculated as estimated soil concentration multiplied by Bv.

^cEstimated concentration in nonvegetative portions; calculated as estimated soil concentration multiplied by Br.

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

^eBv 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.

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

Note: No Kd, soil-to-plant concentration factors, or phytotoxicity information is available for ammonium or silica.

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

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

DW - Dry weight.

Kd - Soil-water distribution coefficient (Baes et al., 1984).

NA - Not available.

ND - Not detected.

UCL - Upper 95% one-tailed confidence interval of the median.

from the pond or fish stocked in the pond), the 95-percent UCL ground water contaminant of potential concern concentrations were compared with available comparison water quality criteria (Table 7.3). Table 7.4 evaluates near-surface water (maximum concentrations from 627) against these same criteria. No available federal (or state) criteria or standards protect terrestrial wildlife from exposure to contaminated water. Available surface water quality criteria to protect freshwater aquatic life include the FWQC (EPA, 1986).

The 95-percent UCL aquatic life ground water concentrations for nitrate and TDS exceed the comparison aquatic life water quality values (Table 7.3), while the ground water concentrations for chloride, iron, manganese, and zinc are below the comparison values. No comparison water quality values are available for ammonium, calcium, magnesium, phosphate, potassium, silica, sodium, strontium, sulfate, uranium, or vanadium.

Concentrations of chloride and TDS in the near-surface water exceed those that are protective for chronic exposure of aquatic life. Near-surface concentrations of other constituents do not exceed available comparison values. However, for the great majority of constituents, no reference values are available.

Although chloride and TDS concentrations in this near-surface water would be toxic to aquatic organisms (Table 7.4), the water is below ground surface, and the organisms probably will not contact this water unless increased rain raises the water table sufficiently for surface expression. In such a case, constituent concentrations should be reduced through dilution. Because the effect of dilution cannot be predicted, additional monitoring of the wash during periods of surface expression would be required to fully evaluate the impact to aquatic organisms, which may occur in the wash when water is present. Note that because surface flow in this portion of the wash is intermittent and dry much of the year, a diverse assemblage of aquatic organisms occurring in the wash is unlikely. Any aquatic organisms in this reach of the wash probably would be migrants from areas containing existing populations. Thus, even if surface expression were to occur, the potential for aquatic organisms to be exposed is considered low, and even if exposure occurred it probably would be of short duration (due to receding water levels through evaporation after rainfall).

Another future hypothetical use of ground water in the area is the irrigation of agricultural crops. Tables 7.3 and 7.4 compare the approximate concentrations in irrigation water using ground water or near-surface water, respectively, with guidelines designed to protect plants (EPA, 1972). Five contaminants of potential concern (iron, manganese, TDS, vanadium, and zinc) have comparison criteria; the 95-percent UCL ground water concentrations for these five contaminants of potential concern are below these guidelines (Table 7.3), with the exception of TDS, which falls in the potentially toxic range. The concentration of sodium in the Cane Valley Wash near-surface water probably would be phytotoxic to all but salt-tolerant plants. The high TDS concentration in this water makes it unusable as an irrigation source (EPA, 1972). Long-term use of this water would cause an excessive absorption of salts in the soil and

Table 7.3 Comparison of contaminants of potential concern in ground water with available water quality values

Contaminant of potential concern	Ground water UCL (mg/L)	Aquatic life water quality value ^a (mg/L)	Water concentration protective of livestock ^b (mg/L)	Concentration in irrigation water protective of plants ^b (mg/L)
Ammonium	370	NA	NA	NA
Calcium	450	NA	NA	NA
Chloride	35	230 ^c	NA	NA
Iron	0.07	1	NA	5.0
Magnesium	320	NA	NA	NA
Manganese	0.17	1.5 ^d	NA	0.20
Nitrate	1200	90 ^e	100	NA
Phosphate	0.6	NA	NA	NA
Potassium	34	NA	NA	NA
Silica	23	NA	NA	NA
Sodium	76	NA	NA	NA
Strontium	3.1	NA	NA	NA
Sulfate	2900	NA	1000	NA
TDS	4300	15,000 ⁱ	10,000 ^g	2000-5000 ^h
Uranium	0.029	NA	NA	NA
Vanadium	0.02	NA	0.1	0.10
Zinc	0.017	1.5 ^f	25	2.0

^aFWQC for protection of freshwater aquatic life via chronic exposure (EPA, 1986) unless specified otherwise.

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

^cFrom EPA (1992c).

^dValue presented represents the lower end of the tolerance range for fresh water aquatic life (EPA, 1986).

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

^fWater hardness-related chronic FWQC (EPA, 1986). Criterion presented was calculated using the average hardness (2340 mg/L) determined from concentrations of calcium and magnesium in alluvial aquifer wells.

^gThe risks with these highly saline waters are so great that they cannot be recommended for use as livestock drinking water under any conditions (NAS, 1974).

^hIrrigation water that can be used for salt-tolerant plants on permeable soils with careful management practices and only occasionally for more sensitive crops (Follett and Soltanpour, 1985).

ⁱNo state or federal standards or criteria are available. Value presented represents the acceptable dissolved solids level for most aquatic organisms in freshwater habitats (Rawson and Moore, 1944; NTAC, 1968).

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

NA - Not available.

TDS - Total dissolved solids.

Table 7.4 Comparison of contaminants of potential concern in near-surface ground water in Cane Valley Wash with available water quality values, Monument Valley UMTRA Project site

Contaminant of potential concern	Concentration at 627 (mg/L)	Aquatic life water quality value ^a (mg/L)	Water concentration protective of livestock ^b (mg/L)
Ammonium	1.6	NAV	NAV
Calcium	46.7	NAV	NAV
Chloride	938	230 ^d	NAV
Iron	NA	1.0	NAV
Magnesium	82	NAV	NAV
Manganese	0.62	1.5 ^e	NAV
Nitrate	ND	90 ^f	100
Potassium	73	NAV	NAV
Silica	133	NAV	NAV
Sodium	9850	NAV	NAV
Strontium	0.43 ^c	NAV	NAV
Sulfate	5990	NAV	1000
TDS	26,600	15,000 ⁱ	10,000 ^h
Vanadium	0.31 ^c	NAV	0.1
Uranium	0.217	NAV	NAV
Zinc	0.081	0.38 ^g	25

Note: Phosphate was not analyzed for in the sample.

^aValues shown are FWQC for the protection of freshwater aquatic life via chronic exposure (EPA, 1986), unless specified otherwise.

^bFrom EPA (1972), unless specified otherwise.

^cAlthough concentration is elevated above plume values, it does not exceed 98-percent confidence interval for concentrations at locations 621 and 622 (frog ponds).

^dFrom EPA (1992).

^eValue presented represents the lower end of the tolerance range for freshwater aquatic life (EPA, 1986).

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

^gWater hardness related chronic FWQC (EPA, 1986).

^hThe risks with these highly saline waters are so great that they cannot be recommended for use as livestock drinking water under any conditions (NAS, 1974).

ⁱNo state or federal standards or criteria are available. Value presented represents the acceptable dissolved solids level for most aquatic organisms in freshwater habitats (Rawson and Moore, 1944; NTAC, 1968).

NA—Not analyzed.

NAV—Not available.

ND—Not detected.

TDS—Total dissolved solids.

would eventually result in phytotoxicity (EPA, 1972). No comparison values are available for the remainder of the contaminants of potential concern. Therefore, it is not possible to evaluate the potential of these compounds to adversely affect plants through irrigation water. The available information suggests that using ground water and near-surface water as a continuous source of irrigation water would result in deleterious effects to plants.

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 the ground water plume well and the maximum concentration in the near-surface water is 5000 micromhos per centimeter ($\mu\text{mhos/cm}$) and 15,730 $\mu\text{mhos/cm}$, respectively. These values are well above the upper end of the acceptable range for agricultural crops, 3000 $\mu\text{mhos/cm}$ (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 effect on the physical characteristics of soils (e.g., increase in colloiddally absorbed sodium, resulting in hard compact soil). The sodium hazard of water is expressed as the sodium adsorption ratio (SAR), which is calculated as the proportion of sodium to calcium plus magnesium in the water. Using the 95-percent UCL ground water concentrations and the maximum concentrations in near-surface water for sodium, calcium, and magnesium, a SAR of 1.6 and 230 were calculated, respectively. The SAR for the near-surface water (230) is well above the upper SAR limit of 10 (Follett and Soltanpour, 1985). Water with a SAR value greater than 10 should not be used as the sole source of irrigation water for long periods of time.

7.4 POTENTIAL IMPACT ON LIVESTOCK

Livestock have been observed grazing on vegetation growing in the Cane Valley Wash floodplain. Ingestion by livestock of vegetation that may have bioconcentrated contaminants from ground water or near-surface water is a potential pathway. However, without additional data (e.g., actual plant tissue concentrations, livestock exposure information), it is difficult to evaluate this exposure pathway.

Based on past and current agricultural activities in the area surrounding the site, ground water might be used in a livestock watering pond. In an attempt to evaluate the potential impact to livestock in this hypothetical scenario, Table 7.3 compares the 95-percent UCL ground water contaminant of potential concern concentrations with approximate drinking water concentrations considered protective of livestock (EPA, 1972). The 95-percent UCL ground water concentrations exceed the comparison water quality criteria for nitrate and sulfate while the 95-percent UCL concentrations for TDS (4300 mg/L),

vanadium (0.02 mg/L), and zinc (0.017 mg/L) are below the comparison criteria (0.1 mg/L and 25 mg/L, respectively). The ground water concentration of nitrate (1200 mg/L) is more than one order of magnitude above the comparison criteria (100 mg/L). If this ground water were the sole source of drinking water for livestock, it would ultimately result in death to the ruminants (e.g., cattle) from methemoglobinemia (Deeb and Sloan, 1975; NAS, 1972). No comparison water quality criteria have been reported for the remainder of the contaminants of potential concern. However, the available information suggests that using ground water containing the 95-percent UCL contaminant of potential concern concentrations as a source of drinking water for livestock would be unacceptable due to nitrate and sulfate concentrations.

As discussed previously, residents hand-dig wells during dry periods to obtain water for livestock. Using ground water below the surface of Cane Valley Wash as a drinking water source for livestock and possibly wildlife would likely result in severe diarrhea from the elevated sulfate concentrations. In addition, edema, physiological dehydration, and death would likely result from chronic ingestion of the elevated sodium levels, and the elevation in TDS is more than threefold greater than levels shown to be lethal to livestock (McKee and Wolf, 1963; NAS, 1974; EPA, 1972). The concentration of vanadium (0.31 mg/L) also exceeds the approximate concentration in water that is considered protective of livestock (Table 7.4).

7.5 LIMITATIONS OF THE ECOLOGICAL RISK ASSESSMENT

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

- Only a small amount of ecological data was collected during this screening.
- Little is known about site-specific intake rates for wildlife or amounts of contaminants taken up by plants. General literature values were used in many cases.
- Only limited ecotoxicological reference data are available.
- Considerable uncertainty is associated with the toxicity of mixtures of contaminants.

7.6 SUMMARY

Water quality values are insufficient to allow comprehensive evaluation of the effect of contaminated ground water or near-surface water on livestock and the environment. Available guidelines suggest no ecological threat exists to plants having roots in contact with the most contaminated ground water in the defined contaminant plume or the near-surface water. However, exposure to these waters through long-term irrigation would result in toxicity to plants. Sodium concentrations can contribute to the total salinity of an irrigation water and may be directly toxic to sensitive crops. However, the primary concern with elevated sodium concentrations is the adverse effect on the physical characteristics of soils (e.g., increase in colloiddally absorbed sodium, resulting in hard compact soil). Although high gypsum soils such as those found at Monument Valley tend to be less sensitive to soil degradation from high-sodium irrigation, the SAR for the near-surface water is more than an order of magnitude greater than reference values. Therefore, no protection would be expected. Water from the most contaminated wells in the currently defined contaminant plume would exceed available water quality values protective of aquatic life for nitrate; for the near-surface water in Cane Valley Wash, the concentrations of chloride and TDS exceed the aquatic life values. However, not enough data are available to adequately evaluate whether exposure to this ground water would actually result in adverse effects to aquatic life.

In the unconsolidated aquifer, watering livestock from the most contaminated wells would be unacceptable due to elevated nitrate and sulfate concentrations. The near-surface water would also not be acceptable as a long-term source of drinking water for livestock due to elevated concentrations of sulfate, TDS, and vanadium.

Consumption of plants that may have bioconcentrated certain contaminants from ground water and near-surface water in Cane Valley Wash is a potential exposure pathway to wildlife and domestic livestock. However, without additional data (e.g., tissue analysis, plant consumption rates, and frequency and duration of exposure), it is difficult to evaluate whether this pathway represents an ecological hazard.

Surface water in the frog ponds currently used by livestock, although fed by the unconsolidated aquifer, is upstream of and not impacted by the contaminant plume. Evaluation of this water suggests that no adverse effects to livestock or vegetation would result from using the water in the frog ponds.

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 Monument Valley site to evaluate the presence of these hazards. Because contaminated ground water is currently not used by area residents, human health is not at risk. However, health risks would be associated with potential future use of contaminated ground water.

The primary risks associated with ground water contamination at the Monument Valley processing site result from the ingestion of nitrate and sulfate by humans and other animals. Using ground water from the most contaminated portion of the plume for drinking purposes will 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, which may enhance the toxicity of nitrate in infants. The upper 11 percent of predicted exposures to manganese could produce neurological symptoms similar to those seen in early manifestations of Parkinson's disease.

Water below ground surface in Cane Valley Wash in the vicinity of the site is unsuitable as a drinking water source for livestock and wildlife and unsuitable for irrigation due to high concentrations of TDS, sulfate, and sodium. Chronic ingestion would produce lethal diarrhea and dehydration in animals and, if used for irrigation, would result in an unacceptable soil buildup of salts that would be phytotoxic to plants grown on those soils.

8.2 LIMITATIONS OF THIS RISK ASSESSMENT

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

- This document evaluates risks associated with exposures only to inorganic contaminants of ground water at the UMTRA site near Monument Valley. As discussed in Section 3.0, potential organic contaminants (those few related to uranium processing) have not been considered.
- The toxicity of any contaminant varies from person to person. For example, normal variability in biochemical factors among individuals, differences in medical history, previous exposures to toxicants, and dietary and exercise habits can all affect susceptibility to chemical toxicity. Using the probability distributions for potential exposure and presenting ranges of exposures that produce toxic effects, this assessment tries to emphasize that variability. However, it is not possible to account for all sources of variability and still present useful and meaningful analyses. Cases in which specific

subpopulations are known to be more sensitive to toxic effects of given constituents have been noted.

- To assess toxicity, standardized reference values developed by agencies such as the EPA are used to determine plant uptake, tissue concentrations in livestock, and toxic effects in humans. These reference values themselves have limitations including the following:
 - Not all constituents elevated above background at the site have toxicity, uptake, or bioconcentration data available.
 - In some cases, data obtained from laboratory animal testing at exposure doses different from those expected at the site were used to determine toxicity. The relationship between dose and response is not always linear, and humans do not always exhibit the same responses as animals.
 - Data used to determine toxicity are generally based on exposure to only the constituent of concern. In reality, exposures generally occur simultaneously to several chemicals. The interactive effects of multiple constituents and the impact of these interactions on expected toxicity generally cannot be accurately assessed from existing data.
- Although considerable effort has been directed at determining plume movement and placing monitor wells in locations that capture maximal contamination, variability in physical systems and models used to determine contaminant plume migration could still result in well placements that do not measure the highest contaminant concentrations or determine the fullest extent of plume impact.
- The results presented here are based on water samples filtered with an 0.45-micrometer filter. The effect of filtration differs for different elements. Filtered samples generally have lower or equal concentrations of a given constituent than unfiltered. Constituents in suspension could be lost with filtration, yet still produce toxic effects if ingested and broken down in the acidic environment of the stomach. Cases where filtration may have had an impact on ground water sample concentrations for a given constituent were considered in interpreting the results.
- Variability can be introduced through sampling and analytical processes. However, the data at UMTRA Project sites have been collected over many years and subjected to rigorous quality assurance procedures. The use of multiple samples introduces high confidence in the reliability and validity of the collected data.
- The drinking water pathway is considered the major determinant of exposure in this assessment. Although other pathways have been screened and determined not to contribute significantly to the total exposure, some of the

factors in these screening calculations have considerable uncertainties; e.g., dermal permeability coefficients and plant uptake factors. The additivity of exposure from these pathways could be significant. When a measurable contribution from other pathways could increase expected exposure significantly enough to alter the predicted toxicity, Section 6.0 notes the alternate source contribution.

By presenting ranges of toxic effects, summaries of available data on health effects and interactions, and outlines of potential limitations, this document provides a reasonable interpretation of potential health risks associated with ground water contamination at this site. This assessment presents both contamination and risk as accurately as possible, based on available data, and conveys areas of uncertainty.

8.3 GROUND WATER CRITERIA

In 1983, the EPA established health and environmental protection standards for the UMTRA Project; in 1987, the EPA proposed revised ground water standards. The UMTRA Project is required to adhere to the 1987 proposed ground water standards until final standards are published. The UMTRA ground water standards consist of ground water protection standards to evaluate disposal cell performance and ground water cleanup standards for existing contamination at processing sites. 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. The proposed standards also allow for supplemental standards or alternate concentration limits (ACL) where appropriate.

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

Nitrate concentrations in plume wells have consistently exceeded the UMTRCA MCL and EPA health advisory levels. Exceedance of the MCL for uranium has occurred in wells 619 and 657, although it is not clear if these uranium levels are related to the mill activities or to subsurface ore bodies. Selenium marginally exceeded the MCL in a 1989 sampling of plume wells; however, the concentrations of selenium are not statistically above background. The near-surface water in Cane Valley Wash (sampling location 627) also exceeds the uranium MCL. Both uranium and vanadium in these samples exceed the health advisory values for children and adults. Uranium values in the plume exceed the health advisory value for children.

Table 8.1 Concentration limits of constituents

Constituent	UMTRA MCL 40 CFR 192.02 (mg/L)	Health advisories 10-kg child, 10-day (mg/L)	Health advisories 70-kg adult lifetime (mg/L)
Chemicals (inorganic)			
Antimony	-	0.015	0.003
Arsenic	0.05	-	-
Barium	1.0	-	2
Boron	-	0.9	0.6
Cadmium	0.01	0.04	0.005
Chromium	0.05	1.0	0.1
Lead	0.05	-	0.015 ^a
Manganese	-	-	-
Mercury	0.002	-	0.002
Molybdenum	0.1 ^b	0.08	0.04
Nickel	-	1.0	0.1
Nitrate	44 ^{c,d}	44 ^{d,e}	-
Selenium	0.01 ^c	-	-
Silver	0.05	0.2	0.1
Strontium	-	25.0	17
Sulfate	-	-	-
Thallium	-	0.007	0.0004
Vanadium	-	0.08	0.02
Zinc	-	6.0	2
Radionuclides			
Radium-226/-228	5 pCi/L	-	-
Uranium (U-234/-238)	30 pCi/L ^f (0.044 mg/L)	0.03 ^{c,f,g}	0.19

^aAction level.

^bExceeded in background and plume wells in unconsolidated aquifer.

^cExceeded in plume wells.

^dEqual 10 mg/L nitrate as nitrogen.

^eUnder review.

^fExceeded in De Chelly well 657.

^gProposed value, under review; expected revision: 1995.

8.4 RISK MITIGATION MEASURES

Because there is potential for serious health effects following short-term use of nitrate-contaminated ground water, this section presents possible ways to restrict access to ground water in order to mitigate risks.

Institutional controls are defined in the proposed ground water standards for UMTRA as mechanisms that can be effectively used to protect human health and the environment by controlling access to contaminated ground water. 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 ground water. Because not all 24 UMTRA sites can be evaluated simultaneously, interim institutional controls are needed before remedial action decisions are made for individual sites.

Institutional controls cannot exclusively depend on markers, fences, or health advisories, as they can too easily be ignored. An education campaign at both the local level and the tribal agency level could curb but would not stop usage. If access were denied, especially to water sources traditionally used in the past, it would be necessary to provide a readily accessible alternative water supply. An alternative supply would need to be installed with the consultation of the local authorities.

Permanent government entities can implement successful institutional controls. 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 for developing water code compliance regulations for the Navajo Nation.

Currently, two permit categories exist for water access: permits for well drilling and permits for water use. Both permits are needed to drill 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 would have to agree to the technical criteria for implementing institutional control of ground water 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 Executive Director of the Division of Natural Resources then reviews the permit to determine the potential for regional, political, or social concerns or impacts.

Establishing interim institutional controls and ensuring the protection of human health and the environment would require a consensus among Navajo Nation governmental agencies and local Navajo governing authorities and chapter

houses. Local authorities should probably be responsible for monitoring new wells to ensure that they have been approved. The governing authorities would also need to be informed of monitoring results and the anticipated duration of contamination problems. Chapter houses are probably the most effective organizations for educating local residents about potential risks and the necessity of access restrictions.

Providing an alternate water source for residents in the Monument Valley site vicinity would help mitigate the possibility of accessing contaminated water in the unconsolidated aquifer by hand-digging wells and ponds. Although hand-digging of wells and ponds occurs at this site, it has not occurred in the vicinity of the present location of the contaminant plume. This type of access is difficult to control; however, reducing the residents' need for additional water supplies by providing good alternative water sources may be the most effective control at this site. The need for a clean alternate water supply is accentuated by the toxic levels of contaminants just below ground surface in Cane Valley Wash. Restricting access to this water source through institutional controls would be a difficult task.

8.5 RECOMMENDATIONS

In general, the proposed ground water standards consisting 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 these standards are either too restrictive or not restrictive enough. When standards are too restrictive, there may be no potential for exposure and a less restrictive ACL may be sought. In other cases, the standards may not be sufficiently protective (e.g., if many contaminants are near the MCL with additive or synergistic adverse health effects).

At Monument Valley, no permanent physical barrier prevents access to contaminated ground water 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 screened because their concentrations fall within nutritional levels (e.g., iron, zinc). It also includes other contaminants such as strontium and vanadium, that were demonstrated to be at concentrations well below adverse health effect 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 contaminated ground water at this site is used. The levels in monitor wells 606 and 655, as well as in other

monitor wells within the contaminant plume in the unconsolidated aquifer, substantially exceed levels at which fatal cases of methemoglobinemia have been reported for infants in the literature. This situation requires implementation of institutional controls as soon as possible, regardless of the remedial action strategy pursued. Development of the De Chelly water resource for use by the valley residents could be accomplished simply by improving access to artesian wells 625 and 613, which may decrease the potential for use of the shallow contaminated water.

This would also reduce the potential for use of hand-dug well 640, which shows some evidence of contamination. The source of elevated uranium and sulfate may not be related to processing, but presents a slight health risk if the well were used continuously.

In the near-term, it is recommended that wells continue to be monitored for ammonium and the contaminants of concern listed in column 4 of Table 3.4. Though ammonium is not considered to pose a health threat, the elevated levels will likely affect the taste and odor of the water, thereby decreasing its usefulness to the residents. Although not all the constituents in column 4 present health risks, the pattern of elevation in these constituents is useful in evaluating the impact of water from the unconsolidated aquifer on downgradient wells and deeper water-bearing zones. Sampling and water-level measurements of background wells in the deeper zones should also be considered to better evaluate water quality in these zones and to verify that the rate of upward flow from the De Chelly Sandstone is sufficient to reduce site-related contaminants.

Monitoring should be included in future ground water work plans to determine the source of contamination in ground water underlying Cane Valley Wash as outlined in Section 3.5.1. Once the source of this contamination is determined, the potential long-term health risks posed by this source should be re-evaluated and any measures to ensure mitigation of these risks should be implemented.

If follow-up investigations indicate the contaminant plume has reached Cane Valley, sampling of plant and animal tissues may be needed to assess bioaccumulation and the potential for food chain transfer.

8.6 FUTURE SITE ACTIVITIES

To resolve outstanding questions with respect to ground water quality, movement, and surface expression at this site, the following monitoring activities should be considered as elements in future site work plans:

- Because wells in the unconsolidated aquifer are screened at varying depths and may not provide water level data appropriate to delineate the water table map, installing additional monitor wells/piezometers with screened sections that bracket the water table is recommended to decrease the uncertainties in assessing the hydraulic gradients and the linear velocities of the uppermost water-bearing zone. Shallow wells near Cane Wash should

be part of this plan to determine the source of contamination in the wash (see Section 3.5.1).

- Upstream monitoring below ground surface in Cane Valley Wash should be considered to determine background water quality of near-surface water in the wash. This will aid in determining the source of contamination near-surface at 627.
- Surface sampling in the wash during periods of surface expression should be considered to determine the potential impact of ground water contamination in this region on expressed surface water.
- Measuring the rate of discharge from surface springs in Cane Valley Wash and north of the tailings site at sampling location 620 would allow accurate assessment of the relative contributions of spring water and downstream ground water seepage to the ephemeral flow in the wash.
- Sampling deeper zone wells (614 and 657) should continue to verify protection from site-related contaminants resulting from the upward movement of ground water over time.

9.0 LIST OF CONTRIBUTORS

The following individuals contributed to the preparation of this report.

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CODE OF FEDERAL REGULATIONS

- 40 CFR 192, "Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings," *Code of Federal Regulations*, Title 40, Part 192, U.S. Environmental Protection Agency, Office of the Federal Register, National Archives and Records Administration, Washington, D.C.

UNITED STATES CODE

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

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