Final Site Observational Work Plan for the UMTRA Project Site at Monument Valley, Arizona

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Plates--will be provided upon request. Email lm.records@lm.doe.gov to request.

Plate 1—Geologic Map of Monument Valley Site

Plate 2—Geologic Cross-Sections of Monument Valley Site

Appendices--will be provided upon request. Email lm.records@lm.doe.gov to request.

Appendix A. Lithologic and Monitor Well Completion Logs . . . . . Vol. I
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Acronyms and Abbreviations

ACL	alternate concentration limits
BIA	U.S. Bureau of Indian Affairs
BLRA	baseline risk assessment
cm	centimeters
cm/s	centimeters per second
cm/year	centimeters per year
COC	contaminants of concern
COPC	contaminants of potential concern
DOE	U.S. Department of Energy
Eh	oxidation-reduction potential
EPA	U.S. Environmental Protection Agency
ESC	expedited site characterization
ET	evapotranspiration
ET/P	evapotranspiration/precipitation
ft	foot (feet)
ft/day	foot (feet) per day
ft/year	feet per year
ft^2	square feet
ft^2/min	square feet per minute
ft^3	cubic feet
ft^3/day	cubic feet per day
gal	gallon(s)
GCAP	Ground Water Compliance Action Plan
GJO	Grand Junction Office
gpm	gallons per minute
GRM	generalized reciprocal method
HEW	U.S. Department of Health, Education, and Welfare
HCl	hydrochloric acid
HQ	hazard quotient
Hz	Herz
ICP–AES	inductively coupled plasma-atomic emission spectrophotemetry
ICP–MS	inductively coupled plasma-mass spectrometry
i.d.	inside diameter
in.	inch(es)
in./year	inches per year
IP	induced polarization
IRIS	Integrated Risk Information System
km	kilometers
L	liter(s)
m	meters
MAP	management action process
MCL	maximum concentration limits
mg/kg	milligrams per kilogram
mg/L	milligrams per liter

Acronyms and Abbreviations (continued)

mL	milliliters
µm	micrometers
µg/L	micrograms per liter
mm	millimeters
Mn	manganese
MSL	mean sea level
mV	millivolts
NEPA	National Environmental Policy Act
NH_4	ammonium
NO_3	nitrate
NRC	U.S. Nuclear Regulatory Commission
O&M	operating and maintenance
OMB	Office of Management and Budget
pCi/L	picocuries per liter
pCi/g	picocuries per gram
PEIS	Programmatic Environmental Impact Statement
ppb	parts per billion
ppm	parts per million
PVC	polyvinyl chloride
RAP	remedial action plan
Rfd	reference dose
RO	reverse osmosis
RRM	residual radioactive material
SO_4	sulfate
SOWP	site observational work plan
Sr	strontium
TAGR	Technical Approach to Groundwater Restoration
TDS	total dissolved solids
TEM	transient electromagnetic
UMTRA	Uranium Mill Tailings Remedial Action (Project)
UMTRCA	Uranium Mill Tailings Radiation Control Act
U	uranium
USDA	U.S. Department of Agriculture
VCA	Vanadium Corporation of America

Executive Summary

Ground water beneath the Monument Valley site was contaminated by former uranium ore-processing operations that were ongoing from 1955 through 1968. Tailing piles, leach areas, an evaporation pond, and other associated contaminated surface materials were removed from the site by January 1994 in accord with 40 CFR Part 192 Subpart A as part of the Uranium Mill Tailings Remedial Action (UMTRA) Surface Project. However, the potential for infiltration of ground-water contaminants remained until that time. Currently, no one is drinking the contaminated ground water.

Site-specific field investigations reveal the alluvial ground water is the aquifer most effected by the former milling operations. Contaminants of concern (COCs) in the alluvial aquifer are identified as nitrate, sulfate, and uranium. Nitrate concentrations exceeding the U.S. Department of Energy (DOE) maximum concentration limit (MCL) of 44.0 milligrams per liter (mg/L) are present in the alluvial aquifer up to a maximum of 4,500-feet (ft) downgradient from the site. Elevated concentrations of COCs are not present in the Shinarump bedrock aquifer. Uranium is present in the De Chelly bedrock aquifer at concentrations that slightly exceeds the 0.044 mg/L uranium MCL; however, the area of impact is small, isolated, and the concentrations appear to be decreasing with time.

DOE's goal is to implement a cost-effective strategy to remediate the ground water at the former Monument Valley mill site that complies with the U.S. Environmental Protection Agency (EPA) ground water standards and protects human health and the environment. The requirements for ground water compliance for UMTRA Project sites, including the Monument Valley site, are found in the Uranium Mill Tailings Radiation Control Act (42 USC §7901 *et seq.*) and EPA's Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings (40 CFR Part 192; 60 FR 2854). The compliance framework was developed in the UMTRA Ground Water programmatic environmental impact statement (DOE 1996c).

The proposed compliance strategy to cleanup the alluvial ground water at the Monument Valley site is no ground water remediation of constituents that do not pose a potential risk and do not exceed EPA standards. For constituents that pose a potential risk or exceed EPA standards or both, the strategy is to perform active ground water remediation using phytoremediation of ammonia-contaminated soils and shallow portions of the aquifer, and distillation of deeper portions of the aquifer, in combination with natural flushing. Information presented in this final site observational work plan supports the proposed compliance strategy in a manner that is consistent with the regulatory compliance framework.

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1.0 Introduction

The Monument Valley Uranium Mill Tailings Remedial Action (UMTRA) Project site in northeastern Arizona (Figure 1–1) is the location of a former uranium mill. Ground water beneath the Monument Valley site was contaminated by milling operations that were ongoing from 1955 through 1968. Tailing piles, leach areas, an evaporation pond, and contaminated surface materials were completely removed from the site by January 1994 in accord with 40 CFR Part 192 Subpart A, as part of the U.S. Department of Energy (DOE) UMTRA Surface Project. However, the potential for infiltration of ground-water contaminants remained until that time.

DOE's goal is to implement a cost-effective compliance strategy that is protective of human health and the environment by remediating contaminated ground water at the Monument Valley site. For site-related constituents that pose a potential risk or exceed the U.S. Environmental Protection Agency (EPA) standards or both, the proposed strategy is to perform active ground water remediation in combination with natural flushing. The proposed compliance strategy is no ground water remediation of site-related constituents that do not pose a potential risk and do not exceed the EPA standards. Sulfate is the only exception to this strategy; details of the compliance strategy for sulfate are given in Section 8.1.2.1.

This final site observational work plan (SOWP) documents the site-specific strategy that will allow DOE to comply with EPA ground water standards at the Monument Valley UMTRA Project site and provides a mechanism for stakeholder participation, review, and acceptance of the recommended remedial alternative. Site-specific data are presented that support the proposed strategy.

Compliance requirements for meeting the regulatory standards at the Monument Valley site are presented in Section 2.0. An overview and history of the former milling operation are reviewed in Section 3.0. Results of field investigations conducted at the site in 1997 are presented in Section 4.0. Site-specific characterization of the geology, hydrology, geochemistry, and ecology are synthesized in the site conceptual model in Section 5.0. Potential human health and ecological risks associated with ground water contamination are summarized in Section 6.0. The proposed compliance strategy and an evaluation of potential remediation technologies to clean up the ground water are presented in Sections 7.0 and 8.0, respectively.

1.1 UMTRA Project Programmatic Documents

The programmatic documents that guide the SOWP include the *UMTRA Groundwater Management Action Process* (MAP) (DOE 1998b), the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailing Remedial Action Ground Water Project* (PEIS) (DOE 1996c), and the *Technical Approach to Groundwater Restoration* (TAGR) (DOE 1993c). The MAP states the mission and objectives of the UMTRA Ground Water Project and provides a technical and management approach for conducting the project. The PEIS is the programmatic decision-making framework for conducting the UMTRA Ground Water Project. DOE will follow PEIS guidelines to assess the potential programmatic impacts of the Ground Water Project, to determine site-specific ground water compliance strategies, and to prepare site-specific

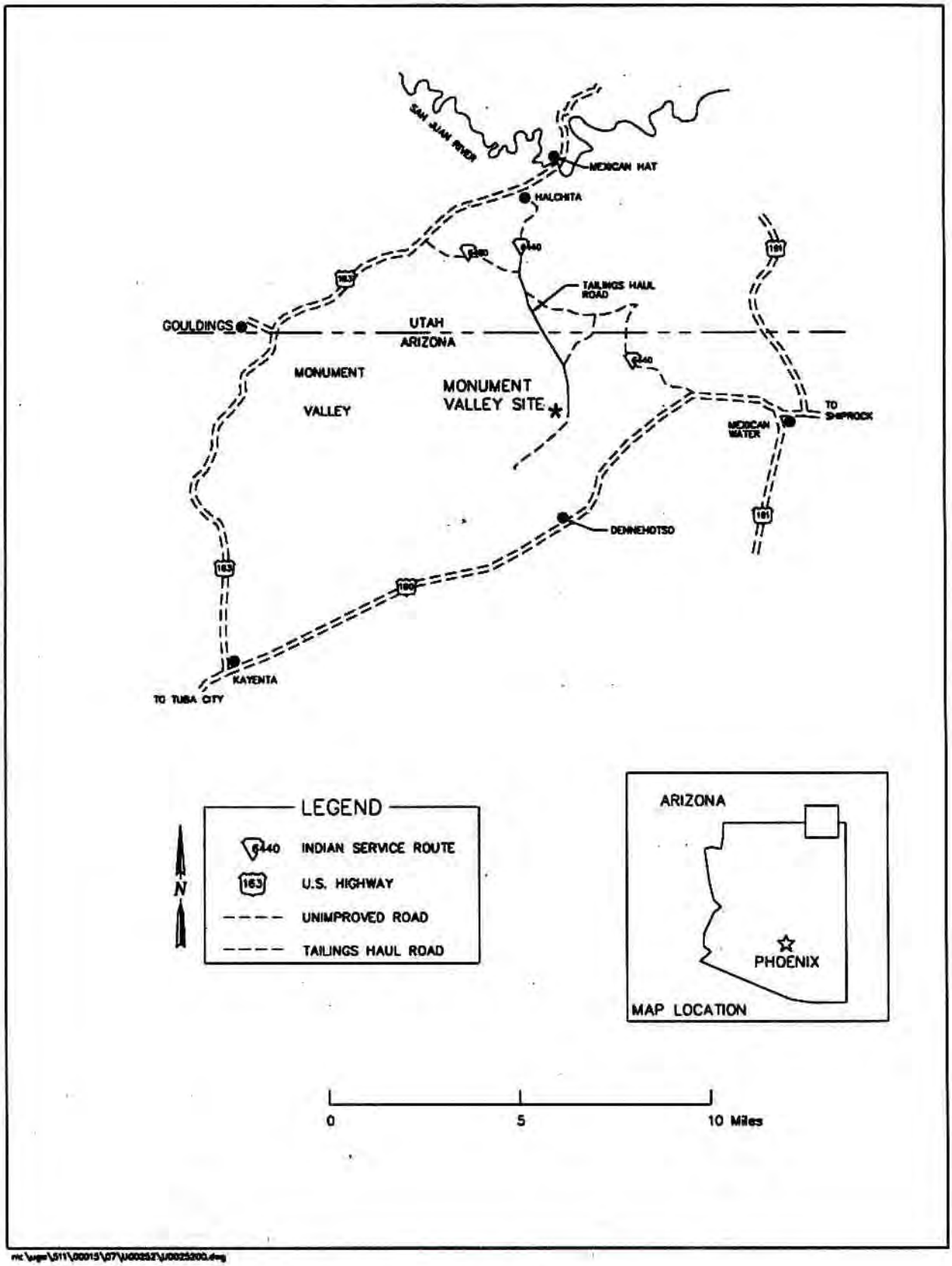

Figure 1–1. Location of the Monument Valley Site

environmental impact analyses more efficiently. Technical guidelines for conducting the ground-water program are presented in the TAGR.

1.2 Relationship to Site-Specific Documents

The surface remedial action plan (RAP) (DOE 1993b) provides site characterization information. This information was updated in developing the SOWP to strengthen the site conceptual model. If an active ground water compliance strategy requiring remedial action is selected for this site, a ground water draft and final Ground Water Compliance Action Plan (GCAP) will be prepared; otherwise, a modification to the surface RAP via a GCAP will suffice.

In 1996, a baseline risk assessment (BLRA) was prepared (DOE 1996b) that identified potential public health and environmental risks at the site. Potential risks identified in the risk assessment are considered and updated in this SOWP to ensure that the proposed compliance strategy is protective of human health and the environment.

After a proposed compliance strategy is identified in the SOWP and described in the GCAP, a site-specific National Environmental Policy Act (NEPA) document (e.g., an environmental assessment) will be prepared to determine the potential effects, if any, of implementing the proposed compliance strategy.

1.2.1 SOWP Revisions

The SOWP is a multiyear process of sequenced document preparation and field data-collection activities consisting of two versions: Revision 0 (draft) and Revision 1 (final).

The draft SOWP was prepared in 1996 and included all previous information about the site, presented a proposed compliance strategy and possible remediation technologies, and defined additional data needs that were required to determine the most likely compliance strategy. Following stakeholder review and resolution of comments, fieldwork was conducted in 1997 to address the data gaps identified in the draft SOWP.

This final SOWP presents the additional data collected in 1997, correlates the data to previous information, updates the site conceptual model, and recommends a final compliance strategy based on the updated site conceptual model.

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2.0 Regulatory Framework

This section identifies the regulatory framework to be applied to the selected ground-water compliance strategy at the former Monument Valley millsite to achieve compliance with Subpart B of EPA health and environmental protection standards for uranium and thorium mill tailings (40 CFR Part 192) and the final rule to the standards published in 60 FR 2854.

2.1 Uranium Mill Tailings Radiation Control Act

The United States Congress passed the Uranium Mill Tailings Radiation Control Act (UMTRCA) (42 USC §7901 *et seq.*) in 1978 in response to public concerns about potential health hazards from long-term exposure to uranium mill tailings. UMTRCA authorized DOE to stabilize, dispose of, and control uranium mill tailings and other contaminated materials at inactive uranium ore-processing sites.

Three UMTRCA titles apply to uranium ore-processing sites. Title I designates 24 inactive processing sites for remediation. It directs EPA to promulgate standards, mandates remedial action in accordance with these standards, stipulates that remedial action be selected and performed with the concurrence of the U.S. Nuclear Regulatory Commission (NRC) and in consultation with the states and Indian tribes, directs NRC to license the disposal sites for long-term care, and directs DOE to enter into cooperative agreements with the affected states and Indian tribes. Title II applies to active uranium mills. Title III applies only to certain uranium mills in New Mexico. The UMTRA Project is responsible for administering only Title I of UMTRCA.

In 1988, Congress passed the Uranium Mill Tailings Remedial Action Amendments Act (42 USC §7922 *et seq.*), authorizing DOE to extend without limitation the time needed to complete ground water remediation activities at the processing sites.

2.1.1 EPA Ground Water Protection Standards

UMTRCA requires EPA to promulgate standards for protecting public health, safety, and the environment from radiological and nonradiological hazards associated with uranium ore processing and the resulting residual radioactive materials (RRM). On January 5, 1983, EPA published standards (40 CFR Part 192) for RRM disposal and cleanup. The standards were revised and a final rule was published January 11, 1995 (60 FR 2854).

The standards address two ground water contamination scenarios: (1) future ground water contamination that might occur from tailings material after disposal cell construction, and (2) the cleanup of residual contamination from the milling process at the processing sites that occurred before disposal of the tailings material (60 FR 2854). The UMTRA Surface Project is designed to control and stabilize tailings and contaminated soil. The UMTRA Ground Water Project addresses ground water contamination at the processing sites and is regulated by Subparts B and C of 40 CFR 192.

2.1.1.1 Subpart B: Standards for Cleanup of Land and Buildings

Subpart B, "Standards for Cleanup of Land and Buildings Contaminated with Residual Radioactive Materials from Inactive Uranium Processing Sites," requires documentation that action at the former ore-processing sites ensures that ground water contamination meets any of the following three criteria:

- Background levels, which are concentrations of constituents in nearby ground water not contaminated by ore-processing activities.
- Maximum concentration limits (MCLs), which are limits set by EPA for certain hazardous constituents in ground water and are specific to the UMTRA Project (Table 2–1).
- Alternate concentration limits (ACLs), which are concentration limits for hazardous constituents that do not pose a substantial hazard (present or potential) to human health or the environment as long as the limit is not exceeded.

Table 2–1. Maximum Concentration Limits of Inorganic Constituents in Ground Water at UMTRA Project Sites

Constituent	Maximum Concentration[a]
Arsenic	0.05
Barium	1.0
Cadmium	0.01
Chromium	0.05
Lead	0.05
Mercury	0.002
Molybdenum	0.1
Nitrate (as N)	10.0[b]
Selenium	0.01
Silver	0.05
Combined radium-226 and radium-228	5 pCi/L
Combined uranium-234 and uranium-238	30 pCi/L[c]
Gross alpha-particle activity (excluding radon and uranium)	15 pCi/L

[a]Concentrations reported in milligrams per liter (mg/L) unless otherwise noted.
[b]Equivalent to 44 mg/L nitrate as nitrate.
[c]Equivalent to 0.044 mg/L, assuming secular equilibrium of uranium-234 and uranium-238.

pCi/L = picocuries per liter.
Reference: 60 FR 2854.

Natural Flushing Standards

Subpart B also allows natural flushing to meet EPA standards. Natural flushing allows natural ground water processes to reduce the contamination in ground water to acceptable standards (background levels, MCLs, or ACLs). Natural flushing must allow the standards to be met within 100 years. In addition, institutional controls and an adequate monitoring program must be

established and maintained to protect human health during the period of natural flushing. Institutional controls would prohibit inappropriate uses of the contaminated ground water. The ground water also must not be a current or projected source of drinking water for a public water system during the period of natural flushing, and beneficial uses of ground water must be protected.

2.1.1.2 Subpart C: Implementation

Subpart C provides guidance for implementing methods and procedures to reasonably ensure that standards of Subpart B are met. Subpart C requires that the standards of Subpart B are met on a site-specific basis using information gathered during site characterization and monitoring. The plan to meet the standards of Subpart B must be stated in a site-specific GCAP. The plan must contain a compliance strategy and a monitoring program, if necessary.

Supplemental Standards

Under certain conditions, DOE may apply supplemental standards to contaminated ground water in lieu of background levels, MCLs, or ACLs (40 CFR Part 192). Supplemental standards may be applied if any of the following conditions are met:

- Remedial action necessary to implement Subpart A or B would pose a significant risk to workers or the public.

- Remedial action to meet the standards would directly produce environmental harm that is clearly excessive, compared to the health benefits of remediation, to persons living on or near the sites, now or in the future.

- The estimated cost of remedial action is unreasonably high relative to the long-term benefits, and the RRM does not pose a clear present or future hazard.

- There is no known remedial action.

- The restoration of ground water quality at any processing site is technically impractical from an engineering standpoint.

- The ground water is classified as limited-use ground water. Subpart B of 40 CFR 192 defines limited-use ground water as ground water that is not a current or potential source of drinking water because total dissolved solids (TDS) exceed 10,000 milligrams per liter (mg/L); there is widespread ambient contamination that cannot be cleaned up using treatment methods reasonably employed in public water supply systems; or the quantity of water available to a well is less than 150 gallons (gal) (570 liters [L]) per day. When limited-use ground water applies, supplemental standards ensure that current and reasonably projected uses of the ground water are preserved (40 CFR Part 192).

- Radiation from radionuclides other than radium-226 and its decay products is present in sufficient quantity and concentration to constitute a significant radiation hazard from RRM.

2.1.2 Cooperative Agreements

UMTRCA requires that remedial action include full participation of the states and Indian tribes that own land containing uranium mill tailings. UMTRCA also directs DOE to enter into cooperative agreements with the states and Indian tribes.

2.2 National Environmental Policy Act

UMTRCA is a major federal action that is subject to the requirements of NEPA (42 USC §4321 *et seq.*). Regulations of the Council on Environmental Quality (to implement NEPA) are codified in 40 CFR Part 1500; these regulations require each federal agency to develop its own implementing procedures (40 CFR §1507.3). DOE-related NEPA regulations are contained in 10 CFR Part 1021, *National Environmental Policy Act Implementing Procedures*. DOE guidance is provided in *Recommendations for the Preparation of Environmental Assessments and Environmental Impact Statements* (DOE 1993a).

Pursuant to NEPA, in 1994 DOE drafted a PEIS for the UMTRA Ground Water Project. The PEIS document was made final in October 1996. The purpose of the NEPA document was to analyze the potential impacts of implementing four programmatic alternatives for ground water compliance at the designated processing sites. The preferred alternative for the UMTRA Ground Water Project was published in a Record of Decision in 1997. All subsequent action on the UMTRA Ground Water Project will comply with the Record of Decision.

2.3 Other Regulations

In addition to UMTRCA EPA ground water standards and NEPA, DOE must also comply with other Federal regulations and executive orders that may be relevant to the UMTRA Project sites. Examples include regulations that require protection of wetlands and floodplains, threatened or endangered species, and cultural resources. Other regulations, for which the State may be delegated authority, include requirements for water discharge and waste management. Executive orders include those related to pollution prevention and environmental justice.

2.4 State/Tribal Regulations

State and tribal regulations must also be complied with where Federal authority has been delegated to the State or where the Navajo Nation exercised the right of sovereignty. Examples include the right of the Navajo Nation to require water-use permits and permits to drill wells.

2.5 DOE Orders

Several environmental, health and safety, and administrative DOE orders that apply to the work being conducted under the UMTRA Ground Water Project. DOE orders prescribe the manner in which DOE will comply with Federal and State laws, regulations, and guidance, and the manner in which DOE will conduct operations that are not prescribed by law. DOE guidance for complying with Federal, State, and tribal environmental regulations are contained in the DOE Order 5400.1 Series, partially superseded by DOE Order 231.1. DOE Order 5400.5 requires protection of the public from radiation hazards. DOE guidance pertaining to NEPA is contained in DOE Order

451.1, and specific guidance pertaining to environmental assessments (EAs) is provided in Recommendations for the Preparation of Environmental Assessments and Environmental Impact Statements (DOE 1993a).

2.6 Agreements

UMTRCA requires that compliance with the ground water standards be accomplished with the full participation of states that are paying part of the costs, and in consultation with Indian tribes on whose lands uranium mill tailings are located. UMTRCA also directs DOE to enter into cooperative agreements with the states and Indian tribes. DOE has negotiated an UMTRA Ground Water cooperative agreement with the Navajo Nation.

The Navajo Nation's proposed secondary cleanup levels for sulfate is 250 mg/L. In concert with the sulfate-to-chloride ratio, this will be adopted as a cleanup goal for the Monument Valley site. See Section 8.1.2.1 for additional information.

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3.0 Site Background

The Monument Valley UMTRA Project site is on the Navajo Indian Reservation (Navajo Nation) in northeastern Arizona, approximately 15 miles south of Mexican Hat, Utah (Figure 1–1). The site, which is accessible by U.S. Bureau of Indian Affairs (BIA) Navajo Service Route 6440, is the location of a former uranium mill that operated from 1955 through 1968. An overview of the site's physical setting and climate, a history of the former milling operation, and a summary of previous investigations is presented in the following sections.

3.1 Physical Setting and Climate

The former millsite is on the west side of Cane Valley, which is drained to the north by Cane Valley Wash. The elevation along Cane Valley Wash is approximately 4,800 ft above mean sea level. The valley is bordered on the east by Comb Ridge, a 600-ft-high escarpment of Navajo, Kayenta, and Wingate Sandstones. On the west side of the valley near the former millsite, the bedrock dips to the east at approximately 5 degrees and rises up to Yazzie Mesa at an elevation of over 5,300 ft. Cane Valley between Comb Ridge and Yazzie Mesa is filled with a reddish-yellow eolian sand and minor amounts of water-transported sand, gravel, and bedrock fragments.

The site is arid, receiving approximately 6.4 inches (in.) of annual precipitation. Most precipitation usually occurs during July through August and December through February. Rainfall during the summer commonly occurs in high-intensity, short-duration storms that are conducive to runoff. Precipitation during the winter, however, usually occurs during low-intensity, longer-duration storms (Cooley et al. 1969). Annual snowfall ranges between 10 and 40 in. The two driest months are generally May and June.

The weather station closest to the Monument Valley site is in Mexican Hat, Utah, about 16 miles north. Climatological data collected from the Mexican Hat weather station for the period 1951 through 1980 indicates an average annual pan evaporation rate of 84.4 in. (DOE 1993b). Pan evaporation rates exceed precipitation every month except January. The highest rates occur from May through August, when pan evaporation exceeds 10 in. per month.

Temperatures show considerable diurnal and seasonal variations. Winters are cold, with temperatures typically below freezing from November through March. Summers are hot, with highs ranging from 90 °F to the low 100s °F.

3.2 Site History

Uranium was discovered in 1942 by Luke Yazzie approximately one-half mile west of the former millsite (Chenoweth 1985). The deposit is a carnotite mineralization in and beneath a paleochannel in the Shinarump Member of the Chinle Formation incised into the underlying Moenkopi Formation and De Chelly Sandstone Member of the Cutler Formation. Vanadium Corporation of America (VCA) acquired mining rights for the deposit from the Office of Indian Affairs in 1943 and named the lease property Monument No. 2. VCA mined the property from 1943 to 1968. Total production was 767,166 tons of ore averaging 0.34 percent U_3O_8 and 1.42 percent V_2O_5. Included in the production estimate are products from a mechanical upgrader, a concentrator, and

a heap leach that operated at various times at the site. The Monument No. 2 mine has produced more uranium than any other mine in Arizona (Chenoweth 1985).

Before 1955, there was no mill at the site. From 1943 to 1946, the ore was shipped to Metal Reserve at Monticello, Utah. From 1947 to 1952, low-grade ore from the mine was mechanically upgraded at a small plant on the bank of the San Juan River near the Mexican Hat bridge. This upgrader is believed to be the prototype for the plant that was built at the Monument Valley site in 1955 (Chenoweth 1985). Ore concentrated from the upgrader was hauled to a mill at Naturita, Colorado.

The upgrader constructed at the Monument Valley site in 1955 consisted of a mechanical separator. In this operation, ore was crushed and sorted by grain size using large amounts of water from two on-site wells (MON–618 and MON–619) in the De Chelly Sandstone. The finer grained material, which was higher in uranium content, was shipped off site for chemical concentration at the Durango, Colorado, mill before March 1963 and later at the VCA mill at Shiprock, New Mexico. No chemicals were used except minor amounts of flocculants (Albrethson 1982). The coarser grained material remained on the site and was piled in the areas identified as the former mill and old tailings pile (Figure 3–1). The mechanical milling operations at the Monument Valley site continued from 1955 to 1964.

In October 1964, batch-leaching equipment was installed at the mill. Batch leaching continued for approximately 3 years, during which approximately 1,000,000 tons of sandy tailings were processed (925 tons per day) in large steel tanks. A separate heap-leaching operation was used on an additional 100,000 tons of low-grade ore in 1966 and 1967.

The millsite was leased from the Navajo Nation until 1968, when the mill closed and the lease expired. Control of the site, structures, and materials reverted to the Navajo Nation at that time.

The mill buildings and milling equipment were removed after 1968. Beginning in 1992, the tailings piles, windblown tailings, contaminated radioactive materials, concrete foundations, and debris were removed and placed in the Mexican Hat UMTRA Project disposal cell, approximately 10 mi north of the former millsite. Relocation of these materials was completed in January 1994.

3.2.1 Sources of Ground Water Contamination from the Milling Operation

Some ground water contamination probably occurred during the mechanical processing period (1955 to 1964) as a result of water draining from stockpiles of the finer grained material prior to shipment off-site for chemical separation and from the coarser material that remained on-site. The primary contaminants would have been relatively soluble components of the ore, such as uranium, calcium, and sulfate (the source of calcium and sulfate would have been gypsum, which was part of the ore body). Infiltration of the contaminated water would have occurred at the former mill and old tailings pile areas designated on Figure 3–1.

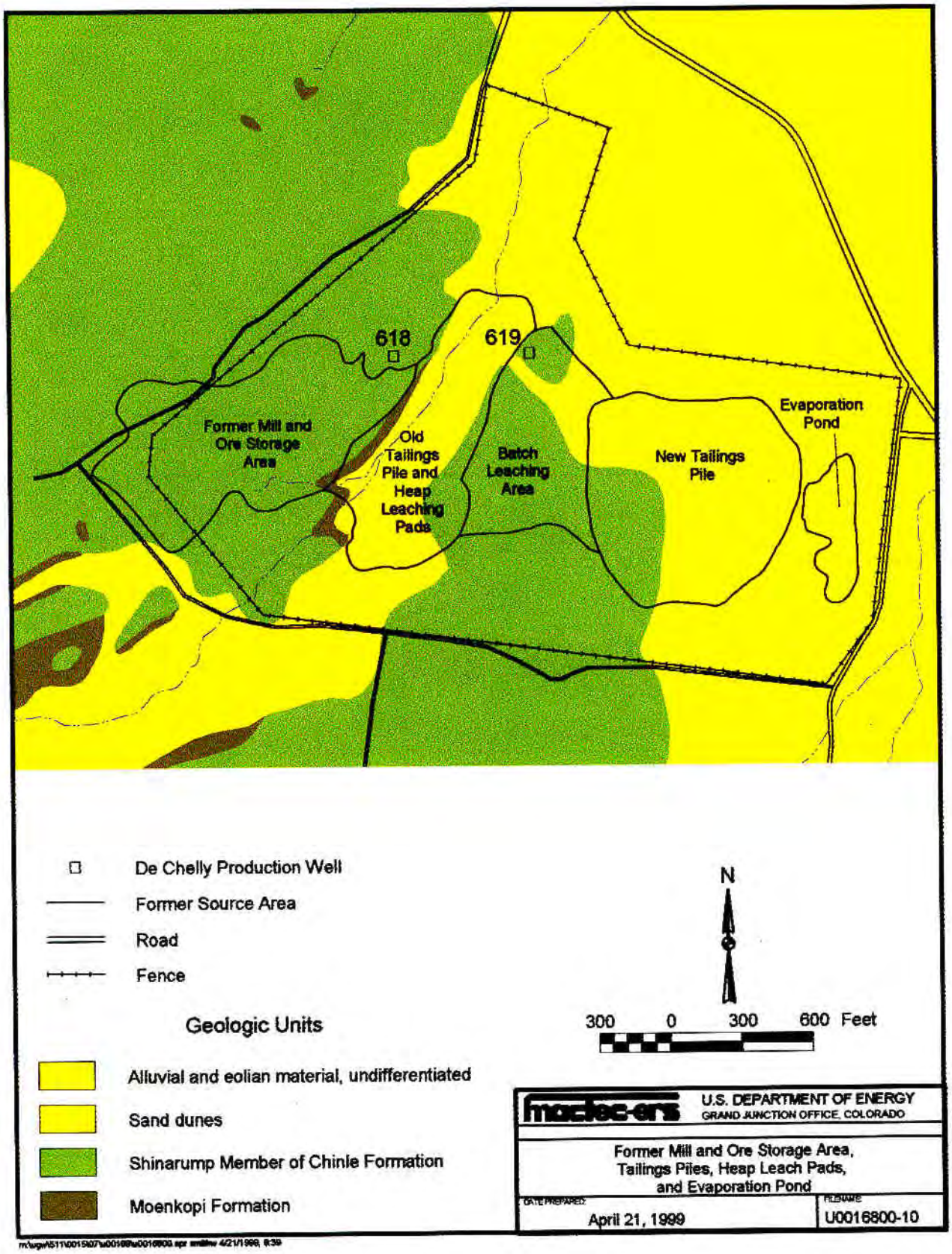

Figure 3–1. Former Mill and Ore-Storage Area, Tailings Piles, Heap-Leach Pads, and Evaporation Pond

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Process chemicals were an additional source of sulfate, nitrate, calcium, and ammonium. Both the batch- and heap-leaching operations used sulfuric acid to leach out uranium and vanadium. The sulfuric acid heap- and batch-leaching solutions were adjusted to pH 4 with ammonia. Quicklime (calcium oxide) was then added to neutralize the pH and produce a bulk precipitate. Later, this bulk precipitate was shipped to the mill at Shiprock, New Mexico, where the uranium and vanadium were extracted. The spent neutralization solution was probably discharged to the new tailings pile and the heap- and batch-leach material was slurried to the new tailing pile (Merritt 1971, DOE 1982) where infiltration of contaminated water would have occurred (Figure 3–1).

3.2.1.1 Quantity Estimates of Process Water and Chemicals

The amount of process water and chemicals (sulfuric acid, ammonia, and nitrate) used at the Monument Valley site from 1964 to 1967 is estimated on the basis of typical usage in uranium mills (Merritt 1971, HEW 1962). The U.S. Department of Health, Education, and Welfare (HEW) report suggests that the amount of water used was approximately 850 gal per ton of reprocessed tailings. The amount of chemical used per ton of ore processed, based on the HEW report and site-specific data compiled by Merritt, is 25 pounds of sulfuric acid, 1 to 30 pounds of ammonia, and 15 to 20 pounds of ammonium nitrate.

3.2.2 Previous Investigations

Merritt (1971) provides detailed descriptions of the uranium concentration process, mill by-products, and process waste streams. Albrethsen and McGinley (1982) summarizes the history of the domestic uranium procurement policies and practices under the Atomic Energy Commission. Chenoweth (1985) documents the history of mining in Monument Valley.

Early geologic and hydrologic studies conducted near the site are reported in Witkind and Thaden (1963), Cooley et al. (1969), Irwin et al. (1971), and James (1973).

Site-specific hydrogeologic and geochemical investigations are described in an engineering assessment (DOE 1981), an Environmental Assessment (DOE 1989), a RAP (DOE 1993b), a water sampling and analysis plan (DOE 1994), a BLRA (DOE 1996b), and the draft SOWP (DOE 1996d).

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4.0 Field Investigation Results

The draft SOWP (DOE 1996d) included all previous information about the site, proposed possible remediation technologies, and defined additional data needs that were required to determine the most likely compliance strategy. Following stakeholder review and resolution of comments, an expedited site characterization (ESC) field investigation was conducted in 1997 to address the data gaps identified in the draft SOWP. Additional field characterization data presented in the following sections were collected to reduce system uncertainties by enhancing the understanding of the site characteristics and thereby ensuring that the appropriate ground water compliance strategy is selected.

Field investigations were optimized by sequencing the activities to achieve a more logical sampling approach. The first activities were based on nonintrusive methods to obtain a more complete and comprehensive understanding of the subsurface environment before more direct characterization methods were employed. The field activities were sequenced as follows: (1) surface geophysical surveys, (2) direct-push ground water sampling and analysis, (3) drilling, soil sampling, and installation of monitor wells, (4) aquifer tests and surface infiltration tests, (5) land surveys of new borings/wells, and (6) ecological and ground water sampling and analyses. Information obtained from each activity was integrated with existing data to revise the site conceptual model and to refine the data collection needs. This integration was performed either concurrently with or before proceeding to the next characterization activity.

Results of the 1997 field investigation are presented in the following sections. All fieldwork and data quality objectives applied to the data collection activities were performed in accordance with the *Work Plan for Characterization Activities at the UMTRA Monument Valley Project Site* (DOE 1997c).

4.1 Surface Geophysical Surveys

Surface geophysical surveys provide a nonintrusive means to rapidly characterize subsurface conditions at the site before more direct sampling methods are employed. Geophysical methods applied to this investigation include seismic refraction, transient electromagnetic (TEM) soundings, and induced polarization (IP) and resistivity soundings. Each method measures a different characteristic physical property. Because some physical properties are interrelated, a combination of methods, such as TEM, IP, and resistivity soundings can be helpful in discriminating a target signal in a noise-field matrix.

Specifics regarding geophysical calculations and modeling, sounding curves, and raw data that support the interpretation of the seismic and electrical methods used are presented in the Monument Valley Geophysical Report (Rogers and Sandberg 1998). Sections 4.1.1 through 4.1.3 provide summaries of the more significant findings.

4.1.1 Seismic Refraction Survey

The seismic refraction method refers to a geophysical technique in which acoustic (sound) waves are used to map subsurface lithologic layers. A source of seismic energy, such as the impact of a sledge hammer on a metal plate resting on the ground surface, produces acoustic waves that

travel in spherical wavefronts down into the subsurface. These seismic waves reflect from, and refract along, boundaries between layers of differing density and seismic velocity. For seismic refraction to detect the top of a layer, the seismic velocity of that layer must be greater than that in the overlying layer. Therefore, the seismic refraction method is suited for determining the depth to, and seismic velocity of, the bedrock surface underlying unconsolidated alluvial deposits.

A buried paleovalley, which may influence the downgradient extent of contamination in both the alluvial and bedrock aquifers, exists beneath the northern part of the processing site. A detailed seismic refraction survey was conducted near and downgradient from the former millsite to provide subsurface information about the presence and extent of the buried paleovalley.

4.1.1.1 Seismic Refraction Procedure

Seismic refraction data were collected along the three parallel traverse lines shown in Figure 4–1; each line was established roughly across and perpendicular to the inferred axis of the buried paleovalley. The first line was near the former old tailings area and was 1,780 ft long. Line 2 was approximately 1,000 ft north from line 1 and was 1,330 ft long. The third line was 670 ft long and was approximately halfway between lines 1 and 2.

Geophones were spaced 10 ft apart along each line using an array (spread) of 24 geophones at a time with an overlap of 10 ft between each spread. A Geometrics model 2401 seismograph was used to record the seismic signals generated by the impact of a sledge hammer on an aluminum plate. The seismic data were processed using the GREMIXa (Interpex, Ltd.) generalized reciprocal method (GRM) computer software package. The GRM uses seismic arrival times at the surface geophones from opposing shots, surface hammer blows forward and reverse of the seismic spread, which travel along the same refractor, along with the reciprocal time between the shots, to calculate the time depth from a surface geophone to the refractor.

The seismic refraction survey was performed according to procedure GP–2(P), “Standard Practice for Acquisition, Reduction, and Display of Refraction Seismic Data” (GJO 1998).

4.1.1.2 Seismic Refraction Results

Seismic profiles for each of the three survey lines are presented in Figures 4–2, 4–3, and 4–4, respectively. The upper portion of each cross-section presents the travel-time curves generated from each shotpoint and geophone spread. The center portion presents the interpreted structure of the subsurface. The lower portion presents the interpreted velocities derived from the field data versus profile distance.

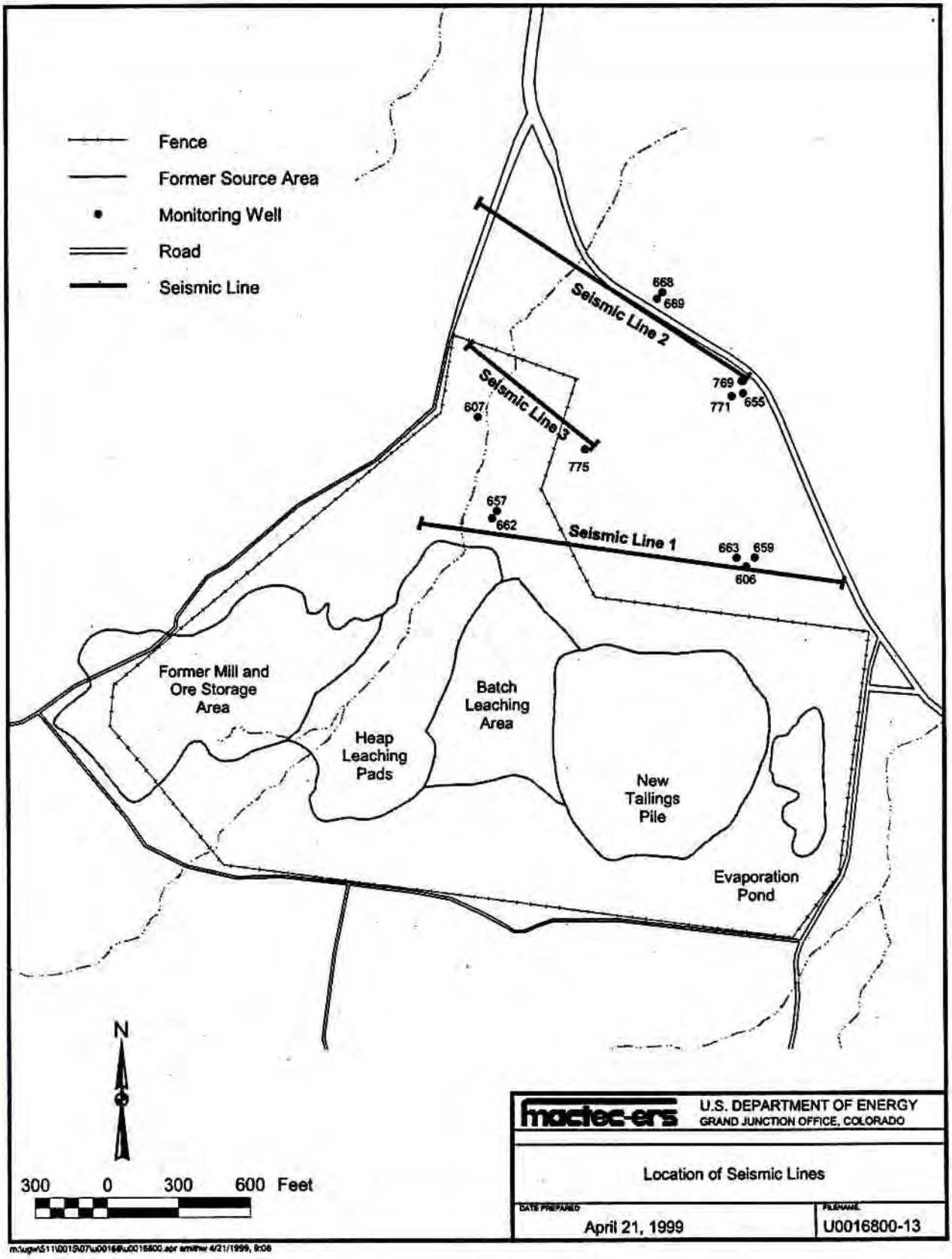

Figure 4–1. Locations of Seismic Lines

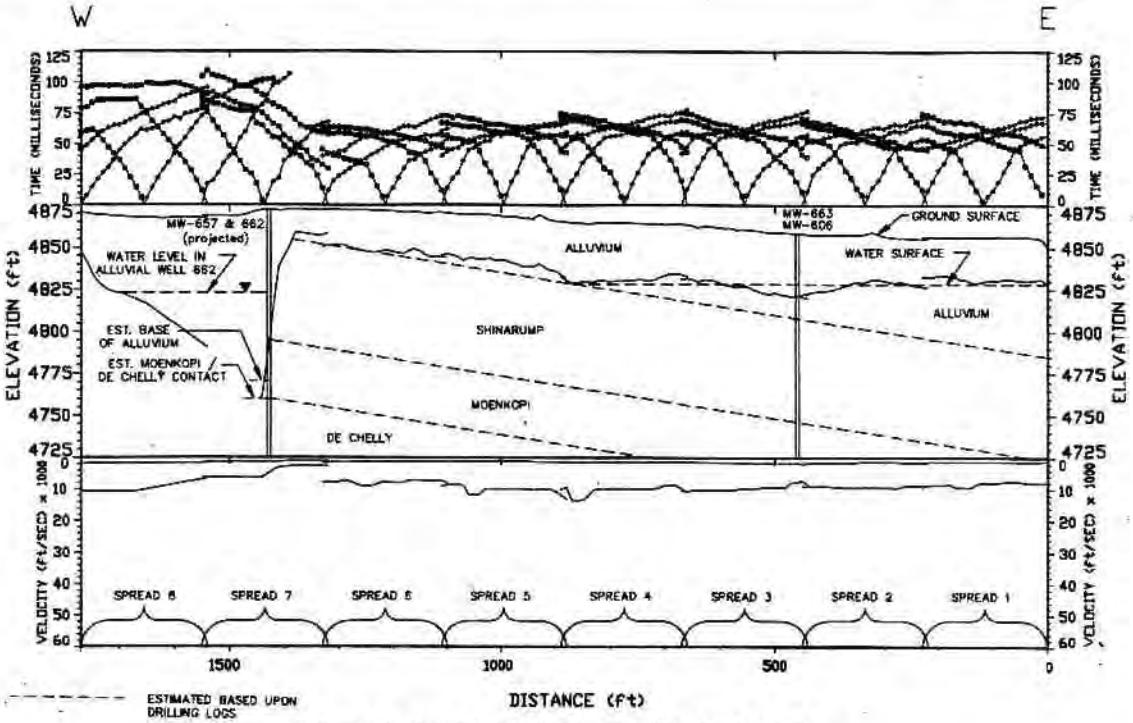

Figure 4–2. Seismic Interpretation for Line 1

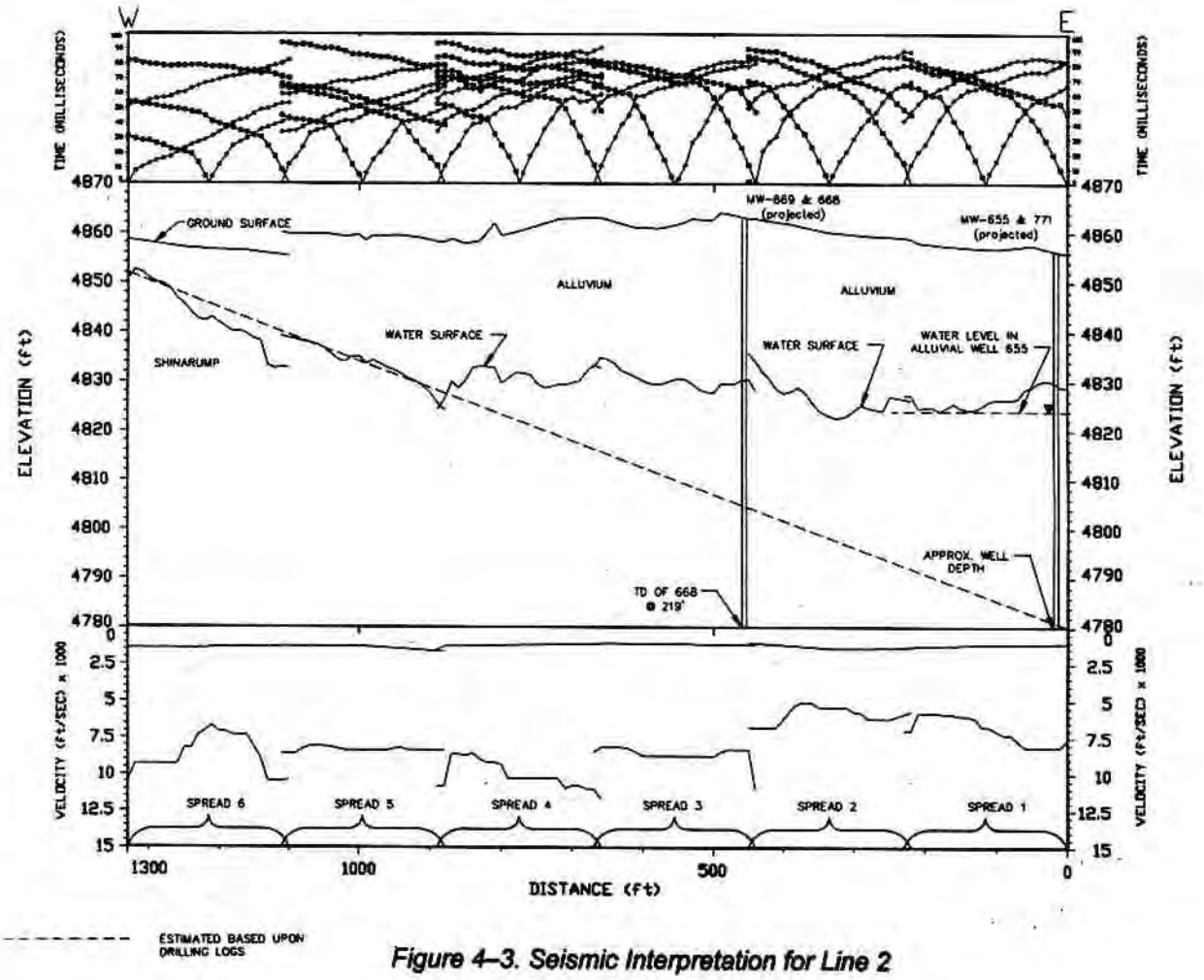

Figure 4–3. Seismic Interpretation for Line 2

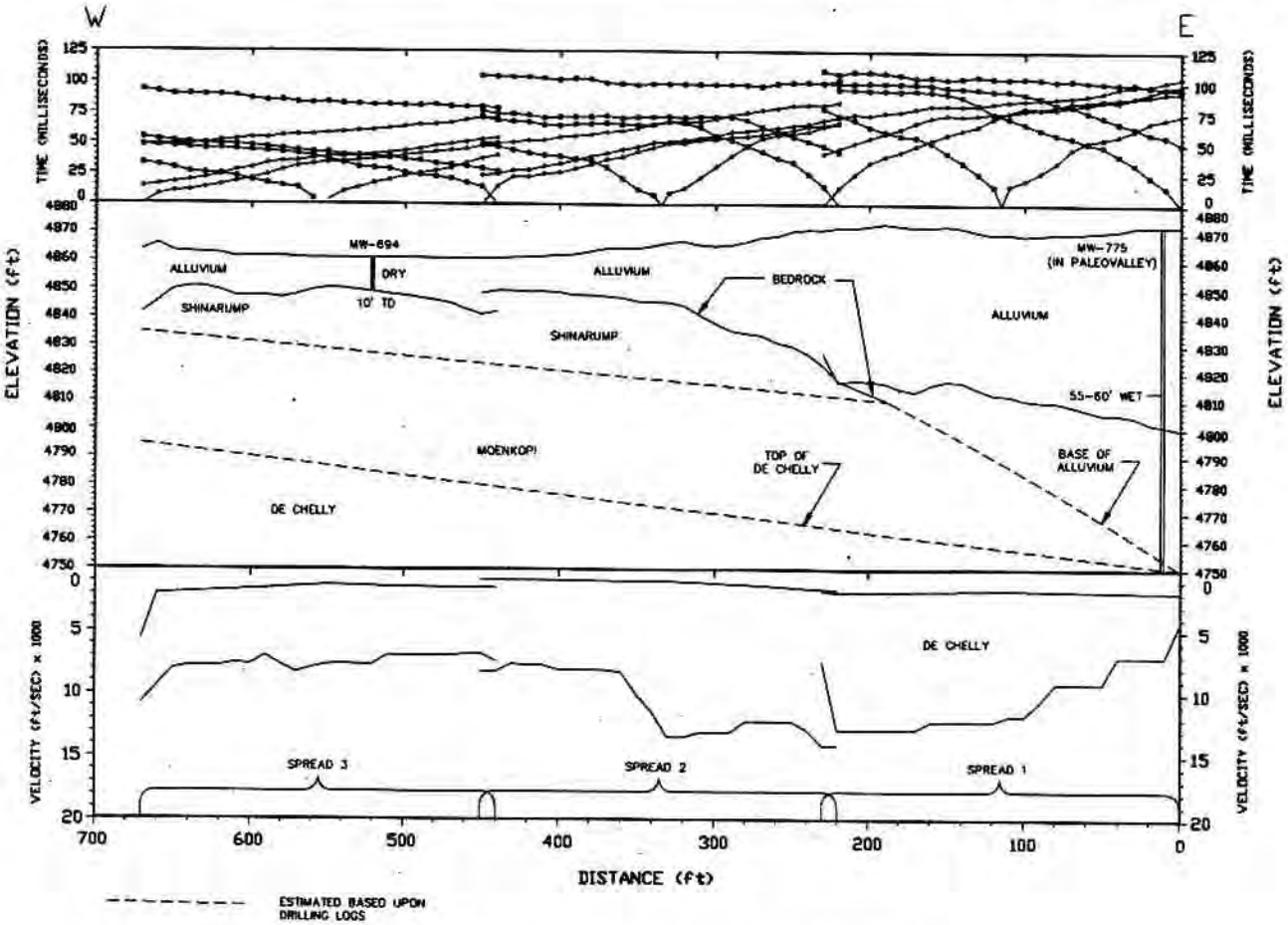

Figure 4–4. Seismic Interpretation for Line 3

The profile of the paleovalley is revealed in seismic line 1 (Figure 4–2). The eastern extent of the paleovalley is defined at the Shinarump outcrop just east of monitor well MON–657 and extends in width approximately 350 ft west to the dipping beds of the Shinarump Member. Examination of seismic lines 2 and 3, which are approximately 1,000 and 500 ft north of seismic line 1, respectively, does not indicate the presence of the buried paleovalley. Therefore, the seismic refraction survey results suggest that the buried paleovalley is not present in the area of lines 2 and 3.

4.1.2 TEM Survey

TEM is a geophysical technique in which a steady-state current within a large transmitting loop is abruptly terminated, causing eddy currents to flow within conductive strata below the loop. These currents decay away with time according to the conductivity (resistivity) and geometry of these strata. A receiver coil is placed at the center of the transmitting loop in a central loop configuration to detect and record the magnetic field resulting from these eddy currents. The data acquired can be mathematically modeled to produce thicknesses and conductivities (resistivities) of subsurface layering.

A TEM survey was conducted at the former millsite to map the downgradient and lateral extent of the contaminant plume in the alluvial aquifer and, as a secondary objective, to provide subsurface information about the nature of the bedrock topography.

4.1.2.1 TEM Survey Procedure

TEM soundings (measurements) were obtained along the seven traverse lines shown in Figure 4–5 to map subsurface resistivity variation associated with changes in ground water ionic concentrations of contaminants and lateral changes in lithology. Data were obtained using the Geonics TEM–47 transmitter and the Geonics Digital PROTEM receiver in the central loop configuration with square transmitting loops 40 meters (m) on a side. Measurements were performed at 285-, 75-, and 30-Herz (Hz) base frequencies at each sounding location.

TEM data were processed initially to produce apparent resistivity versus sample time using the "all time" (ramp-corrected) apparent resistivity formulations provided by the RAMPRES2 (Sandberg 1990) computer software code. An approximate depth section was then created for each TEM traverse by plotting the apparent resistivity at the diffusion depth (Christensen 1995) and contouring the values. The resulting approximation yields a relatively sharp upper boundary for a conductive layer and a diffuse lower boundary.

To improve the depth resolution of the interpretations, one-dimensional layered-earth modeling was employed using a nonlinear least-squares iterative algorithm to fit field data with theoretical data calculated from specific layered-earth parameters using the EINVRT5 computer code (an updated version of EINVRT4, Sandberg 1990). Simultaneous inverse modeling of TEM data with resistivity and IP data was also used to improve layered-earth parameter resolution. EINVRT5 was also used for simultaneous inverse modeling.

4.1.2.2 TEM Results

Apparent resistivity values using the ramp-corrected formulations for gate 5 of the TEM sounding data sets and the 285 Hz base frequency are shown in Figure 4–6. The resulting logarithmic contours indicate ground water contamination in areas of low apparent resistivity. The low apparent resistivity trend appears to originate near the new tailings area near monitor well MON–606 and extends in a northerly direction for approximately 4,500 ft.

Modeling results for the resistivity values measured at sounding station TEM–14, located near alluvial monitor well MON–606, indicate that a low-resistivity layer begins at the top of the alluvial water surface and extends only to the top of the underlying Shinarump Member. This result indicates that ground water contamination is confined to the alluvial aquifer at this location (Rogers and Sandberg 1998). Similarly, modeling results for the resistivity values measured at sounding station TEM–50, located near alluvial monitor well MON–653, indicate that ground-water contamination is restricted to the alluvial aquifer at that location.

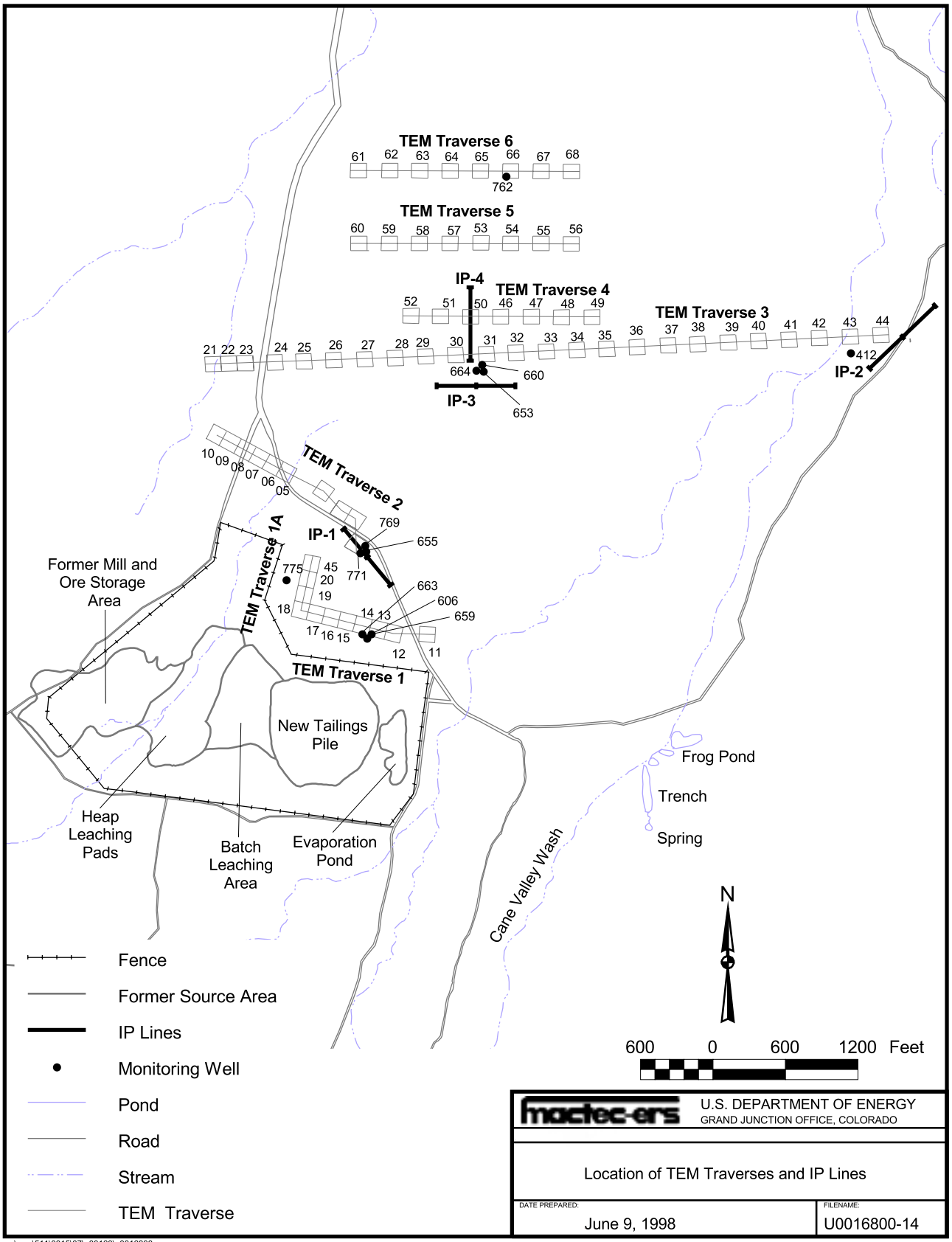

Figure 4-5. Location of TEM Traverses and IP Lines

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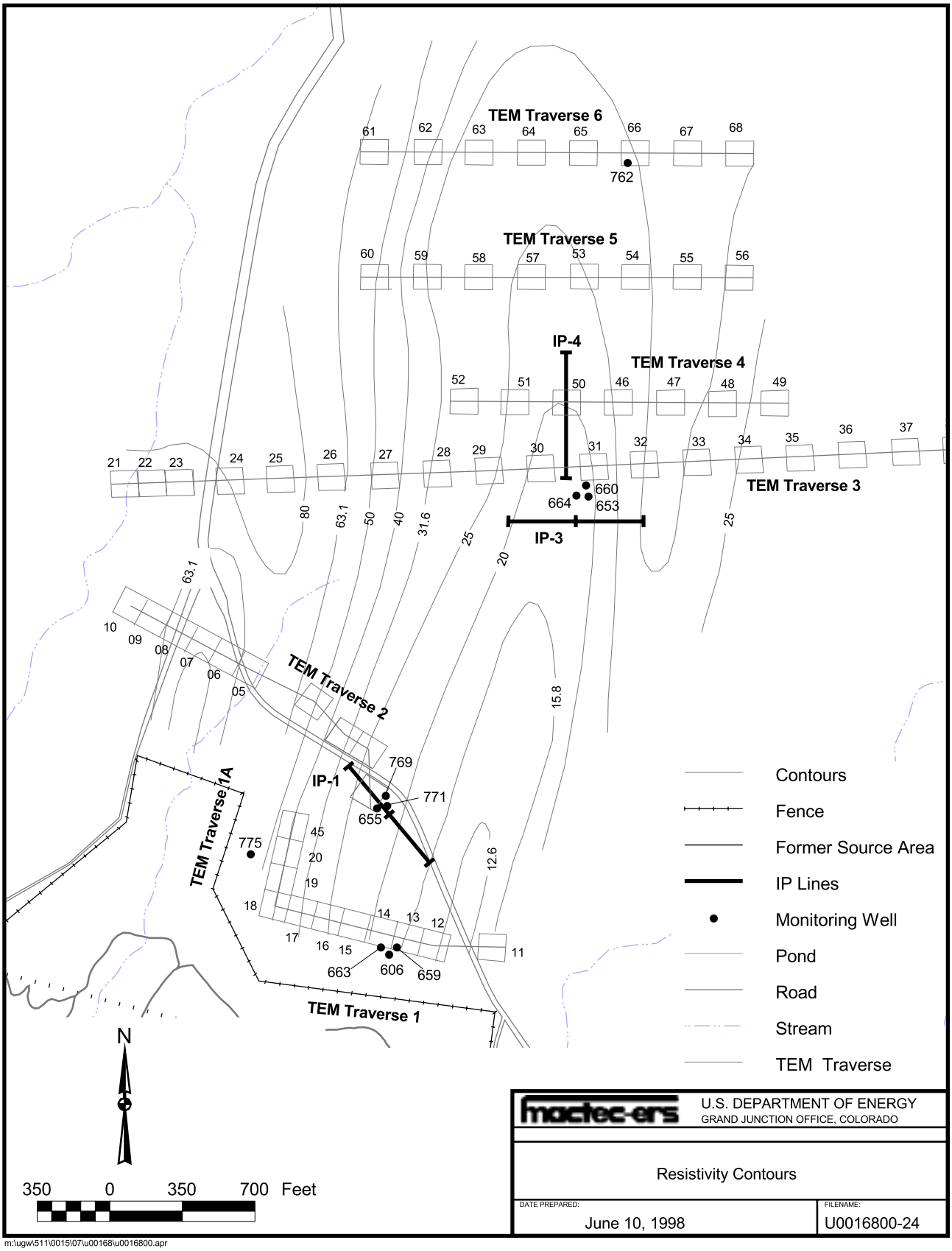

Figure 4-6. Resistivity Contours

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4.1.3 Resistivity/IP Survey

A Resistivity/IP survey was conducted at the former millsite to map the downgradient and lateral extent of the contaminant plume in the alluvial aquifer and, as a secondary objective, to provide subsurface information about the nature of the bedrock topography.

A resistivity survey is a geophysical technique of measuring the electrical resistivity of earth strata. Soundings (measurements) are usually performed using the Schlumberger resistivity array, which consists of a collinear four-electrode array in which current is injected and removed from the outer electrodes, and the resulting voltage (potential) is measured between the inner two electrodes. The separation of the outer electrodes is subsequently increased while the inner potential electrodes are held at a constant separation, which generates a data set of voltage versus current electrode separation. Larger current electrode separations result in a sampling of the electrical resistivity of deeper strata. These data are used to produce layer thickness and resistivities of strata below the sounding location by employing computer modeling methods to interpret the data.

An IP survey is a geophysical technique that measures the storage of electrical charge in strata. A steady-state transmitted electrical current is abruptly terminated. This results in a decay of stored charge over time to a neutral level. This effect produces a continued decaying voltage after current shutoff as a function of time. This IP effect is sampled versus time since transmitter turnoff to produce a reading of chargeability. IP and resistivity data are usually acquired simultaneously by using a bipolar transmitter waveform in which the current is on (+), off, on (–), off, and on (+) again. The receiver obtains the resistivity measurement while the transmitter is on and the IP measurement while the transmitter is off.

4.1.3.1 Resistivity/IP Procedure

Resistivity and IP soundings were obtained at four locations (IP–1 through IP–4) shown in Figure 4–5 using the Schlumberger array at current electrode half-separations ranging from 1.58 m to 100 m at logarithmic increments using 10 per decade. A Phoenix IPT–1 transmitter was used with a 3 kW generator for power. A Zonge GDP–32 general purpose receiver was used to collect the resistivity and IP data in the time domain. Data were obtained using an 8-second waveform.

Resistivity and IP data were plotted in the field for initial data quality inspection using calculated apparent resistivity and observed chargeability versus half-current electrode separation (AB/2). Simultaneous resistivity and IP modeling was performed using EINVRT5 (an updated version of EINVRT4, Sandberg 1990), a nonlinear least-squares inverse modeling computer code.

4.1.3.2 Resistivity/IP Results

The IP method, because of its ability to detect polarization effects due to clay mineralogy in the membrane polarization mechanism, can be used to distinguish conductive layers that result from an increase in ionic concentrations (ground water contamination), from those that result from the presence of clays.

Apparent resistivity and apparent chargeability versus current electrode half-spacing are shown in Figure 4–7 for data collected at location IP–4 near alluvial monitor well MON–653. The data indicate a conductive layer at depth bounded above and below by more resistive layers. This supports the interpretation that the vertical extent of alluvial ground water contamination is constrained by the top of the water table above and by the top of the Shinarump bedrock formation below. Similar curves and results are obtained for data collected at locations IP–1 and IP–3.

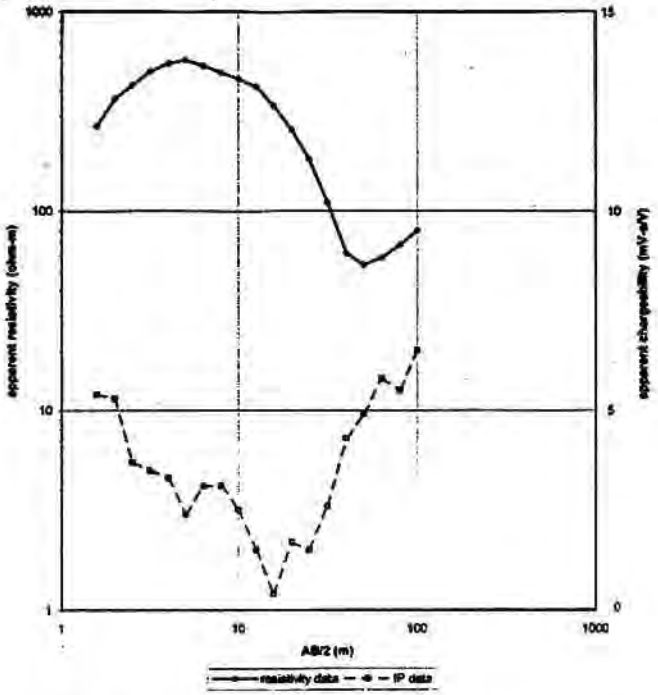

Figure 4–7. IP–4 Resistivity/IP Field Data Curves

4.2 Direct-Push and Hand-Auger Ground Water Sampling

The extent and nature of site-related ground water contamination was determined by delineating the vertical and horizontal distribution of nitrate and sulfate concentrations in the alluvial aquifer using a direct-push sampling device (Hydropunch). Information obtained from the Hydropunch water sampling was supplemented with water samples collected from shallow hand-augered borings located mostly to the east of the site along Cane Valley Wash. The water samples collected by the Hydropunch method and from the shallow hand-augered borings were analyzed for nitrate and sulfate in a mobile laboratory.

The Hydropunch sampling method allows rapid sampling of the ground water from a discrete 2-ft interval. Field analyses provide a quick turnaround time for nitrate and sulfate concentrations so that the site conceptual model can be updated daily and the choice of subsequent sampling locations can be optimized.

4.2.1 Hydropunch Sampling Procedure

A CME–850 track-mounted hollow-stem auger rig was employed to collect ground water samples with the Hydropunch device. With the track-mounted rig centered over the sample location, the

auger was advanced down through the Quaternary alluvium to a depth approximately 2 to 5 ft above the desired sampling point. The Hydropunch device was then inserted into the hollow-stem auger and pressed into the sampling zone of interest. A discrete ground water sample was collected from the device's 2-ft screened interval using a small diameter bailer and analyzed in a mobile field laboratory for sulfate and nitrate by spectrophotometry. Multiple samples were collected at the same location by removing the Hydropunch device and advancing the auger to a depth approximately 2 to 5 ft above the next sampling point. A ground water sample was then collected and analyzed in the same manner as the previous one.

One to four alluvial water samples were collected with the Hydropunch device at each location to profile the contaminant plume as a function of depth. The location and number of samples were determined from results of the surface geophysical survey and from sulfate and nitrate concentrations in ground water samples obtained concurrent with the Hydropunch.

A hand auger was also used to collect ground water samples at selected locations along Cane Valley Wash and near the former source areas where the depth to water is relatively shallow. Samples were collected by first hand-augering a 4-in.-diameter borehole to a depth up to 8 ft below the ground surface. The auger was then removed and a small diameter bailer was used to collect water from the open borehole.

Analytical results of the water sampling were evaluated and integrated with existing data on a day-to-day basis to update the site conceptual model. The updated site conceptual model was used to guide the locations for the next day's sampling activities.

The following procedures were used for the collection and analyses of the water samples:

- LQ–11(P), "Standard Practice for Sampling Liquids," (GJO 1998).
- ESL Procedure 1.3, "Nitrate Analysis," *Environmental Sciences Laboratory Procedure Manual* (1992).
- ESL Procedure 1.5, "Sulfate Analysis," *Environmental Sciences Laboratory Procedure Manual* (1992).

4.2.2 Hydropunch Sampling Results

During the field investigation, 38 ground water grab samples were collected from 17 auger borings by using the Hydropunch. Eleven shallow ground water samples were collected from newly-drilled hand-augered borings. The Hydropunch and hand-auger sample locations are shown in Figure 4–8. In general, locations where only Hydropunch samples were collected are designated by the 600 series of numbers. The 700 series includes new monitor wells and Hydropunch borings that were also completed as monitor wells. The hand-auger locations are designated by the 800 series. The hand-auger locations shown include six historical hand-auger locations as well as eleven new locations where water samples were collected. The information in Figure 4–8 is shown in greater detail in Plate 1, which is provided in the envelope pocket of this SOWP.

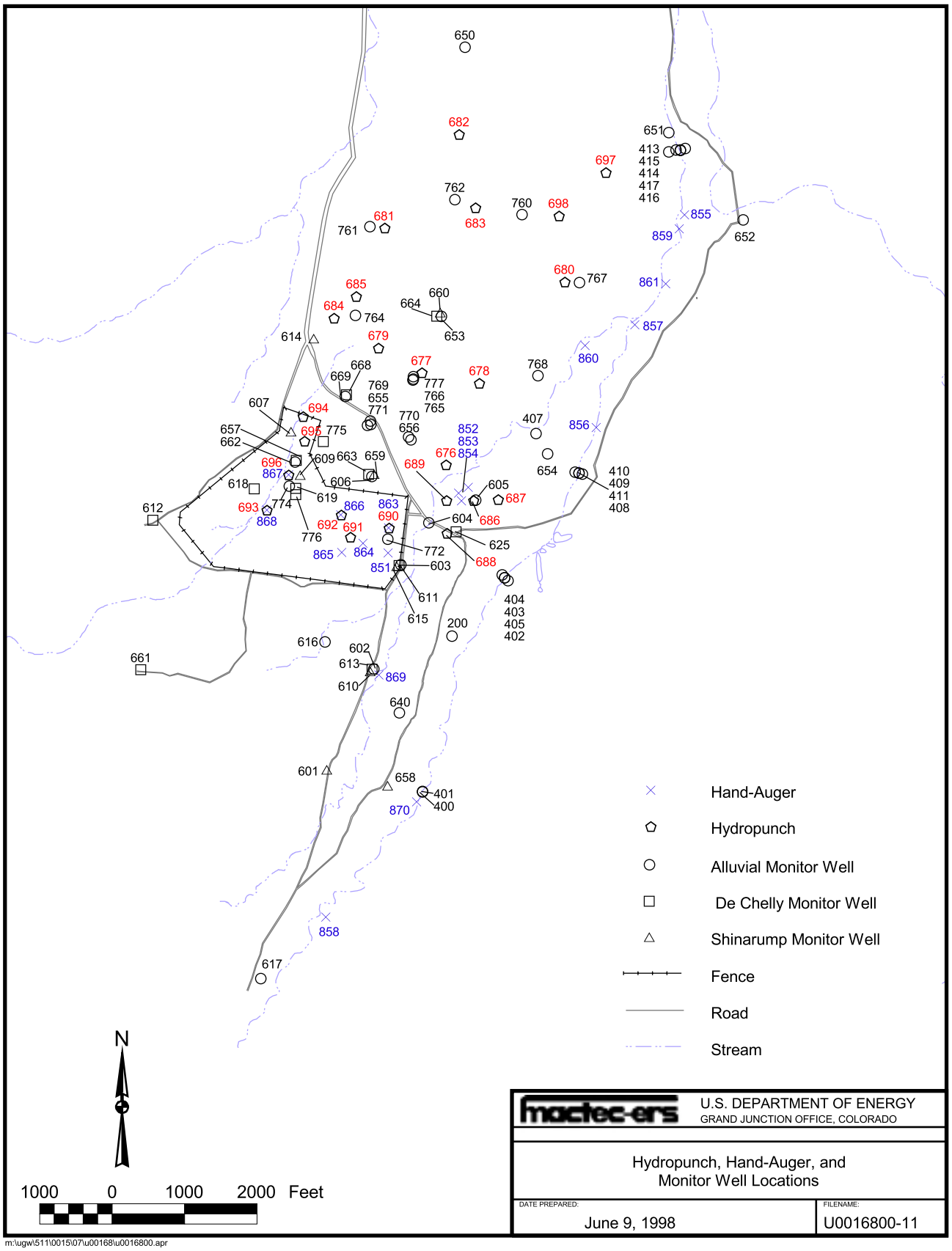

Figure 4-8. Hydropunch, Hand Auger, and Monitor Well Locations

Hydropunch water sample results for nitrate and sulfate field analyses are summarized in Table 4–1. Hand-auger water sample results for nitrate and sulfate field analyses are summarized in Table 4–2.

Table 4–1. Hydropunch Data from the 1997 Field Investigation

Hydropunch Location	From Depth (ft)	To Depth (ft)	Nitrate (mg/L)	Sulfate (mg/L)
MON–676	21	23	98	560
	48	50	9	152
MON–677	38	40	792	1,500
	60	62	726	1,000
	85	87	475	900
MON–678	30	32	673	2,250
	60	62	286	1,130
MON–679	45	47	1210	2,000
MON–680	18	20	30	1,250
	58	60	84	1,065
	69	71	2	70
	89	91	<1	90
MON–681	43	45	95	630
	50	52	53	380
MON–682	38	40	1	195
	65	67	18	345
	90	92	18	370
MON–683	33	35	25	205
	63	65	37	195
	85	87	51	250
MON–685	45	47	75	375
MON–686	10	12	<1	1,175
	35	37	<1	300
	74.5	76.5	<1	225
MON–687	28	30	2	500
MON–688	9	10	<1	260
MON–689	13	15	3	140
	48	50	<1	125
	71	73	<1	75
MON–690	18	20	5	140
	27	28	10	120
MON–696	50	52	74	240
	70	72	38	150
MON–697	21	23	<1	1,000
	53	55	<1	190
	73	75	<1	150
MON–698	18	20	<1	450
	45	47	<1	400

Table 4–2. Hand Auger Water Sample Results from the 1997 Field Investigation

Location Code	Depth (ft)	Nitrate (mg/L)	Sulfate (mg/L)
0851	7	220	600
0852	5	< 1	290
0853	5	1.76	126
0854	5	< 1	420
0855	4	12.1	1,500
0856	4	< 1	70
0857	5	< 1	95
0858	5	5.72	225
0859	4	10.34	470
0860	7	< 1	215
0861	3	< 1	70

At the conclusion of the Hydropunch and shallow ground water sampling, all the newly acquired data were evaluated and integrated with the most recent site conceptual model to determine the optimum location to establish the alluvial monitor well network and to place a monitor-extraction well and a bedrock monitor well.

4.3 Ground Water Well Installations

Information regarding the nature and extent of the alluvial contaminant plume, based on the results of Hydropunch sampling and field analyses, was used to optimize the design of the alluvial monitor well network (see Figure 4–8). The areal extent of the most contaminated portion of the alluvial aquifer, as defined by water samples containing nitrate concentrations exceeding 500 mg/L (Table 4–1), was used to guide the location for a 4-in. monitor-extraction well MON–765. Hydropunch sampling results were also used in combination with the results of the surface geophysical surveys and existing depth-to-bedrock well control to establish the optimum location for a paleovalley bedrock monitor well.

4.3.1 Installation Procedures

Alluvial monitor wells (MON–760 to –762, –764, –766 to –772, –774, and –777) were constructed using 2-in. i.d., flush-joint, threaded polyvinyl chloride (PVC) casing, and slotted PVC screen. The annular space around each casing was filled with sand from the bottom of the borehole to a level 2 ft above the top of the screen. A 3-ft bentonite seal was installed above the filter pack, and the remaining annular space was filled to 2 ft below ground level with an expanding grout mixture. Concrete was used to fill the remaining annulus to ground level and to install the well-cover pad.

The alluvial monitor-extraction well (MON–765) installed near the center of the contaminant plume was constructed using 4-in. i.d., flush-joint, threaded PVC casing and a 30-ft slotted PVC screen. The bottom of the well screen was installed at the bedrock and alluvium contact. The top of the well screen is approximately 15 ft below the alluvial water level. Sand was placed in the annular space from the bottom of the borehole to a depth of 2 ft above the top of the well screen. A 3-ft bentonite seal was installed above the filter sand pack, and the remaining annular space was

filled to a depth of 2 ft below the ground surface with an expanding grout mixture. Concrete was used to fill the remaining annulus to the ground surface and to install the well-cover pad.

A new bedrock monitor well (MON–775) was installed approximately 800 ft downgradient from the former old tailings pile/heap-leach area, along the northeast-trending axis of the paleovalley. This monitor well was constructed by advancing the borehole with a hollow-stem auger through 121 ft of Quaternary material and through several feet of the weathered portion of the Moenkopi Formation. When the auger reached competent Moenkopi Formation, a 5-in. diameter steel casing was cemented in place to prevent migration of contaminants from the alluvial aquifer into the lower De Chelly Sandstone aquifer. The cement was allowed to cure and was tested before the borehole was advanced downward by coring through the remaining section of the Moenkopi Formation and into the upper portion of the De Chelly Sandstone. After coring continued 40 ft into the De Chelly Sandstone the boring was completed with 2-in. i.d., flush-joint, threaded PVC casing and a 25-ft slotted screen.

A second bedrock monitor well (MON–776) was installed 50 ft south of existing production well MON–619 for use as an observation well during an aquifer test. Both the new bedrock boring and existing uncased production well MON–619 were completed using 6-in. i.d. flush-joint, threaded PVC casing and 50-ft slotted screens. The top of the screened interval for both wells was placed approximately 10 ft below the Moenkopi–De Chelly contact. The depth to the Moenkopi and De Chelly contact for well MON–619 was based on the core obtained from the boring for the new bedrock well MON–776. The boring for well MON–776 was advanced by coring to the desired depth, then reaming to the proper diameter to accommodate the well casing and protective well cover.

Detailed well construction procedures are available in the Drilling Statement of Work in the *Work Plan for Characterization Activities at the UMTRA Monument Valley Project Site* (DOE 1997c) and in the procedure that was used for the well installations: LQ–14(P), "Technical Comments on ASTM D 5092—Standard Practice for Design and Installation of Ground-Water Monitor Wells in Aquifers" (GJO 1998). Lithologic and monitor well completion logs are presented in Appendix A.

4.3.2 Alluvial Monitor Well Network

Thirteen 2-in. diameter water wells were installed during the 1997 fieldwork to monitor migration of the contaminant plume in the alluvial aquifer. The locations of the new alluvial aquifer wells are shown in Figure 4–8.

Six of the monitor wells (MON–760, –761, –762, –764, –767, and –768) were installed to monitor the downgradient and lateral extent of the plume boundary. These wells were constructed so that the screened intervals intersect the most likely zone where the highest contaminant concentrations at the plume boundary can be expected to occur. Depths for the screened intervals were based on results of the vertical concentration profiling obtained from the Hydropunch sampling, existing monitor well control, and lithologic information from auger cuttings and split-barrel sampling.

Five monitor wells (MON–766, –769, –770, –771, and –777) were installed to monitor the vertical distribution of contaminants near the center of the plume where the highest nitrate concentrations were detected and for use as observation wells during aquifer tests.

Two monitor wells (MON–772 and MON–774) were installed near the center of the old tailings pile/heap-leach pads and near the eastern edge of the new tailings pile, respectively, to evaluate the potential for residual contaminants in the former source areas. Well MON–774 was also designed to serve as an observation well during an aquifer test.

Construction details such as the screen depth, screen length, total depth of the well, and the geologic formation in which the well is screened are summarized in Table 4–3. Results of the alluvial aquifer tests conducted at wells MON–766, –769, –770, –771, –774, and –777 are provided in Section 4.6. Results of ground water sampling and laboratory chemical analyses are provided in Appendix C.

4.3.3 Alluvial Monitor and Extraction Well

Well MON–765 was installed near the center of the alluvial plume where the highest nitrate concentrations were detected (Figure 4–8). This 4-in. diameter well is used (1) as a ground water sampling well to monitor the vertical distribution of plume contaminants in the middle to lower portion of the alluvial aquifer, (2) as an aquifer test well, and (3) as a potential extraction well during remedial action.

Construction details, such as the screen depth, screen length, total depth of the well, and the geologic formation in which the well is screened are summarized in Table 4–3 and in Appendix A. Results of the alluvial aquifer test conducted at well MON–765 are provided in Section 4.6 and in Appendix B. Results of ground water sampling and laboratory chemical analyses are provided in Appendix C.

4.3.4 Bedrock Monitor Wells

Diamond core holes were drilled approximately 50 ft into the De Chelly bedrock aquifer at two locations. The first location, well MON–775 (Figure 4–8), was drilled to obtain geologic information regarding the characteristics of the buried paleovalley and then completed as a monitor well to evaluate potential uranium contamination in the bedrock aquifer. The second location, well MON–776, was drilled to determine depths to geologic contacts. The core hole was subsequently completed as a bedrock monitor well and used as an observation well during the aquifer test in the De Chelly Sandstone.

Table 4–3. Construction Summary Table—Existing and New Monitor Wells

Location Code	North Coord. (State-Plane)	East Coord. (State-Plane)	Ground Elev. (ft)	Borehole Depth (bls)[a]	Borehole Dia. (in.)	Top of Casing Elev. (ft)	Casing Length (ft)	Casing Diameter (in.)	Screen Depth (bls)	Screen Length (ft)	Flow Code[b]	Zone of Compl.[c]
New Wells Installed in 1997												
0760	2162653	590711	4812.3	77.0	7.6	4814.8	78.0	2.0	55.0	20.0	D	Al
0761	2162488	588611	4832.3	55.5	7.6	4835.0	57.2	2.0	39.0	10.0	D	Al
0762	2162865	589783	4818.1	90.0	7.6	4820.7	57.1	2.0	29.0	20.0	D	Al
0764	2161265	588408	4848.7	52.5	7.6	4851.5	55.3	2.0	47.0	5.0	D	Al
0765	2160368	589204	4845.6	89.0	10.5	4848.5	91.8	4.0	58.6	30.1	D	Al
0766	2160418	589211	4844.8	60.0	7.6	4848.0	60.7	2.0	47.2	10.0	D	Al
0767	2161713	591504	4805.5	65.0	7.6	4808.3	66.8	2.0	43.5	20.0	D	Al
0768	2160426	590931	4817.9	45.0	7.6	4820.7	47.8	2.0	24.4	20.0	D	Al
0769	2159804	588617	4858.3	44.0	7.6	4861.3	47.0	2.0	33.4	10.0	D	Al
0770	2159579	589141	4854.4	65.5	7.6	4857.3	68.4	2.0	54.9	10.0	D	Al
0771	2159742	588575	4860.8	79.0	7.6	4863.3	80.5	2.0	57.4	20.0	D	Al
0772	2158168	588854	4844.7	30.0	7.6	4847.6	30.9	2.0	7.4	20.0	O	Al
0774	2158901	587494	4877.4	55.5	7.6	4880.1	58.2	2.0	45.0	10.0	O	Al
0777	2160383	589206	4845.4	49.0	7.6	4848.2	50.1	2.0	31.8	15.0	D	Al
0775	2159521	587965	4876.5	167.8	10.5	4879.7	170.7	2.0	142.0	25.0	D	Dc
0776	2158791	587590	4880.4	150.2	9.9	4883.3	152.9	6.0	99.5	50.0	O	Dc
Wells Installed Before 1997												
0200	2156826	589741	-	-	-	-	-	-	-	-	U	AL
0400	2154679	589333	4870.7	12.7	2.0	4870.4	12.4	2.0	7.8	4.5	U	AL
0401	2154678	589332	4870.7	6.8	2.0	4870.4	6.5	2.0	4.0	2.0	U	AL
0402	2157594	590516	4840.6	10.1	2.0	4840.3	9.8	2.0	5.2	4.5	U	AL
0403	2157637	590468	4836.6	8.3	2.0	4836.2	8.0	2.0	3.4	4.5	U	AL
0404	2157674	590435	4838.2	8.6	2.0	4837.7	8.0	2.0	3.7	4.5	U	AL
0405	2157637	590468	4836.6	3.7	2.0	4836.5	3.6	2.0	0.7	2.3	U	AL
0407	2159626	590905	4820.4	11.8	2.0	4820.1	11.5	2.0	6.9	4.5	C	AL
0408	2159070	591542	4823.7	8.8	2.0	4823.5	8.6	2.0	3.9	4.5	C	AL
0409	2159084	591495	4821.7	16.0	2.0	4821.5	15.8	2.0	11.1	4.5	C	AL
0410	2159096	591442	4823.7	10.5	2.0	4823.4	10.3	2.0	5.6	4.5	C	AL
0411	2159083	591495	4821.7	4.7	2.0	4821.4	4.4	2.0	2.0	1.6	C	AL
0413	2163573	592962	4784.1	10.5	2.0	4783.9	10.2	2.0	5.6	4.5	C	AL
0414	2163547	592893	4782.4	15.7	2.0	4782.0	15.3	2.0	10.8	4.5	C	AL
0415	2163551	592833	4784.2	9.4	2.0	4783.8	9.0	2.0	4.5	4.5	C	AL
0416	2163523	592736	4785.7	10.5	2.0	4785.3	10.1	2.0	5.2	4.5	C	AL
0417	2163546	592893	4782.4	5.0	2.0	4782.2	4.8	2.0	1.4	3.0	C	AL
0602	2156378	588661	4862.1	35.0	6.6	4864.4	33.9	2.0	19.5	10.0	U	AL
0603	2157813	589037	4847.6	55.0	6.6	4849.4	56.8	2.0	43.0	10.0	U	AL
0604	2158397	589424	4838.7	30.0	6.6	4840.4	31.7	2.0	13.0	15.0	C	AL
0605	2158708	590066	4832.6	32.0	6.6	4835.1	33.5	2.0	14.0	15.0	C	AL
0606	2159034	588634	4861.8	47.0	6.6	4864.7	50.0	2.0	32.0	10.0	D	AL
0616	2156748	587988	4871.1	-	-	4869.5	6.5	-	-	-	U	AL
0617	2152094	587098	4907.8	-	-	4909.1	-	-	-	-	U	AL
0640	2155769	589014	4875.0	-	-	-	-	-	-	-	U	AL
0650	2164970	589923	4791.3	99.5	7.9	4794.3	102.5	4.0	77.5	20.0	D	AL
0651	2163789	592735	4784.6	82.0	7.9	4787.9	85.2	4.0	20.0	60.0	C	AL
0652	2162582	593760	4805.5	56.0	7.9	4808.9	61.4	4.0	34.0	20.0	C	AL
0653	2161250	589596	4834.3	78.0	7.9	4837.1	80.8	4.0	56.0	20.0	D	AL
0654	2159351	591064	4821.6	79.0	7.9	4824.4	81.8	4.0	57.0	20.0	C	AL
0655	2159754	588624	4858.9	60.0	7.9	4862.1	63.2	4.0	38.0	20.0	D	AL
0656	2159545	589175	4853.5	60.0	7.9	4856.3	62.8	4.0	38.0	20.0	D	AL
0662	2159237	587577	4875.8	70.0	7.9	4878.6	72.3	4.0	37.5	30.0	D	AL
0669	2160145	588265	4864.1	56.0	7.9	4867.2	59.1	4.0	34.0	20.0	D	AL
0601	2154981	588018	4881.8	24.0	6.6	4884.9	27.1	2.0	12.0	10.0	U	SR-AL
0607	2159657	587519	4868.0	30.0	6.6	4871.4	30.9	2.0	12.5	10.0	D	SR
0609	2159053	587650	4877.0	15.0	6.6	4880.0	17.0	2.0	7.0	5.0	O	SR-AL
0610	2156339	588612	4862.2	130.5	6.6	4863.2	86.0	2.0	63.0	20.0	U	SR
0614	2160940	587832	4855.6	84.5	8.0	4856.8	71.2	2.0	48.0	20.0	D	SR-AL
0615	2157795	588981	4848.6	110.0	6.6	4850.2	91.6	2.0	68.0	20.0	U	SR
0658	2154764	588857	4877.0	165.0	7.9	4880.0	159.9	4.0	135.0	20.0	U	SR
0659	2159070	588670	4861.7	110.0	7.9	4865.0	112.3	4.0	87.0	20.0	D	SR
0660	2161303	589584	4833.6	155.0	7.9	4836.3	157.8	4.0	133.0	20.0	D	SR
0611	2157811	589017	4848.2	185.0	6.6	4849.3	186.1	2.0	163.0	20.0	U	DC
0612	2158437	585615	5006.2	215.0	6.6	5007.8	216.6	2.0	175.0	20.0	U	DC
0613	2156378	588644	4861.9	160.0	6.6	4864.3	162.4	2.0	138.0	20.0	U	DC
0618	2158868	587017	4922.1	153.0	12.0	4924.8	155.7	12.0	-	-	O	DC
0619	2158877	587587	4886.3	154.4	12.0	4888.6	156.8	6.0	103.9	50.0	O	DC
0657	2159265	587597	4876.6	140.0	7.9	4879.0	140.4	4.0	121.0	15.0	O	DC
0663	2159070	588593	4862.4	217.0	7.9	4865.7	220.3	4.0	175.0	40.0	D	DC
0664	2161256	589537	4834.5	233.0	7.9	4837.4	235.8	4.0	211.0	20.0	D	DC
0668	2160171	588287	4865.0	218.0	7.9	4867.8	217.8	4.0	180.0	20.0	D	DC

[a]bls = below land surface
[b]Flow codes: C = Cross Gradient; D = Downgradient; O = On-site; U = Upgradient
[c]Zones of completion: Al = alluvium; Dc = De Chelly Sandstone Member of the Cutler Formation; Sr = Shinarump Member of the Chinle Formation

Elevated uranium concentrations have been detected in samples from monitor well MON–657, which is completed in the De Chelly Sandstone. Because the downgradient extent of potential contaminant migration in the De Chelly aquifer was unknown, a new bedrock well, MON–775, was installed northeast (downgradient) from well MON–657. The location, based on the results of the geophysical survey, surface geologic relationships, and existing depth-to-bedrock well control, is approximately 800 ft downgradient from the former old tailings pile/heap-leach area, along the northeast-trending axis of the paleovalley. Geologic information obtained from this boring also provides verification of the existence and nature of the paleovalley.

De Chelly ground water collected from production well MON–619 has also had elevated uranium concentrations—approximately 2.5 times the MCL of 0.044 mg/L. However, well MON–619 was an uncased and unscreened borehole that was used as a production well during operation of the mill. Because this well was not a properly installed monitor well, the source of uranium contamination in the De Chelly ground water could not be determined with reasonable certainty. To evaluate the potential extent of uranium contamination in the De Chelly aquifer, production well MON–619 was completed as a monitor well and a new well, MON–776, was installed approximately 50 ft upgradient (south) of existing well MON–619 (Figure 3–1). Well MON–776 is used as a monitor well to obtain samples for uranium analysis and as an observation well during an aquifer test.

Construction details, such as the screen depth, screen length, total depth of the well, and the geologic formation in which the well is screened are summarized in Table 4–3 and in Appendix A. Results of the bedrock aquifer tests at well MON–776 are provided in Section 4.6 and in Appendix B. Results of ground-water sampling and laboratory chemical analyses are provided in Appendix C.

4.4 Sediment and Bedrock Sampling

Both composite and discrete sediment samples were collected during the field investigation. The purpose of the sediment sampling was to obtain lithologic information on the nature of potential lacustrine deposits and alluvial sediments that may influence the migration of contaminants through the alluvial aquifer. Discrete and continuous core samples of bedrock formations were also collected to verify stratigraphy and geologic contacts.

4.4.1 Sediment and Bedrock Sampling Procedures

Composite samples of the auger cuttings were collected every 5 ft during the Hydropunch ground water sampling activities and monitor well installations. Lithologic descriptions of the material were recorded by the site geologist using Unified Soil Classification System terminology in Section SL–24(P) of the *Environmental Procedures Catalog* (GJO 1998).

Discrete subsurface sediment and bedrock samples were collected using a track-mounted hollow-stem auger rig and a split-barrel sampling device. With the track-mounted auger rig centered over the sample location, the auger was advanced down through the Quaternary material to the desired sampling depth. When the auger reached the desired sampling depth, a 3-in. o.d. by 18-in.-long split-barrel sampler was attached to the drive rod and lowered to the top of the interval to be sampled. The barrel was then driven for the length of the sampler or until 6 in. or

less of penetration was achieved after 50 blows with a 140-pound drop hammer having a 30-in. drop. After the split-barrel was full, or no further penetration was possible, the barrel was carefully removed from the borehole and separated from the drive-rod assembly. The barrel was then laid flat on an uncontaminated surface and the head and drive shoe were removed. One-half of the split barrel was removed to expose the sample. The uppermost portion of sample in the split barrel was inspected and the slough was discarded, if present. The remaining sample was considered representative and placed in a stainless steel or aluminum pan, if necessary. The material was described by the site geologist using Unified Soil Classification System terminology in Section SL–24(P) of the *Environmental Procedures Catalog* (GJO 1998).

Continuous core samples of the bedrock formations were collected using a nominal 5-ft long, double tube, swivel-type, NX diamond core barrel and wireline system. Clean water was used as the circulation medium. State-of-the-industry diamond coring practices were used to effect the highest core recovery possible. Recovered core was washed and then placed in boxes within the longitudinal separators, from left to right, as a book would be read, that is, core was placed starting with the shallowest portion of the hole at the upper left corner and ending with core from the deepest portion of the hole in the lower right corner. Spacer blocks were inserted between the cored sections within the longitudinal separators where no recovery was noted. All core boxes, including the lids, were permanently marked showing top and bottom and the beginning and ending depths for the core. All core was described by the site geologist.

All sediment and bedrock sampling was performed in accordance with the following procedures from the *Environmental Procedures Catalog* (GJO 1998):

- SL–6(P), “Technical Comments on ASTM D 1452–80(90)—Standard Practice for Soil Investigation and Sampling by Auger Borings”

- SL–7(P), “Technical Comments on ASTM D 1586–84(92)—Standard Test Method for Penetration Test and Split-Barrel Sampling of Soils”

- SL–19(P), “Technical Comments on ASTM D 2488–93—Standard Practice for Description and Identification of Soils”

- SL–9(P), “Technical Comments on ASTM D 2113–83—Standard Practice for Diamond Core Drilling for Site Investigation”

4.4.2 Sediment and Bedrock Sampling Results

Lithologic descriptions of composite samples of the auger cuttings collected at each Hydropunch and monitor well location shown in Figure 4–8 are presented in the field logs in Appendix A.

Split-barrel samples were collected at the three locations shown on Figure 4–8 that coincide with monitor well MON–760 near the northeast edge of the nitrate plume, monitor well MON–761 near the northwest edge of the nitrate plume, and monitor well MON–774 near the center of the former old tailings/heap-leach area. Lithologic descriptions of the discrete samples collected with the split-barrel sampler are presented in the field logs in Appendix A.

Core samples were recovered from two new De Chelly monitor wells, MON–775 and MON–776, installed near the downgradient extent of the buried paleovalley and near the former old tailings area, respectively. The locations of the bedrock wells are shown on Figure 4–8. Lithologic descriptions of the core are provided in the field logs in Appendix A.

4.5 Subpile Soil Sampling

During the uranium milling operations at the Monument Valley site, several ponds were used for evaporation of milling fluids and for disposal of tailings. The radioactive material has been removed from the site. No radioactive materials exceeding 15 picocuries per gram (pCi/g) radium-226 were left. However, the potential exists for nonradionuclide contaminants to have seeped into the soils. Contaminated soils could contaminate infiltrating water as it passes through them and prolong the ground water cleanup effort.

Soil samples were collected and analyzed to evaluate the distribution of selected site-related constituents in the soils underlying the former tailings piles, heap leach pads, and evaporation ponds. Background soil samples were also collected and analyzed.

4.5.1 Subpile Soil Sampling Procedures

Twenty-six samples from nine soil borings were analyzed. Figure 4–9 shows the locations of the nine soil borings. Three soil borings were in the former location of the new tailings pile, and two each in the former heap-leach pads and evaporation pond. Two background soil borings were upgradient of the site. Each soil boring was hand augered to a depth of 3.5 to 8.5 ft. Samples were double bagged in clean plastic bags and placed in 5-gal plastic buckets for transport to the laboratory.

Lithologic logs of the soil were prepared in the field (Figure 4–10). The upper 1–2 ft was loose fill material that had been placed on the surface and graded after removal of the tailings and was not representative of the subpile soils. Samples were collected at approximately 1-ft intervals below the fill. Figure 4–10 shows the stratigraphic locations of the samples collected for this study.

4.5.2 Sample Preparation Methods

Samples were air-dried (no oven heat) and sieved to less than 2 millimeters (mm). Only a minor amount of material was excluded due to sieving. A petrographic thin section was made of each sample used in the extractions. The thin sections were examined to determine mineralogy and texture. Because some of the minerals of interest are water soluble (e.g., gypsum), the thin sections were cut and polished in oil. Two to three samples from each soil boring were analyzed.

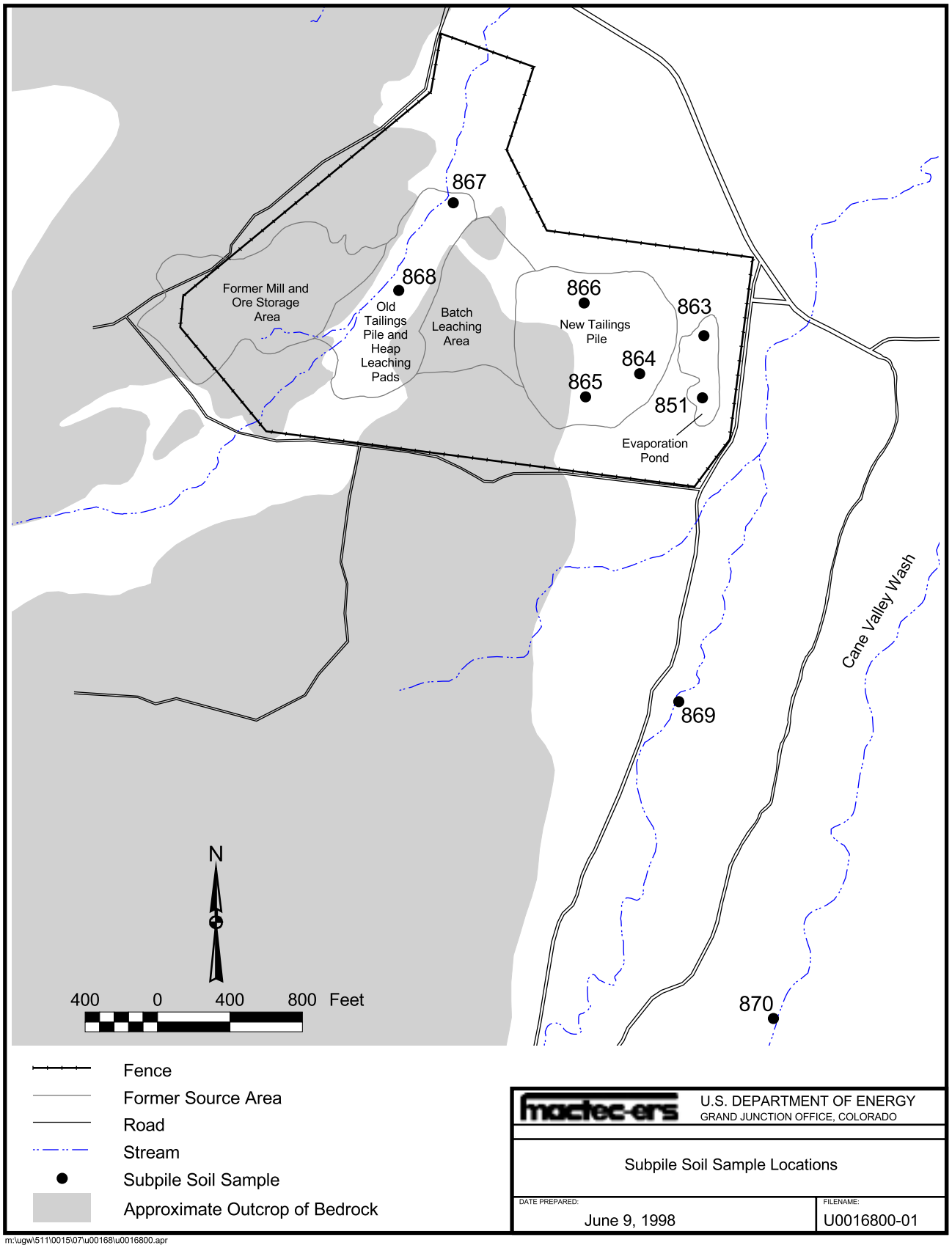

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Figure 4-9. Subpile Soil Sample Locations

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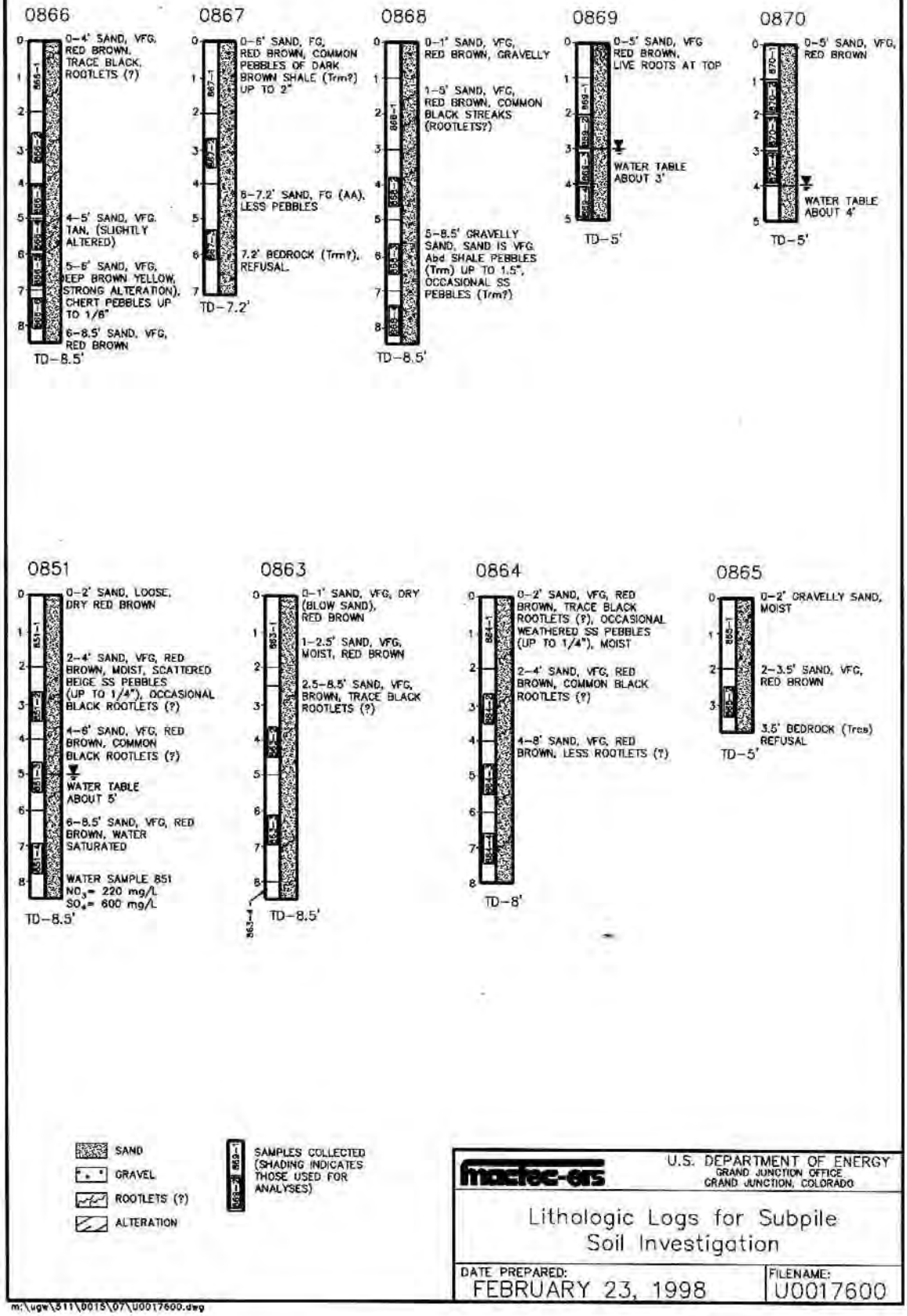

Figure 4–10. Lithologic Logs for Subpile Soil Investigation

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4.5.2.1 Chemical Extraction Methods

Chemical extractions were used to determine the potential mobility of contaminants. Each sample was extracted by using three separate lixiviants, and the residue was completely digested and analyzed. The lixiviants were deionized water, alluvial ground water, and 5-percent hydrochloric acid. Extractions were performed sequentially on the same starting material to avoid variation due to sample heterogeneity. Each extraction was harsher than the preceding one.

Each chemical extraction was related to scenarios that could cause contaminant release at the site. Deionized water was used first to simulate rain water that could leach the subpile sediments through infiltration. Soluble phases, including gypsum, dissolve in deionized water. Alluvial ground water was then used to simulate the water-table rise that could cause ground water to contact contaminated soils. Additional bicarbonate present in the site ground water should release additional uranium that may be sorbed to oxides or silicates. Five-percent hydrochloric acid was then used to remove amorphous ferric and manganese oxyhydroxides. Metals and radionuclides are likely to reside in these oxyhydroxides. The acid treatment also dissolves carbonate minerals and releases any sorbed cations. Although oxyhydroxides are stable in most soils, irrigation practices or other land use could cause reducing conditions in the soils and lead to dissolution of the oxyhydroxides with release of their sorbed constituents. Finally, a complete digestion of the sample residue was performed to determine the total concentrations of the constituents in the soil. Any additional constituents that are contained in recalcitrant mineral phases will be released and analyzed by this method. The results of the subpile samples were compared to the results from the background samples to estimate the degree of contamination.

The extraction procedure consists of the following steps:

- Two grams of soil (accurately weighed) were placed in a centrifuge tube with 100 milliliters (mL) of deionized water, and the contents were shaken on an end-over-end shaker for 4 hours.

- Contents were centrifuged to remove particles less than 2 micrometers (µm) in diameter. Supernatant was decanted into a 200-mL volumetric flask.

- Additional deionized water (about 100 mL) was added. Contents were shaken for 15 minutes, centrifuged, and decanted into the same 200-mL flask.

- The 200-mL flask was filled to volume with deionized water and filtered (0.2 µm filter). Alkalinity, pH, and Eh were measured. The remaining water was preserved and sent to the analytical lab for analyses.

- 100 mL of site ground water were added to the residue in the 100-mL tube and shaken for 4 hours. Composition of the ground water is (micrograms per liter [µg/L]): Mn = 30, Sr = 330, U = 1, V = 10, NH_4 = 9.3, NO_3 = 50, and SO_4 = 35,700.

- Contents were centrifuged to remove particles less than 2 µm in diameter. Supernatant was decanted into a second 200-mL volumetric flask.

- Additional site ground water (about 100 mL) was added. Contents were shaken for 15 minutes, centrifuged, and decanted into the same 200-mL flask.

- The 200-mL flask was filled to volume with site ground water and filtered (0.2 µm filter). Alkalinity, pH, and Eh were measured. The remaining water was preserved and sent to the analytical lab for analyses.

- The procedure was repeated using 5-percent hydrochloric acid.

- The residue was dried, ground, completely digested (microwave digestion with concentrated nitric acid), and analyzed.

- All extracted samples were analyzed for Mn, NH_4, SO_4, NO_3, U, V, and Sr. Nitrate and sulfate were analyzed by ion chromatography. Ammonium was determined by spectrophotometry. Mn, V, and Sr were analyzed by ICP–AES, and U by ICP–MS. The total digestions were analyzed for Mn, U, V, and Sr (NO_3, NH_4, and SO_4 are considered too volatile to provide meaningful results).

- From these data, the amount of each constituent removed during each step was calculated. The total amount of each constituent was also calculated.

4.5.3 Subpile Soil Sampling Results

Lithologic logs for the nine sample locations shown are provided in Figure 4–10. The lithology consists of a red-brown, very fine grained sand. Thin section observations indicate the presence of ferric oxyhydroxides.

Raw data and calculations for the leach analyses are presented in Tables 4–4 through 4–7. Each table lists the data and calculations from the sequential extractions: Deionized water (Table 4–4), ground water (Table 4–5), 5-percent HCl (Table 4–6), and total dissolution (Table 4–7). Bold type in the tables indicates that a concentration was less than the detection limit; for those, the detection limit was used in the calculations.

The extraction of strontium in sample 851–2 is used to illustrate the calculations. The effluent from the deionized water extraction had a strontium concentration of 89.1 µg/L (column 5, Table 4–4).

Two grams of sample were extracted with 200 mL of deionized water:

$$\frac{200\ \text{mL}}{2\ \text{g}} \times \frac{89.1\ \mu\text{g}}{\text{L}} \times \frac{\text{L}}{1{,}000\ \text{mL}} \times \frac{1{,}000\ \text{g}}{\text{kg}} \times \frac{\text{mg}}{1{,}000\ \mu\text{g}} = 8.91\ \text{mg/kg}$$

(Column 6, Table 4–4)

The residuum was then extracted with 200 mL of ground water that had a strontium concentration of 330 µg/L. After this extraction, 332 µg/L of Sr is in the effluent (Column 6, Table 4–5).

Table 4–4. Deionized Water Extraction

Sample ID	Area ID	Mn µg/L	Mn mg/kg	Sr µg/L	Sr mg/kg	U µg/L	U mg/kg	V µg/L	V mg/kg	NH_4 µg/L	NH_4 mg/kg	NO_3 µg/L	NO_3 mg/kg	SO_4 µg/L	SO_4 mg/kg
851-2	EP	1	0.1	89.1	8.91	1	0.1	1270	127	145	14.5	13100	1310	70500	7050
851-3	EP	7.7	0.77	52.5	5.25	1	0.1	877	87.7	41.6	4.16	7270	727	27700	2770
851-4	EP	23.1	2.31	29.2	2.92	1	0.1	544	54.4	178	17.8	1020	102	2120	212
863-2	EP	1.4	0.14	20.9	2.09	1	0.1	103	10.3	46.3	4.63	4760	476	15500	1550
863-3	EP	1	0.1	20.7	2.07	1	0.1	65.5	6.55	41.6	4.16	3800	380	12900	1290
863-4	EP	1	0.1	29.8	2.98	1	0.1	76.7	7.67	50.9	5.09	5190	519	16500	1650
864-2	NT	1	0.1	101	10.1	1	0.1	96.9	9.69	51.1	5.11	1030	103	91900	9190
864-3	NT	1	0.1	29	2.9	1	0.1	89.5	8.95	27.7	2.77	6720	672	14200	1420
864-4	NT	3.7	0.37	14.7	1.47	1	0.1	116	11.6	25.4	2.54	2200	220	3460	346
865-2	NT	1	0.1	50.2	5.02	1	0.1	244	24.4	30.1	3.01	4060	406	31600	3160
866-2	NT	1	0.1	60.6	6.06	1	0.1	353	35.3	1100	110	9730	973	18500	1850
866-3	NT	1.7	0.17	45.8	4.58	1	0.1	106	10.6	1360	136	8360	836	18000	1800
866-4	NT	6.3	0.63	24.8	2.48	1	0.1	98.8	9.88	1840	184	6990	699	4000	400
866-5	NT	5.5	0.55	13.5	1.35	1	0.1	54.3	5.43	2320	232	8240	824	2720	272
866-6	NT	2.8	0.28	11.7	1.17	1	0.1	15.5	1.55	2620	262	8140	814	2240	224
867-2	HL	1	0.1	12.5	1.25	1	0.1	13	1.3	37	3.7	616	61.6	255	25.5
867-3	HL	1.8	0.18	12.5	1.25	1	0.1	13	1.3	34.7	3.47	755	75.5	371	37.1
868-2	HL	1	0.1	10.1	1.01	1	0.1	50.7	5.07	48.6	4.86	8410	841	471	47.1
868-3	HL	1	0.1	9.2	0.92	1	0.1	68	6.8	20.8	2.08	553	55.3	752	75.2
868-4	HL	1	0.1	15.9	1.59	1	0.1	84.5	8.45	25.4	2.54	741	74.1	3470	347
869-2	BG	1	0.1	46.5	4.65	1	0.1	13	1.3	17.5	1.75	268	26.8	3870	387
869-3	BG	1	0.1	38.6	3.86	1	0.1	13	1.3	16.1	1.61	481	48.1	3430	343
869-4	BG	1	0.1	27.1	2.71	1	0.1	13	1.3	37	3.7	8080	808	2430	243
870-2	BG	1.8	0.18	29.5	2.95	1	0.1	13	1.3	34.7	3.47	255	25.5	1280	128
870-3	BG	6.1	0.61	24.8	2.48	1	0.1	13	1.3	23.1	2.31	384	38.4	1130	113
870-4	BG	5.9	0.59	23.8	2.38	1	0.1	13	1.3	25.4	2.54	729	72.9	949	94.9

BG = Background area
EP = Evaporation ponds
HL = Heap-leach pads
NT = New tailings pile
(Shaded cells = detection limit; actual value was lower)

Table 4–5. Ground Water Extraction

Sample ID	Area ID	Mn µg/L	* Mn in GW µg/L	**Mn mg/kg	Sr µg/L	* Sr in GW µg/L	Sr mg/kg	U µg/L	* U in GW µg/L	U mg/kg	V µg/L	* V in GW µg/L	V mg/kg	NH_4 µg/L	* NH_4 in GW µg/L	NH_4 mg/kg	NO_3 µg/L	* NO_3 in GW µg/L	NO_3 mg/kg	SO_4 µg/L	*SO_4 in GW µg/L	SO_4 mg/kg
851-2	EP	1.3	30	-2.9	332	330	0.2	15.9	1	1.49	148	10	13.80	64.2	9.3	5.49	4500	50	445	36400	35700	70
851-3	EP	1.2	30	-2.9	338	330	0.8	4	1	0.30	120	10	11.00	23.3	9.3	1.4	1730	50	168	35900	35700	20
851-4	EP	8.7	30	-2.1	346	330	1.6	5.6	1	0.46	48.1	10	3.81	37.4	9.3	2.81	1340	50	129	36300	35700	60
863-2	EP	1.5	30	-2.9	341	330	1.1	1.4	1	0.04	16	10	0.60	18.7	9.3	0.94	1290	50	124	36200	35700	50
863-3	EP	1	30	-2.9	339	330	0.9	1.6	1	0.06	13	10	0.30	18.7	9.3	0.94	631	50	58.1	35900	35700	20
863-4	EP	1	30	-2.9	330	330	0.0	1.3	1	0.03	13	10	0.30	23.3	9.3	1.4	623	50	57.3	36100	35700	40
864-2	NT	2.9	30	-2.7	338	330	0.8	1	1	0.00	20.8	10	1.08	28	9.3	1.87	717	50	66.7	36100	35700	40
864-3	NT	1	30	-2.9	343	330	1.3	1	1	0.00	22.5	10	1.25	18.7	9.3	0.94	6610	50	656	36200	35700	50
864-4	NT	1	30	-2.9	333	330	0.3	1	1	0.00	18.6	10	0.86	21	9.3	1.17	810	50	76	35700	35700	0
865-2	NT	1	30	-2.9	340	330	1.0	1.1	1	0.01	38.2	10	2.82	21	9.3	1.17	4670	50	462	36400	35700	70
866-2	NT	4.3	30	-2.6	346	330	1.6	1.1	1	0.01	44.1	10	3.41	129	9.3	11.97	821	50	77.1	35900	35700	20
866-3	NT	2.9	30	-2.7	381	330	5.1	1	1	0.00	25.8	10	1.58	101	9.3	9.17	1790	50	174	36000	35700	30
866-4	NT	2.3	30	-2.8	361	330	3.1	1	1	0.00	30.9	10	2.09	211	9.3	20.17	566	50	51.6	35700	35700	0
866-5	NT	7	30	-2.3	355	330	2.5	1	1	0.00	13	10	0.30	295	9.3	28.57	720	50	67	36000	35700	30
866-6	NT	6.5	30	-2.4	360	330	3.0	1	1	0.00	13	10	0.30	377	9.3	36.77	1240	50	119	35800	35700	10
867-2	HL	1	30	-2.9	351	330	2.1	1.1	1	0.01	13	10	0.30	16.3	9.3	0.7	515	50	46.5	35900	35700	20
867-3	HL	1	30	-2.9	345	330	1.5	1	1	0.00	13	10	0.30	21	9.3	1.17	583	50	53.3	36200	35700	50
868-2	HL	1	30	-2.9	353	330	2.3	2.4	1	0.14	13.3	10	0.33	35.1	9.3	2.58	6590	50	654	36000	35700	30
868-3	HL	1	30	-2.9	344	330	1.4	2.8	1	0.18	18.4	10	0.84	9.3	9.3	0	562	50	51.2	35800	35700	10
868-4	HL	1	30	-2.9	341	330	1.1	3.5	1	0.25	22.2	10	1.22	14	9.3	0.47	549	50	49.9	35900	35700	20
869-2	BG	1	30	-2.9	404	330	7.4	1.4	1	0.04	13	10	0.30	17.6	9.3	0.82	1540	50	149	36000	35700	30
869-3	BG	1	30	-2.9	382	330	5.2	1.3	1	0.03	13	10	0.30	21	9.3	1.17	539	50	48.9	36800	35700	110
869-4	BG	1	30	-2.9	360	330	3.0	1.1	1	0.01	13	10	0.30	18.7	9.3	0.94	1190	50	114	37000	35700	130
870-2	BG	1	30	-2.9	364	330	3.4	1	1	0.00	13	10	0.30	23.3	9.3	1.4	451	50	40.1	36400	35700	70
870-3	BG	1	30	-2.9	360	330	3.0	1	1	0.00	13	10	0.30	25.7	9.3	1.64	505	50	45.5	36400	35700	70
870-4	BG	19.5	30	-1.1	362	330	3.2	1.6	1	0.06	13	10	0.30	18.7	9.3	0.94	2910	50	286	36600	35700	90

* Assumes an average value for previous analyses of ground-water samples from well 854.

** Negative extraction indicates uptake. The uptake is subtracted from the HCl step.

BG = Background area

EP = Evaporation ponds

HL = Heap-leach pads

NT = New tailings pile

(Shaded Cells = detection limit; actual value was lower than this value.)

Table 4–6. 5-Percent HCL Extraction

Sample ID	Area ID	Mn µg/L	* Mn Adsorbed mg/kg	Mn mg/kg	Sr µg/L	Sr mg/kg	U µg/L	U mg/kg	V µg/L	V mg/kg	NH_4 µg/L	NH_4 mg/kg	NO_3 µg/L	NO_3 mg/kg	SO_4 µg/L	SO_4 mg/kg
851-2	EP	521	-2.9	49.23	167	16.7	16.2	1.62	459	45.9	72.5	7.25	208	20.8	1500	150
851-3	EP	306	-2.9	27.72	90.3	9.03	4.7	0.47	291	29.1	52.5	5.25	477	47.7	760	76
851-4	EP	691	-2.1	66.97	155	15.5	7.9	0.79	200	20	99.9	9.99	219	21.9	1030	103
863-2	EP	468	-2.9	43.95	86.6	8.66	2.8	0.28	47.9	4.79	40.1	4.01	372	37.2	903	90.3
863-3	EP	342	-3.0	31.2	76.5	7.65	2.6	0.26	37.8	3.78	37.6	3.76	184	18.4	799	79.9
863-4	EP	463	-3.0	43.3	155	15.5	2.5	0.25	55.2	5.52	72.5	7.25	566	56.6	781	78.1
864-2	NT	442	-2.7	41.49	244	24.4	1.3	0.13	64.2	6.42	67.5	6.75	1040	104	1070	107
864-3	NT	513	-3.0	48.3	122	12.2	1.3	0.13	61.1	6.11	65	6.5	794	79.4	1010	101
864-4	NT	549	-3.0	51.9	96.2	9.62	1.2	0.12	50.2	5.02	60	6	550	55	855	85.5
865-2	NT	434	-3.0	40.4	226	22.6	1.6	0.16	109	10.9	57.5	5.75	139	13.9	774	77.4
866-2	NT	843	-2.6	81.73	886	88.6	2.8	0.28	126	12.6	152	15.2	1070	107	1580	158
866-3	NT	190	-2.7	16.29	84.9	8.49	1	0.1	53.6	5.36	90	9	855	85.5	834	83.4
866-4	NT	105	-2.8	7.73	53.7	5.37	1.4	0.14	140	14	102	10.2	1320	132	710	71
866-5	NT	109	-2.3	8.6	48	4.8	1	0.1	21.6	2.16	97.4	9.74	232	23.2	949	94.9
866-6	NT	85.6	-2.4	8.21	48.4	4.84	1.2	0.12	13	1.3	117	11.7	230	23	748	74.8
867-2	HL	566	-3.0	53.6	77.8	7.78	1.4	0.14	13	1.3	35.1	3.51	1670	167	1230	123
867-3	HL	474	-3.0	44.4	69.6	6.96	1.3	0.13	13	1.3	35.1	3.51	786	78.6	786	78.6
868-2	HL	577	-3.0	54.7	67.4	6.74	3.5	0.35	24.3	2.43	35.1	3.51	1170	117	1660	166
868-3	HL	935	-3.0	90.5	89.8	8.98	5.1	0.51	64.8	6.48	40.1	4.01	472	47.2	1820	182
868-4	HL	2000	-3.0	197	164	16.4	9	0.9	118	11.8	47.6	4.76	372	37.2	1630	163
869-2	BG	1390	-3.0	136	898	89.8	2.6	0.26	22.1	2.21	47.6	4.76	554	55.4	1450	145
869-3	BG	921	-3.0	89.1	930	93	2.1	0.21	13	1.3	50.1	5.01	172	17.2	1580	158
869-4	BG	613	-3.0	58.3	224	22.4	1.5	0.15	13	1.3	35.1	3.51	193	19.3	1260	126
870-2	BG	877	-3.0	84.7	193	19.3	1.1	0.11	13	1.3	45.1	4.51	244	24.4	1130	113
870-3	BG	686	-3.0	65.6	182	18.2	1.3	0.13	13	1.3	52.5	5.25	2790	279	1120	112
870-4	BG	682	-1.1	67.15	207	20.7	2.3	0.23	13	1.3	35.1	3.51	133	13.3	1070	107

* Mn that was adsorbed during the ground-water extraction step is assumed to leach back out in this HCl step

BG = Background area

EP = Evaporation ponds

HL = Heap-leach pads

NT = New tailings pile

(Shaded cells = detection limit; actual value was lower than this value.)

Table 4–7. Total Dissolution

Sample ID	Area ID	Mn mg/kg	Sr mg/kg	U mg/kg	V mg/kg	NH_4[a]	NO_3[a]	SO_4[a]
851–2	EP	85.6	7	0.27	15.7	NA	NA	NA
851–3	EP	94.4	4.3	0.24	14.3	NA	NA	NA
851–3	EP	119	4.9	0.22	10.9	NA	NA	NA
863–2	EP	52.1	1.8	0.13	4.4	NA	NA	NA
863–3	EP	38.7	1.6	0.13	4.3	NA	NA	NA
863–4	EP	108	5.5	0.24	7.8	NA	NA	NA
864–2	NT	62.1	3.8	0.18	5.9	NA	NA	NA
864–3	NT	61.6	3.8	0.16	5.7	NA	NA	NA
864–4	NT	73.6	3.9	0.16	5.1	NA	NA	NA
865–2	NT	45	5.3	0.27	9.7	NA	NA	NA
866–2	NT	52.7	2.1	0.15	4.9	NA	NA	NA
866–3	NT	32	2.4	0.13	3.9	NA	NA	NA
866–4	NT	32	5.8	0.18	7.6	NA	NA	NA
866–5	NT	25.7	2.5	0.12	3.4	NA	NA	NA
866–6	NT	26.2	2.6	0.14	3.7	NA	NA	NA
867–2	HL	21.5	1.1	0.1	2.4	NA	NA	NA
867–3	HL	22.1	1.2	0.09	2.5	NA	NA	NA
868–2	HL	47.2	2.4	0.11	3.3	NA	NA	NA
868–3	HL	99.2	1.7	0.13	4.1	NA	NA	NA
868–4	HL	131	4.4	0.72	13	NA	NA	NA
869–2	BG	88.5	9.8	0.38	12.7	NA	NA	NA
869–3	BG	50.6	4.4	0.24	6.1	NA	NA	NA
869–4	BG	39.8	3.8	0.18	4.8	NA	NA	NA
870–2	BG	23.5	1.7	0.12	2.8	NA	NA	NA
870–3	BG	20.4	2.1	0.13	2.8	NA	NA	NA
870–4	BG	17.1	3.1	0.17	3.3	NA	NA	NA

[a] Total digestions were not performed for NO_3, NH_4, and SO_4 due to probability of volatilization.

BG = Background area
EP = Evaporation ponds
HL = Heap-leach pads
NT = New tailings pile

After subtracting out the concentration already in the ground water, the concentration of Sr extracted from the residuum can be calculated:

$$\frac{200\ \text{mL}}{2\ \text{g}} \times \frac{(332-330)\ \mu\text{g}}{\text{L}} \times \frac{\text{L}}{1{,}000\ \text{mL}} \times \frac{1{,}000\ \text{g}}{\text{kg}} \times \frac{\text{mg}}{1{,}000\ \mu\text{g}} = 0.2\ \text{mg/kg}$$

(Column 8, Table 4–5)

For the HCl extraction:

$$\frac{200\ \text{mL}}{2\ \text{g}} \times \frac{167\ \mu\text{g}}{\text{L}} \times \frac{\text{L}}{1{,}000\ \text{mL}} \times \frac{1{,}000\ \text{g}}{\text{kg}} \times \frac{\text{mg}}{1{,}000\ \mu\text{g}} = 16.7\ \text{mg/kg}$$

(Column 7, Table 4–6)

The results of the total digestion (based on 2 grams of sample) are listed on Table 4–7.

Manganese concentrations in the extract decreased when ground water was used as the extractant (columns 3 through 5, Table 4–5). In this case, Mn was transferred to the sediment, which resulted in a negative value for the amount extracted. It was assumed that this Mn adsorbed to the soil and was then desorbed by the subsequent 5-percent HCl extractant. Thus, this amount was subtracted from the mass extracted by HCl (Column 4, Table 4–6), and the values reflect only the amount of Mn present in the original sediment.

On Table 4–8, the concentrations from all four extractions are summed, resulting in the total amount of each constituent that was present in the original sample. Average concentrations of the selected site-related constituents occurring naturally in the earth's crust are provided for reference.

4.6 Hydrologic and Soil Tests

Estimates of the aquifer parameters for both the alluvial and bedrock systems are required to develop a design for a pump-and-treat remedial action and to better understand the hydrogeologic characteristics of the site that could influence migration of contaminants in the ground water. Surface soil permeability, hydraulic conductivity, storage, and specific yield were measured during the field investigation. Results of the measurements are provided in the following sections.

4.6.1 Surface Soil Permeability Tests

Surface permeability tests were conducted on surface and near-surface soils to estimate recharge to the alluvial aquifer through precipitation and to evaluate technologies that rely on land application methods. Details of the test locations, procedures, and data analyses are provided in Appendix B. A map showing the location where each test was performed is also provided in Appendix B.

Table 4–8. Total Amount Extracted

Sample ID	Area	Mn mg/kg	Sr mg/kg	U mg/kg	V mg/kg	NH_4 mg/kg	NO_3 mg/kg	SO_4 mg/kg
851–2	EP	134.93	32.81	3.48	202.4	27.24	1775.8	7270
851–3	EP	122.89	19.38	1.11	142.1	10.81	942.7	2866
851–4	EP	188.28	24.92	1.57	89.11	30.6	252.9	375
863–2	EP	96.19	13.65	0.55	20.09	9.58	637.2	1690.3
863–3	EP	70	12.22	0.55	14.93	8.86	456.5	1389.9
863–4	EP	151.4	23.98	0.62	21.29	13.74	632.9	1768.1
864–2	NT	103.69	39.1	0.41	23.09	13.73	273.7	9337
864–3	NT	110	20.2	0.39	22.01	10.21	1407.4	1571
864–4	NT	125.87	15.29	0.38	22.58	9.71	351	431.5
865–2	NT	85.5	33.92	0.54	47.82	9.93	881.9	3307.4
866–2	NT	134.53	98.36	0.54	56.21	137.17	1157.1	2028
866–3	NT	48.46	20.57	0.33	21.44	154.17	1095.5	1913.4
866–4	NT	40.36	16.75	0.42	33.57	214.37	882.6	471
866–5	NT	34.85	11.15	0.32	11.29	270.31	914.2	396.9
866–6	NT	32.69	11.61	0.36	6.85	310.47	956	308.8
867–2	HL	75.2	12.23	0.35	5.3	7.91	275.1	168.5
867–3	HL	66.68	10.91	0.32	5.4	8.15	207.4	165.7
868–2	HL	102	12.45	0.7	11.13	10.95	1612	243.1
868–3	HL	189.8	13	0.92	18.22	6.09	153.7	267.2
868–4	HL	328.1	23.49	1.97	34.47	7.77	161.2	530
869–2	BG	224.6	111.65	0.78	16.51	7.33	231.2	562
869–3	BG	139.8	106.46	0.58	9	7.79	114.2	611
869–4	BG	98.2	31.91	0.44	7.7	8.15	941.3	499
870–2	BG	108.38	27.35	0.33	5.7	9.38	90	311
870–3	BG	86.61	25.78	0.36	5.7	9.2	362.9	295
870–4	BG	84.84	29.38	0.56	6.2	6.99	372.2	291.9
Crustal Average[a]		950	375	1.8	135	26[b]	89[c]	780[d]

[a] From Mason and Moore 1982.
[b] Crustal average composition of nitrogen cast as NH_4.
[c] Crustal average composition of nitrogen cast as NO_3.
[d] Crustal average composition of sulfur cast as SO_4.

EP = Evaporation ponds
NT = New tailings pile
HL = Heap-leach pads
BG = Background area

4.6.1.1 Surface Soil Permeability Test Results

Hydraulic conductivities (ft/year) calculated from both the E–19 nomographs and the Glover equation are presented in Table 4–9. In most cases, the values are within 10 percent of each other.

Table 4–9. Permeability Test Results

Test No.	Area No.	Nomograph K_s (ft/year)	Glover Eqn K_s (ft/year)
MON–301	2	500.00	392.13
MON–302	2	575.00	427.31
MON–303	1	150.00	127.26

Test No.	Area No.	Nomograph K_s (ft/year)	Glover Eqn K_s (ft/year)
MON–304	1	70.00	72.11
MON–305	1	9.00	8.02
MON–306	1	650.00	677.69
MON–307	3	115.00	126.23
MON–308	3	195.00	199.69
MON–308 DUP	3	160.00	157.27
MON–309	3	150.00	155.20
MON–310	4	65.00	40.56
MON–312	4	165.00	137.61
MON–313	5	160.00	169.68

The test results indicate:

- Saturated conductivity values across the site varied by almost two orders of magnitude. Testing locations MON–305 and MON–306 (both in Area 1) contained the lowest and highest conductivities, respectively.

- The duplicate tests were performed in different boreholes approximately 10 ft apart at location MON–308. As the results indicate, the values are within 20 percent of each other.

- Table 4–10 includes the geometric mean of the surface soil conductivity for the site and for individual areas. The geometric mean for the entire site was 145.8 ft/year based on the nomograph and 131.8 ft/year based on the Glover solution. Area 4 has the lowest permeability rates (geometric mean of 68.92 ft/year), and Area 2 has the highest (geometric mean of 404.97 ft/year). As expected, Area 4 conductivities appear to have been affected by the compacted silty sand layer (probable hard pan layer) approximately 2 ft below the ground surface.

Table 4–10. Summary of Permeability Test Area

Area	No. of Tests	Test Result	Nomograph K_s (ft/year)	Glover Eqn K_s (ft/year)
Site	13[a]	geomean	145.81	131.80
Area 1	4	geomean	88.53	84.04
Area 2	2	geomean	536.19	409.34
Area 3	4[a]	geomean	144.97	151.42
Area 4	2	geomean	103.56	74.71
Area 5	1	NA[b]	160.00	169.68

[a] Includes one duplicate
[b] Not Applicable, only one test was conducted in Area 5.

- The results do not appear to be affected by the equipment used for the test. This was determined by ranking the conductivities in descending order and noting the infiltrometer used to collect the data at that location. There did not appear to be a trend (i.e., there was no evidence that higher or lower conductivities were associated with one of the infiltrometers).

4.6.2 Soil Particle-Size Distribution

Soil particle-size distribution, or soil texture, was characterized as part of a feasibility study of the surface application alternative for ground water remediation. Specifically, the classification of irrigation suitability of regraded areas and rangelands at the Monument Valley site required soil texture data. Soil texture greatly influences the movement and storage of soil water (Hillel 1980).

4.6.2.1 Particle-Size Analysis Procedure

Composite soil samples were collected from a subset of the boreholes that were excavated for the surface soil infiltration tests (Figure B–1): four samples from Area 1 (MON–303 through MON–306), two samples from Area 2 (MON–301 and MON–302), and one sample each from Area 3 (MON–307) and Area 4 (MON–310). Composites consisted of evenly mixed samples taken incrementally from a soil profile to the bottom of the borehole, approximately 4 ft deep.

Soil particle-size fractions were determined using mechanical grain-size analysis (ASTM D–2487) followed by hydrometer analysis of fines (Gee and Bauder 1986). The sand fractions were separated using ASTM sieve sizes 10, 20, 100, and 200. Silt and clay fractions were determined using a 152H hydrometer with slurry temperatures controlled in a water bath. Soil aliquots weighing between 4 and 60 grams were mixed with a 4 percent, 125-mL sodium hexametaphosphate dispersing solution using a blender.

4.6.2.2 Soil Texture Results

Soil texture results are summarized in Table 4–11. In most sampling locations, soil profiles consisted of uniform, reddish-brown coppice dune sand with over 80 percent fine sand. According to the U.S. Department of Agriculture (USDA) (Soil Survey Staff 1975) system, these soils are classified as sand. Two exceptions were MON–301 and MON–310. MON–301, a subsoil in the pond area (Area 2) contained 23 percent silt and is classified as a loamy sand.

MON–310 was sampled in the Cane Wash area (Area 4). The upper 2 ft of the profile consisted of reddish-brown sand. A light grey compacted sand was observed from approximately 2 ft below the surface to the bottom of the borehole.

4.6.3 Aquifer Tests

Aquifer tests were conducted to determine the hydraulic parameters of the alluvial aquifer. An aquifer test was also completed in the De Chelly aquifer to define the hydraulic parameters and determine if the alluvial aquifer is hydraulically connected to the De Chelly near the paleochannel in the southwest portion of the site. Details of the test procedures and data analyses are provided in Appendix B.

Table 4–11. Monument Valley Soil Particle-Size and Texture Classification

Sample Location	Soil Texture[a] and Classification[b]						
	coarse sand (%)	medium sand (%)	fine sand (%)	silt (%)	clay (%)	USDA	ASTM
Area 1: Soil Borrow							
MON–303	1	2	95	2	0	sand	SP
MON–304	1	6	92	1	0	sand	SP
MON–305	2	3	92	3	0	sand	SP
MON–306	2	1	97	0	0	sand	SP
Area 2: Tailings and Ponds							
MON–301	1	2	72	23	2	loamy sand	SM
MON–302	0	1	87	11	1	sand	SP
Area 3: Undisturbed North							
MON–307	0	0	100	0	0	sand	SP
Area 4: Cane Wash							
MON–310	0	2	96	2	0	sand	SP

[a]Mechanical grain-size analysis using ASTM sieve sizes 10, 20, 100, and 200, followed by hydrometer analysis of fines (ASTM D–2487, SSSA 1986).
[b] Soils are classified by both USDA (1975) and Unified (ASTM D–2487) systems. Within the Unified system, SM = silty sands, poorly graded sand-silt mixtures, and SP = poorly graded sands, gravely sands, little or no fines.

4.6.3.1 Previous Investigations

1985 Investigation

Slug tests were conducted in eight wells screened in the alluvial aquifer, four wells screened in the Shinarump aquifer, and seven wells screened in the De Chelly aquifer during 1985. Table 4–12 lists the methods used to analyze the slug test data and presents results from those tests.

In addition to the slug tests, one aquifer test was performed at well MON–651, which is screened in the alluvium. For this aquifer test, a well point was installed to collect water-level data 21 ft from pumping well MON–651. MON–651 is in Cane Wash (predominantly fluvial deposits) and is not considered representative of the alluvial aquifer near the nitrate plume (predominantly eolian deposits).

A flow rate of 13.6 gallons per minute (gpm) was sustained over an 11-hour time period during the test. Water level in the observation well actually increased during the test and provided inconclusive results after data analysis. Hydraulic conductivities were estimated using drawdown and recovery data from the pumping well. Table 4–12 contains results of data analysis from this aquifer test. Data collected from pumping wells during aquifer tests may not provide representative estimates of hydraulic parameters of an aquifer, because a number of the assumptions that are associated with the analytical methods are not met.

1992 Investigation

Notes in the technical notebook reference indicate aquifer tests in 1992 were conducted using wells MON–619 and MON–668 as pumping wells, both of which are screened in the De Chelly aquifer. Table 4–13 lists the observation wells, screened elevations, and distances to the pumping wells for both tests.

Table 4–12. Summary of Hydraulic Conductivity from 1985 Aquifer and Slug Tests

Formation	Well I.D.	Bouwer–Rice Method	Hvorslev Method	Ferris–Knowles Method	CBP Method	Arithmetic Mean K	Geometric Mean K
AL	MON–602	4.3×10^{-6}	3.3×10^{-6}	NA	NA	3.8×10^{-6}	NA
AL	MON–604	1.7×10^{-5}	NA	NA	NA	1.7×10^{-5}	NA
AL	MON–605	3.6×10^{-5}	4.6×10^{-5}	NA	NA	4.1×10^{-5}	NA
AL	MON–651	NA	NA	NA	NA	2.2×10^{-4}[a]	NA
AL	MON–653	3.3×10^{-5}	4.5×10^{-5}	NA	NA	3.9×10^{-5}	NA
AL	MON–655	3.9×10^{-6}	4.6×10^{-6}	NA	NA	4.3×10^{-6}	NA
AL	MON–656	2.8×10^{-5}	3.2×10^{-5}	NA	NA	3.0×10^{-5}	NA
AL	MON–662	2.0×10^{-5}	2.9×10^{-5}	NA	NA	2.5×10^{-5}	NA
AL	MON–669	5.4×10^{-6}	9.5×10^{-6}	NA	NA	7.5×10^{-6}	NA
DC	MON–610	2.2×10^{-6}	2.7×10^{-6}	9.5×10^{-7}	3.1×10^{-6}	2.2×10^{-6}	2.0×10^{-6}
DC	MON–612	NA	NA	1.8×10^{-6}	2.9×10^{-6}	2.4×10^{-6}	NA
DC	MON–657	NA	NA	1.1×10^{-6}	4.1×10^{-6}	2.6×10^{-6}	NA
DC	MON–661	NA	NA	NA	2.1×10^{-6}	2.1×10^{-6}	NA
DC	MON–663	NA	NA	2.1×10^{-7}	3.2×10^{-7}	2.7×10^{-7}	NA
DC	MON–667	NA	NA	3.2×10^{-5}	6.0×10^{-6}	1.9×10^{-5}	1.4×10^{-5}
DC	MON–668	NA	NA	4.4×10^{-7}	1.7×10^{-6}	1.1×10^{-6}	8.6×10^{-7}
SR	MON–601	5.5×10^{-6}	NA	NA	NA	5.5×10^{-6}	NA
SR	MON–658	1.4×10^{-5}	NA	4.9×10^{-6}	2.3×10^{-5}	1.4×10^{-5}	1.2×10^{-5}
SR	MON–659	1.2×10^{-5}	NA	4.5×10^{-6}	2.1×10^{-5}	1.3×10^{-5}	1.0×10^{-5}
SR	MON–660	9.4×10^{-5}	NA	7.2×10^{-5}	NA	8.3×10^{-6}	NA

[a]K calculated using the Chow and Theis Recovery method based on a single aquifer test in well MON–651.

AL = alluvium
CBP = Cooper-Bredehoeft-Papadopulos method
DC = De Chelly Member of the Cutler Formation
K = Hydraulic conductivity (ft/s)
SR = Shinarump Member of the Chinle Formation

Source: Monument Valley RAP, Appendix F, 1993b.

Table 4–13. Aquifer Test Well Construction Specifics for the 1992 Field Investigation

Well No.	Well Type	Aquifer	Top of Screen Elevation (ft above MSL)	Bottom of Screen Elevation (ft above MSL)	Approx. Dist. from Pumping Well (ft)
MON–619	Pumping	De Chelly	NA	NA	NA
MON–657	Observation	De Chelly	4,762.21	4,747.21	40
MON–662	Observation	Alluvial	4,842.31	4,812.31	40
MON–668	Pumping	De Chelly	4,686.71	4,666.71	NA
MON–669	Observation	Alluvial	4,831.78	4,811.78	10
MON–655	Observation	Alluvial	4,822.39	4,802.39	550
MON–663	Observation	De Chelly	4,689.41	4,649.41	1,150

MSL = Mean Sea Level.
NA - Not applicable, well not screened at time of 1992 test.

The well MON–619 test included a step test run at pumping rates of 9 gpm and 14 to 16 gpm. A graph indicates the pumping well water level was lowered only 0.2 and 0.5 ft, respectively, at these pumping rates. During the actual aquifer test, which was run at a pumping rate of 39 gpm, questionable drawdown was monitored in observation wells MON–657 and MON–662. The water level initially increased less than 0.5 ft once the pump was started, returned to the static water level, and was followed by a fluctuation in the water level that resulted in less than 0.2 and 0.3 ft of drawdown during the test period in wells MON–657 and MON–662, respectively.

Questionable drawdown data were also collected from wells MON–669, –655, and –663 during the aquifer test in which well MON–668 was pumped at a rate of 15 gpm. The water levels in wells MON–663 and MON–669 both increased (0.8 and 0.06 ft above the static water level, respectively) during the initial pumping period and, as a result, no drawdown was detected. At the third observation well, MON–655, there was no response to pumping (i.e., the water level did not fluctuate from the static level).

The lack of valid drawdown data from observation wells during these two tests did not allow for the calculation of representative aquifer parameters. The site technical notebook did not contain any calculations using the data from well MON–619 and well MON–668 tests. Ground water seepage velocity calculations presented in Appendix F of the RAP (DOE 1993b) used data generated from the 1985 tests only.

Table 4–14 provides the seepage velocity results for the alluvial, Shinarump, and De Chelly aquifers. The velocities are based on hydraulic conductivity ranges from the 1985 test data only. Effective porosity and hydraulic gradient estimates were based on site conditions during the 1985 field effort, as presented in Appendix F of the RAP.

Table 4–14. Ground Water Seepage Velocity Estimations Based on the 1985 Field Investigation

Aquifer	Estimated Effective Porosity	Hydraulic Gradient	Hydraulic Conductivity Range (ft/day)	Seepage Velocity Range (ft/day)
Alluvial	0.25	0.011	0.28 to 19	0.01 to 0.84
Shinarump	0.10	0.010	0.39 to 8.1	0.04 to 0.80
De Chelly	0.10	0.011	0.018 to 2.8	0.002 to 0.3

Source: Monument Valley RAP, Appendix F, 1993b.

4.6.3.2 1997 Investigation Results

655 Alluvial Test

The initial data at well MON–655 indicated that a pumping rate over 0.6 gpm could not be sustained for an extended (greater than 24 hours) period of time. The first aquifer test lasted 70.7 hours and resulted in 12 ft of drawdown in the pumping well and only 0.1 ft of drawdown in observation well MON–769 and no response to pumping was measured in observation well MON–771 (both wells are approximately 50 ft from the pumping well).

Analysis of the data from observation well MON–769 provided inconclusive results. Recovery data were analyzed to estimate the hydraulic conductivity of the alluvial aquifer in the vicinity of well 655 (Table 4–15). Analysis of the step test data resulted in a pumping well specific capacity of 0.009 square feet per minute (ft^2/min).

Fine-grained sand was observed in the water discharging from the pump during the first test. A screen slot size of 0.051 in. had been selected according to the information provided in the completion record for this well. However, the aquifer material at this location consists of a fine-grained sand in which the grain diameters range from 0.002 to 0.01 in. Consequently, there is the potential for the aquifer material to pass through the well screen and reduce well efficiency. The well was redeveloped in an attempt to increase the efficiency and the flow potential.

Additional development did not increase the efficiency of the well, and a flow greater than 0.6 gpm was not sustainable for an extended time. A second test was completed to compare the hydraulic conductivity from the initial test. A slug test was also completed at this well to compare to the hydraulic conductivity estimated from the aquifer tests and the slug test performed on the same well in 1985. Since the sustainable flow rate could not be increased, it is expected that the specific capacity calculated from the first step test (0.009 ft^2/min) would not change significantly.

During the second aquifer test, approximately 0.25 ft of drawdown was measured in observation well MON–771, with insignificant drawdown detected in well MON–769 (which is opposite of the response in the first test). In the pumping well, there was approximately 18 ft of drawdown, which suggests the development attempt may have actually decreased the well efficiency. Table 4–16 provides results from the analyses of data collected during the December 1997 aquifer tests and slug tests. Analysis of data collected from the observation well during the aquifer test indicate that the hydraulic conductivity ranges from 1.7 to 3.2 ft/day (geometric mean of 2.4 ft/day); the analysis of data collected during the recovery of the pumping well suggested a value of 0.06 ft/day. Analysis of the data provided a specific yield estimate of 0.001.

Analyses of the slug test data indicated that hydraulic conductivity ranged from 0.20 to 0.32 ft/day. The 1985 data analyses suggested a hydraulic conductivity ranging from 0.34 to 0.40 ft/day for well MON–655.

The seepage velocity, which represents the rate at which water actually moves through the aquifer pore spaces, can be calculated using the following formula:

$$V_S = \frac{K}{n_e} \times \frac{dh}{dl}$$

where v_s is the seepage velocity (ft/day), K is the hydraulic conductivity (ft/day), n_e is the effective porosity (dimensionless), and dh/dl is the horizontal hydraulic gradient (dimensionless). If the hydraulic conductivity ranges from 1.7 to 3.2 ft/day (using aquifer test results), the seepage velocity for the alluvial aquifer in the vicinity of well MON–655 ranges from 0.075 to 0.141 ft/day. This value is based on an estimated effective porosity of the alluvial aquifer of 0.25 (DOE 1993b) and average horizontal hydraulic gradient of 0.011 (September 1997 data).

Table 4–15. Summary of Pump Test Analyses 9/97—Alluvial Aquifer Monument Valley Field Investigation

Test Location	Well Type	AQ	Theis Analysis		Cooper/Jacob		Neuman Semi-Log			Theis/Jacob Rec	
			T (ft^2/min)	K (ft/day)	T (ft^2/min)	K (ft/day)	T (ft^2/min)	K (ft/day)	Sy	T (ft^2/min)	K (ft/day)
Well 655	PUMP	Alluv	NA	NA	NA	NA	NA	NA	NA	3.17E–04	0.011
	PP										
Well 765	PUMP	Alluv	NA	NA	NA	NA	NA	NA	NA	9.70E–04	0.025
	PP										
Well 777	OBS	Alluv	0.461	11.9	0.302	7.8	0.322	8.3	0.71	INC	INC
	PP										
Well 766	OBS	Alluv	0.411	10.6	0.486	12.5	0.581	14.9	0.057	INC	INC
	FP										

Hydraulic Conductivity Geometric Mean Summary

Well 777	Well 766	Well 765 Cluster*
K (ft/day)	K (ft/day)	K (ft/day)
9.2	12.5	10.7

AQ = The aquifer the well is screened in
FP = Well fully penetrating
K = Hydraulic conductivity
INC = Data provided inconclusive results
NA = Not applicable
OBS = Observation Well
PP = Well partially penetrating
PUMP = Pumping well
Sy = Specific yield
T = Transmissivity
* = Well 765 recovery data not included

Table 4–16. Summary of Pump Test Analysis 12/97—Alluvial Aquifer Monument Valley Field Investigation Slug Tests

Test Location	Well Type	AQ	Flow Rate	Theis Analysis		Cooper/Jacob		Neuman Semi-Log			Theis/Jacob Rec		Slug Tests	
													Bower/Rice	Hvorslev
				T (ft^2/min)	K (ft/day)	T (ft^2/min)	K (ft/day)	T (ft^2/min)	K (ft/day)	Sy	T (ft^2/min)	K (ft/day)	K (ft/day)	K (ft/day)
Well 655 Cluster														
Well 771	OBS PP	Alluv	0.55 GPM	0.093	3.2	0.075	2.6	0.051	1.7	0.001	INC	INC	NA	NA
Well 655	PUMP PP	Alluv	0.55 GPM	NA	NA	NA	NA	NA	NA	NA	0.002	0.060	0.32	0.20
Well 765 Cluster														
Well 777	OBS PP	Alluv	3 GPM	0.450	11.6	0.445	11.4	0.482	12.4	0.41	INC	INC	NA	NA
Well 777	OBS PP	Alluv	3.25 GPM	0.868	22.2	0.752	19.3	1.062	27.3	0.37	0.681	17.4	NA	NA
Well 766	OBS PP	Alluv	3 GPM	1.130	28.9	0.717	18.3	0.794	20.4	0.015	INC	INC	NA	NA
Well 766	OBS PP	Alluv	3.25 GPM	1.370	35.3	1.320	34.0	0.910	23.4	0.017	0.993	25.5	NA	NA
Well 765	PUMP PP	Alluv	3.25 GPM	NA	NA	NA	NA	NA	NA	NA	0.003	0.077	NA	NA

Hydraulic Conductivity Geometric Mean Summary

Flow Rate	Well 771 AQ Test K (ft/day)	Well 655 Slug Test K (ft/day)	Well 777 AQ Test K (ft/day)	Well 766 AQ Test K (ft/day)	Well 765 Rec Test K (ft/day)	Well 765 Cluster* K (ft/day)
0.55 GPM	2.4	0.25	NA	NA	NA	NA
3.0 GPM	NA	NA	11.8	22.1	NA	16.1
3.25 GPM	NA	NA	21.2	29.1	0.077	24.9
3 & 3.25 GPM	NA	NA	16.5	25.9	0.077	21.5

AQ = The aquifer the well is screened in
INC = Result inconclusive
K = Hydraulic conductivity
NA = Not applicable
OBS = Observation well
PP = Well partially penetrating well
PUMP = Pumping well
Sy = Specific yield
T = Transmissivity
* = Values do not include pumping well recovery data or slug test data

765 Alluvial Test

Results of short-term tests indicated that a flow of 1.1 gpm could be sustained from well MON–765 over an extended period of time. As a result, this flow rate was used during the first aquifer test (which lasted 60.6 hours) at the MON–765 location. In response to this flow rate, approximately 0.1 ft of drawdown was observed in the observation well (MON–777) located 15 ft from the pumping well, which had a drawdown of more than 35 ft. This suggests a very steep cone of depression was created during the testing period. Less than 0.1 ft of drawdown was noted in the observation well 52.5 ft away (well MON–766).

Results of the analysis of data collected from the aquifer tests completed in September 1997 are presented in Table 4–15. Based on the analysis of data collected from the observation wells, the hydraulic conductivity ranged from 7.8 to 14.9 ft/day (geometric mean of 10.7 ft/day). A hydraulic conductivity of 0.025 ft/day was calculated from the recovery data collected from the pumping well. A specific capacity of 0.004 ft^2/min was calculated from analysis of the data collected during the step test.

Well MON–765 was further developed in an attempt to increase its efficiency and corresponding flow rate. This well was installed with 0.010 in. screen slot, which is better suited for the particle size in the subsurface material compared to the well MON–655 design. A step test conducted after development suggested a flow rate of 3.0 gpm could be sustained for an extended period of time. The specific capacity increased to 0.0155 ft^2/min after well development.

The 3 gpm test lasted 19.9 hours. During that time, a drawdown of approximately 38 ft was observed in the pumping well, and approximately 0.2 ft of drawdown was measured in observation wells MON–777 and MON–766. Time allowed for another short-term (less than 24 hours) test, this time using a flow rate of 3.25 gpm. This 20.8-hour test resulted in approximately 43 ft of drawdown in the pumping well, and again approximately 0.2 ft of drawdown was measured in both observation wells.

Table 4–16 shows the results of the analysis of data collected during the December 1997 aquifer tests. Analyses of the data collected from the observation wells during the 3 gpm test indicate the hydraulic conductivity ranges from 11.4 to 28.9 ft/day (geometric mean of 16.1 ft/day); the analyses of data collected during the 3.25 gpm indicate a range of 19.3 to 35.3 ft/day (geometric mean of 24.9 ft/day). Data collected from the pumping well during the recovery phase of the 3.25 gpm test suggest a hydraulic conductivity of 0.077 ft/day.

When a hydraulic conductivity range of 11.4 to 35.3 ft/day is used, the seepage velocity for the alluvial aquifer in the vicinity of well MON–765 ranges from 0.50 to 1.55 ft/day. These calculations are based on an effective porosity of 0.25 and a horizontal hydraulic gradient of 0.011.

De Chelly Test

As previously mentioned, hydraulic conditions in the De Chelly aquifer near the site are believed to range from confined to semiconfined, depending on the location. Based on the step test

completed at well MON–619, a pumping rate of 70 gpm could be maintained during a multiday aquifer test. Step-test data indicated a specific capacity of 0.79 ft^2/min.

During a 90-hour test, approximately 12 ft of drawdown was measured in the pumping well. In observation wells MON–776, –774, and –668, approximately 8 ft, 5 ft, and 3 ft of drawdown were measured, respectively. The fact that 3 ft of drawdown was measured 1,450 ft from the pumping well in another well screened in the De Chelly suggests the aquifer is confined outside of the paleochannel.

Another significant finding during this test was the rapid response to pumping in well MON–774 (located in the paleochannel and screened in the alluvial aquifer) water levels. This response verifies the hydraulic connection between the alluvial aquifer and the underlying De Chelly within the boundaries of the paleochannel.

During analysis of the data from observation well MON–776, a break in the slope of the drawdown versus time (log) data suggests the effect of a hydrologic boundary. A similar trend was noted in drawdown data from the pumping well and observation well MON–774. This break in the slope of the data may represent the time when the cone of depression reached the far (western) boundary of the paleochannel. Data from observation well MON–668 showed only a slight break in the slope.

Table 4–17 shows the hydraulic conductivity values that resulted from analysis of data collected during the De Chelly test. Analysis of data collected from the two observation wells screened in the De Chelly resulted in a hydraulic conductivity ranging from 0.6 to 4.3 ft/day. Data collected during the recovery test suggested a hydraulic conductivity ranging from 1.2 to 2 ft/day. Taking into account all the results, the hydraulic conductivity geometric mean for the De Chelly aquifer near well MON–619 is 1.6 ft/day. Storativity estimates ranged from 8.3×10^{-5} to 4.7×10^{-4}.

A hydraulic conductivity range of 0.6 to 4.3 ft/day, an assumed effective porosity of 0.15 (DOE 1993b), and a measured horizontal hydraulic gradient of 0.014 result in a seepage velocity that ranges from 0.06 to 0.4 ft/day. This range is comparable to the range determined by the 1985 slug test data (0.002 to 0.3 ft/day).

Summary of the Alluvial Aquifer Tests

- Tests were initially completed on the alluvial aquifer in September 1997. Analysis of the step test data resulted in very low well efficiencies for the two pumping wells. After additional well development, aquifer tests were re-run in December 1997 at those two locations.

- Data collected during the December 1997 aquifer tests suggest that the hydraulic conductivity ranges from 1.7 to 3.2 ft/day in the vicinity of well MON–655. Based on this hydraulic conductivity range, the seepage velocity ranges from 0.075 to 0.141 ft/day.

Table 4–17. Summary of Aquifer Test Analyses – De Chelly Aquifer Monument Valley Field Investigation

Test Location	Well Type	AQ	Theis Analysis			Jacob/Cooper			Hantush			Theis Jacob Rec	
			T (ft^2/min)	K (ft/day)	S	T (ft^2/min)	K (ft/day)	S	T (ft^2/min)	K (ft/day)	S	T (ft^2/min)	K (ft/day)
Well 619	PUMP	DeC	NA	NA	NA	NA	NA	NA	NA	NA	NA	0.69	2.00
	PP												
Well 776	OBS	DeC	***1.05***	***3.00***	***4.72E–04***	1.51	4.30	8.36E–05	0.33	1.00	227E–04	0.70	2.00
	PP												
Well 668	OBS	DeC	0.37	1.10	4.20E–04	0.63	1.80	3.22E–04	0.21	0.60	3.17E–04	0.40	1.20
	PP												
Well 774	OBS	Alluv	***2.35***	***58.50***	NA	0.69	17.10	NA	NA	NA	NA	0.83	14.80
	FP												

Hydraulic Conductivity Geometric Mean Summary

Well 619	Well 776	Well 668	Well 774*	Well 619 Cluster**
K (ft/day)	K (ft/day)	K (ft/day)	K (ft/day)	K (ft/day)
2.00	2/25	1.09	15.91	1.61

Values in bold italics generated from data in which there was a questionable fit
AQ = The aquifer the well is screened in
All graphs contained in Appendix B
FP = Well fully penetrating
INC = Result inconclusive
K = Hydraulic conductivity
NA = Not applicable
OBS = Observation well
PP = Well partially penetrating
PUMP = Pumping well
S = Storativity
T = Transmissivity

* = Well 774 values did not include Theis data
** = Data did not include alluvial (774) results

DeChelly Aquifer assumed to be 500 ft thick in the vicinity of well 619
Wells 619, 776, and 668 corrected for partial penetration, well approx. 150 ft deep
Well 774 completed in alluvium, pumping well completed in De Chelly (directly underlying alluvium at this location)

- In the vicinity of well MON–765, data indicate the hydraulic conductivity ranges from 11.4 to 35.3 ft/day (geometric mean of 21.5 ft/day). Based on the analyses of data collected during the December 1997 tests, the seepage velocity ranges from 0.50 to 1.55 ft/day.

- The specific yield, based on December 1997 test data, ranges from 0.015 to 0.41.

- The specific capacities of wells MON–655 and MON–765 are 0.009 and 0.016 ft^2/min, respectively. The specific capacity of well MON–655 appears to have been influenced by improper well construction (i.e., slot openings too large), which may be responsible for its low efficiency. As a result, hydraulic parameter estimates based on data collected at this location may not be representative.

Summary of the De Chelly Aquifer Tests

- Analysis of data from the well MON–619 test resulted in a hydraulic conductivity range of 0.6 to 4.3 ft/day (geometric mean of 1.6 ft/day). Using this conductivity range, the seepage velocity ranges from 0.06 to 0.4 ft/day.

- Analyses of data suggest the storativity ranges from 8.3×10^{-5} to 4.7×10^{-4}. According to the step test data, the specific capacity of well MON–619 is 0.79 ft^2/min.

- During the De Chelly test, drawdown was noted in the observation well located in the paleochannel and screened in the alluvial aquifer. This response indicates a direct hydrologic connection between the alluvial and De Chelly aquifers in this region of the site.

- The De Chelly aquifer appears to be unconfined to semiconfined in the vicinity of the paleochannel and mostly confined in other regions of the site.

4.7 Plant Ecology Investigation

Plant ecology plays an important role in surface and ground water remediation at the Monument Valley site. Successful revegetation of the millsite and tailings areas can control soil loss and improve the value of the land resource (Munshower 1996). By applying a technique called phytoremediation, plants may be used to extract and treat ground water and soil contaminants such as ammonium and nitrate for a fraction of the cost of traditional pump-and-treat techniques (Kim and Ondrey 1996; Kim 1996). Because of high evapotranspiration/precipitation (ET/P) ratios in desert ecosystems, revegetation can also prevent leaching of soil contaminants and thus help contain ground water contamination sources (Weand and Hauser 1997). By pumping nitrate-contaminated ground water for irrigation of revegetation areas, the land application alternative (Baumgartner et al. 1996) may accelerate plant establishment, plant productivity, evapotranspiration (ET), and nitrogen extraction. Conversely, plants that root into the plume or are irrigated with plume water are potential exposure pathways for humans and ecological receptors.

The plant ecology of the former millsite, tailings area, and surrounding areas was characterized to address the following issues:

- Human health and ecological risks associated with site-related contaminated ground water.
- Soil water balance effects on recharge and discharge components of the hydrological system.
- Feasibility of the phytoremediation alternative.
- Feasibility of the land application alternative for ground water remediation.

The plant ecology investigation consisted of

- A plant species survey.
- Estimates of the percent cover and age structure of phreatophyte populations.
- Evaluations of the composition, relative abundance, and distribution of plant associations.
- Vegetation mapping.

4.7.1 Plant Species Survey

The former millsite, tailings area, pond area, and the area delineated by the extent of the nitrate and sulfate plumes (Section 5.3) were traversed on June 24, 1997, to identify plant species. The results of the plant species survey (Table 4–18) became the foundation for the plant ecology investigation; all succeeding ecological characterization and applications build on interpretations of the species composition and associations. The occurrence and relative abundance of certain plant species provide a measure of the health of the ecosystem. Knowing the species and their physiological and ecological tolerances provides evidence of environmental conditions that are of importance for understanding the site hydrology, potential human health and ecological risks, and the feasibility of phytoremediation and land application alternatives.

4.7.2 Phreatophyte Cover and Age Structure

Phreatophytes (literally “well plants”) at the Monument Valley site may act as natural pump-and-treat systems for ground water nitrates. Two phreatophyte populations grow over the plume area: black greasewood and fourwing saltbush. Black greasewood (*Sarcobatus vermiculatus*) is an obligate phreatophyte; it requires a permanent ground water supply. Black greasewood can transpire water from aquifers as deep as 18 m below the land surface (Nichols 1993). Fourwing saltbush (*Atriplex canescens*) is a facultative phreatophyte; it takes advantage of ground water when present but can tolerate periods of low water availability. The rooting depth of fourwing saltbush may exceed 8 m (Foxx et al. 1984). This section describes methods and results of sampling to determine the percent cover and age of black greasewood populations potentially growing into the nitrate plume. Cover estimates for fourwing saltbush are presented in Section 4.7.3.

Table 4–18. Plants Growing on the Reclaimed Tailings and Plume Areas at the Monument Valley Site

Scientific Name[a]	Acronym[b]	Common Names[c]
Shrubs		
Artemisia filifolia Torr.	ARFI	sand sagebrush, old-man sagebrush
Atriplex canescens (Pursh) Nutt.	ATCA	fourwing saltbush, cenizo, chamizo
Atriplex confertifolia (Torr. & Frem.) Wats.	ATCO	shadscale, spiny saltbush, sheep fat
Chrysothamnus nauseosus (Pall.) Britt.	CHNA	rubber rabbitbrush, chamisa
Ephedra torreyana S. Wats.	EPTO	joint fir, Mormon tea, Brigham tea
Gutierrezia sarothrae (Pursh) Britt. & Rusby	GUSA	broom snakeweed,
Haplopappus pluriflorus (Gray) Hall	HAPL	jimmyweed, jimmy goldenbush
Lycium pallidium Miers	LYPA	tomatillo, desert wolfberry
Opuntia phaeacantha Engelm.	OPPH	prickly pear, many-spined cactus
Poliomintha incana (Torr.) Gray	POIN	bush mint, rosemary-mint, purple sage
Sarcobatus vermiculatus (Hook.) Torr.	SAVE	black greasewood, chico, chicobush
Senecio douglasii DC.	SEDO	threadleaf groundsel, creek senecio
Tamarix ramosissima Ledeb.	TARA	tamarisk, salt cedar, tamarisco
Yucca angustissima Engelm.	YUAN	narrowleaf yucca, fineleaf yucca
Grasses		
Aristida purpurea Nutt.	ARPU	Purple threeawn, wiregrass
Bromus tectorum L.	BRTE	cheatgrass brome, downy brome
Festuca microstacys Nutt.	FEMI	small fescue, vulpia
Hilaria jamesii (Torr.) Benth.	HIJA	galleta, curly grass
Oryzopsis hymenoides (R. & S.) Ricker	ORHY	Indian ricegrass, sand bunchgrass
Sporabolis airoides (Torr.) Torr.	SPAI	alkali saccaton
Sporabolis cryptandrus (Torr.) Gray	SPCR	sand dropseed
Sporabolus contractus A.S. Hitchc.	SPCO–1	spike dropseed
Sporabolus giganteous Nash	SPGI	giant dropseed
Forbs		
Tripterocalyx carneus (Greene) Galloway	TRCA	wooton sandverbena
Chenopodium album L.	CHAL	common lambsquarter, goosefoot
Ambrosia acanthacarpa Hook.	AMAC	bur ragweed
Amsinkia tessellata Gray	AMTE	rough fiddleneck
Arabis L. species	AR sp.	rockcress mustard
Astragalus L. species	AS sp.	milkvetch, locoweed
Datura wrightii Regel	DAWR	sacred datura, angels trumpet
Descurainia pinnata (Walter) Britt.	DEPI	pinnate tansey-mustard
Erigeron L. species	ER sp1.	daisy
Eriogonum Michx. species	ER sp2.	wild buckwheat, skeletonweed
Kochia scoparia (L.) Schrader	KOSC	kochia, summer cypress
Lepidium L. species	LE sp.	pepperweed, peppergrass
Lupinus L. species	LU sp.	lupine
Machaeranthera Nees. species	MA sp.	aster
Oenothera albicaulis Pursh	OEAL	white-stemmed evening primrose
Plantago patagonica Jacq.	PLPA	wooly plantain
Salsola iberica Sennen & Pau	SAIB	Russian thistle, tumbleweed
Sphaeralcea coccinea (Pursh) Rydb.	SPCO–2	scarlet globemallow, falsemallow
Sphaeralcea parvifolia A. Nels	SPPA	Nelson globemallow

[a]The scientific nomenclature for genera, species, and authorities is consistent with Voss (1983) and the choices of Welsh et al. (1987).
[b]Acronyms combine the first two letters of the genus and species names.
[c]English and Spanish common names are from a variety of sources (Mayes and Lacy 1989; Dodge 1985; Elmore and Janish 1976; Dunmire and Tierney 1995; and Whitson 1992).

4.7.2.1 Black Greasewood Cover

A line intercept method (Bonham 1989) and high-resolution aerial photography were used to estimate black greasewood cover. Field measurement methods were abandoned because of widespread injury and mortality in the black greasewood population, apparently as a consequence of herbicide spraying during surface remediation of the site. The potential greasewood cover as represented in a February 1995 photograph, and not the current condition, was needed for water balance and phytoremediation evaluations.

Line transects equivalent to 30 m long were located on the photograph using a baseline and transect sampling scheme. A baseline equivalent to 177 m long was placed along a road northwest to southeast through the center of the greasewood population. Starting points were randomly selected for transects extending both north and south of the baseline. Random numbers were also used to select starting points along each transect for the 30-m intercept lines and to select an azimuth for the direction of each line. The distance *d* of greasewood canopy intercepted by a randomly placed 30-m line was measured and percent cover for that line was estimated (Bonham 1989):

$$Percent\ \ Cover\ = \frac{\Sigma d}{30m} \times 100$$

High-precision measurement of intercept distances on the photograph was achieved using a sliding table assembly with a lead-screw motion and a binocular microscope with 10-to-70 power zoom and a cross-hair eyepiece. The photograph was attached to the sliding table assembly. The assembly was connected to a digital position readout with a glass linear encoder. The lead screw moves the 10 by 30 centimeter sliding table 1.0 centimeter per 10 revolutions. The encoder transducer provided a digital output of the sliding table position at a resolution of 0.001 mm. Such high resolution encoders are typically used for machining tools. The digital position readout has an LED display that changes instantaneously to indicate the exact position of the encoder.

Five transects on the north side of the baseline and three on the south side fell within the boundaries of the greasewood population (Table 4–19). A total of n = 29 lines extending from these eight transects fell within the population boundaries. The mean percent canopy cover for the greasewood stand was 37.1 with a standard error of 2.8 (Table 4–19). Because the 1995 photograph was taken before the population was sprayed with herbicides, these values are considered to be reasonable estimates of the potential cover of black greasewood for purposes of evaluating the site water balance and the phytoremediation alternative.

4.7.2.2 Black Greasewood Age Structure

Black greasewood is considered to be a good candidate for phytoremediation of ground water nitrates at the Monument Valley site. However, because the greasewood has been decimated by herbicide spraying and heavy grazing, the population will have to be restored in the plume area to achieve acceptable nitrate uptake rates. Therefore, the feasibility of the phytoremediation alternative is dependent on rapid establishment and growth of greasewood transplants in

Table 4–19. Black Greasewood Cover Estimates

Transect[a]		Line Intercept[b]							
		Transects North of Baseline				Transects South of Baseline			
	Starting	Starting		Intercept Distances		Starting		Intercept Distances	
Number	Point (m)	Point (m)	Azimuth[c]	Canopy Measurements (m)[d]	Total (m)	Point (m)	Azimuth[c]	Canopy Measurements (m)[d]	Total (m)
1	20.49	2.23	99.51	1.25, 2.57, 2.33, 1.97	8.12	26.61	207.46	2.69, 2.74, 6.50	11.93
		57.13	16.13	3.58, 4.36, 3.04, 1.55	12.53	43.95	8.26	5.43, 3.58, 3.40, 6.56	18.98
		62.73	354.94	1.73, 1.37, 1.43, 3.70	8.23	64.77	318.32	2.74, 1.31, 0.54, 1.91	6.50
2	66.89	6.64	64.37	1.55, 0.42, 4.12, 3.88, 6.62, 0.66	17.24	15.11	254.92	1.73, 1.25, 2.51, 3.76, 3.28, 3.82, 3.64	19.99
		37.38	310.33	1.91, 1.13, 0.60	3.64	30.38	205.40	1.25, 0.66, 2.03, 2.92, 0.54, 4.65, 1.37	13.43
		77.37	344.66	1.97, 2.09	4.06	84.59	122.81	2.57, 0.66, 0.48, 1.73	5.43
		119.09	287.92	4.06, 0.95, 1.25, 1.91, 3.64	11.81				
		142.59	170.33	1.43, 1.37, 1.61, 1.73	6.15				
3	98.02	6.66	325.14	1.85, 4.83	6.68	23.40	46.60	1.19, 2.57, 7.88, 1.73	13.37
		36.86	215.38	1.79, 4.48, 2.74, 1.01, 2.33, 1.61	13.96	271.25	228.17	0.54, 0.24, 1.73, 1.07, 2.09	5.67
		85.77	4.27	3.58, 0.66, 2.21, 0.30, 2.63	9.37				
		91.61	244.86	2.03, 3.76, 0.90, 0.54, 5.01	12.23				
4	143.04	27.17	240.95	1.73, 0.18, 1.07, 1.31, 1.19, 2.69, 0.54	8.71				
		59.26	274.68	1.49, 1.07, 0.48, 9.79, 1.07	13.90				
		72.69	197.38	2.74, 3.46, 3.76, 2.09, 1.13, 1.85, 0.42	15.45			Percent Cover Summary Statistics:	
		111.40	182.71	2.45, 1.31, 1.49, 1.91, 2.74, 0.42	10.32			mean	37.14
								variance	231.27
5	162.15	12.59	337.01	1.49, 0.84, 2.57, 7.64	12.53			std. dev.	15.21
		51.28	10.55	3.70, 3.58, 3.28, 2.57, 3.58, 1.13	17.84			s.e.(mean)	2.82
		77.77	253.84	2.15, 0.72, 1.01, 1.97	5.85			95% C.I.	5.78
		94.46	247.85	0.60, 5.13, 1.07, 3.58, 1.79, 1.91	14.08			95% LCL	31.36
		147.02	169.11	2.74, 2.09, 0.78, 0.95, 2.92, 3.88	13.37			95% UCL	42.93

[a] Transect starting points were random distances along a 177-m baseline through the center of the stand. Transects extended both north and south of starting points.

[b] Lines for measuring black greasewood canopy intercept started at random distances and extended at random azimuths from transects.

[c] Degrees clockwise from north.

[d] Each measurement represents the length of canopy of an individual black greasewood "contacted" by the intercept line.

overgrazed and denuded areas overlying the plume. A few volunteer greasewood plants have established in the tailings area. The age and size of these volunteer plants were evaluated as an indication of growth rates.

Three black greasewood plants and two fourwing saltbush plants that volunteered in the tailings subpile soils were sampled (Table 4–20). For all five plants, plant height, the long diameter of the canopy, and the short diameter of the canopy were measured. Cross sections of the primary stem of each plant were cut and prepared for analysis using the methods of Fritts and Swetnam (1989). Stem sections cut at an oblique angle in the field were recut at a transverse angle. Specimens were polished with a power sander using sequentially finer grades of sandpaper until vascular cells were discernible under magnification. Entire cross sections were examined for locally absent and double rings and then the rings were counted.

Once greasewood plants become established in disturbed areas, reproduction occurs primarily as sprouting from underground stems that spread laterally from mature plants. This cloning of nurse plants was observed in the subpile soil area. The density of new greasewood plants (Table 4–20), mostly likely clones, were counted within a 6-m radius of the three larger nurse plants.

Table 4–20. Canopy Measurements and Annual Growth Rings of Black Greasewood and Fourwing Saltbush

Plant Number[a]	Height (m)	Long Diameter (m)	Short Diameter (m)	Canopy Volume[b] (m^3)	Clone Density[c] (100 m^2)	Annual Growth Rings
SAVE1	1.35	2.64	2.03	5.68	1.8	4
SAVE2	1.47	2.31	2.16	5.76	3.5	4
SAVE3	1.45	2.97	1.83	6.19	14.2	4
ATCA1	1.02	1.47	1.32	1.55	NA	4
ATCA2	0.89	1.52	1.01	1.07	NA	4

[a]Plant numbers include the genus/species acronyms given in Table 4–18.
[b]Canopy volume was calculated as the area of an ellipse—pi × (long diameter/2) × (short diameter/2)—multiplied by plant height. This overestimate of the volume suffices for comparative purposes.
[c]Seedlings within a 6-m radius of nurse plants were assumed to be clones.
m^2 = square meters.
NA = Not applicable.

4.7.3 Plant Associations and Vegetation Mapping

A plant association is a unit of classification that defines a particular type of plant community. An association generally has a consistent floristic composition, a uniform appearance, and a distribution that reflects a certain mix of environmental factors that can be shown to be different from other associations. Classifying and mapping plant associations helped to delineate land management units at the Monument Valley site with respect to (1) ecological condition; (2) potential for applying the phytoremediation alternative; (3) revegetation potential; (4) irrigation suitability; and (5) likely vegetation response to irrigation with plume water.

The association is a synthesis of local examples of vegetation called stands. For the purpose of defining plant associations at the Monument Valley site, a modified relevé method was used to characterize stands, and then stands were grouped into associations using simple ordination and gradient analysis techniques (e.g., Barbour et al. 1987).

4.7.3.1 Relevé Sampling and Results

The sampling unit, or stand, was defined as an area of approximately 1 hectare (2.5 acres). Most sampling units were well locations within in the plume area (Table 4–21). Several well locations were subjectively selected for sampling because, as a group, they appeared to represent the range of vegetation types in the area. This semiquantitative sampling method consisted of walking through the stand and compiling a list of all plant species present, then walking through the stand again and assigning species to cover classes. Percent cover was not measured precisely. A species was placed in one of six cover classes: less than 1 percent, 1 to 5 percent, 5 to 25 percent, 25 to 50 percent, 50 to 75 percent, and 75 to 100 percent.

Table 4–21. Locations for Relevé Sampling of Plant Stands at Monument Valley

Stand Number	Location of Hectare-Size Stand
606E	East of well MON–606
655	Centered at well MON–655
656	Centered at well MON–656
662	Centered at well MON–662
663W	West of well MON–663
664	Centered at well MON–664
676	Centered at well MON–676
678	Centered at well MON–678
684	Centered at well MON–684
694	Centered at well MON–694
695NE	Northeast of well MON–695
766E	East of well MON–766
766W	West of well MON–766
DR	Crest of a dune ridge between wells MON–764 and MON–664
MSNE	Within the northeast corner of the mill site fence

Relevé data were first organized in a primary data table; stands and species were listed in the order in which they were observed and sampled. A second, differentiated table was generated (Table 4–22) where species were grouped according to growth form (shrubs, forbs, and grasses), and stands were grouped with similar species composition and species abundance. Rows (species) were shifted and columns (stands) were also shifted until groupings of characteristic species emerged. *Sarcobatus vermiculatus* (black greasewood), *Atriplex canescens* (fourwing saltbush), *Haplopappus pluriflorus* (jimmyweed), and *Poliomintha inicana* (bush mint) were considered to be characteristic species for defining associations because they dominate some stands but are nonexistent in others. Some species occurred rarely in only one or two stands (e.g., *Chrysothamnus nauseosus* [rabbitbrush]), and others occurred so often (e.g., *Sporabolus cryptandrus* [sand dropseed]) as to be of little value in differentiating stands.

4.7.3.2 Indirect Gradient Analysis Results

No clear breaks between groups of stands were apparent in the differentiated table that could be used to define associations (Table 4–22). In contrast, the ordering of stands suggests that the

importance of species varies along a continuum. The continuum view of plant associations holds that changes in the abundance of species from stand to stand reflect the physiological tolerance of species to changes in environmental gradients. A simple indirect gradient analysis technique was used to help identify possible environmental drivers, or trigger factors, that may be of overriding importance in controlling spatial distributions of plant associations.

In Figure 4–11 the importance (measured as percent cover) of the most abundant species is plotted by stand in the same order as in Table 4–22. A subset of stands from Table 4–22 that appeared to best capture changes in species abundance was subjectively selected for inclusion in Figure 4–11. Most of the cover data are based on the relevé results. The only exception is that line intercept results were used for cover of black greasewood at stand 606E. Some artistic license was used to draw the species abundance curves from discrete cover data.

An analysis of Figure 4–11 leads to the following inferences:

- The indirect gradient analysis supports the view that associations of species vary across the Monument Valley site as a continuum rather than as discrete units.
- A small subset of species dominate the continuum.
- The abundance curves suggest that some dominant species are associated—have similar distributions—in Monument Valley plant communities.
- Segments of the continuum represented by peaks in associated species can be used to delineate plant associations for the purpose of mapping vegetation units.

4.7.3.3 Vegetation Mapping Results

The relevé results and inferences from the indirect gradient analysis provided a means for delineating plant associations that were used as vegetation mapping units (Table 4–23). Mapping unit names contain the two most dominant species in the plant association. Associations overlap—a given stand may occur in more than one association—because there are no discrete boundaries between associations. Russian thistle and bur ragweed occur in all associations. Over much of the site, the presence of these species is indicative of a history of overgrazing. Highly disturbed areas that were regraded during surface remediation activities and are dominated by Russian thistle and bur ragweed were placed in a separate mapping unit. Remediated areas that remain denuded were also placed in a separate mapping unit.

Table 4–22. Differentiated Table of Relevé Data Showing Species Groups or Associations

	Relevé Number														
Genus Species	**606E**	**656**	**676**	**678**	**766E**	**684**	**DR**	**766W**	**664**	**695NE**	**694**	**655**	**663W**	**MSNE**	**662**
Shrubs															
Artemisia filifolia	1	1	+												
Sarcobatus vermiculatus	2	2	1	1	+									+	
Atriplex confertifolia	1	1	+	1										+	
Senecio douglasii	+	+	1												
Atriplex canescens		2	1	1	1	1	1	1	1	1	+	+	+	+	+
Haplopappus plurifloris		1	2	1	2	2	2	1	1	1		1	+		
Ephedra torreyana				+		+	+	+	+	1	1				
Poliomintha incana						+	+		1	2	1	+			
Gutierrezia sarothrae			+	+	+	+		+				+	+		
Yucca angustissima		+			+		+		+				+		
Opuntia phaeacantha		+			+	+							+		
Chrysothamnus nauseosus													+		
Tamarix ramosissima															
Lycium pallidium															
Grasses															
Bromus tectorum	+	+	1	+	+	1		+			+	+	+	+	
Oryzopsis hymenoides	+		+	+	+	+	+	+	1	1	1	+	+	+	
Sporabolis cryptandrus	+	+	+	+	+	+	+	+	+	1		+	+		
Festuca microstacys	+											+			
Sporabolus contractus		+		+	+			+	+	+			+		
Sporabolis airoides				+	+	+		+	+				+		
Hilaria jamesii				+											
Sporabolus giganteous										+	+				+
Aristida purpurea															
Forbs															
Salsola iberica	2	2	1		+	1	2	1	+	+	1	2	2	2	1
Ambrosia acanthacarpa	1	1			+	+		1	1	+	+	1	1		+
Descurainia pinnata	+	+	+	+	+	+	+	+	+	+		+			
Sphaeralcea coccinea		+		+		+						+		+	
Plantago patagonica		+	+	+		+						+	+		
Amsinkia tessellata		+	+		+	+					+	+			
Astragalus sp.			+	+	+										
Arabis sp.			+	+	+		+	+	+	1					+
Eriogonum sp.				+	+	+									
Erigeron sp.				+	+										
Machaeranthera sp.				+									+		
Lepidium sp.															
Kochia scoparia															
Sphaeralcea parvifolia															+
Datura wrightii											+				
Lupinus sp.									+						
Oenothera albicaulis															

Cover Classes: (+) <1 percent, (1) 1 to 5 percent, (2) 5 to 25 percent, (3) 25 to 50 percent, (4) 50 to 75 percent, (5) 75 to 100 percent

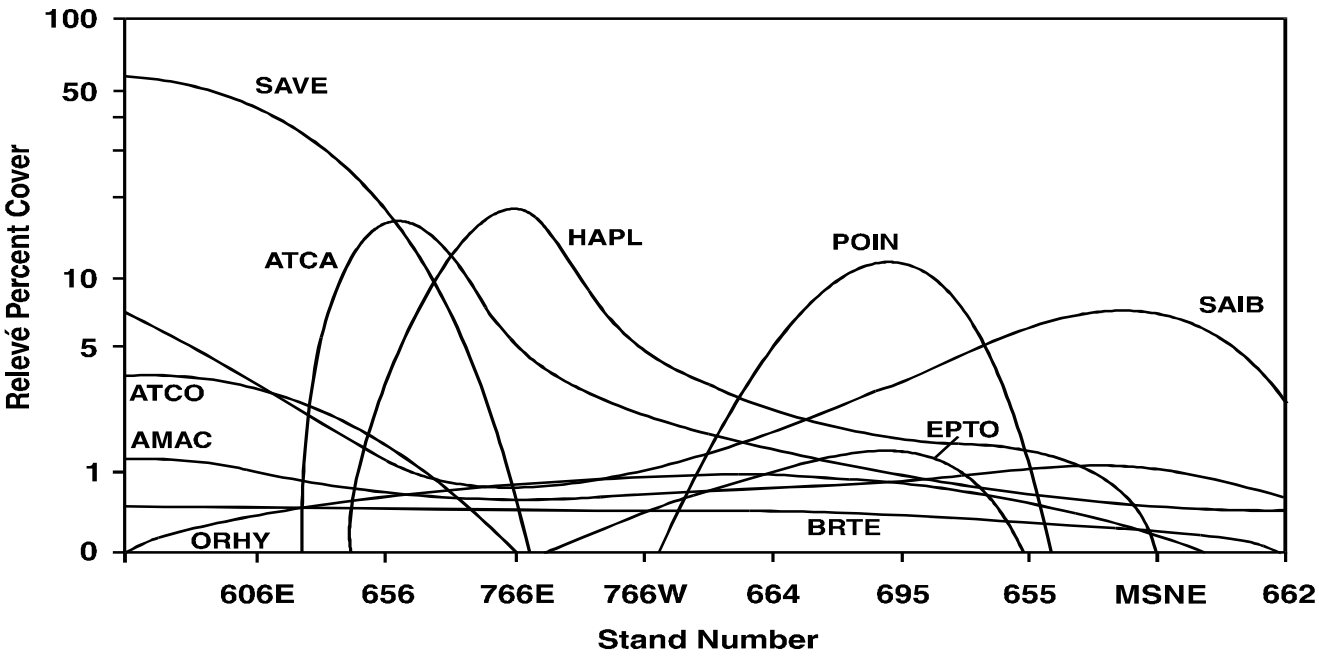


Figure 4-11. Indirect Gradient Analysis of Monument Valley Plume Vegetation

Production of the vegetation map (Figure 4–12) involved

(1) Mapping stand (well) locations on a 1995 aerial photograph.

(2) Identifying vegetation patterns in the photograph, under magnification, that were consistent with the plant associations (Table 4–23).

(3) Outlining mapping unit boundaries using a combination of stand locations and vegetation patterns.

(4) Returning to the field to check the reliability of the photograph interpretation.

Table 4–23. Plant Associations Used as Mapping Units for Monument Valley Site Vegetation

Map Unit	Plant Association	Dominant Species	Stands
SAVE/ ATCO[a]	Greasewood / Shadscale	*Sarcobatus vermiculatus / Atriplex confertifolia*	606E, 656, 676, 681
ATCA/ HAPL	Fourwing saltbush / Jimmyweed	*Atriplex canescens / Haplopappus pluriflorus*	656, 676, 678, 766E, 684, DR, 766W, 664, 695NE
POIN/ EPTO	Bush mint / Joint fir	*Poliomintha inicana / Ephedra torreyana*	664, 695NE, 694
SAIB/ AMAC	Russian thistle / Bur ragweed	*Salsola iberica / Ambrosia acanthacarpa*	655, 663W, MSNE, 662
Bare	Denuded	NA	NA

[a]The SAVE/ATCO unit enclosed two distinctly different vegetation patterns when examined on the aerial photograph and thus was split. The two new units differ with respect to the size and abundance of *Sarcobatus vermiculatus* and not the species composition.
NA = Not applicable

4.8 Land Surveys

At the conclusion of the site investigation fieldwork, physical coordinates and elevations for each new monitor well, Hydropunch location, surface infiltration test location, and hand-auger soil and water sample location were determined by a registered land surveyor. The survey team followed standard contractor survey practices and procedures.

4.9 Ground Water Sampling and Analysis

Each new monitor well was allowed to sit undisturbed for at least 40 hours after final completion before it was developed. Development was performed according to the *Work Plan for Characterization Activities at the UMTRA Monument Valley Project Site* (DOE 1997c). After the wells were properly developed, ground water samples were collected from the new monitor well network and selected existing wells and submitted to the Grand Junction Office (GJO) Analytical Laboratory for analyses.

4.9.1 Ground Water Sampling Procedures

Ground water sampling was performed in accordance with the *Addendum to the Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1996a) and the *Environmental Procedures Catalog* (GJO 1998). The following specific procedures from the *Environmental Procedures Catalog* (GJO 1998) were used for ground water sampling:

- GN–8(P), “Standard Practice for Sample Labeling.”
- GN–9(P), “Standard Practice for Chain-of-Sample-Custody and Physical Security of Samples.”
- GN–13(P), “Standard Practice for Equipment Decontamination.”
- LQ–3(P), “Standard Practice for Purging Monitor wells.”
- LQ–11(P), “Standard Practice for Sampling Liquids.”
- LQ–12(P), “Standard Practice for the Collection, Filtration, and Preservation of Liquid Samples.”
- LQ–2(T), “Standard Test Method for the Measurement of Water Levels in Ground Water Monitor Wells.”
- LQ–4(T), “Standard Test Method for the Field Measurement of pH.”
- LQ–5(T), “Standard Test Method for the Field Measurement of Specific Conductance.”
- LQ–6(T), “Standard Test Method for the Field Measurement of the Oxidation-Reduction Potential (Eh).”

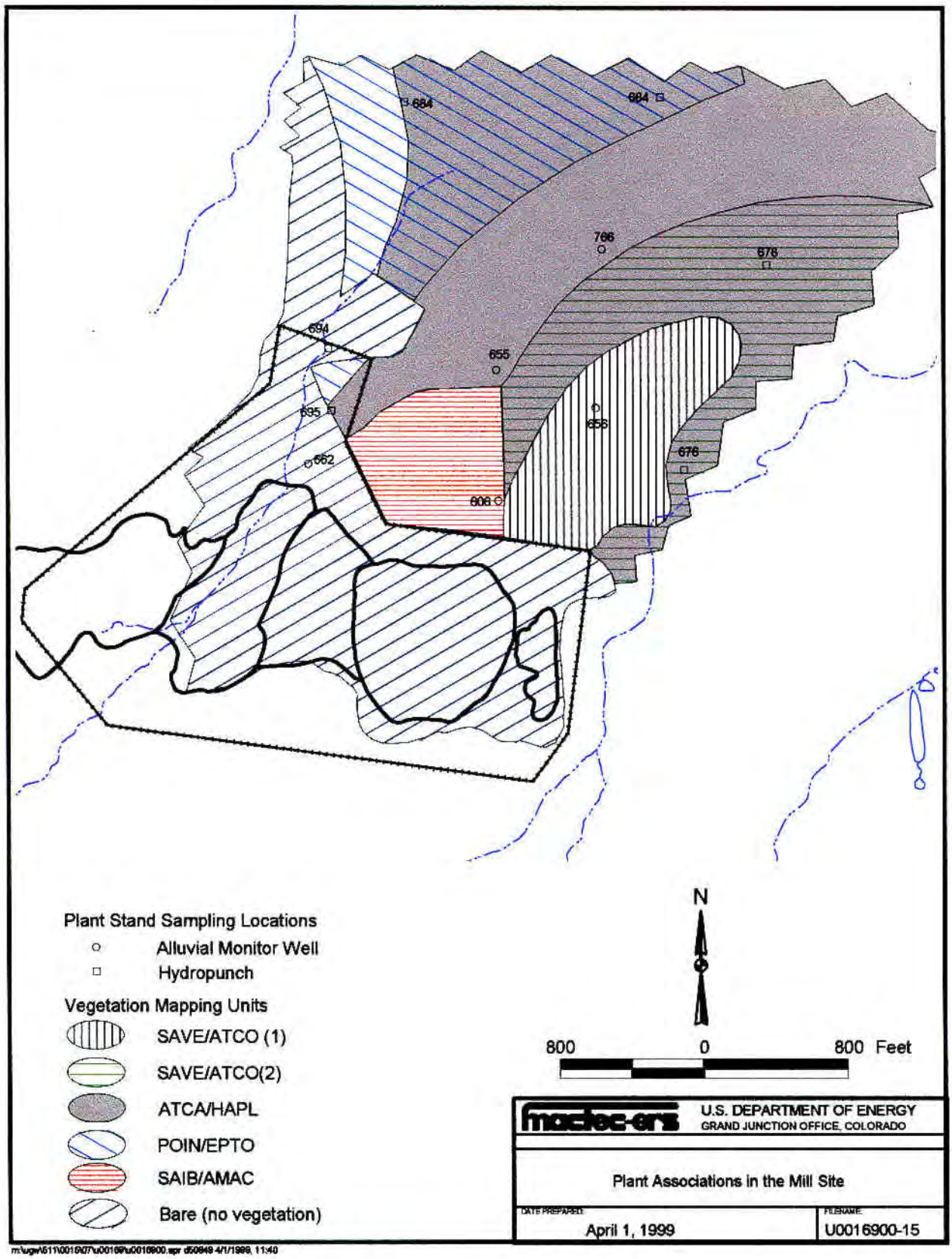


Figure 4–12. Plant Associations in the Millsite

- LQ–7(T), “Standard Test Method for the Field Measurement of Alkalinity.”
- LQ–8(T), “Standard Test Method for the Field Measurement of Temperature.”
- LQ–9(T), “Standard Test Method for the Field Measurement of Dissolved Oxygen.”
- LQ–10(T), “Standard Test Method for Turbidity in Water.”

4.9.2 GJO Analytical Laboratory Sample Analysis Results

A minimum of 10 percent of the samples collected and analyzed were field quality-control samples. Field quality-control samples included equipment blanks, trip blanks, check samples, and duplicates. These samples were submitted for the same analyses as the other field samples.

Analyses of ground water samples submitted to the GJO Analytical Laboratory were also checked for accuracy through internal laboratory quality-control checks, such as blind duplicates, splits, and known standards as specified in relevant EPA guidelines or the contractor’s *Handbook of Analytical and Sample-Preparation Procedures Volumes I, II, and III* (Rust Geotech, undated).

Final analytical results were entered into the SEE_UMTRA database and an independent data validation assessment was performed (DOE 1997a). Results of the analyses are presented in Appendix C.

5.0 Site Conceptual Model

This section presents an interpretation of the site characterization data collected in 1997, correlates the data to previous information, and provides the most current understanding of the extent and magnitude of contamination, exposure pathways, and risk to public health and the environment. These data are integrated into the following site conceptual model to support the proposed ground water compliance strategy and remediation objectives.

5.1 Geology

5.1.1 Regional Setting

The Monument Valley site is in Cane Valley, which is in the eastern part of the larger feature known as Monument Valley that straddles the Monument Upwarp in northeastern Arizona and southeastern Utah. The regional setting of the site is shown in Figure 5–1. Comb Ridge, about 1.5 miles (2.5 kilometers [km]) east of the site, flanks the east side of Cane Valley and is the expression of Comb Monocline where rocks of Triassic and Jurassic age dip 10 to 20 degrees eastward off the Monument Upwarp. Cane Valley, drained by the north-flowing Cane Valley Wash, is floored by unconsolidated material of Quaternary age that consists of dune sand, alluvial material (sand and gravel), and fine-grained sediments that are probably lake-bed deposits (clay or sandy clay). Resistant, gray to tan sandstone that dips eastward at about 4 to 6 degrees flanks the west side of Cane Valley. Several canyons have been incised through the sandstone exposing older reddish siltstones and sandstones of Triassic and Permian age.

Cane Valley is at an elevation of about 4,800 ft (1,500 m) in the area of the site. To the east, Comb Ridge rises abruptly to an elevation of about 5,600 ft (1,700 m). The slopes that gradually rise to the west to elevations of about 5,400 ft (1,650 m) are, from north to south, Yazzie Mesa, Main Ridge, and South Ridge (Figure 5–1).

5.1.2 Stratigraphy

Rocks of Permian to Jurassic age crop out on and within 2 miles (3 km) of the Monument Valley site. Below the unconsolidated Quaternary deposits, the principal bedrock formations affecting the site ground water are, from oldest to youngest: Permian Cutler Formation, Triassic Moenkopi Formation, and Triassic Chinle Formation. These formations, with several of their members, are shown in the schematic stratigraphic section for the site in Figure 5–2. The same formations and several overlying formations exposed in Comb Ridge are shown in Figure 5–3, which is a west-to-east cross section through the site region.

Characteristics of the principal rock units, from oldest to youngest, that are exposed or penetrated by boreholes at the site are described below. Following the description of rock units is a description of unconsolidated Quaternary material that covers much of the site and fills Cane Valley. A detailed geologic map of the site and immediately surrounding area that was investigated in this characterization is shown in Plate 1. Four geologic cross sections (Plate 2) of the site area show stratigraphic relations from west to east across the west part of Cane Valley in profiles from north (A to A') to south (D to D'). Lithologic description of the bedrock and Quaternary material penetrated by the boreholes drilled for monitor wells and Hydropunch

sampling are presented in Appendix A. Included in the borehole lithologic descriptions in Appendix A are descriptions of lithology from the eight boreholes (608B, 610 through 614, 775, and 776) that were cored.

5.1.2.1 De Chelly Sandstone Member of Cutler Formation

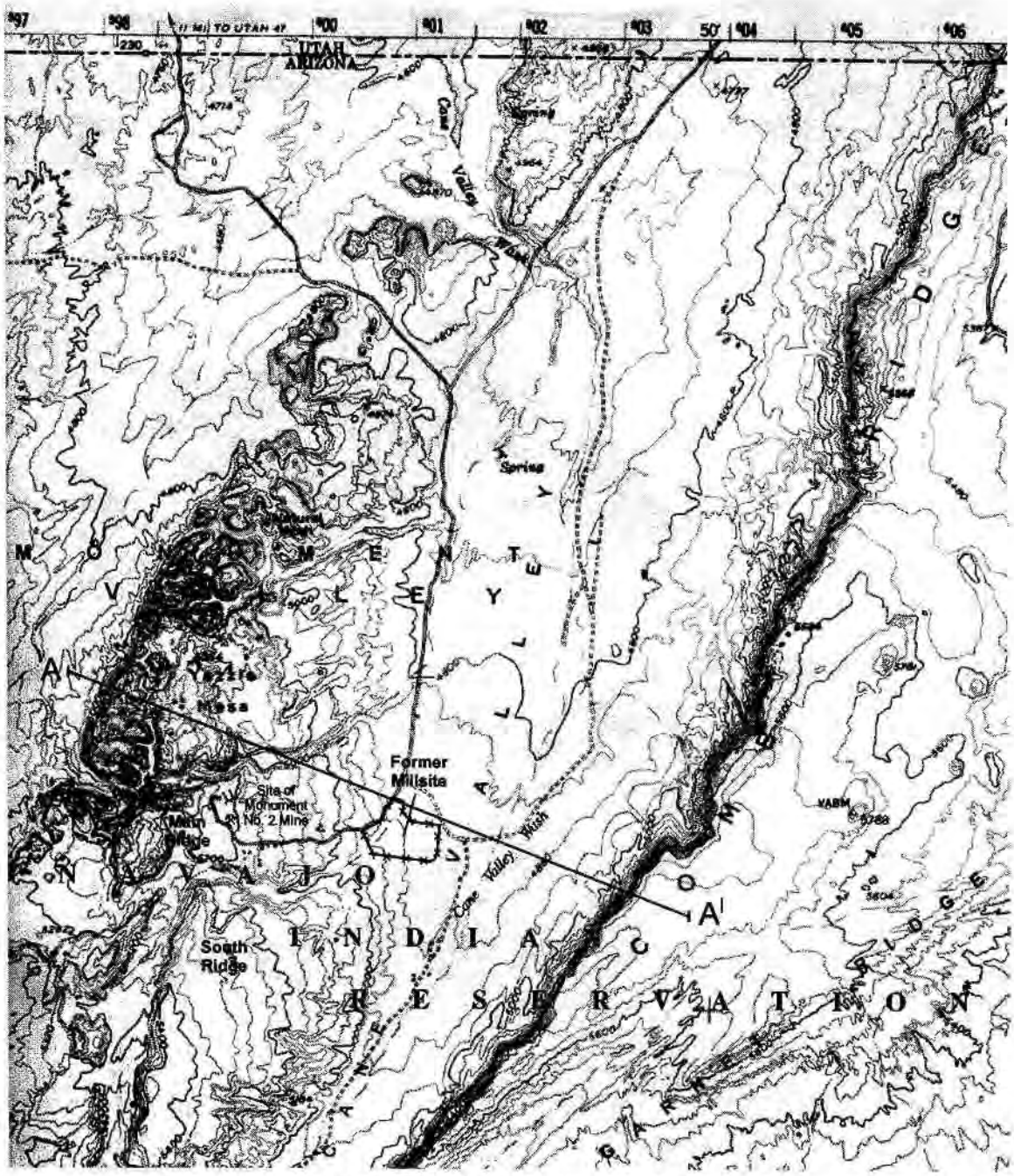
The De Chelly Sandstone Member is about 500 ft (150 m) thick and is the uppermost member of the Cutler Formation. The De Chelly is underlain by the low-permeability siltstones of the Organ Rock Tongue. Sandstone of the De Chelly is light reddish brown (5YR 6/4), fine-grained, quartzose, and poorly sorted. Grains range in diameter from 0.06 to 0.50 mm and are subrounded to round, with a few larger grains that are angular because of authigenic quartz overgrowths (Witkind and Thaden 1963). Most of the grains are colorless quartz with a thin iron oxide film coating each grain imparting the reddish color. Small amounts of microcline, plagioclase feldspar, chalcedony, muscovite, biotite, and zircon are scattered at random throughout the sandstone. The sandstone is friable and weakly cemented by chalcedony, calcium carbonate, and iron oxide. Massive trough crossbedding is characteristic of this eolian sandstone. Crossbed orientation shows little variation and the strike and dip typically is N70°E and 25°SE, respectively.

A prominent and distinct disconformity with almost no relief marks the top of the De Chelly Sandstone. Above the disconformity is the dark red sandstone and siltstone of the Hoskinnini Member of the Moenkopi Formation. The disconformity is widespread and readily identified in core from deep holes (boreholes 608B, 611, 612, 613, 775, and 776) in the site area and in outcrops in canyons west of the site.

5.1.2.2 Hoskinnini Member of Moenkopi Formation

The Hoskinnini Member is only about 10 to 15 ft (3 m) thick in the site area and is the lower member of the Moenkopi Formation (Figure 5–2). Originally named as an uppermost member of the Cutler Formation, the Hoskinnini Member was reassigned as the basal member of the Moenkopi Formation by Stewart (1959). The Hoskinnini Member sediments are generally coarser grained than the overlying main body of the Moenkopi Formation. The top of the Hoskinnini Member was placed at the top of a tan to gray-green, medium-grained sandstone bed about 1.5 ft (45 centimeters [cm]) thick that is overlain by about 11.5 ft (3.5 m) of reddish-brown siltstone and very fine-grained sandstone. Core from borehole 776 penetrated a 12- to 13-ft thickness of the Hoskinnini Member.

Basal Hoskinnini strata are considered to be a reworked zone composed partly of the underlying De Chelly sediments (Witkind and Thaden 1963). The bottom 2 to 5 ft (0.6 to 1.5 m) of Hoskinnini Member consists of a medium- to coarse-grained, massive- to even-bedded sandstone of light reddish brown (2.5YR 6/4) to reddish brown (5YR 4/3) in color that is mottled in places. Sand grains consist of quartz, chert, and plagioclase feldspar that are stained with a light coating of iron oxide and cemented mainly by white calcium carbonate. Grains range from subangular to subrounded with the coarser grains being more angular. The basal mottled sandstone grades



Modified from the USGS 15' Dennehotso, Arizona, topographic map, 1952 ed.

0 1 2 miles

Explanation

- Improved Graded Road
- Unimproved Road
- A——A' Location of Regional Geologic Cross Section
- Fenced Site Boundary

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Figure 5–1. Regional Setting of Monument Valley, Arizona, Site

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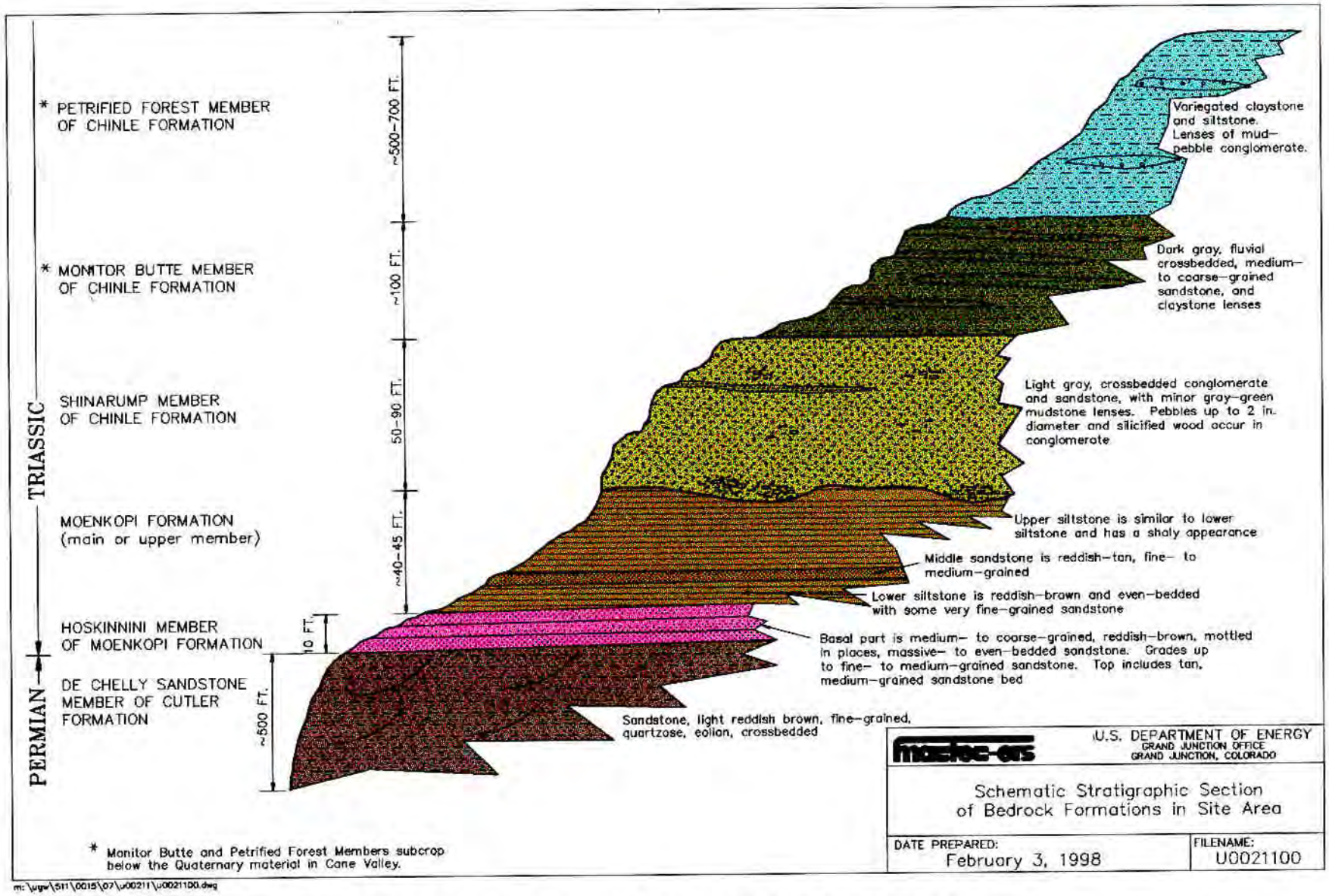

Figure 5–2. Schematic Stratigraphic Section of Bedrock Formations in Site Area

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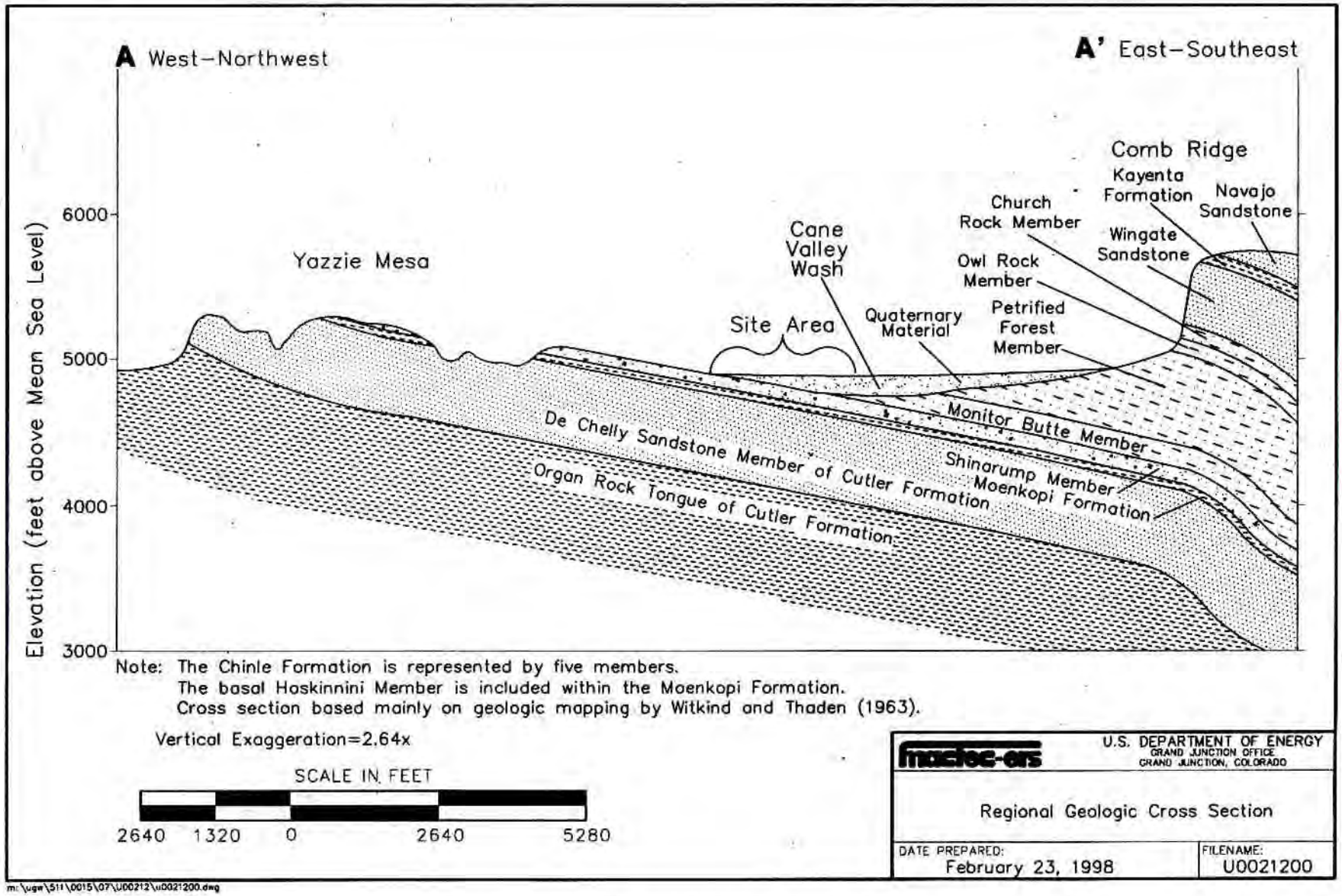

Figure 5–3. Regional Geologic Cross Section

upward into a fine- to medium-grained sandstone 1 to 2 ft (0.3 to 0.6 m) thick that contains tan whorls as evidence of continued mixing and reworking. Above this zone of mixing, one or more thin, medium-grained, tan sandstone beds separated by thin red siltstone may be present. The top of the Hoskinnini Member was placed at the top of the highest tan sandstone bed. The thin Hoskinnini Member in the site area is near the eastern edge of occurrence of the member, which was laid down in a generally quiet tidal flat environment (Witkind and Thaden 1963). The distinctive mixing and reworked zone on the lower Hoskinnini occurs in core from borehole MON–775, which was drilled through Quaternary material into the immediately underlying bedrock consisting of 7 to 8 ft (2.1 to 2.5 m) of the lower part of the Hoskinnini Member.

5.1.2.3 Moenkopi Formation (Main or Upper Member)

The main part of the Moenkopi Formation conformably overlies the Hoskinnini Member and is about 40 to 45 ft (12 to 14 m) thick in the site area. Gray (1961) informally divided the main part of the Moenkopi Formation into three members: a lower siltstone, a middle sandstone, and an upper siltstone. This informal subdivision describes the Moenkopi Formation exposed southwest of the site along the south side of Main Ridge (Plate 1). In those outcrops, the lower siltstone member is about 11.5 ft (3.5 m) thick and consists of even-bedded reddish brown siltstone and very fine-grained sandstone. Cores from this interval (from site boreholes 608B, 610 through 614, and 776) are similar and consist of dark brown (7.5YR 4/4) to dark reddish brown (5YR 3/2) interbedded siltstones and sandstones. Distinctive features present in outcrop that indicate deposition in a nearshore mud flat environment include ripple marks, raindrop pits, and mud cracks.

A fine- to medium-grained, reddish-tan, fluvial sandstone bed about 2 ft (0.6 m) thick overlies the lower siltstone member. This ledge-forming sandstone is laterally continuous and probably correlates to the middle sandstone member as described by Gray (1961). Sand grains are subangular to angular, coated with a film of brown iron oxide, composed mainly of colorless quartz, cemented by calcium carbonate, and range in diameter from 0.1 to 0.3 mm (Witkind and Thaden 1963).

Approximately 29 ft (9 m) of even-bedded siltstone, very fine-grained sandstone, and silty shale beds constitute the upper siltstone member, which is similar in composition to the lower siltstone member. The thin, even-bedded character of this unit give a shaly appearance to this member. Present everywhere in the uppermost Moenkopi is a bleached zone 3 to 7 ft (1 to 2 m) thick immediately below the disconformity at the base of the Shinarump Member of the Chinle Formation. In both outcrop and in core from the site, the bleached zone varies in color from gray (5Y 6/1) to light gray (5Y 7/1). The bleaching was a result of humic acid, a reductant present in the ground water during or soon after the deposition of the Shinarump Member.

5.1.2.4 Shinarump Member of Chinle Formation

The basal member of the Chinle Formation, the Shinarump, is composed of a heterogeneous combination of mainly light gray (10YR 7/2), firmly cemented, crossbedded, conglomerate and sandstone and minor mudstone beds. These sediments were deposited in a series of meandering channels that trended to the northwest. In the site area, the resistant member is 50 to 90 ft (15 to 28 m) thick and forms an irregular, hummocky slope that dips eastward at approximately 4 to

6 degrees where it is exposed in the western part of the site. In the subsurface, in the central and eastern parts of the site, the eastward dip of the Shinarump becomes shallower at only 2 to 3 degrees. The Shinarump grades upward into alternating sandstone and claystone beds of the Monitor Butte Member.

Conglomerate generally defines the scoured base of the member and is composed of mostly rounded pebbles of less than 2 in. (5 cm) in diameter. Average pebble size is ¾ to 1 in. (2 to 2.5 cm). Pebbles are predominantly quartz with smaller amounts of quartzite and chert. Color of pebbles may be white, red, black, green, and yellow. Brown, silicified wood fragments several inches long are common; some parts of original tree trunks as large as 2 ft (0.6 m) in diameter and 5 ft (1.5 m) long are present. Conglomerate grades into medium- and coarse-grained sandstone, which forms the majority of the member. Fine-grained sandstone beds are rare, and a few lenses of gray-green mudstone up to 2 ft (0.6 m) thick are present throughout the Shinarump Member.

The basal fluvial channels of the Shinarump have contained important vanadium and uranium deposits in the Colorado Plateau area. This depositional environment hosted vanadium and uranium mineralization at the Monument No. 2 Mine just west of the site. The Shinarump at Monument No. 2 Mine is much thicker than normal for the area owing to scouring of the basal channel completely through the underlying Moenkopi Formation and into the top of the De Chelly Sandstone Member (Witkind and Thaden 1963).

Intensive exploration for similar thick areas in the Shinarump that denoted possible mineralized channels was conducted in the 1950s and 1960s north and south of the processing site in Cane Valley and along its west flank. One such area of intensive exploratory drilling occurred in the site area and apparently found a west-northwest trending mineralized channel. This exploration effort reportedly consisted of 81 boreholes in which a total of approximately 19,600 ft (6,000 m) was drilled (unpublished uranium exploration map of Oljeto-Monument Valley area). The narrow channel is about 1,000 ft (300 m) long and is about 500 ft (150 m) north of the frog pond area. Surface evidence of the intensive drilling that defined this channel is no longer apparent; however, the drilling likely occurred in an area several thousand feet across in the vicinity of the frog ponds. Depths of boreholes exploring for the basal Shinarump in this area were at least 200 ft (60 m) and could have been as much as 300 ft (90 m) in places where thick Shinarump channel(s) are located. It is likely that some of these boreholes were deep enough to have penetrated the upper part of the De Chelly Sandstone, particularly in the area of the Shinarump channel where scouring greatly reduced the thickness of the Moenkopi Formation.

The thickest Shinarump in the site area found during monitor well drilling was in well MON–664 where the member is approximately 90 ft (28 m) thick. In this borehole, the underlying Moenkopi Formation, which is typically about 50 ft (15 m) thick in this area, is only about 20 ft (6 m) thick. This indicates that a basal channel of the Shinarump has cut down about 30 ft (9 m) into the Moenkopi Formation.

5.1.2.5 Monitor Butte Member of Chinle Formation

The Monitor Butte Member is composed mainly of sandstone, which is fluvial, crossbedded, medium- to coarse-grained, and occurs in dark gray lenses. The thickness of the member is about

100 ft (30 m); however, it is not exposed at the site because it is covered by Quaternary alluvial and eolian material on the floor of Cane Valley.

Bedrock at total depth of monitor well MON–650 at the north end of the site may possibly be in the lower part of the Monitor Butte Member. Two other wells (MON–660 and MON–664) possibly may have penetrated the Monitor Butte Member; however, it is uncertain because lithologic information for these wells is scant and vague. One other well that could have penetrated the lowermost part of the Monitor Butte is well MON–625; however, the total depth of this hole is uncertain and borehole lithologic information is nonexistent. The uncertain location of the subcrop contact of the Shinarump Member and overlying Monitor Butte Member of the Chinle Formation is shown in Plate 1. This contact is inferred from the eastward dip (4 to 6 degrees) of the top of the Shinarump Member bedrock surface and the thickness of Quaternary material present in Cane Valley.

5.1.2.6 Petrified Forest Member of Chinle Formation

Variegated claystone and siltstone compose the bulk of the Petrified Forest Member, which is 500 to 700 ft (150 to 220 m) thick—more than half of the thickness of the Chinle Formation. Minor sandstone and mud-pebble conglomerate beds also are present in the member. The Petrified Forest Member also is not exposed at the site, but it subcrops in the east part of the site in the center of Cane Valley beneath Quaternary material.

Soft red sandstone bedrock at total depth of monitor well MON–652 just east of Cane Valley Wash in the northeast part of the site is in the lower part of the Petrified Forest Member. This well, shown in cross section A to A’ (Plate 2), is the only one at the site that penetrates the Petrified Forest Member. The uncertain location of the subcrop contact of the Monitor Butte and Petrified Forest Members of the Chinle Formation is shown in Plate 1. This subcrop contact is inferred from the assumed eastward dip (2 to 6 degrees) and thickness (about 100 ft [30 m]) of the Monitor Butte Member.

The two members of the Chinle Formation overlying the Petrified Forest Member (in ascending order), Owl Rock and Church Rock Members, crop out east of the site on the east side of Cane Valley along the west-facing slope of Comb Ridge. These members and the overlying sandstones in the Wingate Sandstone, Kayenta Formation, and Navajo Sandstone that form Comb Ridge (Figure 5–3) are east and up-section from the site and do not affect site ground water.

5.1.2.7 Quaternary Material

Thick, unconsolidated Quaternary material consisting of alluvial (sand and minor gravel), eolian (fine- and very fine-grained sand), and minor lacustrine (sandy clay) deposits fill Cane Valley in the site area. Thickness of the Quaternary material in the site area is typically as much as 90 ft (28 m), as determined by borehole and geophysical data and shown on Plate 2 in cross sections A to A’ and B to B’ (both perpendicular to the strike of Cane Valley). The Quaternary thickness exceeds 100 ft (30 m) in several places: 102 ft (31 m) at well MON–650 at the north end of the site, and 120 and 122 ft (37 and 38 m) at wells MON–657 and MON–775, respectively, in a deep paleovalley in the southwest part of the site. This paleovalley, cut through the Shinarump Member and into the Moenkopi Formation, shown on Plate 2 in cross sections C to C’ and D to D’,

appears to contain the thickest Quaternary deposits (possibly up to 130 ft [40 m]) in the site area. Except for the small area of the paleovalley that contains the thickest Quaternary material at the site, the axis of thickest Quaternary deposits in Cane Valley trends north-northeast and is about 2,000 ft (620 m) west of Cane Valley Wash.

The extent of Quaternary deposits is shown in the geologic map on Plate 1. Active and partly stabilized sand dunes that are as much as 15 ft (5 m) high cover much of the valley immediately north-northeast of the processing site. This area and other smaller areas of sand dunes are mapped separately.

The character and variability of the Quaternary material was determined by description of split barrel samples and auger cuttings from the boreholes drilled during the summer of 1997 and from lithologic descriptions of previous drilling included in the SOWP, Rev. 0 (DOE 1996d). Most commonly, the material is well sorted, fine- to very fine-grained, quartzose sand that was deposited by eolian processes. Color ranges from light tan to reddish brown, and typically is yellowish red (5YR 5/6) to reddish yellow (5YR 6/6). Descriptions of the material, generalized from the borehole lithologic logs in Appendix A, are shown on Plate 2 in the four cross sections (A to A' through D to D').

Less common constituents of the Quaternary material are coarse sand with pebbles, gravelly sand, coarse sandy gravel, clayey sand, clayey silt, and sandy clay. The coarser sand and gravelly material was deposited by fluvial processes in minor stream channels and in alluvial fans that occasionally spread into Cane Valley. Pebbles as large as 1 in. (2.5 cm) long occur in these fluvial deposits. At the base of the Quaternary material, coarse deposits up to several feet thick that contain fragments of underlying bedrock often occur. The narrow, upper end of the deep paleovalley contains the coarsest Quaternary material (well MON–657, Appendix A) found during drilling at the site. Elsewhere, the coarser material where it occurred above the base of the Quaternary is rare, only several inches thick, and its sporadic occurrence indicates that the thin layers are discontinuous.

The finer material consisting of sand and silt with varying amounts of clay was deposited in ponded water environments, such as that of a shallow lake and an abandoned stream channel. Occurrence of the clayey material (usually indicated by stickiness in samples) is sporadic and in thin layers (several inches to no more than several feet thick) scattered throughout the thickness of the Quaternary material. Clayey sand and silt usually has some gray or green coloration, but is typically light brownish gray, greenish olive, or pinkish gray. Distribution of the clayey layers coincides with the north-northeast trending wide band of thickest Quaternary deposits; boreholes along the east and west flanks of the valley, generally where the Quaternary thickness is less than 50 ft (15 m), did not pass through clayey layers. Clay layers penetrated by several boreholes are shown on Plate 2 in cross sections A to A' and B to B'. No thick, extensive layer of clay was found. Instead, thin clayey layers at various depths (most commonly from 40 to 60 ft [12 to 19 m]) were found that extend (can be correlated) for distances of hundreds of feet, but not on the order of thousands of feet.

In the vicinity of the fenced garden plot area just north of water supply well MON–625, a light gray clayey layer is present at a depth of less than 10 ft (3 m) in well MON–688 and hand auger hole 854. In the same area, at wells MON–686 and MON–605 (Plate 1) greenish-white water was

noted in association with a gray clayey sand layer at a depth of about 31 ft (9.5 m). In this area and in others around the site where one or more clayey layers are present, the clay layers may locally perch ground water and may channel ground water movement between clay layers.

Quaternary material in large areas along the floor of Cane Valley adjacent to Cane Valley Wash are covered by a thin white crust. This crust is composed of gypsum (hydrous calcium sulfate) or gypsite (an earthy variety of gypsum containing sand and silt) that forms as an efflorescent deposit by evaporation of the shallow (within a few feet of the surface) ground water in this area and deposition (crystallization) of its contained salts.

Calcification (formation of hardpan composed mainly of calcium carbonate) has occurred in places just below the surface of the Quaternary material. One place in the site where this hardpan is exposed is along the east bank of the main tributary to Cane Valley Wash about 500 ft (150 m) north of well MON–654 (Plate 1). Here, the hardpan is white, well indurated, and about 3 ft (1 m) thick.

5.1.3 Structure

Bedrock units in the site area strike north to north-northwest, and their eastward dip varies from 2 to 6 degrees. The variation in angle of dip across the site area was determined by (1) field mapping and surveying the elevation of the basal contact of the Shinarump Member and (2) plotting the elevation of the basal Shinarump Member contact from deep boreholes. Contouring of these elevations results in a structure map of the base of the Shinarump Member. This map shows that the dip of bedrock in the west part of the site (generally west of the former new tailings pile) is 4 to 6 degrees and the dip becomes less steep (2 to 3 degrees) in the east part of the site. This relationship is shown on Plate 2 in the cross sections A to A’ through D to D’.

A pervasive primary joint system is well exposed in the Shinarump Member and older rocks at the site and on the dip slope up Main Ridge to the west. Joints in this system are vertical, spaced about 3 ft (0.9 m) apart, and strike N50–60W. Calcite commonly coats the joint surfaces and minor slickensides occur sporadically.

A minor fault with a displacement of 2 ft (0.6 m) and the same orientation as the primary joint system was seen in the Moenkopi Formation just west of the processing site area. Strong joint control (and possibly a minor fault) occurs in the ridge along the east side of the paleovalley in the area of well MON–619. Just south of this well, the sandstone ridge of the Shinarump Member abruptly drops down about 10 ft (3 m). Quaternary material covers and obscures the contact between the two sandstone ridge segments (Plate 1). A joint surface on this contact just south of well MON–619 strikes N60W, but no slickensides were seen. Because no other definitive evidence for displacement could be found, the displaced ridge may be only the result of differential erosion along the primary joint system.

A vertical system of secondary and tertiary joint systems are present that strike approximately due east and N40E, respectively. These systems along with the primary joint system and minor faults may channel ground water flow in bedrock in the site area.

5.1.4 Bedrock Topography and Geomorphology

A deep, narrow, northeast-trending paleovalley that presently is filled by Quaternary material over most of its length cuts through the south part of the site. The paleovalley was incised into bedrock by an ancestral tributary drainage to Cane Valley Wash in wetter climatic conditions that occurred during parts of the Pleistocene epoch. Additional boreholes drilled in 1997 immediately preceded by a geophysical seismic refraction survey resulted in a more complete understanding of the location and depth of the paleovalley in the site.

Drainages to the west of the site on Yazzie Mesa, Main Ridge, and South Ridge have incised narrow canyons up to 200 ft (60 m) deep (Figure 5–1). Topographic relief on the Shinarump Member sandstones on the dip slope between the incised Drainages is typically only 20 to 40 ft (6 to 12 m). The drainage canyon that separates Main Ridge from South Ridge and exposed the uranium ore body at the Monument No. 2 Mine continues eastward and northeastward to the site where it becomes a paleovalley (or paleodrainage) filled with Quaternary eolian and fluvial material. This paleovalley crosses the site where the old tailings pile and heap-leaching pads were constructed during milling operations (Figure 3–1). Southwest of this processing area, the paleovalley is filled by dune sand and obscured for a distance of about 1,000 ft (300 m) southwestward to the point where the paleovalley rejoins the present intermittent drainage.

Cross section D to D' in the vicinity of well MON–619 (Plate 2) and in the vicinity of well MON–657 (Plate 1 in the work plan [DOE 1997c]), indicate the steep-walled character of the paleovalley and the Quaternary fill thickness of between 60 and 100 ft (18 and 30 m). Seismic refraction survey line 1 (Figure 4–2) also shows the steep-walled paleovalley in the vicinity of well MON–657. Depth of incision in this segment of the paleovalley may have reached only into the lower part of the Moenkopi Formation, into the sandstone and siltstone of the Hoskinnini Member. The actual base of the paleovalley at well MON–657 is probably in the lower Moenkopi rather than the De Chelly Sandstone—previous rotary drilling of this borehole after passing through Quaternary sands and gravels drilled through at least 5 ft (1.5 m) of what was interpreted as Moenkopi rock fragments before entering the De Chelly Sandstone.

North of the well MON–657 area, the axis of the buried paleovalley bends slightly to the northeast and is near well MON–775, which passed through about 120 ft (37 m) of Quaternary material before penetrating the Hoskinnini Member. Here, as in the area of well MON–657, the base of the paleovalley is probably in more resistant sandstones in the lower part of the Hoskinnini Member. In this area, the depth of the paleovalley decreases and the valley walls are less steep, as shown on Plate 2 in cross section C to C' and in seismic refraction survey line 3 (Figure 4–4).

North of well MON–775, the paleovalley axis bends more easterly, and the north edge of the paleovalley appears to be in the area of the well cluster MON–655, –769, and –771. Bedrock is at a depth of 43 ft (13 m) in well MON–769 and at 79 ft (24 m) in well MON–771, which is only about 60 ft (19 m) to the south. These wells mark the steep north edge of the paleovalley, which probably extends 30 to 40 ft (9 to 12 m) deeper just to the south along its axis. Seismic refraction survey line 2 (Figure 4–3) is north and west of the paleovalley.

The position of the buried paleovalley north and east of well cluster MON–655, –769, and –771 is not known. It is likely that the paleovalley axis continues in an east-northeast direction and then bends northward to join the paleodrainage that drained the ancestral Cane Valley located 500 to

1,000 ft (150 to 300 m) west of the present position of Cane Valley Wash. The ancestral drainage of Cane Valley was at a base level much lower than at present. The current drainage, Cane Valley Wash, leaves Cane Valley about 3.5 miles (5.6 km) north of the site through a narrow valley cut in bedrock that drains northwestward and eventually into Gypsum Creek and the San Juan River. The ancestral drainage of Cane Valley Wash was probably located about 2.5 mi (4 km) farther north at the north end of Cane Valley. This drainage also drained into Gypsum Creek and could have provided a much lower base level for Cane Valley and its tributaries. The lower base level would allow for incision prior to filling the valley with alluvial and eolian material. The ancestral drainage of Cane Valley Wash could have been blocked by landslides from the west flank of Comb Ridge or by a combination of eolian deposition during a drying climate and landslide/alluvial fan processes. Blocking of this drainage outlet likely created short periods of internal drainage in Cane Valley resulting in brief formation of lakes and deposition of fine-grained lacustrine or clayey deposits.

The presence of active and partly stabilized sand dunes in and along the sides of Cane Valley indicates that wind erosion and deposition are the dominant geomorphic factors in the site area. Geomorphic factors of secondary importance are brief, infrequent episodes of heavy rainfall events associated with the summer and fall monsoonal period that spread alluvial material down and along the intermittent drainages.

Areas of active to partly stabilized dunes are typically oriented north-northeast, reflecting the prevailing wind direction from the south-southwest. The presence of coppice dunes up to 8 ft (2.5 m) high in several areas in the floor of Cane Valley indicate that active wind erosion by deflation is occurring. Calcified rhizoliths that stand up in relief frequently occur around the edges of stabilized dune deposits, also indicators of active deflation.

5.2 Hydrology

The three main aquifers onsite are the alluvial, Shinarump, and De Chelly aquifers (in descending order), with the Shinarump and De Chelly separated by the Moenkopi Formation and its lowermost Hoskinnini Member. The alluvium is predominantly an unconfined aquifer, which is underlain by the unconfined and leaky confined Shinarump. The main confining unit is the Upper Moenkopi, which overlies the leaky confined Hoskinnini and De Chelly. The Hoskinnini and De Chelly appear to be hydrologically connected, and are described as a single unit in some of the earlier boring logs. In the region of the site containing the quarternary paleochannel, the Shinarump and Upper Moenkopi Formation have been eroded away, providing a direct hydrological connection between the alluvial and De Chelly aquifers. Each of the three main aquifers will be discussed separately in detail.

5.2.1 Alluvial Aquifer

The alluvial aquifer consists mainly of windblown fine- to medium-grained sand deposits which vary in thickness from 0 to 120 ft. The thickest deposits were encountered in the paleovalley area where the Shinarump and Moenkopi has been eroded away. Outside of the paleochannel region, the alluvial material is generally thicker near the axis of the valley, and tapers to very thin deposits near the western and eastern boundaries of the site where no alluvium is present adjacent to bedrock exposures.

There is a broad range of the depth to ground water in the alluvial aquifer across the site. One well (MON–654) screened in the alluvium appears to be under artesian conditions. Potential cause for this occurrence will be discussed in the ground water vertical gradient section. Excluding the well MON–654 location, the depth to alluvial ground water generally ranges from 8 ft (wells MON–602 and –604, located along Cane Wash) to 50 ft (wells MON–662 and –669) below the ground surface. In the area of the nitrate plume, alluvial ground water is encountered between 30 to 40 ft below the ground surface.

Figure 5–4 is the ground water elevation contour map for the alluvial aquifer based on August 1997 water levels. Alluvial ground water generally flows north in the site vicinity. The average horizontal gradient was calculated using water-level elevations measured in wells MON–603 and MON–653 (Table 5–1). These two wells were chosen because they are rather far apart (3,482 ft) and the direct distance between them trends parallel the direction of ground water flow. Historically (water levels have been measured since 1985) the horizontal gradient has been 0.011, which is the same as the gradient calculated using the August 1997 water-level data. The gradient is higher at the southern end of the site (0.012) than the northern portion (0.007) as evidenced by the closer contour spacing in Figure 5–4.

Work completed prior to 1997 suggested the alluvial aquifer hydraulic conductivity ranged from 0.28 to 19 ft/day. The 1997 field investigation, which was focused on the portion of the alluvial aquifer containing the nitrate plume, suggested an average hydraulic conductivity of 21.5 ft/day.

Assuming an effective porosity of 0.25 and a hydraulic gradient range of 0.007 to 0.012, the ground water velocity ranges from 0.6 to 1.0 ft/day. At these velocities the nitrate plume would have taken from 15 to 25 years to reach its present location (furthest extent is approximately 5,600 ft according to ground water quality data presented in Section 5.3.3.1). In the vicinity of the plume the average gradient is 0.0095, which results in a groundwater seepage velocity of 0.82 ft/day. At this velocity, it would take approximately 22 years for the above background nitrate plume to reach its present extent.

Recharge to the alluvial aquifer is the result of the infiltration of precipitation and from upward leakage from the underlying aquifers. This area receives approximately 6.4 in. of precipitation annually, with the majority of the precipitation resulting from isolated thunderstorms during the

Table 5–1. Horizontal Gradient Calculations—Monument Valley Field Investigation

Alluvial Aquifer				***Shinarump Aquifer***				***De Chelly Aquifer***			
Distance Between Wells MON–605 and MON–653 = 3,482 ft				***Distance Between Wells MON–601 and MON–659 = 4,141 ft***				***Distance Between Wells MON–661 and MON–664 = 6,350 ft***			
Date	**603 GW Elev (ft)**	**653 GW Elev (ft)**	**Horizontal Gradient**	**Date**	**601 GW Elev (ft)**	**659 GW Elev(ft)**	**Horizontal Gradient**	**Date**	**661 GW Elev (ft)**	**664 GW Elev (ft)**	**Horizontal Gradient**
10/9/86	4838.19	4799.69	0.011	10/9/86	4870.27	4828.20	0.010	10/19/85	4895.79	4805.01	0.014
3/26/87	4838.16	4798.87	0.011	3/26/87	4870.45	4827.92	0.010	4/27/86	4897.24	4805.54	0.014
5/8/87	4838.6	4799.95	0.011	5/8/87	4870.76	4828.67	0.010	10/9/86	4897.81	4805.49	0.015
4/22/88	4838.81	4800.01	0.011	11/22/92	4870.73	4824.90	0.011	3/26/87	4896.32	4804.16	0.015
11/21/92	4838.51	4799.76	0.011	2/18/93	4871.31	4828.70	0.010	5/8/87	4896.42	4805.53	0.014
2/18/93	4838.86	4799.82	0.011	6/29/93	4870.69	4828.89	0.010	12/9/93	4896.68	4801.04	0.015
6/29/93	4838.43	4800.19	0.011	12/9/93	4870.65	4828.31	0.010	4/20/94	4896.66	4814.67	0.013
12/9/93	4837.98	4800.27	0.011	4/21/94	4870.92	4828.68	0.010	12/14/94	4895.85	4806.52	0.014
4/20/94	4838.53	4800.31	0.011	12/7/94	4870.54	4828.62	0.010	4/19/95	4896.79	4806.71	0.014
12/7/94	4838.23	4800.2	0.011	12/14/94	4870.53	4828.33	0.010	11/15/95	4896.63	4806.44	0.014
4/19/95	4838.66	4800.22	0.011	4/19/95	4871.00	4828.74	0.010	8/19/97	4898.87	4807.35	0.014
11/15/95	4838.04	4800.12	0.011	11/15/95	4870.18	4828.48	0.010			**AVG**	**0.014**
1/14/97	4839.02	4802.2	0.011	1/12/97	4870.74	4829.85	0.010				
8/19/97	4838.61	4801.97	0.011	8/19/97	4870.47	4829.31	0.010				
		AVG	**0.011**			**AVG**	**0.010**				

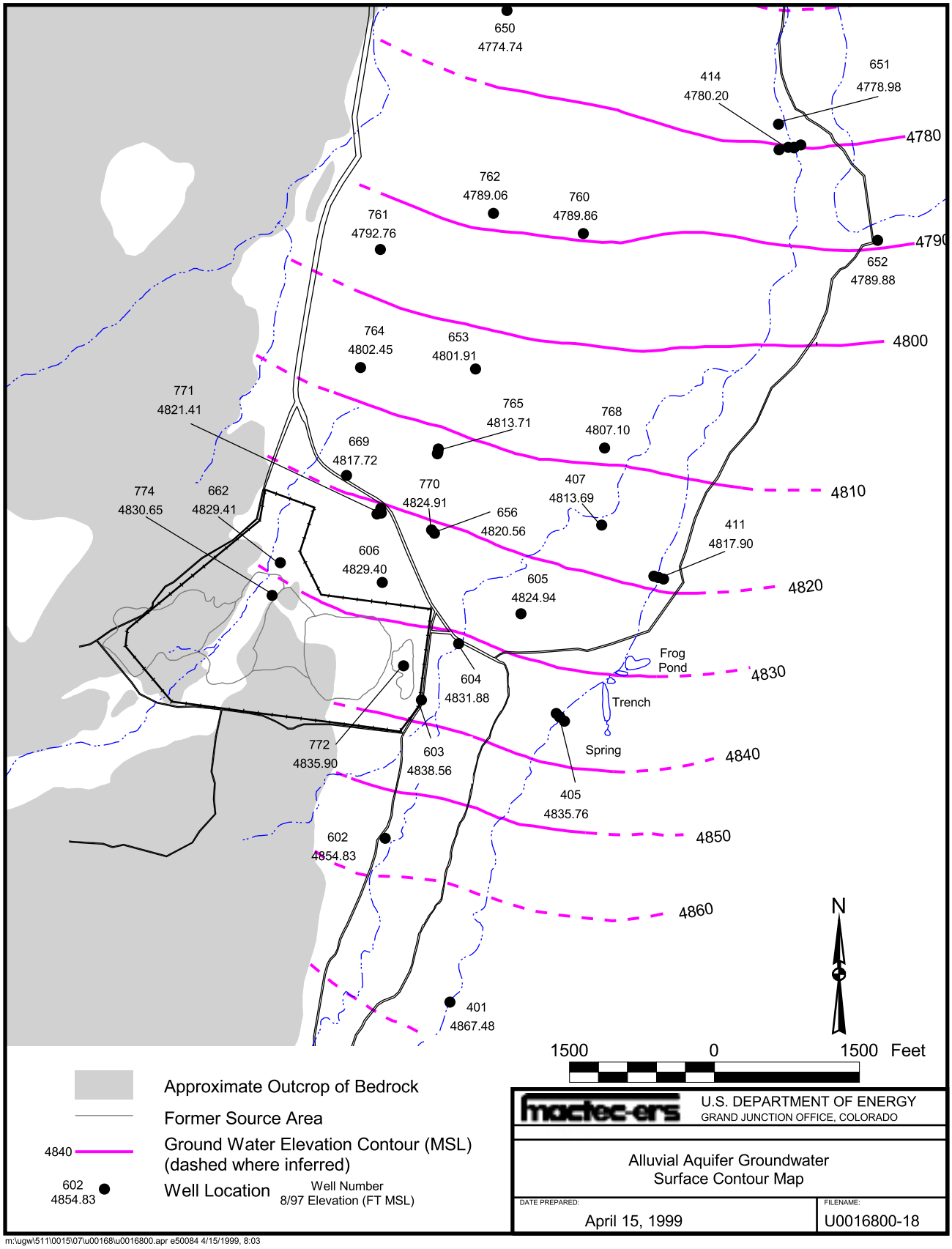

Figure 5-4. Alluvial Aquifer Potentiometric Surface Contour Map

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late summer and early fall. Using the Thornthwaite and Mather (1957) method, an estimated 1.6 in. of the annual 6.4 in. is available for recharge and runoff on a yearly basis. However, only a fraction of the annual precipitation actually enters the aquifer due to loss from evaporation and plant uptake.

Discharge from the alluvial aquifer is primarily the result of ET and evaporation. Pumping from the alluvial aquifer is limited because of the poor water quality and the lower yields when compared to the deeper aquifers.

5.2.2 Shinarump Aquifer

The Shinarump aquifer consists of lenticular deposits of sandstone and conglomerate with occasional thin mudstone layers. Consistent with most alluvial fan deposition, the conglomerate is near the base of the deposit that generally grades upward into the finer grained deposits. The Shinarump forms an exposured bedrock slope west of the site, and to the east the Shinarump aquifer underlies the alluvial aquifer. Thickness of the Shinarump ranges from 0 to 90 ft, and thins north of the site. In some areas where the Shinarump has been eroded, it has been replaced by alluvial material.

Shinarump ground water generally occurs under semiconfined conditions, with the finer-grained upper portions of the unit possibly acting as a confining unit. Ground water may also be under unconfined conditions in the few portions of the site where Shinarump crops out. Depth to ground water ranges from 7 ft (well MON–610) to 50 ft (well MON–614) below ground surface.

Ground-water flow is to the north-northeast according to the ground-water contour map generated using September 1997 water level data (Figure 5–5). As shown in Table 5–1, the average horizontal gradient historically has been 0.010, and the August 1997 water-level data revealed the gradient was the same. This gradient was calculated using water-level data collected from well MON–601 and well MON–659 (located 4,141 ft north of well MON–601). Historical water-level data is contained in Appendix B.

According to the analysis of slug test data, the hydraulic conductivity ranges from 0.4 to 8 ft/day. Assuming an effective porosity of 0.25 and using the horizontal gradient of 0.010, the ground-water seepage velocity ranges from 0.02 to 0.32 ft/day.

Recharge to the Shinarump aquifer is from the infiltration of precipitation in outcrop areas, and to a smaller extent leakage from the underlying De Chelly aquifer. Discharge from the Shinarump appears to be limited to the alluvial aquifer.

5.2.3 De Chelly Aquifer

The De Chelly aquifer consists of fine-grained sandstone that is approximately 500 ft thick in the site area. Ground water is generally semiconfined, and may be unconfined in areas where the main confining unit, the overlying Upper Moenkopi, has been eroded.

The potentiometric surface elevation of the De Chelly aquifer is higher compared to the ground surface elevation along portions of the eastern boundary, resulting in artesian conditions at wells

MON–611, –613, and –625. The maximum depth to De Chelly ground water at other areas of the site is approximately 165 ft, in the vicinity of well MON–661.

Similar to the alluvial and Shinarump aquifers, the De Chelly ground water flow direction is towards the north. As shown on Figure 5–6, there is a higher hydraulic gradient to the south of the site (0.018) compared to the north of the site (0.011). Using water-level data collected from wells MON–661 and MON–664 (6,350 ft apart), the average horizontal gradient across the area historically has been 0.014. Water-level data (Appendix B) collected in August 1997 data suggests a horizontal gradient of 0.014 (Table 5–1).

Analysis of data collected from a 1985 aquifer test indicated a hydraulic conductivity of 6 ft/day. The subsequent test completed during the 1997 field investigation suggested a hydraulic conductivity on the order of 2 ft/day. Using these two values as the range, the ground water seepage velocity ranges from 0.19 to 0.56 ft/day. These calculations were based on an assumed effective porosity of 0.15 and the average hydraulic gradient of 0.014.

Recharge to the De Chelly is mainly a function of precipitation in the vicinity of the site. Outcrops of De Chelly Sandstone located to the west and south of the site tend to enhance recharge into the aquifer. Discharge is the result of vertical leakage into overlying units (to be discussed in the next section) and by domestic and stock use.

5.2.4 Aquifer Interaction

There are three well clusters (wells MON–606/663/659, MON–653/664/660, and MON–603/611/615) located at the site in which wells are screened in the alluvial, Shinarump, and De Chelly aquifers. Water-level data collected at these locations were used to calculate the vertical gradients and ground water flow velocities between the three aquifers.

5.2.4.1 Vertical Gradients

Gradients were calculated by taking the difference of the measured water levels and dividing that value by the difference between the mid-point elevations of the screened intervals for the respective wells. A negative value represents an upward flow direction. Table 5–2 provides the ground water elevations and resulting gradients for the three clusters at various times since 1985. Water-level measurements collected within 48 hours of each other at any cluster location were assumed to be valid and are included in the table.

It should be noted that these gradient calculations may underestimate the actual gradient. For instances where De Chelly wells were under artesian conditions, the ground water elevation was measured at the top of the well casing when, in fact, the water level was actually higher.

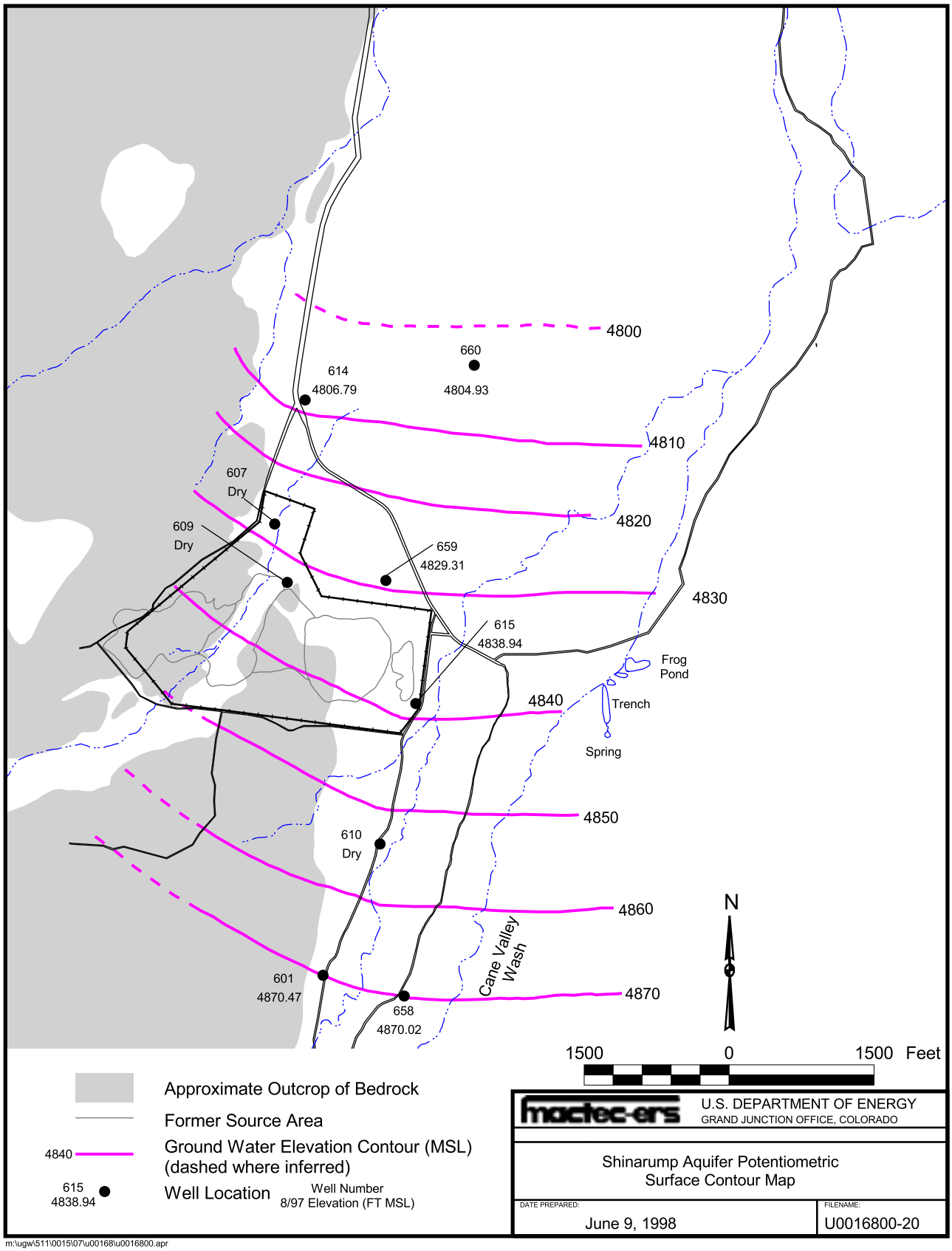

Figure 5-5. Shinarump Aquifer Potentiometric Surface Contour Map

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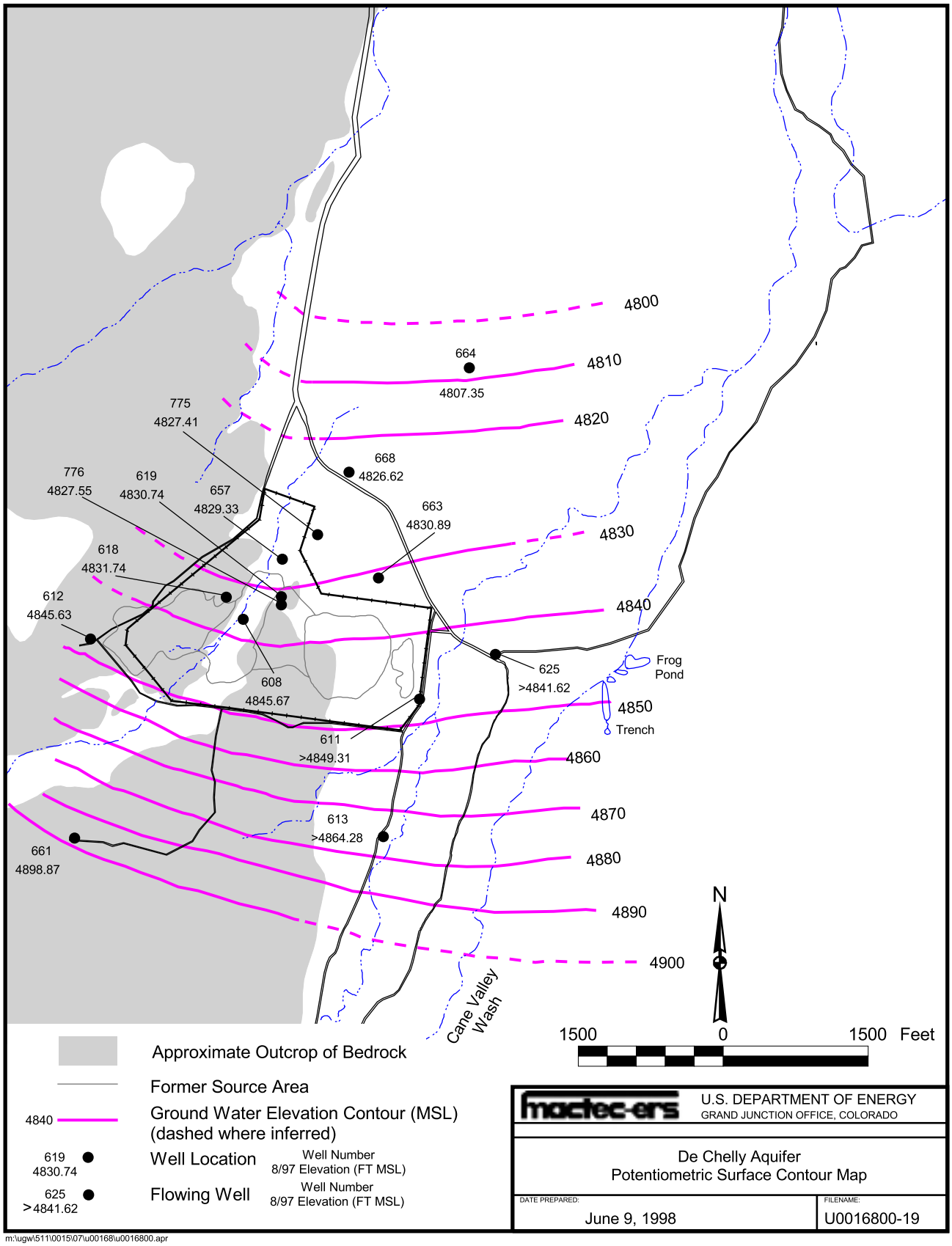

Figure 5-6. De Chelly Aquifer Potentiometric Surface Contour Map

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Table 5–2. Ground Water Vertical Velocity Calculations–Monument Valley Field Investigation

Well MON–606/663/659 Cluster												
				Scrn Midpt			Screen Elev Difference					
Well	Aquifer	TOS Elev	BOS Elev	Elev			Dc/Al	Sr/Al	Dc/Sr			
MON–606	Al	4831.31	4821.31	4826.31			156.90	59.68	97.22			
MON–659	Sr	4776.63	4756.63	4766.63								
MON–663	Dc	4689.41	4649.41	4669.41								
	Ground Water Elevation			Head Difference			Vertical Gradient			Vertical Specific Discharge		
	Well 606	Well 663	Well 659	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr
Date	Al	Dc	Sr	Diff	Diff	Diff	Grad	Grad	Grad	ft/day	ft/day	ft/day
10/9/86	4828.13	4832.38	4828.20	-4.25	-0.07	-4.18	-0.027	-0.001	-0.043	5.31E–07	9.38E–05	5.23E–07
3/26/87	4828.42	4831.71	4827.92	-3.29	0.50	-3.79	-0.021	0.008	-0.039	4.11E–07	-6.70E–04	4.74E–07
5/8/87	4828.49	4832.23	4828.67	-3.74	-0.18	-3.56	-0.024	-0.003	-0.037	4.67E–07	2.41E–04	4.45E–07
12/1/89	4827.11	4831.08	4827.45	-3.97	-0.34	-3.63	-0.025	-0.006	-0.037	4.96E–07	4.56E–04	4.54E–07
1/27/91	4828.57	4832.21	4828.53	-3.64	0.04	-3.68	-0.023	0.001	-0.038	4.55E–07	-5.36E–05	4.60E–07
2/21/92	4828.25	4832.09	4828.35	-3.84	-0.10	-3.74	-0.024	-0.002	-0.038	4.80E–07	1.34E–04	4.67E–07
2/18/93	4828.58	4831.63	4828.70	-3.05	-0.12	-2.93	-0.019	-0.002	-0.030	3.81E–07	1.61E–04	3.66E–07
6/29/93	4828.81	4826.52	4828.89	2.29	-0.08	2.37	0.015	-0.001	0.024	-2.86E–07	1.07E–04	-2.96E–07
12/9/93	4828.19	4829.27	4828.31	-1.08	-0.12	-0.96	-0.007	-0.002	-0.010	1.35E–07	1.61E–04	1.20E–07
4/20/94	4828.55	4831.19	4828.68	-2.64	-0.13	-2.51	-0.017	-0.002	-0.026	3.30E–07	1.74E–04	3.14E–07
12/8/94	4828.50	4831.84	4828.62	-3.34	-0.12	-3.22	-0.021	-0.002	-0.033	4.18E–07	1.61E–04	4.03E–07
4/19/95	4828.69	4832.21	4828.74	-3.52	-0.05	-3.47	-0.022	-0.001	-0.036	4.40E–07	6.70E–05	4.34E–07
11/15/95	4828.43	4832.29	4828.48	-3.86	-0.05	-3.81	-0.025	-0.001	-0.039	4.82E–07	6.70E–05	4.76E–07
8/19/97	4829.60	4832.28	4829.31	-2.68	0.29	-2.97	-0.017	0.005	-0.031	3.35E–07	-3.89E–04	3.71E–07
			Avg	**-2.90**	**-0.04**	**-2.86**	**-0.018**	**-0.001**	**-0.029**	**3.63E–07**	**5.07E–05**	**3.58E–07**
Well MON–653/664/660 Cluster												
				Scrn Midpt			Screen Elev Difference					
Well	Aquifer	TOS Elev	BOS Elev	Elev			Dc/Al	Sr/Al	Dc/Sr			
MON–653	Al	4778.94	4758.94	4768.94			154.18	77.78	76.40			
MON–660	Sr	4701.16	4681.16	4691.16								
MON–664	Dc	4624.76	4604.76	4614.76								
	Ground Water Elevation			Head Difference			Vertical Gradient			Vertical Specific Discharge		
	Well 653	Well 664	Well 660	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr
Date	Al	Dc	Sr	Diff	Diff	Diff	Grad	Grad	Grad	ft/day	ft/day	ft/day
10/19/85	4799.39	4805.01	4800.91	-5.62	-1.52	-4.10	-0.036	-0.020	-0.054	7.15E–07	1.56E–03	5.13E–07
4/28/86	4799.88	4805.54	4802.81	-5.66	-2.93	-2.73	-0.037	-0.038	-0.036	7.20E–07	3.01E–03	3.41E–07
10/9/86	4799.69	4805.49	4802.62	-5.80	-2.93	-2.87	-0.038	-0.038	-0.038	7.38E–07	3.01E–03	3.59E–07
3/26/87	4798.87	4804.16	4801.88	-5.29	-3.01	-2.28	-0.034	-0.039	-0.030	6.73E–07	3.10E–03	2.85E–07
5/8/87	4799.95	4805.53	4802.86	-5.58	-2.91	-2.67	-0.036	-0.037	-0.035	7.10E–07	2.99E–03	3.34E–07
8/13/92	4799.74	4805.66	4802.73	-5.92	-2.99	-2.93	-0.038	-0.038	-0.038	7.53E–07	3.08E–03	3.66E–07
11/19/92	4799.76	4805.59	4802.76	-5.83	-3.00	-2.83	-0.038	-0.039	-0.037	7.42E–07	3.09E–03	3.54E–07
2/18/93	4799.82	4803.16	4802.86	-3.34	-3.04	-0.30	-0.022	-0.039	-0.004	4.25E–07	3.13E–03	3.75E–08
6/29/93	4800.19	4801.44	4802.90	-1.25	-2.71	1.46	-0.008	-0.035	0.019	1.59E–07	2.79E–03	-1.83E–07
12/9/93	4800.27	4801.04	4802.96	-0.77	-2.69	1.92	-0.005	-0.035	0.025	9.80E–08	2.77E–03	-2.40E–07
4/20/94	4800.31	4814.67	4803.33	-14.36	-3.02	-11.34	-0.093	-0.039	-0.148	1.83E–06	3.11E–03	1.42E–06
12/8/94	4800.20	4806.56	4803.22	-6.36	-3.02	-3.34	-0.041	-0.039	-0.044	8.09E–07	3.11E–03	4.18E–07
4/19/95	4800.22	4806.71	4803.23	-6.49	-3.01	-3.48	-0.042	-0.039	-0.046	8.26E–07	3.10E–03	4.35E–07
11/15/95	4800.12	4806.44	4803.07	-6.32	-2.95	-3.37	-0.041	-0.038	-0.044	8.04E–07	3.03E–03	4.21E–07
1/14/97	4802.20	4807.56	4805.32	-5.36	-3.12	-2.24	-0.035	-0.040	-0.029	6.82E–07	3.21E–03	2.80E–07
8/19/97	4801.97	4807.35	4804.93	-5.38	-2.96	-2.42	-0.035	-0.038	-0.032	6.85E–07	3.04E–03	3.03E–07
			Avg	**-5.58**	**-2.86**	**-2.72**	**-0.036**	**-0.037**	**-0.036**	**.7 11E–07**	2.94E–03	3.40E–07

Table 5–2. (continued) Ground Water Vertical Velocity Calculations–Monument Valley Field Investigation

Well MON–603/611/615 Cluster												
				Scrn Midpt			Screen Elev Difference					
Well	Aquifer	TOS Elev	BOS Elev	Elev			Dc/Al	Sr/Al	Dc/Sr			
MON–603	Al	4805.56	4795.56	4800.56			125.21	29.27	95.94			
MON–615	Sr	4781.29	4761.29	4771.29								
MON–611	Dc	4685.35	4665.35	4675.35								
	Ground Water Elevation			Head Difference			Vertical Gradient			Vertical Specific Discharge		
	Well 603	Well 611	Well 615	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr
Date	Al	Dc	Sr	Diff	Diff	Diff	Grad	Grad	Grad	ft/day	ft/day	ft/day
4/23/85	4838.31	4849.31	4838.62	-11.00	-0.31	-10.69	-0.088	-0.011	-0.111	1.38E–06	8.47E–04	1.34E–06
6/4/85	4838.25	4849.31	4838.58	-11.06	-0.33	-10.73	-0.088	-0.011	-0.112	1.38E–06	9.02E–04	1.34E–06
3/26/87	4838.16	4849.31	4838.23	-11.15	-0.07	-11.08	-0.089	-0.002	-0.115	1.39E–06	1.91E–04	1.39E–06
5/8/87	4838.60	4849.31	4838.96	-10.71	-0.36	-10.35	-0.086	-0.012	-0.108	1.34E–06	9.84E–04	1.29E–06
11/22/92	4838.51	4849.31	4838.89	-10.80	-0.38	-10.42	-0.086	-0.013	-0.109	1.35E–06	1.04E–03	1.30E–06
6/29/93	4838.43	4849.31	4838.67	-10.88	-0.24	-10.64	-0.087	-0.008	-0.111	1.36E–06	6.56E–04	1.33E–06
12/9/93	4837.98	4849.31	4838.31	-11.33	-0.33	-11.00	-0.090	-0.011	-0.115	1.42E–06	9.02E–04	1.38E–06
4/20/94	4838.53	4849.31	4838.85	-10.78	-0.32	-10.46	-0.086	-0.011	-0.109	1.35E–06	8.75E–04	1.31E–06
12/7/94	4838.23	4849.31	4838.59	-11.08	-0.36	-10.72	-0.088	-0.012	-0.112	1.39E–06	9.84E–04	1.34E–06
4/19/95	4838.66	4849.31	4839.01	-10.65	-0.35	-10.30	-0.085	-0.012	-0.107	1.33E–06	9.57E–04	1.29E–06
1/12/97	4839.02	4849.31	4839.43	-10.29	-0.41	-9.88	-0.082	-0.014	-0.103	1.29E–06	1.12E–03	1.24E–06
8/19/97	4838.61	4849.31	4839.03	-10.70	-0.42	-10.28	-0.085	-0.014	-0.107	1.34E–06	1.15E–03	1.29E–06
			Avg	**-10.87**	**-0.32**	**-10.55**	**-0.087**	**-0.011**	**-0.110**	**1.36E–06**	8.84E–04	**1.32E–06**
							Hydraulic Gradient			Vertical Specific Discharge		
Site Wide Averages:							Dc/Al	Sr/Al	Dc/Sr	Dc/Al	Sr/Al	Dc/Sr
							ft/ft	ft/ft	ft/ft	ft/day	ft/day	ft/day
							-0.045	**-0.017**	**-0.055**	7.80E–07	1.39E–03	6.25E–07

Notes : Al = Alluvial Aquifer
Bos Elev = Bottom of screen elevation (MSL)
Dc = De Chelly Aquifer
Grad = Gradient
Scrn Midpt = Screen midpoint elevation (MSL)
Sr = Shinarump Aquifer
TOS Elev = Top of screen elevation (MSL)

Results

As Table 5–2 shows, ground water movement has historically been upward from the De Chelly, through the Shinarump, and into the alluvial aquifer at each of the well cluster locations. The average hydraulic gradient between the De Chelly and the Shinarump is -0.055; between the Shinarump and the alluvium the average gradient is -0.017; and between the De Chelly and the alluvium the average gradient is -0.045. A negative gradient value indicates the flow direction is vertically upwards.

Since 1985, the gradient appears to have reversed direction on an infrequent basis. These gradient reversals may have resulted from inaccurate water-level measurements, or the water levels may have been influenced by the pumping of water supply wells during mine reclamation work.

Recent data (collected during the 1997 field investigation) do not indicate a significant difference in hydraulic gradients compared to historical data for the well cluster locations. The data also indicate that hydraulic gradients do not fluctuate seasonally.

It should be noted that the report of a gradient reversal (Appendix F of the RAP) at the MON–603/611/615 cluster location after 1989 was in error. Wells MON–611 and MON–615 were mislabeled in the field at some point between 1987 and 1989, ultimately resulting in an apparent reversal of the gradient. This error has been corrected in the field and in the data base.

5.2.4.2 Ground Water Flow Vertical Velocities

Table 5–2 also provides the ground water vertical specific discharge estimates for ground water flow between the alluvial, Shinarump, and De Chelly aquifers. Specific discharge between the various aquifers was determined using different formulas based on either the presence or absence of a confining unit.

Ground water specific discharge between the De Chelly and the Shinarump aquifers was calculated using the following formula:

$$q = ((h_1 - h_2) / b) K$$

where q = the groundwater vertical specific discharge, or flux (ft/day). The h_1 term is the hydraulic head measured in the Shinarump aquifer overlying the Moenkopi confining unit, while h_2 is the hydraulic head of the De Chelly aquifer below the confining unit. The b term refers to the thickness of the Moenkopi confining unit (estimated to be 40 ft) and K is the hydraulic conductivity of the Moenkopi, which has an estimated horizontal hydraulic conductivity range from 10^{-4} to 10^{-5} ft/day (Golyn 1995). Vertical hydraulic conductivity is generally an order of magnitude lower compared to the horizontal conductivity. As a result, the vertical hydraulic conductivity of the Moenkopi is estimated to range from 10^{-5} to 10^{-6} ft/day. Using the midpoint of this range, the Moenkopi vertical hydraulic conductivity is estimated to be $5x10^{-6}$ ft/day.

A different approach was used to determine the vertical specific discharge between the Shinarump and the alluvial aquifers since there is no confining unit between these two aquifers. The following formula was used:

$$q = (dh / dl) K$$

where the dh/dl term represents the hydraulic gradient between the Shinarump and the alluvial aquifers (values listed in Table 5–2), and K represents the hydraulic conductivity of the Shinarump aquifer. The horizontal hydraulic conductivity of the Shinarump ranges from 0.4 to 8 ft/day. Using the same method as described above to estimate the Moenkopi conductivity, the estimated vertical hydraulic conductivity of the Shinarump used to calculate the specific discharge is 0.08 ft/day.

The formula used to determine the vertical specific discharge between the alluvial and De Chelly aquifers is the same as described for vertical ground water flow between the De Chelly and the Shinarump; however, different values for the thickness and conductivity are used depending on the head loss between the Shinarump and the alluvial aquifers. If there was a small head loss through the Shinarump (less than 0.5 ft), then the thickness and conductivity of the Moenkopi (40 ft and $5x10^{-6}$ ft/day, respectively) are used. Less than 0.5 ft of head loss was measured at both the 606/659/663 and 603/615/611 clusters.

At the 653/660/664 cluster, the head loss through the Shinarump was larger than 0.5 ft (an average of 2.9 ft), and the thickness and conductivity terms in the equation were estimated by calculating the total thickness of the units combined and the respective average vertical conductivity. Based on the field data, the average thickness of the Shinarump is 70 ft. To determine the vertical specific discharge between the alluvium and De Chelly aquifer for this location, a total thickness of 110 ft with a vertical conductivity of 1.4×10^{-5} ft/day were used for the b and K terms, respectively.

Results

Results are included in Table 5–2, with positive specific discharge values representing upwards flow. As shown in Table 5–2, ground water flows upward from the De Chelly towards the alluvial aquifer at all three locations where the data were collected.

In addition to the calculated vertical gradients and respective specific discharges there is additional evidence which supports vertical ground water flow from the De Chelly to the alluvium. Well MON–654, which is located along the eastern portion of the site and screened in the alluvial aquifer, has been observed to be under artesian conditions. The water contained in this well is of De Chelly type, suggesting the artesian flow conditions are a direct result of flow from the De Chelly aquifer (Section 5.3.1.1).

According to the geologic cross-sections in this region of the site the confining Moenkopi is present, and there does not appear to be a direct connection between the alluvial aquifer and the underlying De Chelly. One possible explanation for the influence from the De Chelly may be associated with past drilling activity in this immediate region of the site. Incomplete records from uranium exploration activity indicate potentially 80 boreholes were drilled in the immediate area of the present location of well MON–654. There are no details for the depth of each hole; however, on average each hole was approximately 180 ft deep and extended into the De Chelly aquifer. It is likely these boreholes were not properly abandoned, providing a number of conduits for the De Chelly ground water to vertically migrate into the alluvium over time.

5.2.5 Water Balance

Part of the characterization of the ground water flow system requires the development of a water balance which identifies the components of the flow system, presents the magnitudes and directions of the components, and provides a check for numerical modeling results. The focus of this water balance is the ground-water flow associated with the alluvial aquifer and represents one interpretation of the data collected from the site at this time.

Figure 5–7 shows the boundaries (which encompass a total area of 50,140,000 ft^2) established for the water balance. The most upgradient head boundary is set equal to the average hydraulic head (4,850 ft MSL) measured in well MON–602, while the most downgradient head (4,775 ft MSL) is based on the average hydraulic head measured in well MON–650. The eastern and

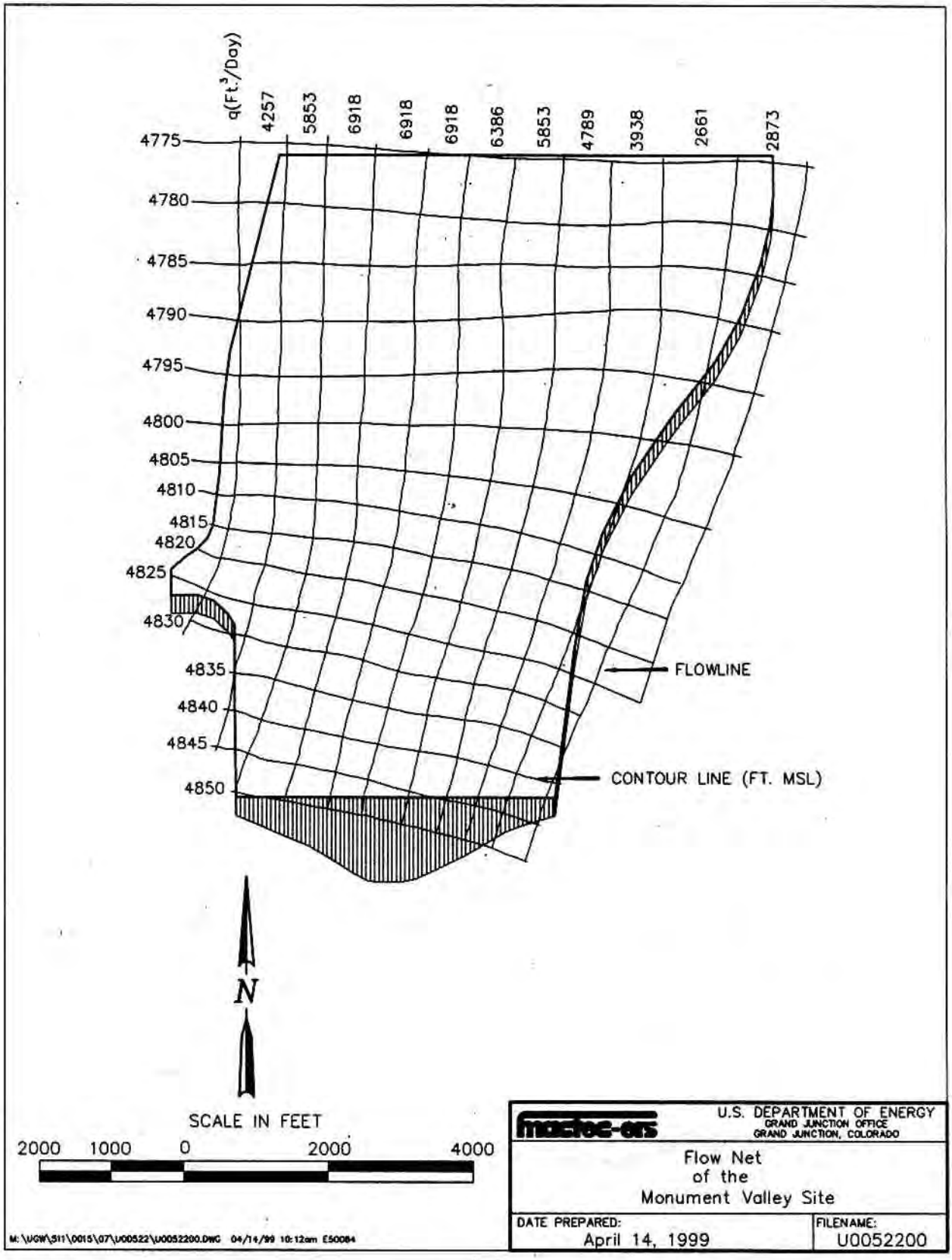

Figure 5–7. Flow Net for the Monument Valley Site

western boundaries were established in the vicinity of the site where Comb Ridge to the east and Shinarump outcrops to the west start to influence the saturated thickness of the alluvial aquifer.

Flow through the alluvial aquifer is estimated by determining the ground water flow entering and exiting the system. Inflow includes flow across the southern boundary, flow across the eastern and western boundaries into the site, ground water migrating from the underlying De Chelly, and recharge from precipitation. Outflow includes flow through the northern boundary and ground water loss through ET.

With the exception of the frog ponds, there are no perennial surface water bodies onsite. The washes that trend north-south through the site transport water only during intense storm events, and the water quickly infiltrates into the alluvium. This surface runoff is not considered to be an additional component of recharge beyond the previously mentioned infiltration of precipitation. There are no direct measurements of natural recharge available for the site.

Ground water discharge is primarily a function of evaporation and ET. Loss due to evaporation is taken into account in the recharge determination. Only an estimated 10 to 20 percent of the annual precipitation is estimated to actually infiltrate and provide recharge to the alluvial aquifer (Stephens 1994).

ET, which is the major component of ground water discharge from the alluvial aquifer, has not been measured directly at the site. However, literature values are available for similar hydrologic systems and plant communities. An estimate of ET for this water balance is based on literature values and the dominant type of plant encountered at the site during a vegetation survey (Section 5.4.2).

Assumptions made in developing the water balance include:

- The flow system for the alluvial aquifer is assumed to be unconfined across the site, with a hydraulic conductivity within the flow system assumed to be one order of magnitude lower in the vertical direction compared to the horizontal direction.

- The total discharge is estimated for the entire thickness of the alluvial aquifer, where flow is assumed to be nearly horizontal. Upgradient and downgradient boundaries of the flow system are assumed to have fixed heads, with discharge through the aquifer assumed to be steady-state.

- Monument Valley is similar to other sites within the arid southwestern United States, therefore measured recharge rates in other parts of the southwest are similar to those at Monument Valley.

- Flow into the site along the east and west is dependent upon respective watershed areas.

5.2.6 Water Balance Calculations

5.2.6.1 Ground Water Flow Across the Southern and Northern Boundaries

Figure 5–7 is a flow net constructed to estimate the groundwater flow across the southern and northern boundaries of the site. Flowlines were drawn perpendicular to the groundwater contours which were used to construct the flowtubes. The hydraulic conductivity calculated in the vicinity of well 765, in conjunction with the hydraulic gradient and saturated thickness (Figures 5–4 and 5–8), were used to determine the flow rates for each flowtube. The resulting flow rates for these tubes ranged from 2661 to 6918 ft^3/day. Based on the flow net, along the southern boundary the hydraulic conductivity estimates ranged from 23 to 44 ft/day, and along the northern boundary the conductivity estimates ranged from 20 to 46 ft/day. The saturated thickness at the northern and southern boundaries is shown by the cross-sections in Figure 5–9.

5.2.6.2 Ground Water Flow Across the Eastern and Western Boundaries

There are no direct measurements of ground-water flow across the east and west boundaries. To estimate flow from these regions, the surrounding areas upgradient of the site between the Shinarump outcrops to the west and Comb Ridge to the east are divided into six different watersheds (WS1 through WS6 as shown on Figure 5–10). Ground water flow entering the site across the southern boundary is a function of flow (in the form of ground water flow resulting from the infiltration of precipitation) predominantly from watershed area WS5 and partially from WS4. Flow associated with the remaining area of WS4, and all of WS1, WS2, and WS3 are contributors to ground water flow through the western boundary, while ground water flow coming into the site from the east is a function of flow originating from WS6.

Once the watersheds were established, a net recharge based on precipitation applied to each watershed was estimated. This was completed by comparing the ground water flow entering the site at the southern boundary and the ground water flow leaving the site at the northern boundary, and determining the flux necessary to provide the flow from the respective watershed areas. This value, which represents a net recharge flux, was then applied to the watershed areas which are responsible for contributing flow through the eastern and western boundaries of the site.

Along the eastern recharge boundary, it is estimated the flux would be consistent across the entire length of the boundary, since there does not appear to be any variation along this boundary. However, along the western boundary there appear to be three distinct recharge zones. It is estimated that one-half of the flow originating from the western boundary is the result of flow from the paleochannel. The remainder of the flow is split between the zones to the north and south of the paleochannel on the western boundary. A difference between the northern and southern zones is the result of the boundary's proximity to the bedrock outcrops, with the northern zone having a lower recharge flux (further distance away from the outcrops) compared to the southern zone's flux (directly adjacent to the bedrock outcrops).

As a result, the paleochannel recharge zone is assigned a flux which ranges from 0.049 to 0.058 ft/day, while the zones to the north and south fluxes range from 0.0036 to 0.0043 ft/day and 0.010 to 0.012 ft/day, respectively. The flux assigned to the eastern boundary is estimated to

be approximately the same as the flux assigned to the northern zone of the western boundary. The eastern boundary recharge flux is estimated to be between 0.0034 and 0.0040 ft/day.

5.2.7 Flux Across the Water Table Boundary – Recharge from Precipitation

No data have been collected to quantify the amount of recharge from precipitation at the site. As a result, this parameter is estimated from literature values. Stephens (1994) presents a comparison of field studies completed in basins in the semi-arid areas of the western United States. The Monument Valley site may be considered analogous to the sites described by Stephens (1994) because of the low annual precipitation measured at Monument Valley (approximately 6.4 inches per year [in./year]) in combination with a rather high annual evaporation rate (estimated to be 84.4 in./year [Cooley 1970]).

According to Thornthwaite and Mather (1957), of the 6.4 inches of annual precipitation measured near the vicinity of the site, only 1.6 inches is available for recharge to the alluvial aquifer and runoff (data contained in Appendix B). Of the 1.6 inches, it is assumed that one-half of this amount acts as runoff, leaving 0.8 inches of annual precipitation available as recharge. This value (0.00018 ft/day) represents the flux to apply to the site area.

5.2.8 Flux Across the Water Table Boundary – Flow from the De Chelly

Geochemical data collected from the frog ponds and samples from wells 654 and 767 indicate De Chelly-type water has migrated into the alluvial aquifer (Section 5–3). The area of the alluvial aquifer believed to be influenced by the De Chelly groundwater is shown on Figure 5–11.

In the vicinity of the ponds and these wells, subsurface data do not suggest the absence of any confining units which may explain the migration of the water from the De Chelly to the alluvial system (Cross-section B, Plate 2). One explanation is the possible presence of the improperly abandoned exploration boreholes, as previously discussed. Based on the information available at this time, there is insufficient data to estimate the flow rate of De Chelly groundwater into the alluvial aquifer.

5.2.9 Evapotranspiration

In deep, fine- to medium-textured soils, ET can account for almost all infiltration in upland arid areas where healthy, late-successional vegetation dominates. However, overgrazing of Cane Valley rangelands has greatly reduced leaf area; the removal of soil via transpiration is less than would be expected for healthy rangeland. Therefore, localized recharge (downward movement of soil water below the influence of plants) in the plume vicinity may exceed 10 percent of the precipitation volume.

Phreatophyte populations downgradient of the mill site are ground water discharge zones (Section 4.7.3.3). Development of a conceptual water balance model for the site requires estimates of ET rates for these phreatophyte communities. Given the depth to ground water and the presence of a top layer of dune sand that can act as a capillary barrier to vertical tension gradients, the evaporation component of discharge is likely insignificant (less than 0.00009 ft/day, less than 0.00009 vapor flow).

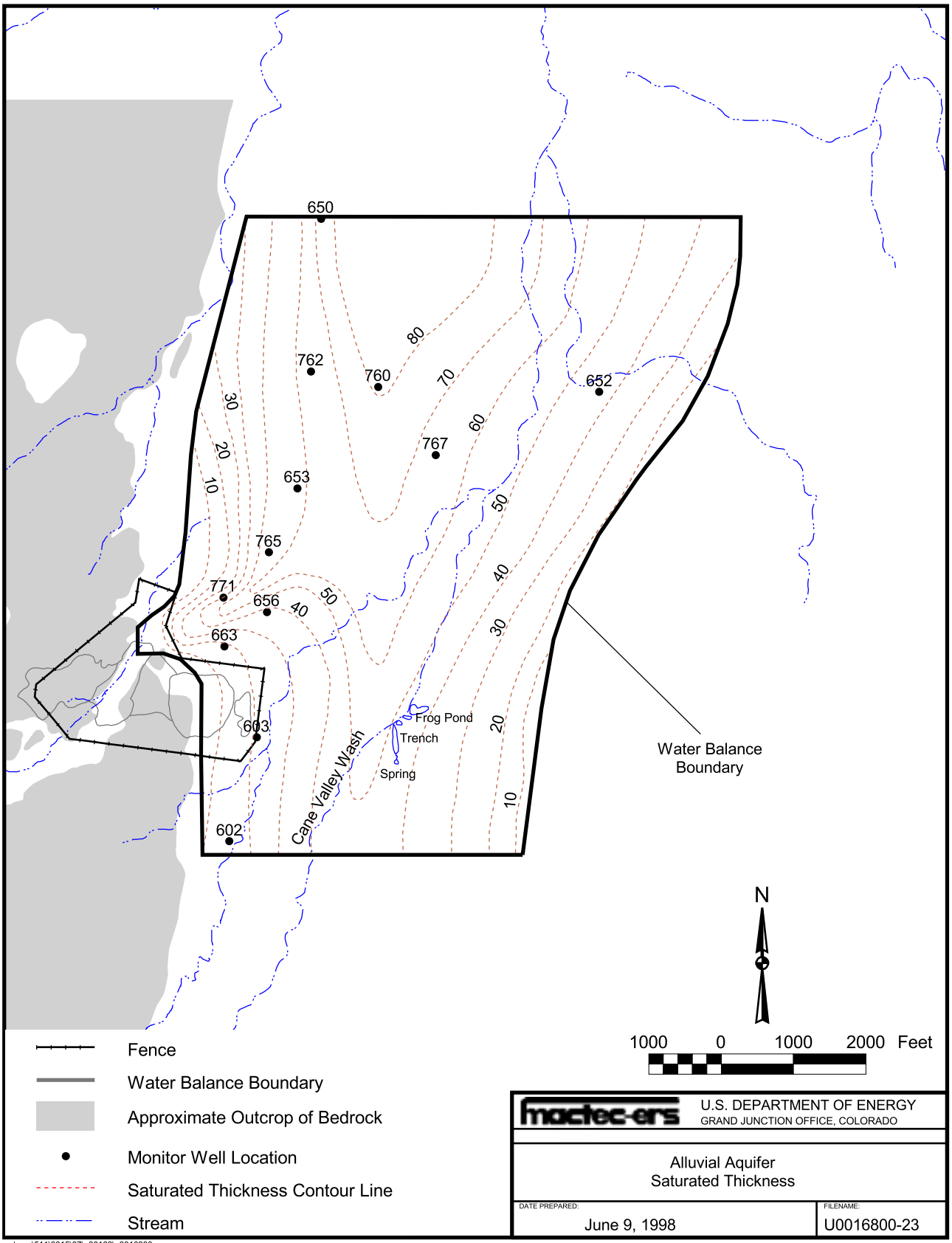

Figure 5-8. Saturated Thickness

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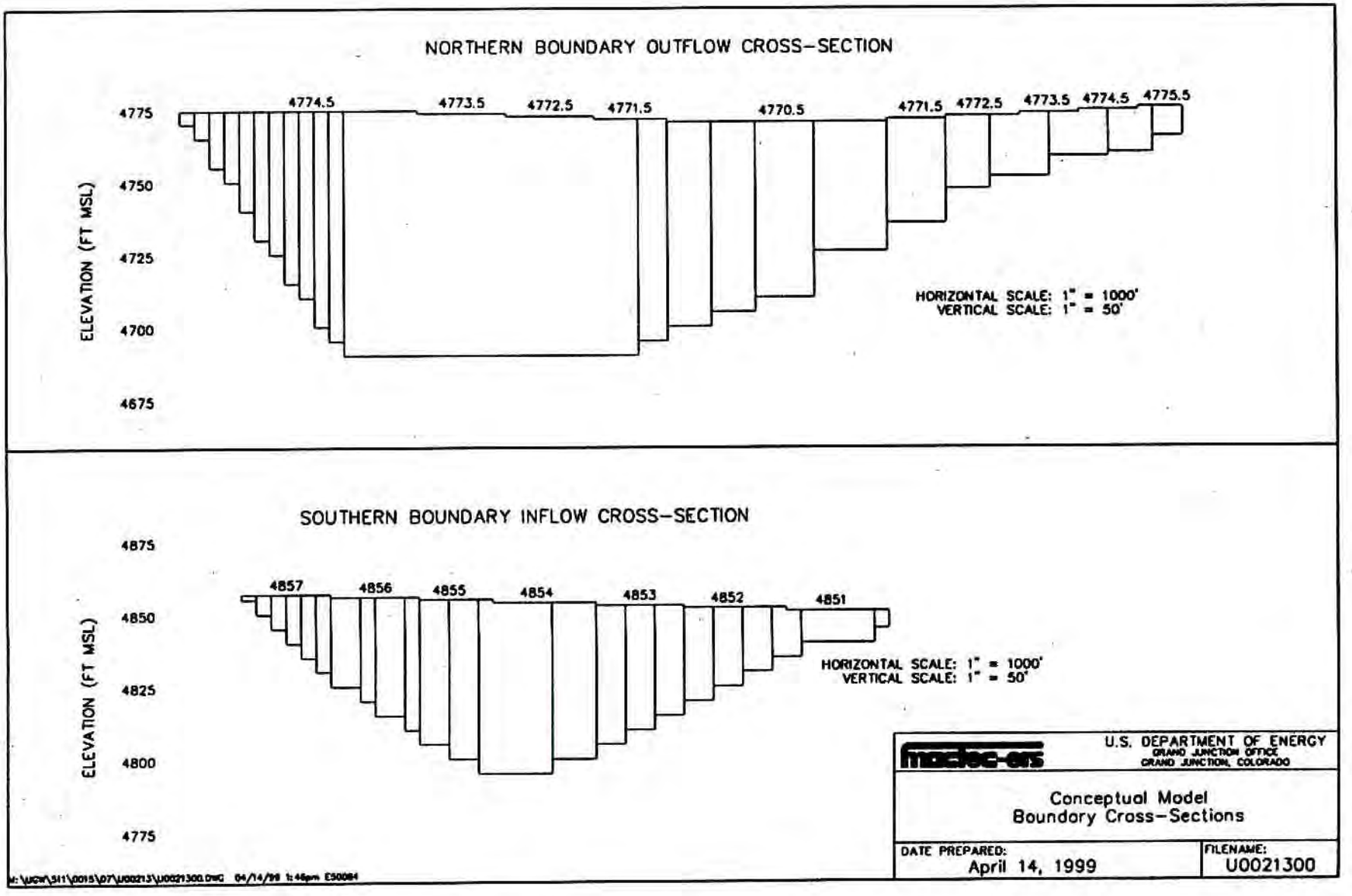

Figure 5–9. Ground Water Flow Model Boundary Cross Sections

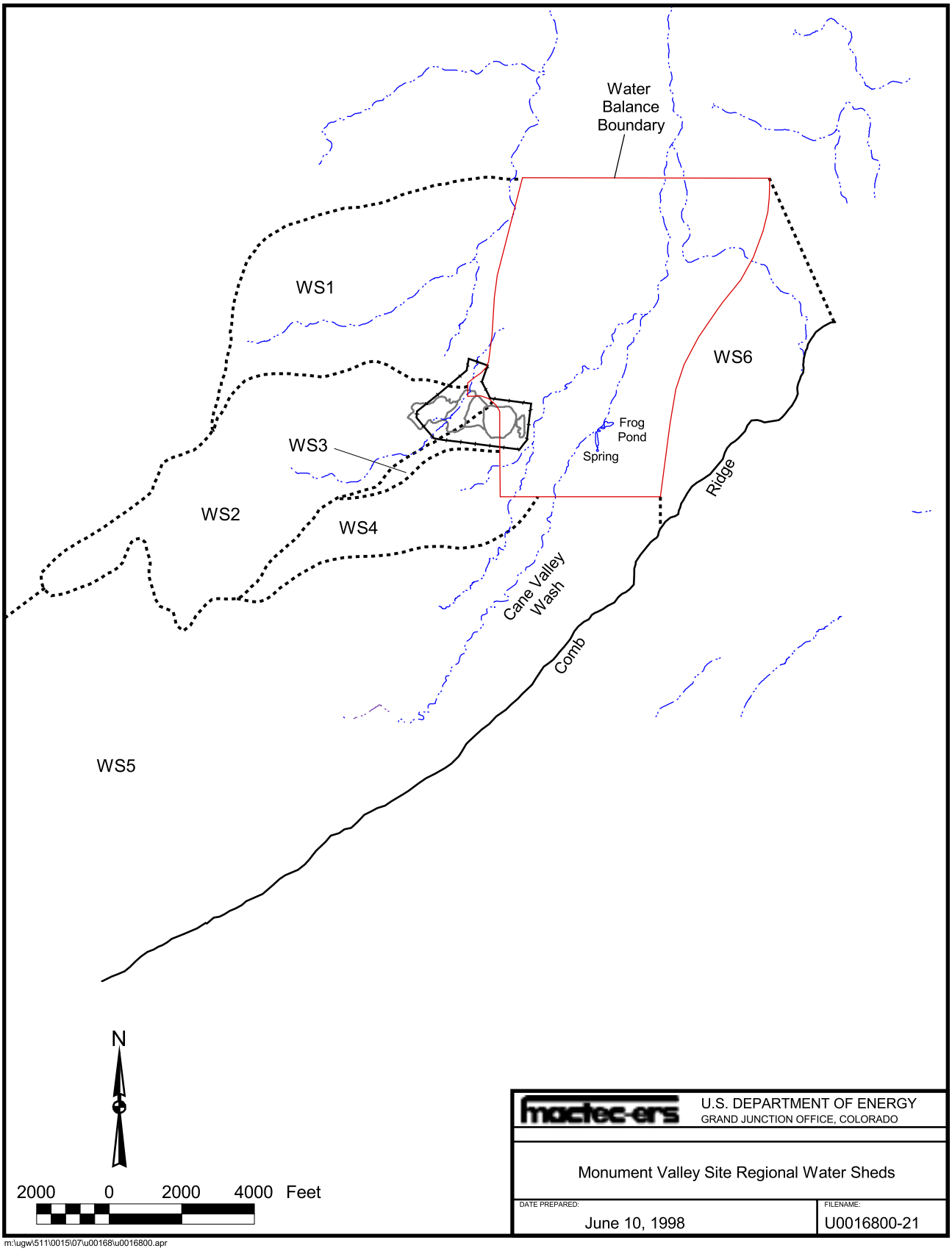

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Figure 5-10. Monument Valley Site Regional Water Sheds

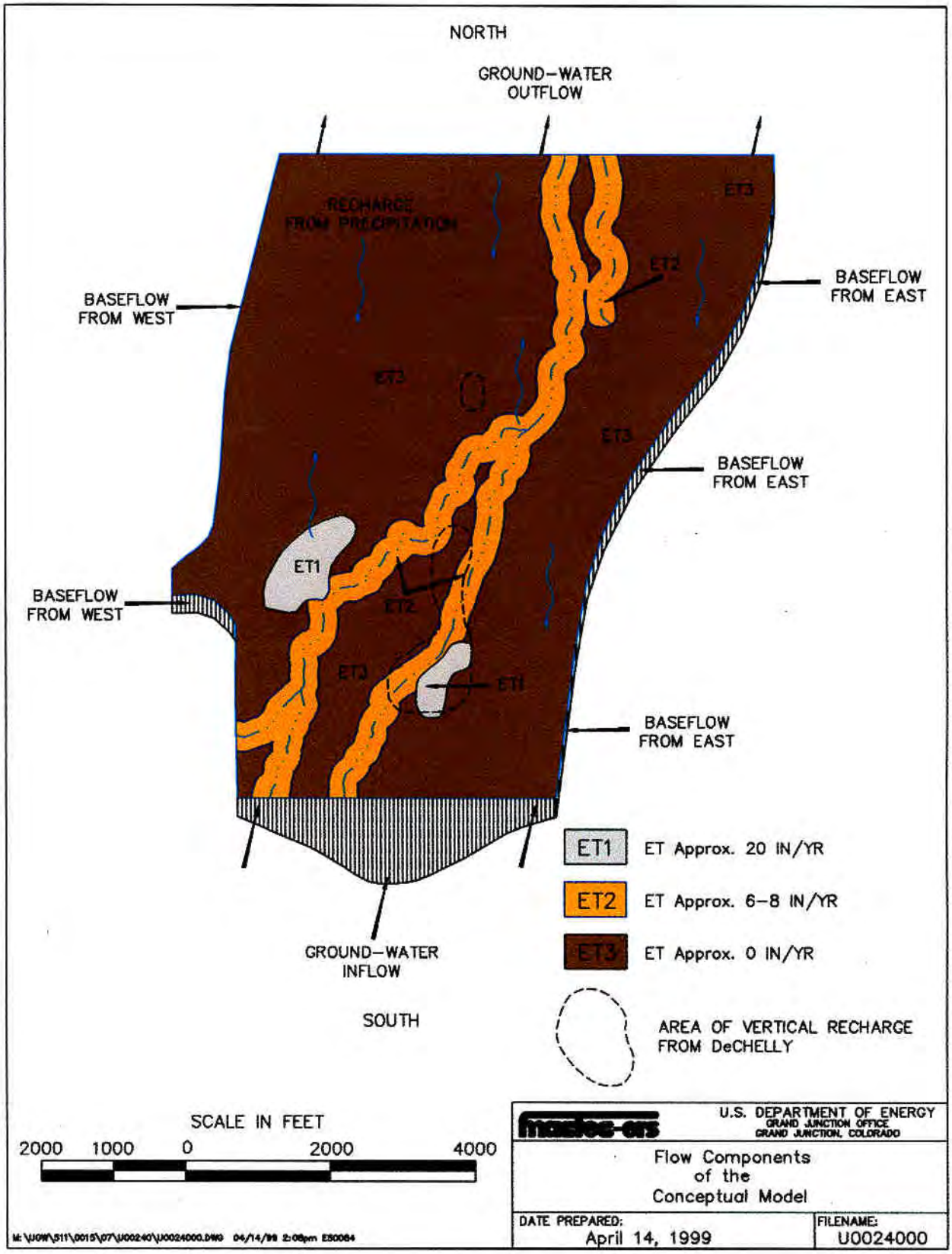

Figure 5–11. Flow Components of the Conceptual Model

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As shown on Figure 5–11, two distinct areas have been delineated (based on current plant communities) from which ground water is discharged via ET. Area ET1 is dominated by greasewood, which has a deep root system that taps directly into the ground water. Recent studies suggest that transpiration from healthy greasewood populations can range from 0.002 to 0.014 ft/day (Nichols 1993 and Branson et al. 1981). The greasewood community located northeast of the former location of the new tailings pile (Area ET1) has been subjected to over-grazing. As a result, it is believed that the greasewood in this region may be less efficient at transpiring ground water, with a flux more on the order of 0.0035 to 0.0045 ft/day.

Area ET2 is limited to areas adjacent to the washes which have plant communities with lower ET rates, with estimated fluxes ranging from 0.00089 and 0.0013 ft/day. The remainder of the site has been designated as Area ET3, which is not considered to be an area of ground water discharge. These fluxes assigned to areas ET1 and ET2 represent maximum potential losses of ground water from the system. The actual volume of ground water removed from the system is dependent upon plant root depth, and may be considerably lower compared to the estimated volume.

5.2.9.1 Discussion and Results

The goal of this water balance is to describe the various steady-state flow components which dictate the ground water flow in the vicinity of the Monument Valley site, based on the current field data. Table 5–3 presents the field water balance by summing the various components, while Figure 5–11 provides a conceptual model to graphically display the components.

Table 5–3. Results of the Steady-State Water Balance for Monument Valley

Flow Component	Inflow (ft^3/day)	Outflow (ft^3/day)
Southern Boundary	44,379	0
Northern Boundary	0	57,363
Eastern Boundary	2,876	0
Western Boundary - Zone 1	2,112	0
Western Boundary - Zone 2	2,112	0
Western Boundary - Zone 3	4,423	0
Recharge from Precipitation	8,929	0
ET loss from Zone ET1*	0	4,480
ET loss from Zone ET2*	0	4,236
Flow from the De Chelly	Insufficient Data	0
TOTAL	64,831	66,079

*These values represent the maximum potential losses

As Table 5–3 shows, there is a 1.9 percent mass balance error in the water balance, with the outflow exceeding the inflow by 1,248 ft^3/day. This difference may be attributed to the inflow of groundwater to the alluvial aquifer from the underlying De Chelly.

5.2.9.2 Conclusions

- A steady-state field water balance was derived for the Monument Valley site. Inflows to the system include: 1) ground water inflow from the south, east, and west, and 2) recharge from precipitation over the entire site. Discharge from the flow system occurs as: 1) ET in areas dominated by the greasewood population and along the drainages, and 2) downgradient ground water discharge to the north.

- A number of the flow components (ground water flow from the east and west, precipitation recharge rates, and ET rates) were estimated because no direct measurements have been made. For some flow components, a flux range was established which represented the estimated minimum and maximum fluxes. Average values for one data set collected in August 1997 were used in preparing Table 5–3.

5.3 Geochemistry

DOE collected ground water quality data from the former processing site and vicinity from April 1985 through September 1998. These data are accessible in the SEE_UMTRA database. The most recent information available was used to assess surface-water and ground water quality. The nature and extent of site-related constituents occurring above natural background concentrations are evaluated and the fate and transport of the site-related constituents in the ground water are summarized in the following sections.

5.3.1 Natural Background

Background water quality is defined as the quality the water would have if uranium milling activities had not taken place. The water quality prior to the milling operations is inferred by characterizing the water quality in areas upgradient of the site that are unaffected by process contamination. Surface waters, soils, and ground water from the alluvium, Shinarump Member of the Chinle Formation, and the De Chelly Sandstone Member of the Cutler Formation were evaluated.

5.3.1.1 Background Surface Water

The only permanent surface water present in the vicinity of the project area occurs east of the former mill site in what is referred to as the Cane Valley frog ponds (Figure 5–12). The frog ponds consist of two man-made ponds constructed during the 1950s and 1960s when the mill was in operation (Hammack 1993). The ponds are situated roughly in a north to south direction along the drainage axis of Cane Valley Wash. Water was supplied by a concrete-lined cistern at the southern pond. The sides of the northern pond were lined with wooden planks braced by ore from the mines. The wooden planks, ore from in and around the northern pond, and evidence of the concrete cistern at the southern pond were subsequently removed during completion of the surface remediation activities at the former mill site in April 1994.

Presently, the southern pond is contained in a long, narrow, and deep bulldozer cut in a large sand dune. The bulldozer cut intersects the alluvial ground water which provides some recharge

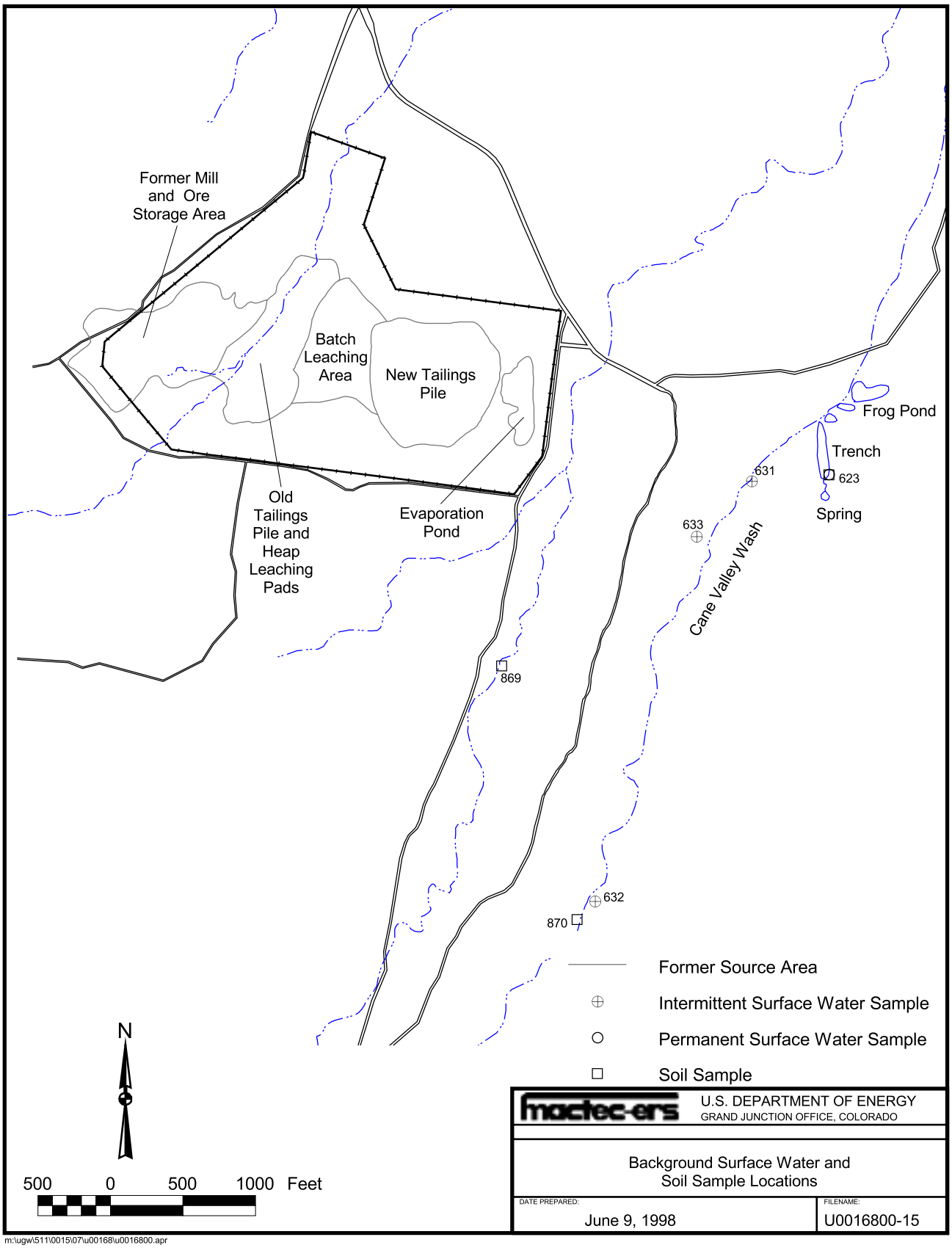

Figure 5-12. Background Surface Water and Soil Sample Locations

to both ponds throughout the year. Geochemical similarities between the pond water and ground water from the De Chelly bedrock aquifer water suggest the ponds may also be receiving recharge through former uranium exploration boreholes that penetrated the artesian bedrock aquifer in the immediate area. The exploration boreholes were probably not properly abandoned, thereby allowing artesian flow from the De Chelly aquifer into the alluvium.

Water quality analyses for permanent surface water samples collected from 1993 through 1997 at the southern frog pond (MON–623) are summarized in Table 5–4. The background surface water sample location is shown on Figure 5–12 and the analytical results are presented in Appendix D.

Background water quality results are interpreted using the Piper diagram (Piper 1944) presented in Figure 5–13. Permanent background surface water collected from the southern frog pond (well MON–623) is characterized by a predominance of calcium and magnesium cations with lesser amount of sodium (Figure 5–13). This calcium-magnesium-carbonate type water closely resembles the chemistry of water from the De Chelly aquifer (Section 5.3.1.5), suggesting that artesian flow from the bedrock aquifer may be providing local recharge to the pond.

TDS concentrations in the frog ponds average 332 mg/L and range from 255 to 420 mg/L. The average sulfate to chloride ratio is 4.4. Nitrate is present at an average concentration of 0.5 mg/L and ranges from 0.1 to 1.0 mg/L. Commonly detected trace constituents include iron, manganese, strontium, radium-226, uranium, and zinc. On average, the water pH is above neutral (pH 7.9) and the redox condition is oxidizing (oxidation-reduction potential 342 millivolts [mV]).

Most of the surface flow along Cane Valley Wash and other small drainage channels in the vicinity of the site is ephemeral (duration of flow less than one month) as a direct result of local precipitation. Natural scours created by ephemeral flow along Cane Valley Wash are common and many intersect the shallow ground water forming small pools which may contain standing water for prolonged periods of up to several weeks or more (intermittent). In response to evaporation and transpiration the pools get smaller and eventually go dry. These small intermittent pools have been observed to occur just upstream of the frog ponds and downstream for several miles.

Water quality analyses for surface water samples collected from 1995 through 1997 at three intermittent pools located upstream from the frog ponds (MON–631, –632, and –633) are summarized in Table 5–4. The background surface water sample locations are shown on Figure 5–12 and the analytical results are presented in Appendix D.

Evident in the background intermittent surface water results presented in the Piper diagram (Figure 5–13) is the predominance of the sodium cation with lesser amounts of magnesium and calcium. The predominant anion is carbonate (reported as alkalinity in Table 5–4) with lesser amounts of sulfate and chloride. This sodium-carbonate type water is also characterized by relatively high concentrations of TDS which average 1,951 mg/L and range from 890 to 2,230 mg/L. Water in these small intermittent pools is subject to severe effects from evaporation, which tends to increase the concentrations of trace and major elements while keeping their relative proportions constant. For example, while sulfate and chloride concentrations increase, the ration of sulfate to chloride concentration will remain approximately the same. Thus, surface

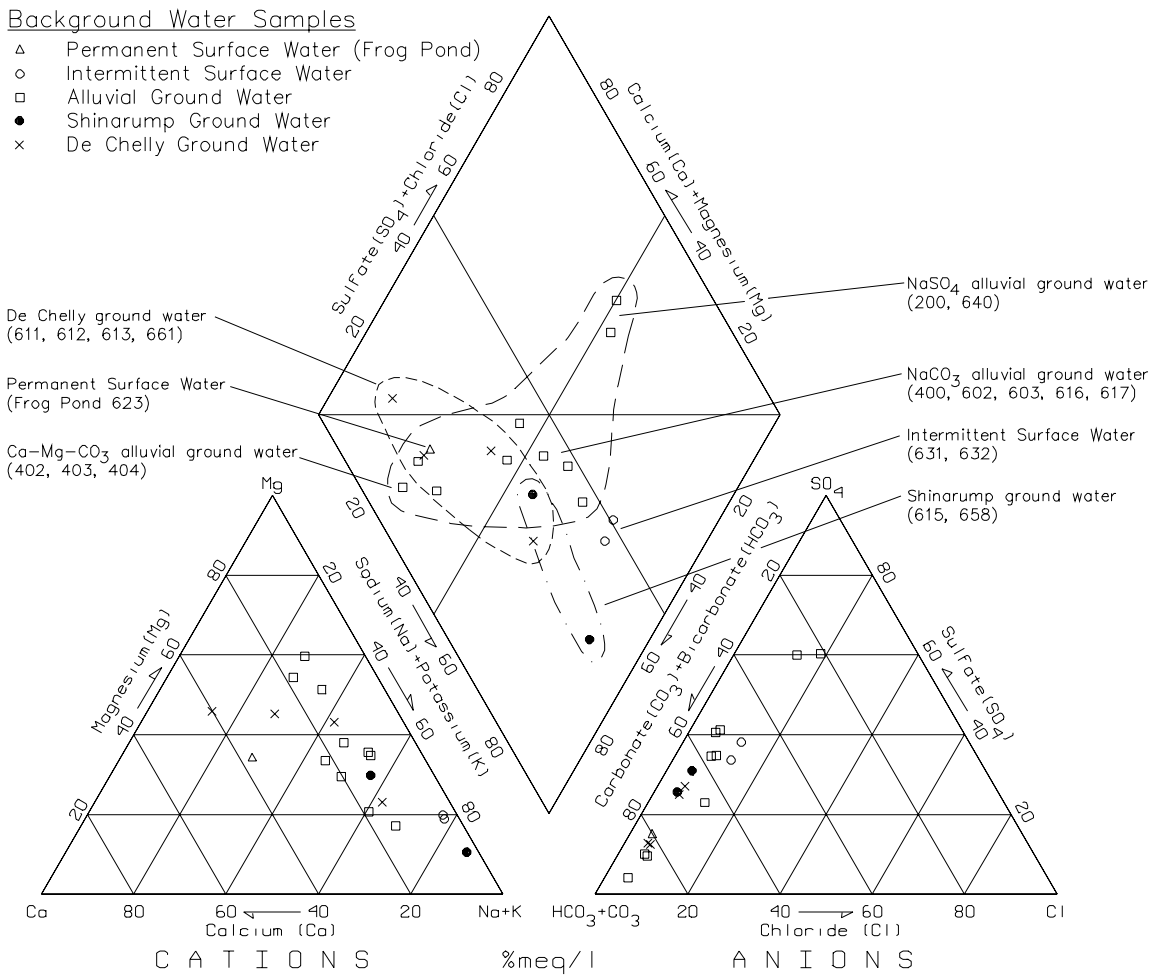

Figure 5-13. Piper Diagram of Background-Water Chemistry

water in the pools along Cane Valley Wash tend to have naturally occurring high concentrations ratio of sulfate to chloride concentration will remain approximately the same. Thus, surface of TDS and major ions including sulfate, chloride, magnesium, sodium, and alkalinity, as compared to the permanent background surface waters (Table 5–4). Commonly detected trace constituents in the background intermittent surface water include manganese, molybdenum, selenium, strontium, uranium, vanadium, and radium-226. On average, the water pH is above neutral (pH 8.6) and the redox condition is oxidizing (oxidation-reduction potential 317 mV).
The average sulfate to chloride ratio is 8.1. Nitrate (expressed as NO_3) is present at an average concentration of 0.5 mg/L and ranges from 0.4 to less than 1.0 mg/L.

5.3.1.2 Background Sediment and Soil Chemistry

In the area of the frog ponds and Cane Valley Wash, the ground water in the alluvial aquifer is commonly within a few feet of the surface. Capillary action keeps the sediments in the bottom of the wash wet, and evaporation and transpiration by plants of the capillary water results in the precipitation and accumulation of a 1- to 3-mm-thick crust of salts over most of the surface of the wash.

Table 5–4. Background-Water Quality for Permanent and Intermittent Surface Water

	Permanent[a]			Intermittent[b]		
Analyte	**FOD[c]**	**Mean[d]**	**Range[e]**	**FOD[c]**	**Mean[d]**	**Range[e]**
Major (mg/L)						
Ammonium as NH_4	3/8	.0479	.0126–.1	½	.665	<.1–1.28
Calcium	10/10	54.16	28.9–171	5/5	19.7	12.8–28
Chloride	7/7	8.9643	5.9–15	5/5	155.13	3.2–238
Magnesium	10/10	30.35	22–76.1	5/5	92.9778	50.7–132
Nitrate	3/8	.4716	.0744–1	1/5	.4836	.352–<1
Potassium	9/9	6.4956	2.2–29.7	1/1	27.7	27.7–27.7
Sodium	10/10	43.7	35–54.5	5/5	570.2	180–917
Sulfate	8/8	39.2375	24.9–70	4/4	422.3	95–573
Metal (mg/L)						
Arsenic	0/1	.0025	<.005–<.005			
Iron (Filtered)	0/3	.015	<.03–<.03	0/4	.015	<.03–<.03
Iron (Unfiltered)	2/2	16.6	.1–33.1			
Manganese (Filtered)	5/5	.0602	.01–.16	2/4	.021	<.01–.05
Manganese (Unfiltered)	4/4	.315	.02–.99			
Molybdenum	0/5	.005	<.01–<.01	4/4	.0728	.017–.12
Selenium	0/2	.0003	<.0002–<.001	1/1	.0002	.00024–.00024
Strontium	11/11	.6031	.407–1.35	2/2	.58	.38–.78
Uranium	9/11	.0028	<.001–.0063	5/5	.0088	.002–.0274
Vanadium	1/12	.0095	<.004–.06	4/5	.0136	<.01–.02
Zinc (Filtered)	0/2	.025	<.05–<.05			
Zinc (Unfiltered)	2/3	.116	<.05–.213			
Other						
Alkalinity as $CaCO_3$ (mg/L)	8/8	261.625	197–371	3/3	795	629–911
Redox Potential (mV)	4/4	341.5	146–438	2/2	316.5	198–435
Silica (mg/L)	5/5	72.92	14.9–295			
Sulfate/Chloride	6/6	4.3558	3.0329–5.9322	4/4	8.05	1.5924–29.6875
Total Dissolved Solids (mg/L)	7/7	332.1429	255–420	5/5	1951	890–2230
Total Phosphorus as PO_4 (mg/L)	2/2	.29	.1–.48			
pH (s.u.)	8/8	7.865	7.28–9.05	4/4	8.61	8.22–9.32
Radiologic (pCi/L)						
Lead–210 (Filtered)	0/2	.4175	<.75–<.92	0/1	.565	<1.13–<1.13
Radium–226	½	.0775	<.07–.12	1/1	1.55	1.55–1.55
Radium–228	0/2	.3	<.5–<.7	0/1	.35	<.7–<.7

[a]Upstream permanent surface water at the southern end of the frog pond; sample location MON–623.
[b]Upstream intermittent pools; sample locations MON–631, –632, and –633.
[c]Frequency of detection (number detected/number analyzed).
[d]Arithmetic mean based on averages from each location; one-half the detection limit used for values below detection.
[e]Minimum and maximum value detected; < indicates value below detection limit.

In some areas downstream of the frog ponds, the area covered by salts is more than 300 ft (100 m) wide. Field observations of these salts indicate that they are very soluble. Wind transport of the salts has been observed, inferring that wind ablation and dissolution of the salts during rains precludes the formation of thick salt deposits in the wash.

Background soils and sediments were collected at one surface location in 1993 and at two hand-auger locations in 1997 (Figure 5–12). The 1993 sample (MON–623) was collected near the southern frog pond, which is located upgradient from the former vicinity property site associated with the northern pond. Samples were collected in 1997 from background locations MON–869 and –870 which were established further upgradient from the site. Three samples were collected from each of these hand-auger locations at depths ranging from 1 to 5 ft below ground level. Lithologic logs and analytical results for these background samples are presented in Section 4.5.

Background concentrations of selected site-related constituents are summarized in Table 5–5 for soil and sediments samples collected at the three upgradient locations. For comparison, concentrations observed in background soils and sediments for the western United States are also presented (Shacklette and Boerngen 1984). Results indicate that average levels of manganese, strontium, uranium, and vanadium at the Monument Valley site are slightly low as compared to average concentrations in typical background soils and sediments for the western United States, possibly reflecting the sandy, well sorted nature of the sediments derived from windblown sands.

Table 5–5. Comparison of Background Concentrations of Selected Constituents in Soil and Sediment Samples at the Monument Valley Site to the Western United States

Constituent	Monument Valley Site[a] (mg/kg)			Western United States (mg/kg)		
	Mean	Standard Deviation	Range	Mean	Standard Deviation	Range
Manganese	123.7	53.3	84.8 – 224.6	380	1.98	30 – 5,000
Nitrate	301.9	314.3	1.8 – 941.3	NA[b] (89)	NA	NA
Strontium	64.1	44.4	25.8 – 116.0	200	2.16	10 – 3,000
Sulfate	477.3	185.8	291.9 – 771.0	NA (780)	NA	NA
Uranium	0.5	0.2	0.3 – <1.0	2.5	1.45	0.68 – 7.9
Vanadium	8.1	3.9	5.7 – 16.5	70	1.95	7 – 500

[a]Background locations MON–623, –869, and –870.
[b]NA = not analyzed, value in parenthesis indicates average crustal abundance (Mason and Moore 1982).

5.3.1.3 Background Water Quality in the Alluvial Aquifer

Background water quality data for the alluvial aquifer near the former processing site is inferred by examining results of water samples collected from 1985 through 1997 at six upgradient monitor wells MON–400, –402, –403, –404, –602, and –603 and at four upgradient private wells MON–200, –616, –617, and –640). The ten background alluvial well locations are shown in Figure 5–14. Water quality results are presented in Appendix C and summarized in Table 5–6.

Water quality results presented in the Piper diagram (Figure 5–13) indicate that the background alluvial water is either a sodium-carbonate type, sodium-sulfate type, or calcium-magnesium

carbonate type. The areal distribution of the different types of natural background alluvial waters, upgradient of the site, is shown in Figure 5–14. The furthest upgradient waters are characterized by a predominance of the sodium cation, with one type dominated by the carbonate anion and the other type dominated by the sulfate anion. The sodium-carbonate type water is present in samples collected at monitor well locations MON–400, –602, –603, and private wells MON–616 and MON–617. Sodium-sulfate type water occurs at private wells MON–200 and MON–640 with relatively higher concentrations of sodium, chloride, calcium, and magnesium (Appendix C), which reflect the effects of a higher degree of local ET or reflect the effects of being in local contact with relatively higher concentrations of naturally occurring soluble salts in the alluvium.

Calcium-magnesium-carbonate type water occurring at monitor wells MON–402, –403, and –404, which are located just upgradient of the frog ponds (Figure 5–14), is geochemically similar to water from the De Chelly aquifer (Figure 5–13). This geochemical signature suggests that the alluvium in this area of the site may be receiving recharge through former uranium exploration boreholes that penetrated the artesian bedrock aquifer in the immediate area. A similar situation is believed responsible for the calcium-magnesium-carbonate type water in the frog ponds (Section 5.3.1.1).

The alluvial ground water is further characterized by an average sulfate-to-chloride ratio of 4.9 (Table 5–6). TDS concentrations average 627 mg/L and range from 294 to 1,590 mg/L. The highest TDS concentrations are associated with the sodium-sulfate type waters reflecting local ET effects. Nitrate is present at an average concentration of 6.4 mg/L and range from less than 0.04 to 47 mg/L. Commonly detected trace constituents include aluminum, arsenic barium, iron, molybdenum, selenium, strontium, uranium, vanadium, zinc, bromide, fluoride, lead-210, radium-226, radium-228, and thorium-230. On average, the water pH is above neutral (pH 8.0) and the redox condition is oxidizing (oxidation-reduction potential 299 mV).

5.3.1.4 Background Water Quality in the Shinarump Member of the Chinle Formation

Background water quality in the Shinarump Member can be evaluated by examining analytical results of water samples collected from 1985 through 1997 at upgradient monitor wells MON–615 and MON–658. Locations of the background monitor wells are shown in Figure 5–14. Monitor wells MON–601 and MON–610 are also upgradient, however monitor well MON–601 is screened across both the alluvium and Shinarump Member and therefore is not considered representative of background water quality in the Shinarump and monitor well MON–610 is dry. Available water quality results for monitor wells MON–615 and MON–658 are presented in Appendix C and summarized in Table 5–6.

Ground water in the Shinarump aquifer is characterized as a sodium-carbonate type (Figure 5–13). TDS concentrations average 325 mg/L and range from 301 to 370 mg/L, which are lower than observed in the alluvial aquifer (Table 5–6). The average pH of 8.1 is above neutral and the redox condition, on average, is oxidizing (oxidation-reduction potential 126 mV).

Table 5–6. Background Water Quality for the Alluvial, Shinarump, and De Chelly Aquifers

Analyte	Alluvium Aquifer[a]			Shinarump Aquifer[b]			De Chelly Aquifer[c]		
	FOD[d]	Mean[e]	Range[f]	FOD[d]	Mean[e]	Range[f]	FOD[d]	Mean[e]	Range[f]
Major (mg/L)									
Ammonium as NH_4	10/40	0.0941	.0053–.6	2/8	0.0387	.0078–.1	4/29	0.0399	<.004–<.1
Calcium	61/61	33.9081	4.5–77.3	9/9	9.36	2.45–17.5	35/35	18.9542	6.34–28.6
Chloride	53/53	37.8947	9.7–125	8/8	8.936	7–11	30/30	7.4088	3.11–34.7
Magnesium	61/61	48.3448	13–107	9/9	13.965	1.61–22.8	35/35	19.7379	8.37–31.9
Nitrate	49/54	6.4141	<.044–47	5/8	2.622	<.014–8.86	29/32	5.1136	.66–22
Potassium	56/56	1.7278	.5–3.2	9/9	1.9133	.93–2.91	34/34	3.0978	1.82–5.29
Sodium	61/61	118.1984	34.2–299	9/9	91.35	64.1–119	35/35	39.586	7.15–92.1
Sulfate	57/57	185.0842	12.7–668	8/8	66.37	53.1–77	31/31	38.0292	11.6–66.3
Metal (mg/L)									
Aluminum	8/27	0.1847	<.05–.8	4/5	0.35	<.1–.7	8/16	0.3403	<.05–.8
Arsenic	7/41	0.0043	<.001–.01	1/7	0.0048	<.005–<.01	4/25	0.0054	<.0015–.022
Barium	21/38	0.0906	.01–.4	2/6	0.16	<.1–.4	10/23	0.1131	.036–.5
Cadmium	3/39	0.0009	<.00013–<.005	0/7	0.0005	<.001–<.001	2/27	0.0007	<.00013–<.005
Iron (Filtered)	6/31	0.0251	<.01–.18	4/6	0.055	<.03–.13	5/20	0.0475	<.03–.25
Iron (Unfiltered)	10/12	0.1475	<.01–.51	1/1	0.08	.08–.08	3/3	1.5025	.04–2.95
Lead	2/26	0.003	<.0015–<.01	0/5	0.005	<.01–<.01	1/18	.004	<.0015–<.01
Manganese (Filtered)	8/38	0.0312	<.0015–.1	2/6	0.0375	<.01–.08	5/25	0.0255	<.0015–.21
Manganese (Unfiltered)	1/22	0.0043	<.0015–<.01	0/1	0.005	<.01–<.01	2/8	0.1746	<.0015–.42
Mercury	1/10	0.0001	<.0002–<.0002	0/4	0.0001	<.0002–<.0002	0/11	0.0001	<.0002–<.0002
Molybdenum	17/50	0.0135	<.0049–.17	3/7	0.098	<.01–.17	11/30	0.0559	<.0049–.18
Selenium	15/40	0.0086	.00094–.0318	0/7	0.0019	<.001–<.005	9/27	0.0021	.00033–.005
Strontium	47/55	0.6405	.05–1.65	5/8	0.143	.0471–.35	28/35	0.2796	<.1–.68
Uranium	59/66	0.0049	<.001–.021	7/9	0.0042	<.0003–.0099	38/39	0.0036	.0004–.0096
Vanadium	23/63	0.025	<.0019–.7	2/8	0.1122	<.01–.7	9/38	0.1304	<.0019–.8
Zinc (Filtered)	11/26	0.1447	<.001–1.6	4/6	0.0293	<.005–.085	11/21	0.0467	<.001–.725
Zinc (Unfiltered)	12/14	0.0292	.004–.099	1/1	0.101	.101–.101	4/5	0.0248	.004–.05
Other									
Alkalinity as $CaCO_3$	51/51	322.2229	172–539	8/8	197.7667	168–220	31/31	164.7556	77–235
Bromide(mg/L)	2/8	0.6292	<.1–<2	0/1	1	<2–<2	0/5	0.7889	<.1–<2
Fluoride (mg/L)	14/15	0.435	.1–1	4/4	0.4333	.3–.5	13/13	0.3595	.2–.6
Redox Potential (mV)	9/9	299.2	96–462	3/3	126.25	–27–446	13/13	184.1875	108–466
Silica (mg/L)	27/27	25.8921	5–54	5/5	10.025	8–12	20/20	9.7139	5–11.5
Sulfate/Chloride	51/51	4.8551	1.0948–11.9841	8/8	7.6304	5.8127–10.4054	27/27	4.9404	1.7003–15.425
Sulfide (mg/L)	0/14	0.1484	<.01–<1	0/4	0.0425	<.01–<.1	1/12	0.2408	<.01–4.9
Total Dissolved Solids (mg/L)	49/49	627.2409	294–1590	8/8	324.9667	301–370	30/30	232.7137	118–370
Total Phosphorus as PO_4 (mg/L)	9/15	0.1813	<.1–.6	5/5	0.3875	.2–1	2/14	0.1143	<.1–.3
pH (s.u.)	53/53	8.0179	6.9–9.04	8/8	8.0693	7.44–8.57	32/32	8.0523	7.37–9.36
Radiologic (pCi/L)									
Lead–210 (Filtered)	11/13	0.9218	0–5.8	5/5	0.7992	.1–1.3	10/14	0.4425	0–1.2
Lead–210 (Unfiltered)	4/6	2.276	<.77–5.7				3/3	5.3875	1.35–7.2
Radium–226	36/38	0.6084	0–5.7	6/6	1.0163	.2–2.17	23/25	0.2865	0–2.4
Radium–228	32/37	0.8815	0–7.7	4/6	0.375	.2–1.6	19/25	0.7	0–4.9
Thorium–230	12/12	0.8896	0–6.3	3/3	0.1	0–.3	9/9	0.2125	0–1.1

[a]Upgradient alluvial monitor wells MON–400, –402, –403, –404, –602, –603, and upgradient alluvial private wells MON–200, –616, –617, and –640.
[b]Upgradient Shinarump monitor wells MON–615 and –658.
[c]Upgradient De Chelly monitor wells MON–611, –612, –613, and –661.
[d] Frequency of detection; number of samples above detection limit/number of samples analyzed.
[e]Arithmetic mean based on averages from each location; one–half the detection limit used for values below detection.
[f]Minimum to maximum value detected: < indicates value below detection limit.

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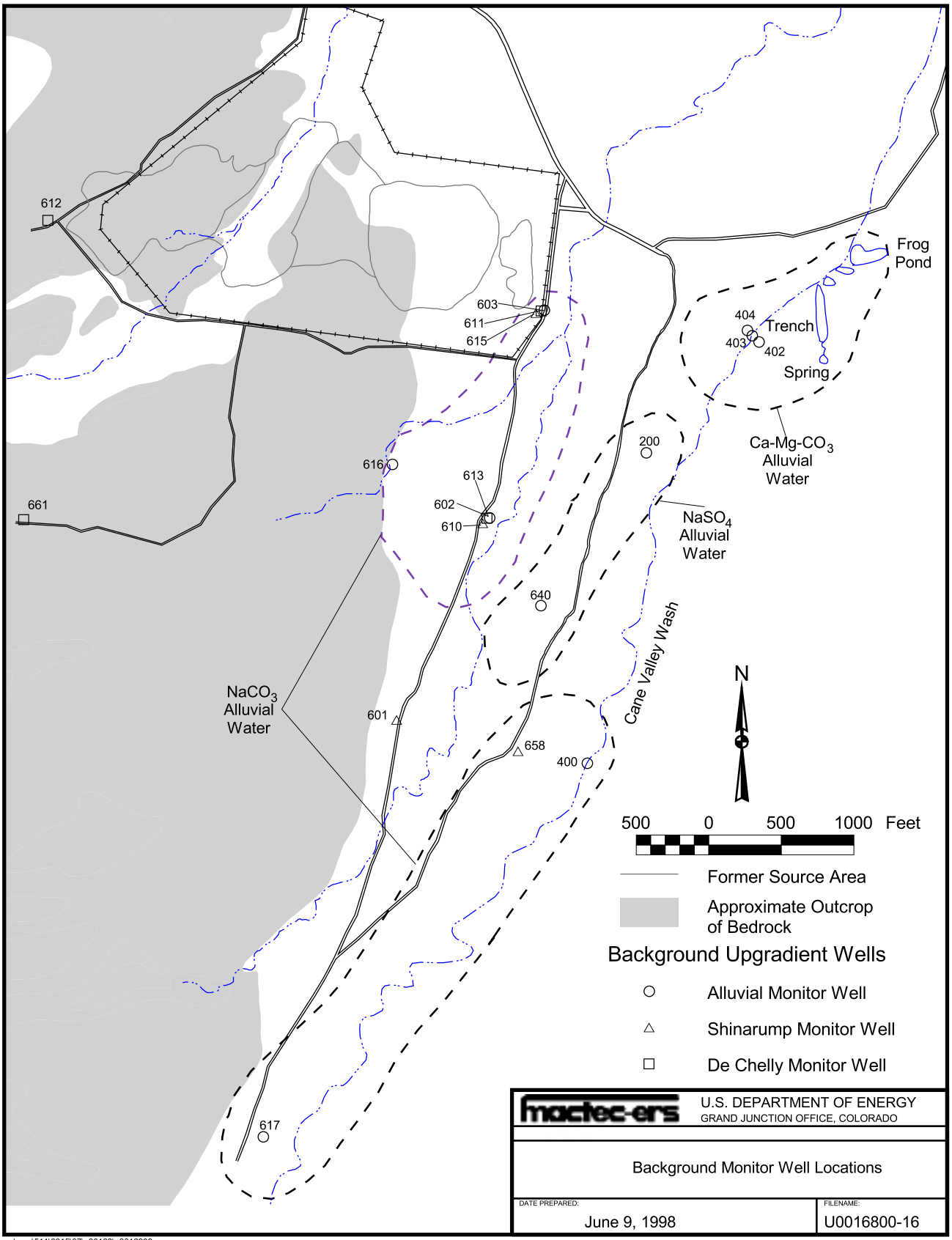

Figure 5-14. Background Monitor Well Locations

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Commonly detected trace constituents include aluminum, barium, iron, manganese, molybdenum, strontium, uranium, vanadium, zinc, fluoride, lead-210, radium-226, radium-228, and thorium-230. Nitrate is present at an average concentration of 2.6 mg/L with a range from less than 0.014 to 8.9 mg/L. The average sulfate-to-chloride ratio is 7.6, which is somewhat higher than that found in the alluvial and De Chelly background waters, although lower than in the contaminated portions of the alluvial aquifer.

5.3.1.5 Background Water Quality in the De Chelly Sandstone Member of the Cutler Formation

Background water quality for the De Chelly Sandstone aquifer can be evaluated by examining analytical results from water samples collected from 1985 through 1997 at upgradient monitor well locations MON–611, –613, and –661. The four upgradient De Chelly monitor well locations are shown in Figure 5–14. Water quality results are presented in Appendix C and summarized in Table 5–6.

Ground water in the De Chelly aquifer can be characterized as a calcium-magnesium-carbonate type (Figure 5–13). The water is chemically similar to that in portions of the alluvial and Shinarump aquifers, but has somewhat less sodium and is more dilute. TDS concentrations average 233 mg/L and range from 118 to 370 mg/L, which is lower than observed in the alluvial and Shinarump aquifers. On average, the water pH is above neutral (pH 8.1) and the redox condition is oxidizing (oxidation-reduction potential 184 mV).

Commonly detected trace constituents include aluminum, arsenic, barium, iron, manganese, molybdenum, selenium, strontium, uranium, vanadium, zinc, fluoride, lead-210, radium-226, radium-228, and thorium-230. Nitrate is present at an average concentration of 5.1 mg/L with a range from 0.7 to 22.0 mg/L. The average sulfate-to-chloride ratio is 4.9, which is consistent with other background waters.

5.3.2 Source Areas

Manganese, nitrate, strontium, sulfate, uranium, and vanadium were identified in the BLRA (DOE 1996b) as the most significant site-related constituents occurring in the alluvial aquifer. The nature of which these site-related constituents are associated with the former processing operations and the extent to which they are available to be dispersed in ground water downgradient from the site are evaluated in the following sections.

5.3.2.1 Former Tailing Piles and Evaporation Pond

Three former source areas of potential ground water contamination exist at the site: (1) the old tailings pile and heap-leach area, (2) the new tailings pile, and (3) the evaporation pond. The location of these former source areas are shown in (Figure 3–1). The old tailings pile was composed of the sand tailings that were a residual product of the mechanical upgrading of ore. The upgrading process used water that contained a minor amount of flocculents but no other processing chemicals. Thus, tailings solutions in the old pile basically were water-equilibrated to minerals in the ore. Heap leaching of these old tailings occurred in the area where they were stored. Old tailings were placed on the heap-leach pad and sulfuric acid was added to the tailings.

Heap-leach pads were lined to collect the leachate which contained sulfuric acid. By contrast, the new tailings pile contained sand tailings and processing solutions. The processing solutions contained sulfate, nitrate, and ammonium from the processing chemicals. The evaporation pond was probably used to retain seepage from the new tailings pile.

The degree of contamination in the former source areas can be estimated from analyses of pore fluids that were collected prior to the contaminated material being removed by the surface remediation project. Pore-fluid samples were collected at the new tailings pile and the evaporation pond in 1985 and 1986 using suction lysimeters. The lysimeters were installed at different depths at each location. At the new tailings pile the lysimeters were installed at 5, 10, 15, and 20 ft (1.5, 3, 4.5, and 6.1 m) below the tailings surface (lysimeters 814, 815, 816, and 817, respectively). At the evaporation pond area east of the new tailings pile the lysimeters were installed at 15 and 20 ft (4.5 and 6.1 m) below the surface (lysimeters 805 and 804, respectively). Although no pore fluids were sampled from the old tailings pile, water-leaching tests were performed on tailing samples and the results provided an estimate of the pore-fluid composition. Analytical results for selected pore fluids and water-leachate samples are summarized in Table 5–7. Included in Table 5–7 is the range in concentrations of selected constituents in natural background alluvial ground water (Section 5.3.1.3) for comparison to the pore-fluid results.

Results of the leachate analyses presented in Table 5–7 demonstrate that water in contact with the former old tailing pile probably contained primarily calcium and sulfate and several metals and trace constituents including aluminum, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, uranium, and vanadium. These constituents were probably derived from the dissolution of ore-associated minerals in the tailings, including gypsum (calcium sulfate), uranyl vanadates, and minor amounts of copper-bearing minerals (Witkind and Thaden 1963). The dissolution of the mineral gypsum (hydrous calcium sulfate) may explain the predominance of both calcium and sulfate in the old tailings leachates.

Pore fluid from the new tailings pile can be characterized as an ammonium-nitrate and calcium-sulfate solution, reflecting the presence of gypsum in the ores, the dissolution of other calcium-bearing minerals in the ores, and the addition of sulfuric acid and ammonium nitrate to the processing solutions. The new tailings fluids were also acidic as indicated by the relatively low pH (4.3). Chloride levels are notably low in these solutions and about the same as in background ground waters. Metals and trace elements include aluminum, barium, chromium, cobalt, copper, iron, manganese, molybdenum, nickel, strontium, uranium, vanadium, and zinc. These elements were derived from the dissolution of the ores. In general, concentrations of site-related constituents in the pore-fluid solutions associated with the new tailings pile increased with increasing depth in the pile, reflecting seepage of the solutions from the base of the pile, and infiltration of precipitation into the top of the pile.

Solutions beneath the evaporation pond area differ from those in the new tailings pile. Notably, the acidity of the solutions has been reduced by reactions with carbonate minerals in the subsoil. Also, the ammonium and sulfate concentrations decreased while the sodium concentrations increased. These changes reflect reactions of the tailings fluids with the subsoil resulting in a calcium-sodium-nitrate-sulfate solution. As with the tailing pore fluids the chloride

Table 5–7. Chemistry of Tailings Solutions and Leachates

Analyte[a]	Leachate (old tailings pile)[b]	Tailings solution (new tailings pile)[c]	Evaporation Pond Solution[d] (subsoil)	Alluvial Ground Water Background Range
Major				
Ammonia as NH_4		1200	47	.0053–.6
Calcium	1110	626	425	4.5–77.3
Chloride	6.16	20	45	9.7–125
Magnesium	96.5	78.3	135	13–107
Nitrate		530	570	<.044–47
Potassium	48.3	21.9	5.21	.5–3.2
Sodium	96.5	70.7	362	34.2–299
Sulfate	2890	4510	1610	12.7–668
Metal				
Aluminum	11.7	5.5	0.9	<.05–.8
Antimony		<.003	<.003	<.003–<.06
Arsenic	0.308	<.01	<.01	<.001–.01
Barium	0.142	0.3	0.5	.01–.4
Cadmium	0.115	<.001	<.001	<.00013–<.005
Chromium	0.241	0.05	0.03	<.0053–.04
Cobalt		1.41	0.11	<.03–.06
Copper	0.554	0.14	0.04	<.01–.03
Iron	1.21	1.86	0.04	<.01–.18
Lead	0.431	<.01	<.01	<.0015–<.01
Manganese	35	3.94	0.77	<.0015–.1
Mercury		<.0002	<.0002	<.0002–<.0002
Molybdenum	<.0616	0.32	0.24	<.0049–.17
Nickel		1.1	0.15	<.04–.13
Selenium	<.0616	<.005	<.005	.00094–.0318
Silver		<.01	<.01	<.01–<.01
Strontium		0.7	1.6	.05–1.65
Tin		<.005	<.005	<.005–<.01
Uranium	0.739	0.0753	1.08	<.001–.021
Vanadium	6.16	1.08	0.91	<.0019–.7
Zinc		3.86	0.129	<.001–1.6
Other				
Alkalinity as $CaCO_3$		2	44	172–539
Fluoride			1.7	.1–1
Redox Potential	535			96–462
Silicon	6.77			
Total Dissolved Solids		6850	2900	294–1590
pH[e]	6.3	4.3	6.72	6.9–9.04
Radiologic				
Radium–226		13	4.7	0–5.7

[a]Leachate data in mg/kg except for pH (s.u.) and redox potential (mV); tailings and evaporation data in mg/L except for pH (s.u.), redox potential (mV), and radium-226 (pCi/L).
[b]Old tailings pile leachate data are maximum values from location MON51–0504, sampled on 9/1/83.
[c]New tailings pile solution data are maximum values from locations MON01–0814 to –0817 sampled on 10/28/85 and 4/27/86.
[d]Evaporation pond subpile soil solution data are maximum values from locations MON01–0804 to –0805 sampled on 10/28/83 and 4/27/86.
[e]pH data are minimum values.

concentrations are notably low (45 mg/L) and are also within the range observed in natural background alluvial ground water. The metals and trace elements that are present in the tailings solutions are also present in the evaporation pond area.

Overall, the former tailings and evaporation pond solutions contain much greater proportions of ammonium, calcium, nitrate, potassium, and sulfate than are present in background ground waters (Table 5–7). Trace elements including manganese, uranium, and vanadium are also present above background concentrations. Thus, these are the constituents most likely to be present in the subpile soils and dispersed in the ground water downgradient from the former source areas.

5.3.2.2 Subpile Soils

The Monument Valley site had several periods of uranium milling activities. During these activities, mill tailings, heap-leach residues, and various processing chemicals were stored in unlined ponds. Any tailings and residuals in the soils that exceeded 15 pCi/g radium-226 were removed from the site during the surface remediation which was completed in 1994. However, site-related inorganic constituents detected in relatively high concentrations in pore fluid samples collected from the former source areas (Section 5.3.2.1) suggest that some of these constituents may have leached into the soils below the storage ponds and gone undetected during the radiometric assessment for the tailings removal.

Samples of the soils directly beneath the former sources areas were collected and analyzed for manganese, nitrate, strontium, sulfate, uranium, and vanadium, all of which were identified in the BLRA (DOE 1996b) as the most significant site-related constituents occurring in the alluvial aquifer to determine if these areas are likely to be continuing sources of ground water contamination. Ammonium was also analyzed because it is present in ground water and will oxidize to NO_3. Seven on-site locations and two background locations were sampled (Figure 4–9). Selected soil samples were subjected to three sequential leachings in the laboratory (Section 4.5). Each leach represents a scenario that might cause the mobilization of contaminants from soils into the ground water. The first leach is deionized water which represents the effect that relatively clean rain or snow would have as it percolates through the soils. The second leach is uncontaminated ground water representing the effect that a high water table might have if it were to contact contaminated soils. The third leach is 5-percent hydrochloric acid (HCl), which will remove carbonate minerals and iron and manganese oxyhydroxides. These phases are believed to be the main metal and uranium scavengers in the soils. Although it is not likely that water of this acidity would ever contact the soils, the removal of oxyhydroxides might occur if land uses changed significantly (for example, agricultural use could cause changes in redox conditions that would influence mineral dissolution). The 5-percent HCl is considered a worst-case scenario. The residue after 5-percent HCl was completely digested to allow calculation of the total contaminant present in each soil sample.

The leachates were analyzed for the following site-related constituents: ammonium (NH_4), manganese (Mn), nitrate (NO_3), sulfate (SO_4), strontium (Sr), uranium (U), and vanadium (V). Each site-related constituent is discussed separately below.

Manganese

Manganese ground water chemistry is controlled largely by oxidation state. More oxidized conditions lead to the stability of manganese oxyhydroxide solid phases which precipitate as coatings on sand particles. Less oxidized conditions, typically occurring in fetid swampy areas, anaerobic sediments as in marshes, wetlands, and boggy areas as described for much of the site with shallow ground water, will dissolve Mn-oxyhydroxide phases and mobilize Mn.

Little Mn was extracted from any of the subpile soil samples with deionized water (Figure 5–15). Manganese concentrations increased in the soil samples when treated with ground water suggesting that the soils may have oxidized the dissolved Mn to form Mn-oxide precipitates or that Mn was adsorbed (Table 4–5). Manganese was removed from all soils by 5-percent HCl and during total digestion. There are no obvious differences between the amounts of Mn leached from the on-site soils and those of the background areas (Figure 5–15). In fact, the shallowest sample in background boring 869 had the second highest HCl-leachable Mn.

The total amount of Mn in the on-site soils ranged from 32 to 328 milligrams per kilogram (mg/kg) whereas the background soils contained 85 to 225 mg/kg (Table 4–8, Figure 5–15). All of the samples contain significantly less Mn than the earth's crustal average of 950 mg/kg (Table 4–8). It is concluded that concentrations of site-related Mn are not significant in the subpile soils and it is unlikely that there is a contribution to ground water.

Ammonium

Ammonium is typically found as a structural ion in feldspars and as an exchangeable ion in smectitic clays. In clays, it has a preference over most other cations for interlayer exchange sites.

Soil boring 866 from beneath the new tailings pile is anomalous in leachable NH_4, most of which was leached by deionized water (Figure 5–16). The soils at 866 were strongly altered compared to any other borings. The alteration consisted of deep yellow and red-brown coloration probably due to the abundance of iron oxides and oxyhydroxides; clay minerals are also present. This alteration may be an artifact of interaction with mill processing fluids. No contaminants of potential concern (COPCs) were clearly associated with this altered zone.

The extractable NH_4 from boring 866 ranged from 137 to 310 mg/kg whereas the values from background ranged from 7 to 9 mg/kg (Table 4–8, Figure 5–16). These values suggest that a leachable source of ammonium exists beneath the former new tailings pile. This source, however, does not underlie the entire former new tailings pile as indicated by lower levels observed in borings 864 and 865. The evaporation pond may also be a source for some NH_4 as indicated by leachable concentrations up to 31 mg/kg.

In the altered soils of boring 866 the ammonium is probably either adsorbed to oxyhydroxide mineraloids or in ion-exchange sites on clay minerals. The high concentrations (up to 310 mg/kg) in 866 compared to background (9 mg/kg) and average crustal concentrations (less than 26 mg/kg; see Table 4–8), indicate that the NH_4 originated from mill processing fluids. This NH_4 may be oxidizing in the shallow soil environment and contributing NO_3 to ground water.

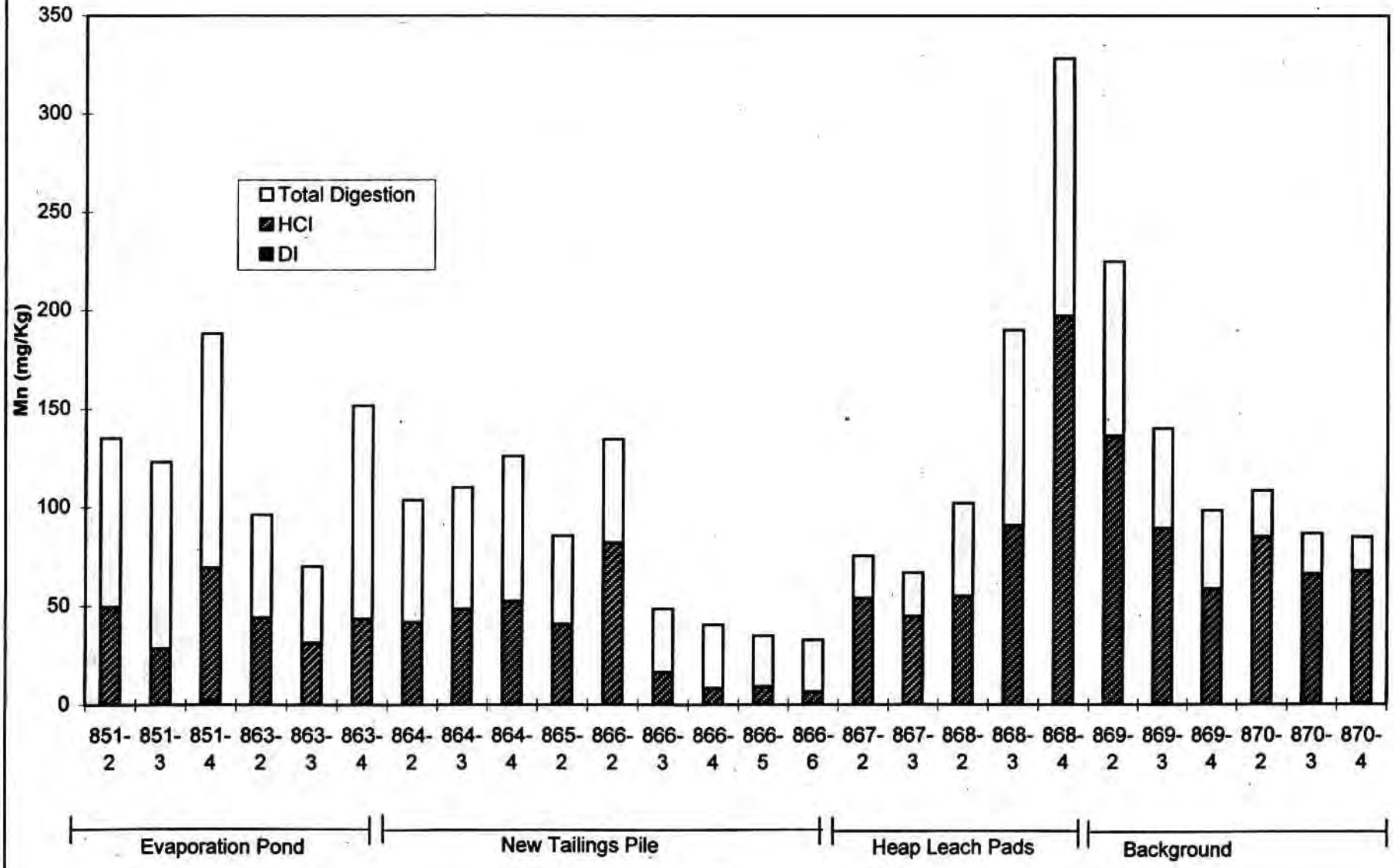

Figure 5–15. Manganese

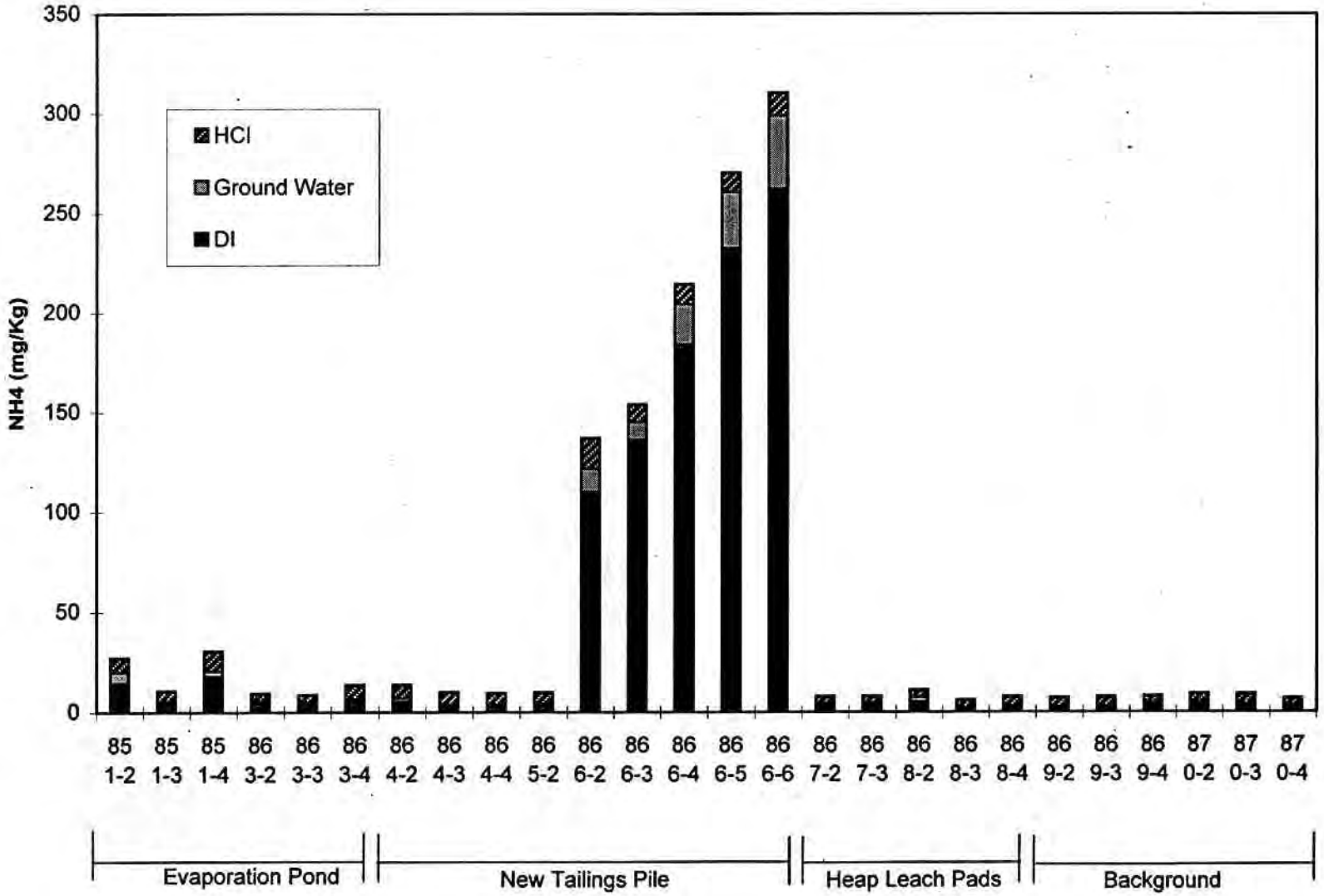

Figure 5–16. Ammonium

During the uranium milling operation, fluids high in NH_4 content may have seeped into some of the soils underlying the tailings. Because of its ability to partition to mineral phase, NH_4 would have built up relatively high soils concentrations as it established an equilibrium distribution with this fluid. As fresh water percolates through the soils, it will leach the NH_4. Concentrations will decline over time.

Nitrate

Nitrate is usually mobile in ground water systems. Nitrate can be produced by oxidation of reduced forms of nitrogen. Nitrate can also be reduced by the action of microbes.

Some nitrate was leached from all soil samples by deionized water (Figure 5–17). No distinct patterns were present. Boring 866, which had high NH_4 concentrations, also had relatively high NO_3 (up to 1,157 mg/kg); however, one of the background samples (869–4) had a similar concentration (941 mg/kg). Most of the NO_3 in 866 and in background sample 869–4 was leachable with deionized water.

Most of the NO_3 concentrations, including those in background samples, exceed the average concentration in the earth's crust (less than 89 mg/kg) suggesting that much of the nitrate is anthropogenic or due to shallow soil microbial processes. The amount of NO_3 leached by deionized water was higher on average in the soils beneath the site than in background soils indicating that some nitrate is probably due to the milling process. The nitrate may be the result of oxidation of NH_4 that has been fixed in cation exchange sites.

Sulfate

Sulfate is usually mobile in ground water systems. With high concentrations of Ca under high evaporation conditions, gypsum ($CaSO_4 \cdot 2H_2O$) can form. Gypsum is soluble and will readily redissolve upon contact with more dilute water.

Up to 9,190 mg/kg of SO_4 was leachable by deionized water from soils beneath the mill site (Figure 5–18). The soils under the evaporation ponds and the new tailings pile have higher concentrations of sulfate than the heap-leach or background areas. There appears to be higher SO_4 near the ground surface as illustrated by the depth profiler for borings 851, 864, and 866 (Figure 5–18).

The sulfate is probably due to the presence of gypsum as indicated by its ability to readily leach in deionized water. No gypsum was identified in thin sections despite an effort to retain gypsum by cutting the sections in oil. The small amount (less than 1 percent) of gypsum could have been missed, however. Alkali salt deposits containing gypsum appear as white crusts and are common in the desert environment near Monument Valley. While there appears to be an increased concentration of SO_4 in the soils near the mill, SO_4 concentrations of this magnitude are probably not uncommon in nearby uncontaminated areas.

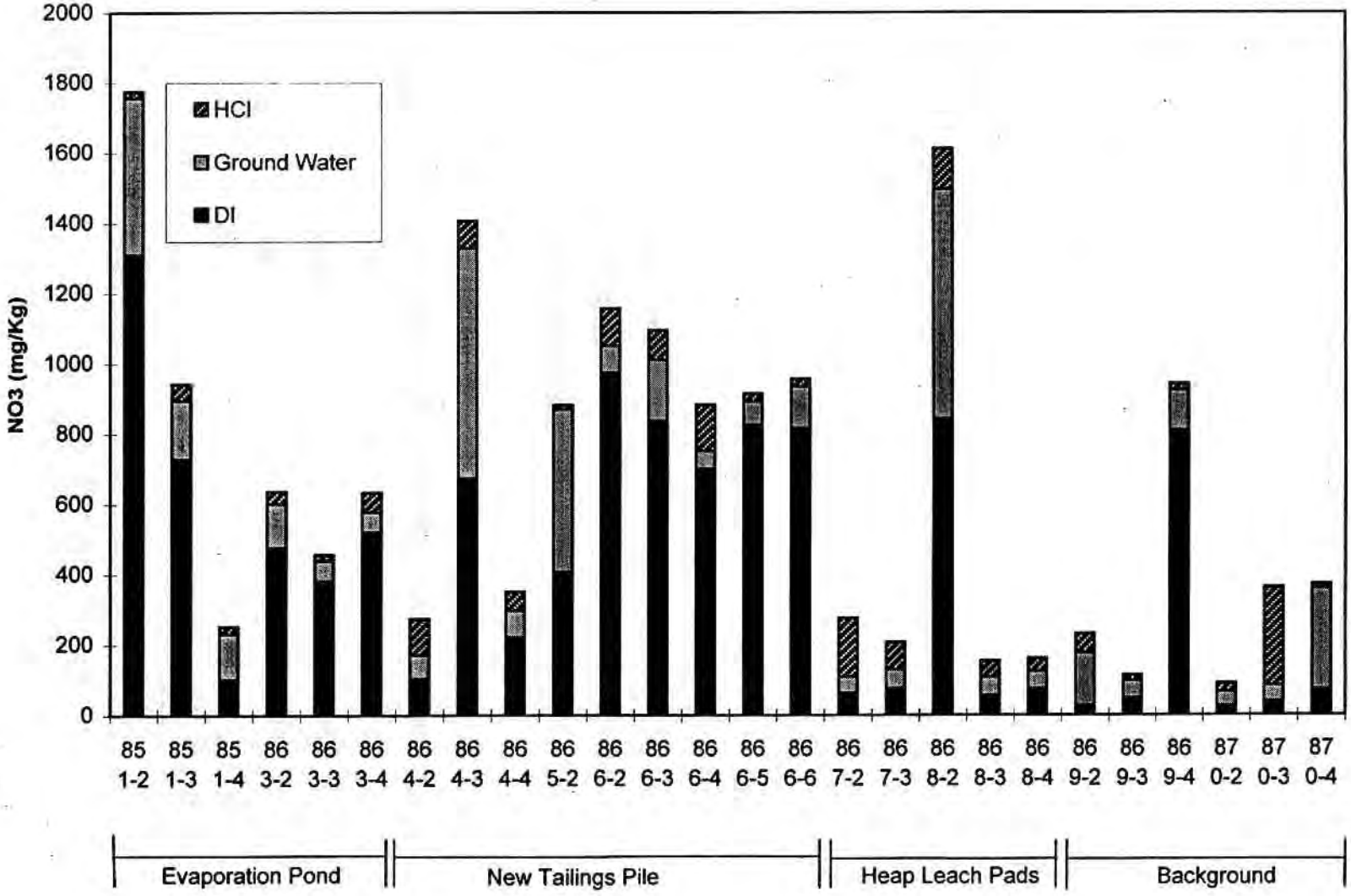

Figure 5–17. Nitrate

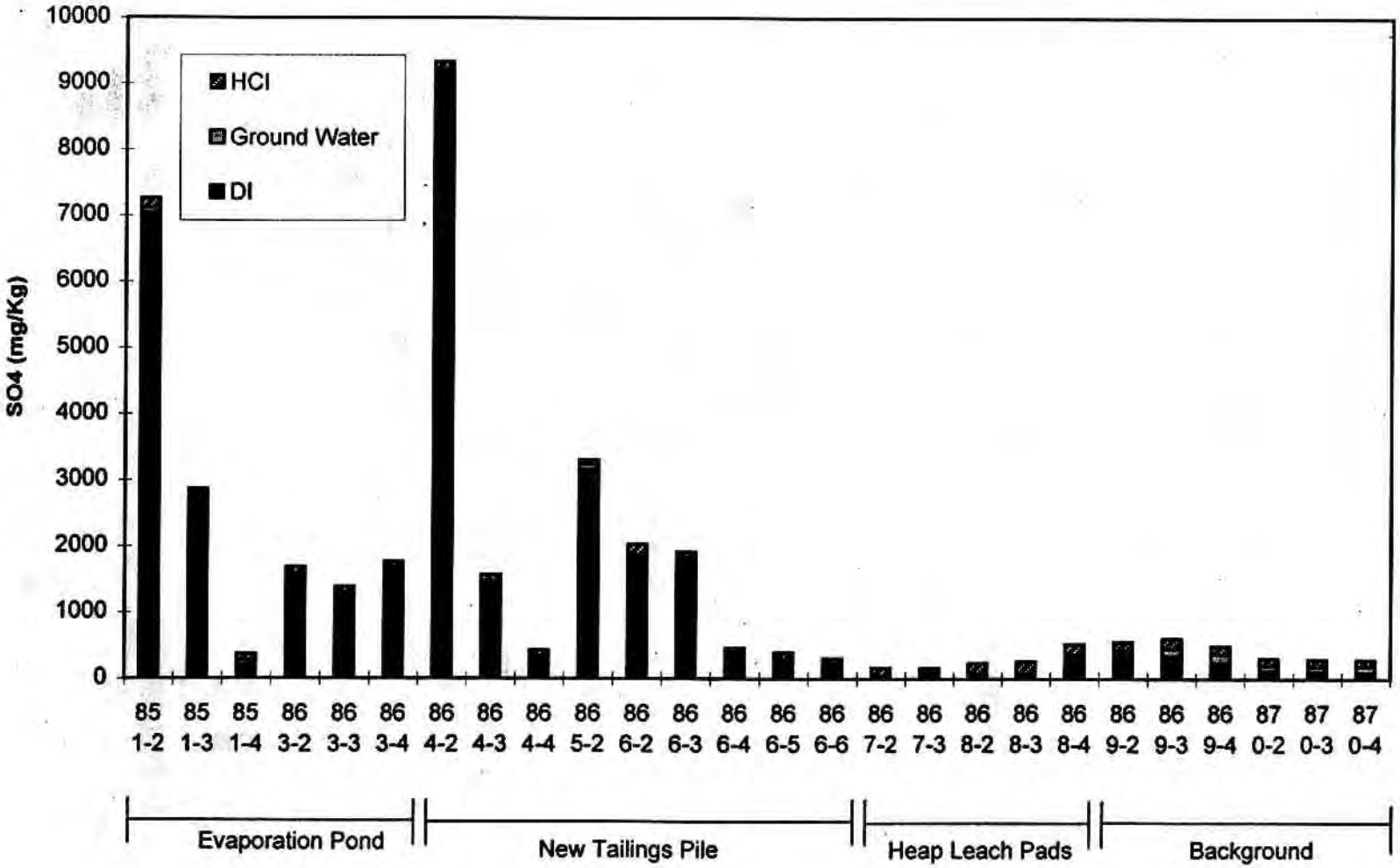

Figure 5–18. Sulfate

Strontium

Strontium is relatively mobile in ground water, however, it will substitute for calcium in carbonate and sulfate minerals. Sr^{+2} is the dominant ionic species present.

Strontium concentrations in samples from the millsite are similar to or lower than those from the background samples (Figure 5–19). The maximum concentration (112 mg/kg) was observed in background soil sample 869 and is lower than the average value for the earth's crust of 375 mg/kg (Table 4–8).

Little Sr was extracted by deionized water or ground water; most of the extracted Sr was during the HCl step. Strontium often substitutes for calcium in calcite which would first dissolve during the HCl leaching. Thus, it is reasonable that the Sr is contained in carbonate minerals.

The occurrence of Sr in a non-water leachable form and at higher concentrations in background than on site indicates that is it unlikely that subpile soils are contributing Sr contamination to ground water.

Uranium

Uranium is mobile in most ground water due to the presence of aqueous carbonate, a strong complexing agent. Uranium often is sequestered by adsorption to Fe oxyhydroxides contained in soils. Under strong reducing conditions it can precipitate as uraninite (UO_2).

No U above the detection limit was extracted by deionized water (Figure 5–20, Table 4–4). The 5-percent HCl leach was the most effective at removing U. Only two samples (851–2 and 868–4) had U concentrations above the average crustal concentration of 1.8 mg/kg (Table 4–8). The extractability of U in the 5-percent HCl leach suggests an association with ferric oxyhydroxides.

With the exception of boring 851, and possibly 868, the millsite samples are comparable in U composition to background samples. While some subpile soil U may be millsite related, as suggested by the elevated concentration in boring 851, the concentrations are not appreciably higher than background. It is unlikely that the subpile soils are contributing significant amounts of U to ground water.

Vanadium

Vanadium is often adsorbed by iron and manganese oxyhydroxides under ground water conditions. It also substitutes for cations in clay minerals and in manganese oxides. Under strongly reducing conditions vanadium minerals will precipitate.

Vanadium concentrations in the subpile soils are elevated over the background samples (Figure 5–21). Two samples from a boring at the evaporation ponds have concentrations of 202 and 142 mg/kg which are slightly higher than the crustal average of 135 mg/kg; all other concentrations are below the crustal average (Table 4–8). Much of the V is leachable in deionized water.

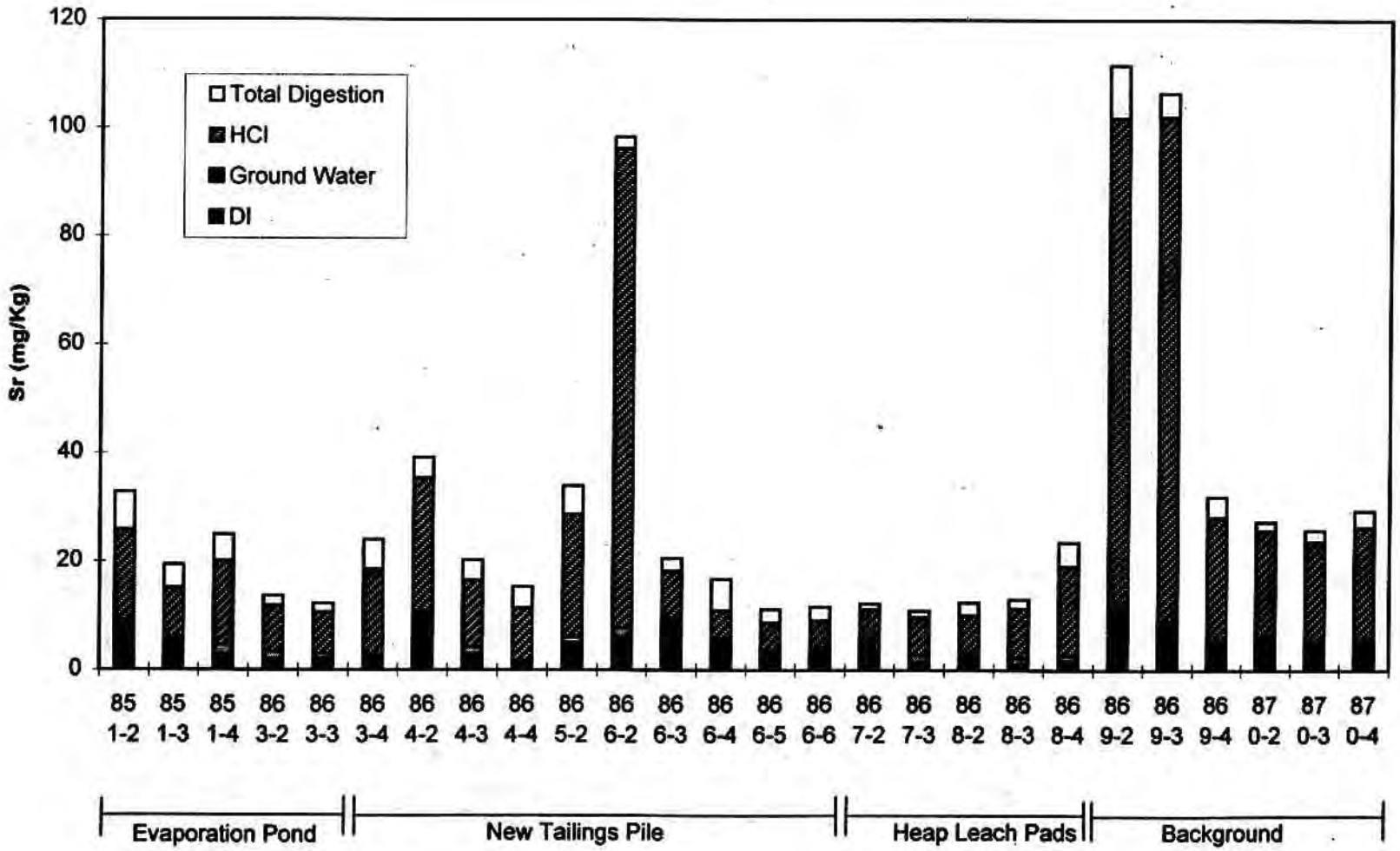

Figure 5–19. Strontium

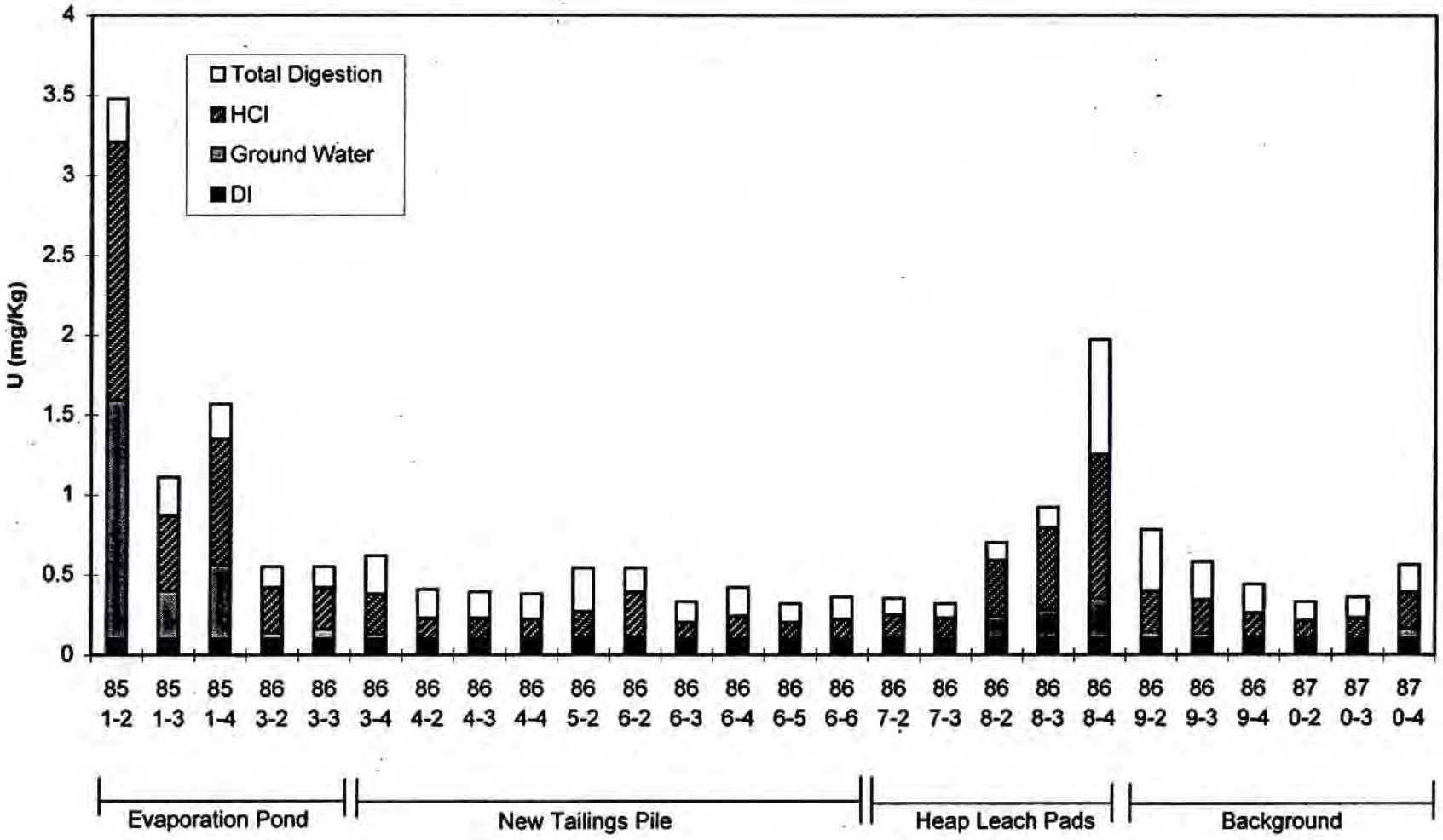

Figure 5–20. Uranium

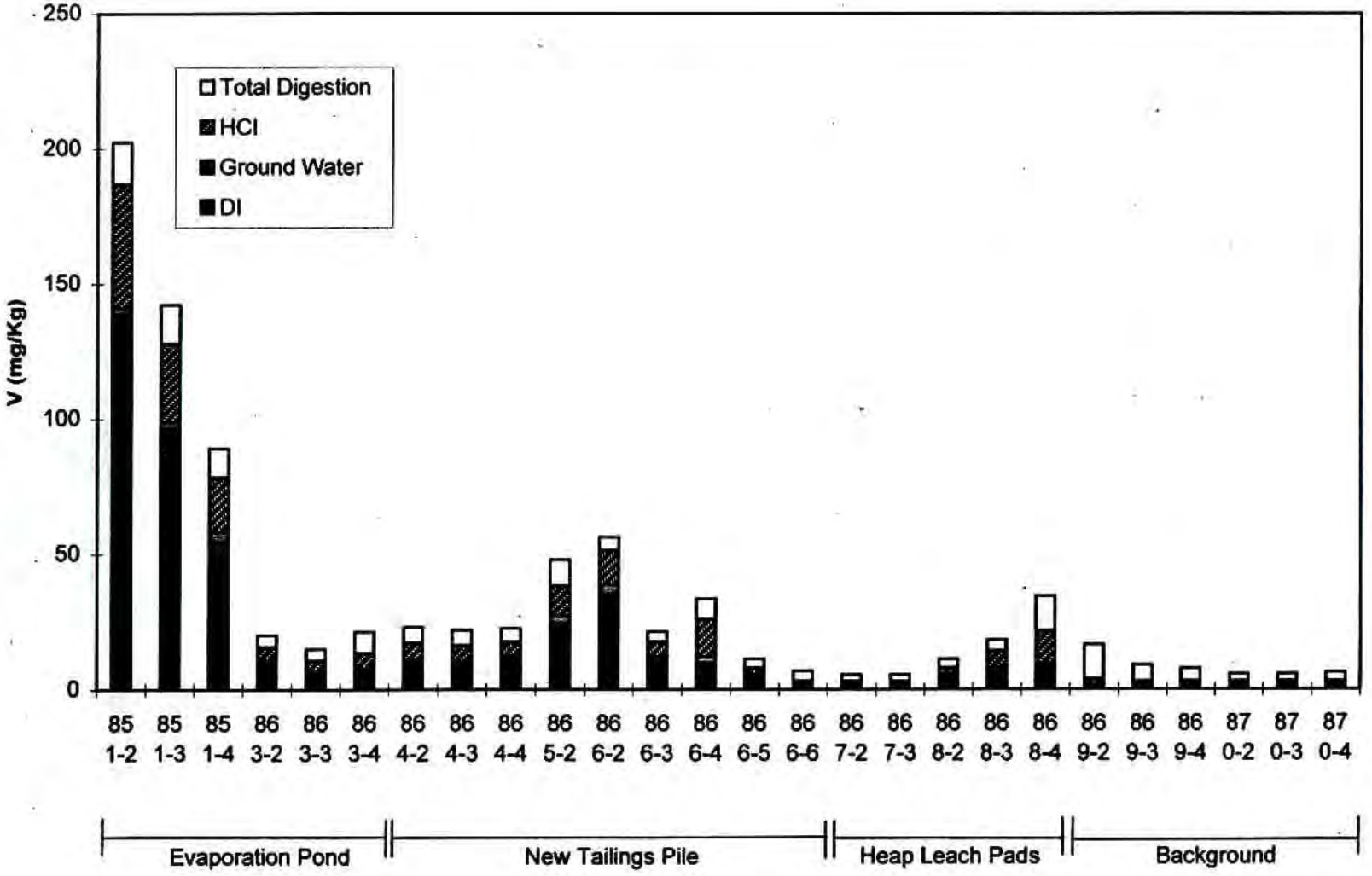

Figure 5–21. Vanadium

Similar to U, boring 851 from the evaporation pond area has elevated V concentrations. Unlike U, however, the V leachable by deionized water in sample 851–2 (127 mg/kg) is over 98 times the highest background value, which is at the detection limit of 13 mg/kg (Table 4–4).

These data suggest that the subpile soils may be contributing some V to the ground water system. Since little V is present in the near field ground water system, the V contribution must either be small or the V is reabsorbing to the aquifer sediments.

Subpile Soil Summary

Leaching experiments on subpile and background soils indicate that Mn, Sr, and U are probably not being leached from subpile soils at concentrations that will contaminate ground water. Sulfate appears to have elevated concentrations in the subpile soils but this observation may result from an inadequate sampling of the background soils. Vanadium concentrations are elevated in the subpile soils but do not appear to be contaminating ground water.

Ammonium is anomalously high over at least a 6.5 ft interval at one location in soils beneath the northern portion of the former new tailings pile. NH_4 does not have an MCL, however, it can oxidize to NO_3, which does have an MCL. NH_4 may have persisted at the millsite due to its strong affinity for ion exchange sites, while NO_3 would have readily flushed out. It is possible that the NH_4-rich soils are generating NO_3 which then enters the ground water system. The sampling was too sparse to determine the lateral extent of the NH_4-rich soils.

5.3.3 Nature and Extent of Contamination

Although some ground water contamination of the relatively soluble components of the ore probably occurred during the mechanical processing period from 1955 to 1964, the majority of ground water contamination probably resulted from discharged process chemicals used during operation of the mill from 1964 to 1967 (Section 3.2).

Ground water contamination from the mechanical processing would have occurred at the former mill and old tailings pile (Figure 3–1). The old tailings pile was composed of the sand tailings which were a residual product of the mechanical upgrading of ore. The upgrading process used water that contained a minor amount of flocculents but no other processing chemicals. Thus, tailings solutions in the old tailings pile basically were water-equilibrated to minerals in the ore. Old tailings were placed on the adjacent heap-leach pad and sulfuric acid added to the tailings. The heap-leach pads were lined to collect the leachate that contained sulfuric acid.

Process chemicals contributing sulfate, nitrate, and ammonium to the ground water contamination would have occurred at the new tailings pile. The new tailings pile contained both sand tailings and processing solutions. Immediately to the east of the new tailings pile was an evaporation pond. The specific purpose of the evaporation pond is unknown, but it may have been used to retain seepage from the new tailings pile (Figure 3–1).

5.3.3.1 Impact to the Alluvial Aquifer

The water quality results observed in 1997 are summarized in Table 5–8 for all on-site and downgradient alluvial monitor wells. The range in natural background for each constituent, based on the upgradient water quality results previously presented in Section 5.3.1.3, is included in Table 5–8 for comparison.

Ammonium, calcium, nitrate, sulfide, and manganese are the site-related constituents most prevalent in the alluvial aquifer as indicated by the relatively high frequency (greater than 50 percent) of samples that exceed the upper range in natural background (Table 5–8). Other site-related constituents are present at concentrations above the upper range in natural background, however they occur less frequently. For example, sulfate exceeds the upper range in natural background in approximately 44 percent of samples, while magnesium exceeds 41 percent, potassium 35 percent, iron 33 percent, uranium 29 percent, strontium 24 percent, and gross alpha 12 percent.

Ammonium and nitrate also provide the greatest contrast to natural background concentrations in the alluvial ground water. That is, the maximum ammonium concentration of 254 mg/L detected in ground water collected at monitor well MON–606 is 423 times the upper range in natural background. The maximum nitrate concentration of 1,030 mg/L, also detected in ground water collected at MON–606, is 22 times the upper range in natural background.

Maximum concentrations of calcium, magnesium, and sulfate occur at moderate levels between 5 to 11 times the upper range in natural background. All of these maximum concentrations were detected in water collected from monitor well MON–771.

Maximum concentrations detected for potassium, strontium, uranium, and gross alpha occur at levels that are less than 5 times the upper range in natural background. The maximum potassium and strontium concentrations were detected in water from monitor well MON–771 while the maximum uranium and gross alpha concentrations were detected in water from monitor well MON–774.

Nitrate and uranium are the only site-related constituents that exceed a MCL. Nitrate frequently exceeds the 44 mg/L MCL while uranium only slightly exceeds the 0.044 mg/L MCL at one isolated location (MON–774).

Alluvial ground water collected from the two most contaminated locations, monitor wells MON–606 and MON–771, is chemically similar to the tailings pore fluids in that the water contains much greater proportions of calcium and sulfate than are present in background alluvial ground water (Figure 5–22). Other chemical similarities to the tailings pore fluids exist. For example, the chloride concentrations in the plume waters are also notably low (5 to 106 mg/L) and consistent with the range observed in natural background alluvial ground water (10 to 125 mg/L). This combination of relatively high sulfate concentrations due to the addition of sulfuric acid to the process solutions, and chloride occurring in concentrations that are consistent with the range in natural background, results in relatively high sulfate-to-chloride ratios for the

Table 5–8. Comparison of On-Site and Downgradient Ground Water to Natural Background Ground Water Quality in the Alluvial Aquifer

Analyte	On-site and Downgradient				Percent Exceeding Upper Range in Background[e]	Background Range[c]
	FOD[a]	Mean[b]	Range[c]	Max Well[d]		
Major (mg/L)						
Ammonium as NH_4	15/17	53.4558	<.007–254	0606	53	.0053–.6
Calcium	17/17	142.5106	9.18–559	0771	59	4.5–77.3
Chloride	17/17	24.55	5.24–106	0768	0	9.7–125
Magnesium	17/17	115.2712	6.31–600	0771	41	13–107
Nitrate	14/17	216.9482	<.014–1030	0606	65	<.044–47
Potassium	17/17	8.4952	.959–50.1	0771	35	.5–3.2
Sodium	17/17	115.2412	21–251	0653	0	34.2–299
Sulfate	16/16	755.2625	26.7–3540	0771	44	12.7–668
Metal (mg/L)						
Aluminum	0/7	0.0536	<.05–<.2	0655	0	<.05–.8
Arsenic	1/13	0.003	<.005–.007	0606	0	<.001–.01
Barium	1/13	0.0692	<.1–.3	0606	0	.01–.4
Cadmium	0/17	0.0005	<.001–<.001	0774	0	<.00013–<.005
Iron (Unfiltered)	5/6	1.8908	<.03–10.2	0606	33	<.01–.51
Manganese (Filtered)	2/7	0.0364	<.01–.19	0606	14	<.0015–.1
Manganese (Unfiltered)	6/7	0.0536	<.01–.21	0606	86	<.0015–<.01
Mercury	0/7	0.0001	<.0002–<.0002	0669	0	<.0002–<.0002
Molybdenum	0/11	0.005	<.01–<.01	0669	0	<.0049–.17
Selenium	10/17	0.0028	.001–.0107	0762	0	.00094–.0318
Strontium	17/17	1.2372	.0979–4.95	0771	24	.05–1.65
Uranium	16/17	0.0165	<.001–.069	0774	29	<.001–.021
Vanadium	4/17	0.0131	<.013–.0542	0772	0	<.0019–.7
Zinc (Filtered)	5/7	0.0314	.006–.07	0650	0	<.001–1.6
Zinc (Unfiltered)	6/6	0.0348	.019–.057	0662	0	.004–.099
Other						
Alkalinity as $CaCO_3$ (mg/L)	17/17	217.2353	161–312	0656	0	172–539
Bromide (mg/L)	3/4	0.35	.2–.5	0655	0	<.1–<2
Fluoride (mg/L)	4/7	0.1929	<.1–.4	0650	0	.1–1
Redox Potential (mV)	17/17	95	–86–219	0669	0	96–462
Silica (mg/L)	8/8	30.675	12.4–131	0662	13	5–54
Sulfate/Chloride	16/16	29.3	3.1–111.7	0771	63	1.1–12
Sulfide (mg/L)	4/7	4.3243	<.1–11.7	0662	57	<.01–<1
Total Dissolved Solids (mg/L)	17/17	1505.7059	232–5800	0771	35	294–1590
Total Phosphorus as PO_4(mg/L)	6/7	0.3071	<.1–.8	0606	14	<.1–.6
pH (s.u.)	17/17	7.5582	6.9–8.37	0650	0	6.9–9.04
Radiologic (pCi/L)						
Gross Alpha	4/17	8.8735	<2.45–27.17	0774	12	0–14
Gross Alpha (Excluding Uranium)	4/17		0–.49	0764		
Gross Beta	7/17	12.4594	<2.69–61.89	0771	18	0–20
Lead–210 (Filtered)	1/17	0.4721	<.63–1.14	0662	0	0–5.8
Radium–226	15/17	0.2685	.11–.49	0767	0	0–5.7
Radium–228	2/17	0.4794	<.3–1.2	0762	0	0–7.7
Thorium–230	7/7	0.3429	0–.7	0655	0	0–6.3

[a]Frequency of detection; number of samples above detection/number of samples analyzed.
[b]Arithmetic mean based on most current sampling; one-half the detection limit used for values below detection.
[c]Minimum to maximum value detected; < indicates value below the detection limit.
[d]Monitor well location containing the maximum observed concentration.
[e]Percent of on-site and downgradient sample results that exceed the upper range in natural background ground water.

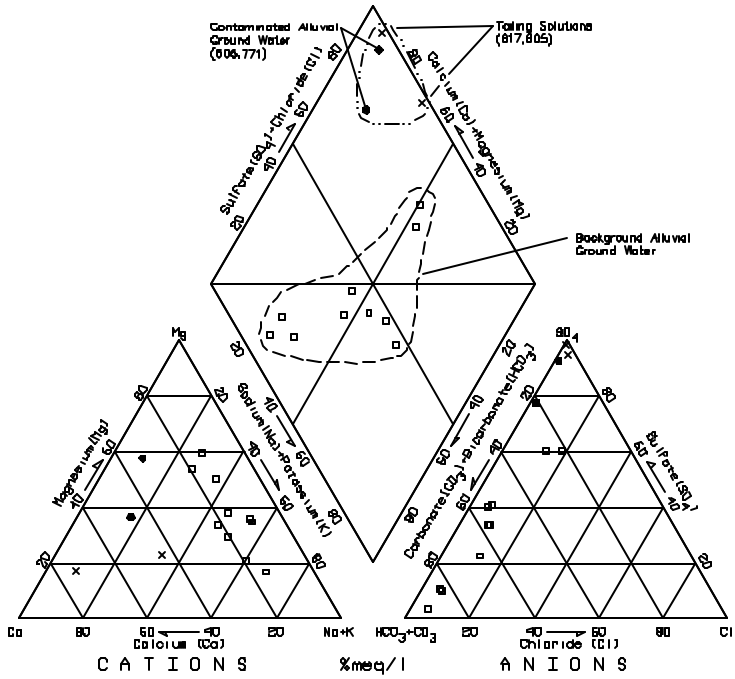

Figure 5–22. Piper Diagram of Contaminated Alluvial Ground-Water Chemistry

portion of the alluvial aquifer that is contaminated. The maximum sulfate-to-chloride ratio of 112 detected in ground water collected at monitor well MON–771 is approximately 9 times the upper range in natural background. TDS concentrations in the alluvial plume average 1,506 mg/L and range from 232 to 5,800 mg/L; values which are higher on average than background waters. On average, the water pH is above neutral (pH 7.6) and the redox condition is oxidizing (oxidation-reduction potential 95 mV).

Areal Extent of Contamination in the Alluvial Aquifer

Nitrate is especially useful as an indicator chemical to discriminate site-related contaminated ground water from alluvial background waters because it occurs in relatively low concentrations in background ground water (Section 5.3.1.3), is associated in relatively high concentrations with the former tailings pore fluids (Table 5–7), and is highly mobile in alluvial ground water under almost all conditions, thus it is a conservative estimate of the extent of site-related contamination. The MCL allowable for nitrate contamination at a DOE facility of 44 mg/L is considered to be representative of the boundary of contamination and is considered sufficient for use in defining the maximum extent of site-related contamination in the alluvial aquifer (DOE 1997c).

The maximum areal extent of contamination in the alluvial aquifer is revealed by examining nitrate results from ground water samples collected using the Hydropunch direct-sampling method during the 1997

ESC field investigation (Section 4–2). The highest nitrate concentrations obtained at locations where multiple Hydropunch samples were collected to vertically profile the plume are presented in Figure 5–23a in order to map the greatest lateral and longitudinal extent of contamination. Nitrate results obtained from water samples collected from several hand-auger borings and from the ground water sampling campaign in August and September 1997 are also presented in Figure 5–23a to obtain the most comprehensive coverage possible.

It is apparent from the 44 mg/L nitrate boundary delineated in Figure 5–23a, that the leading edge of the plume has migrated approximately 4,500 ft (0.85 miles) north of the former mill site. The northerly direction of plume migration is consistent with the direction of the ground water flow in the alluvial aquifer (Figure 5–4). A linear ground water flow velocity of 150 ft/year is estimated assuming nitrate contamination first entered the alluvial aquifer at the start of the 1967 milling operation (4,500 ft/30 years).

A mass of relatively high nitrate is delineated by concentrations greater than 500 mg/L which begins near the former new tailings pile and extends approximately 2,600 ft (0.5 miles) downgradient. Thus, the primary source of nitrate contamination in the alluvial aquifer appears to be related to process fluids draining from the former new tailings pile with lesser amounts of contamination contributed by leakage from the evaporation pond to the east and from the former old tailings pile and heap-leach areas to the west.

The boundary of the nitrate plume as defined by the most recent sampling data collected in August 1998 is shown in Figure 5–23b. The 1998 sampling did not include the temporary locations MON–680 and –678 that were sampled in 1997, therefore, the areal extent of the plume to the northeast does not appear as broad as that shown on Figure 5–23a. In general, nitrate concentrations appear to have decreased since 1997 by more than 150 mg/L near the former source area at wells MON–606 and –792 and increased by more than 10 mg/L at downgradient locations MON–656, –771, –669, and –764.

Sulfate concentrations in the alluvial aquifer exhibit a similar geochemical dispersion pattern as nitrate. The 1997 sulfate plume, revealed in Figure 5–24a by concentrations greater than 600 mg/L, also appears to originate near the downgradient edge of the former new tailings area. The sulfate plume as defined by the sampling results from August 1998 is shown in Figure 5–24b. As with the nitrate plume, the northeast boundary appears less extensive in 1998 because the temporary locations MON–680 and –678 were not available for sampling. Changes in sulfate concentrations also show a trend similar to the nitrate concentrations, with sulfate levels increasing downgradient and decreasing near the former source area.

Uranium, calcium, and strontium also tend to be mobile in the alluvial ground water under the conditions at the site, as indicated by their respective downgradient concentrations presented in Figures 5–25, 5–26, and 5–27, respectively. Similarly, ammonium concentrations (Figure 5–28) exhibit a downgradient dispersion pattern, however the dispersion is less extensive, reflecting the removal of ammonium from solution by adsorption on the aquifer matrix. Distribution of other site-related constituents such as manganese and vanadium presented in Figures 5–29 and 5–30, respectively, do not exhibit a downgradient migration pattern in the alluvial aquifer.

Vertical Extent of Contamination in the Alluvial Aquifer

The vertical extent of contamination in the alluvial aquifer is best visualized by examining the concentration profiles presented in Figures 5–31, 5–32, and 5–33. Nitrate concentrations (Figure 5–23a) measured in ground water samples obtained by the Hydropunch method during the most recent site characterization and analytical results from the 1997 ground water sampling campaign were used to prepare the concentration profiles. Cross-section A to A' (Figure 5–31) starts at the former new tailings area and continues approximately 6,600-ft north to the most downgradient monitor well MON–650. The highest nitrate concentration of 1,030 mg/L occurs in alluvial ground water at monitor well MON–606 located near the former new tailings area. Nitrate concentrations decrease to the 44 mg/L MCL approximately 4,500 ft north (downgradient) near monitor well MON–762, defining the maximum downgradient longitudinal extent of the plume.

Nitrate concentrations tend to gradually increase as a function of depth in the most downgradient area of the plume. This is evident at locations MON–683 and MON–762 where the nitrate concentration of 25 mg/L detected near the top of the aquifer progressively increases first to 38 mg/L near the middle of the aquifer and then to 51 mg/L at the bottom of the aquifer. Conversely, closer to the former new tailings source area the nitrate concentrations tend to gradually decrease as a function of depth in the aquifer. For example, at MON–765 and MON–677 the highest nitrate concentration is 792 mg/L detected near the top of the aquifer. Concentrations progressively decrease first to 726 mg/L and 641 mg/L in water samples collected from the middle of the aquifer and then to 475 mg/L nitrate at the bottom of the aquifer. It is also apparent in the cross-section A to A' that the alluvial ground water from the entire saturated section located between the former new tailings area at MON–606 (approximately 10 ft in thickness) to downgradient monitor well MON–653 (approximately 50 ft in thickness) is contaminated above the 44 mg/L nitrate MCL.

Downgradient lateral dispersion of the nitrate plume to the west of the site is limited by the Shinarump sandstone where the alluvial water intersects the Shinarump in subcrop as shown in cross-section B to B' (Figure 5–32). Along this western edge of the plume, for example at location MON–669, the nitrate concentrations are close to the 44 MCL. Dilution of the plume water from surface recharge along the west margin of Cane Valley, where the eastward dipping Shinarump sandstone crops out, probably contributes to these relatively low concentrations.

The lateral downgradient extent of contamination in the alluvial aquifer to the east of the site is identified in cross-section B to B' by the non-detectable nitrate concentrations observed in water samples collected at locations MON–768 and MON–860. These non-detectable nitrate concentrations provide evidence that the plume does not extend under Cane Valley Wash. A similar relationship is observed near the downgradient leading edge of the plume as shown in cross-section C to C' (Figure 5–33) where non-detectable nitrate concentrations are associated with ground water samples collected at eastern locations MON–760, –698, and –697.

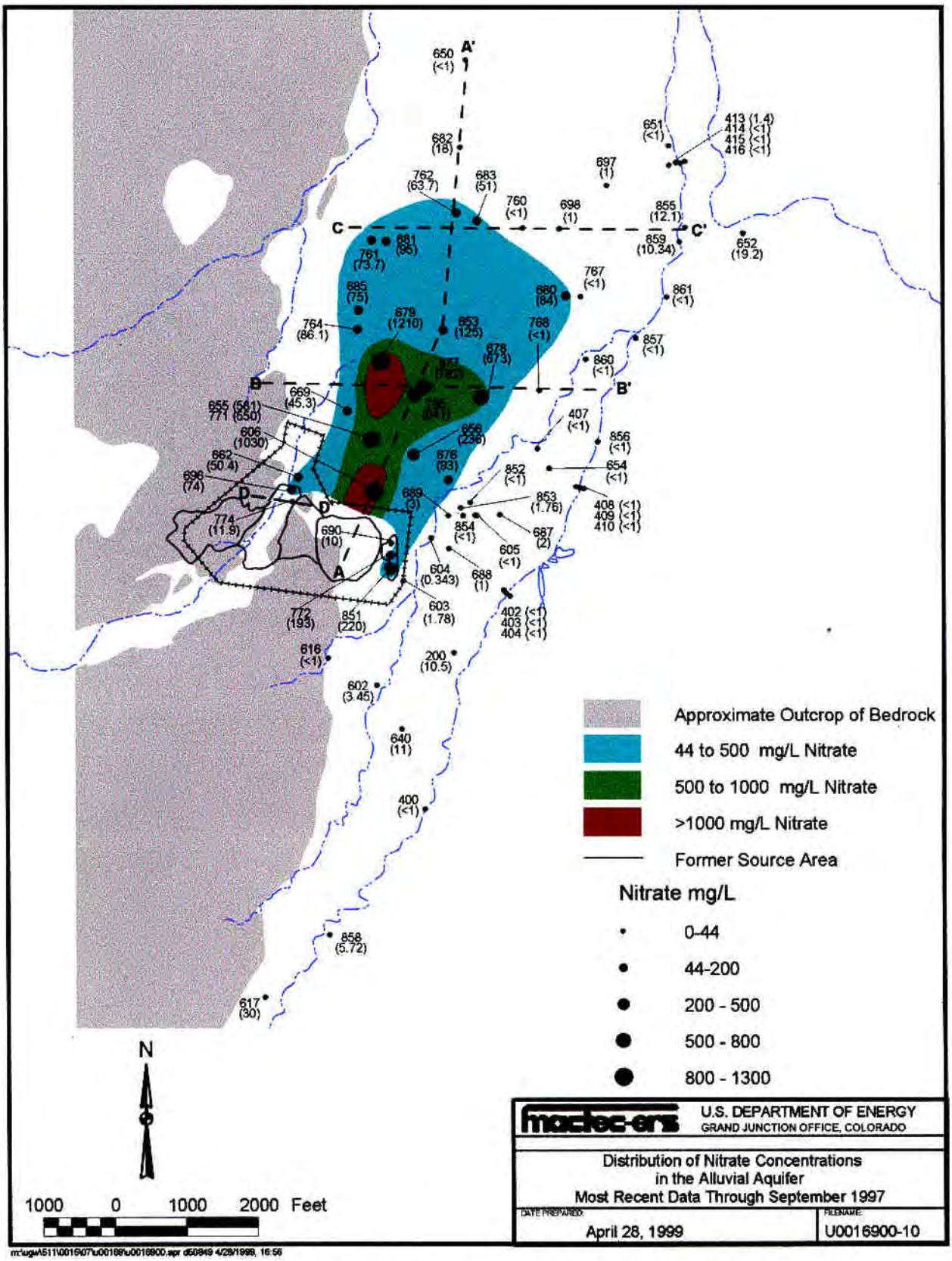

Figure 5–23a. Distribution of Nitrate Concentrations in the Alluvial Aquifer - Data Through September 1997

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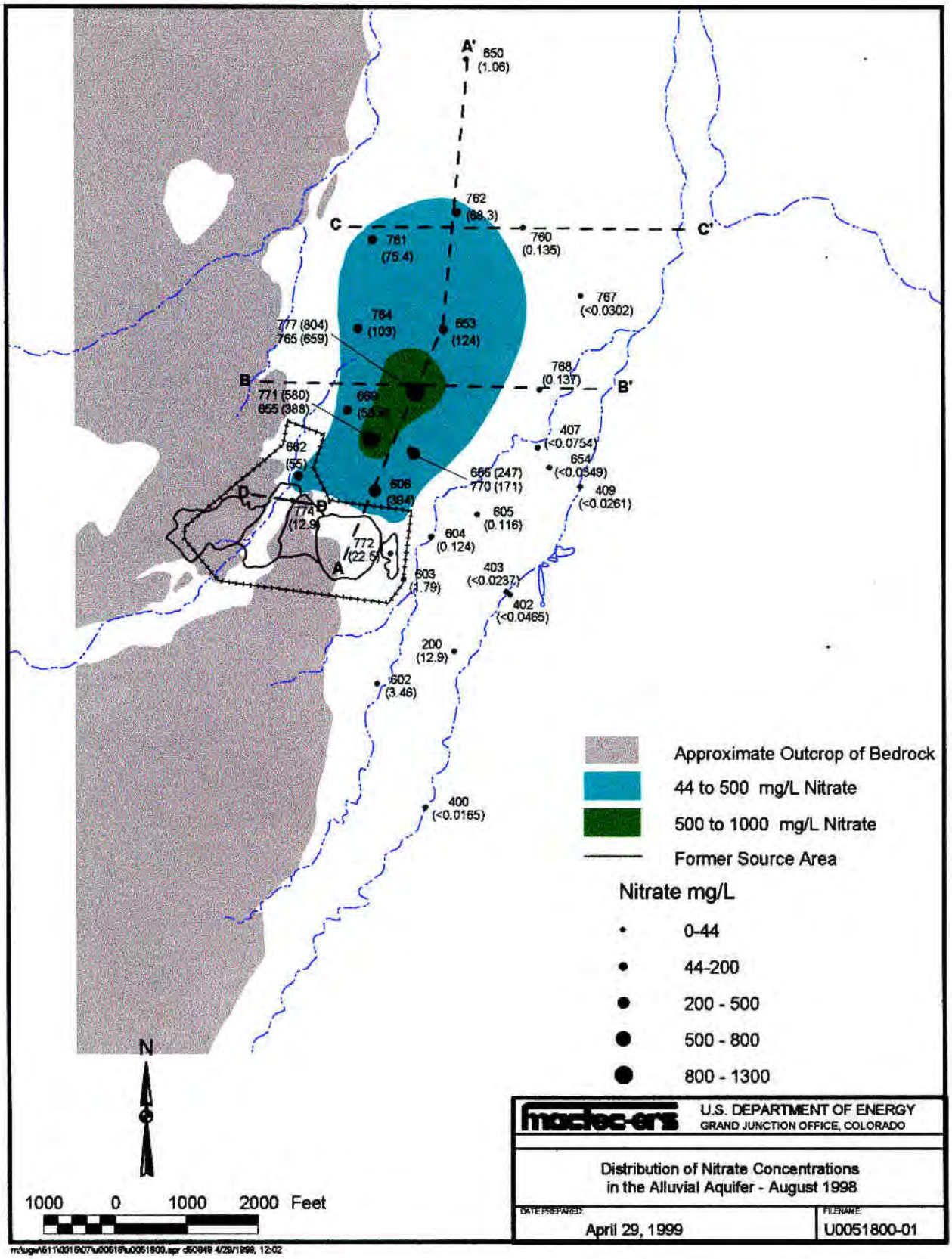

Figure 5–23b. Distribution of Nitrate Concentrations in the Alluvial Aquifer - August 1998

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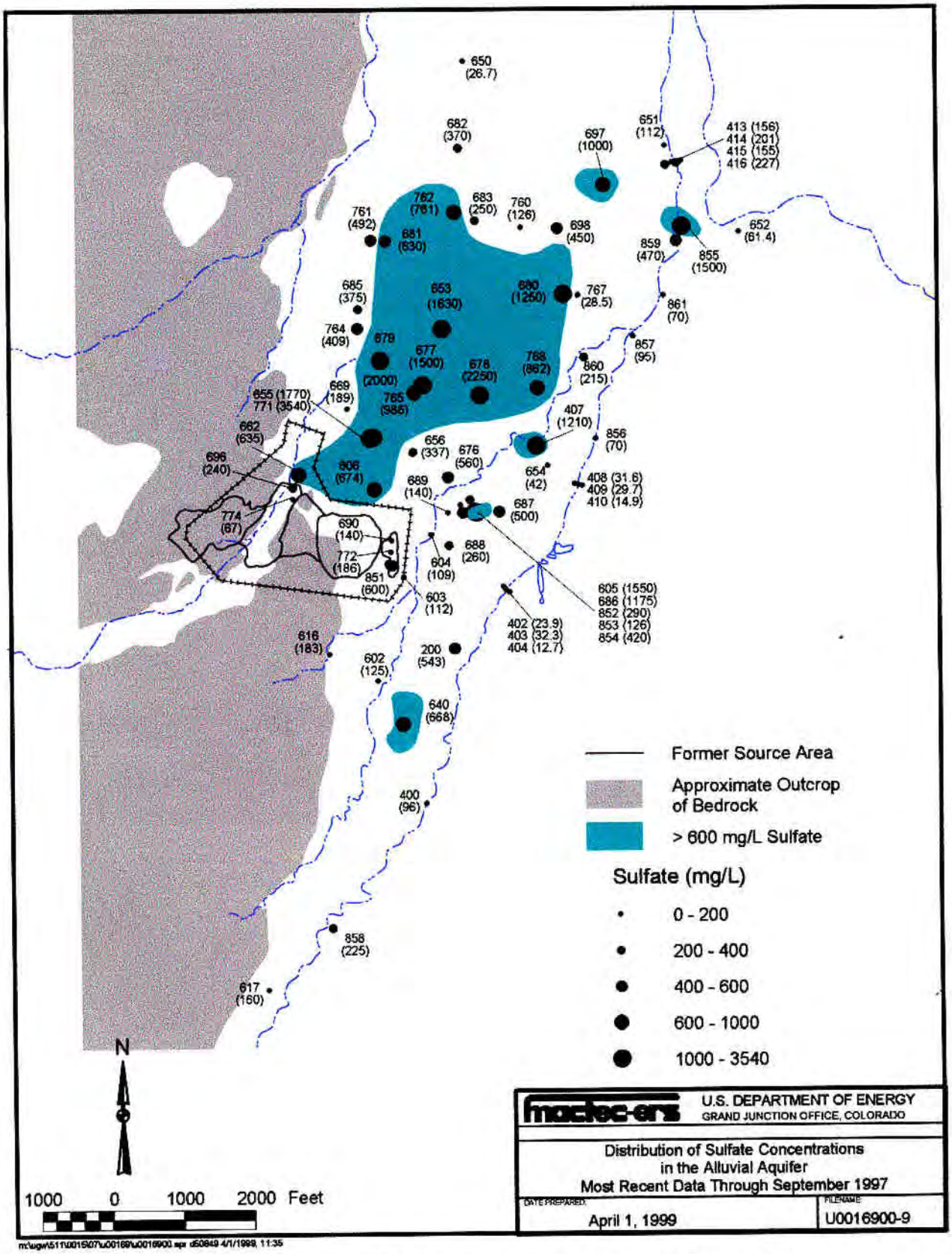

Figure 5–24a. Distribution of Sulfate Concentrations in the Alluvial Aquifer - Data Through September 1997

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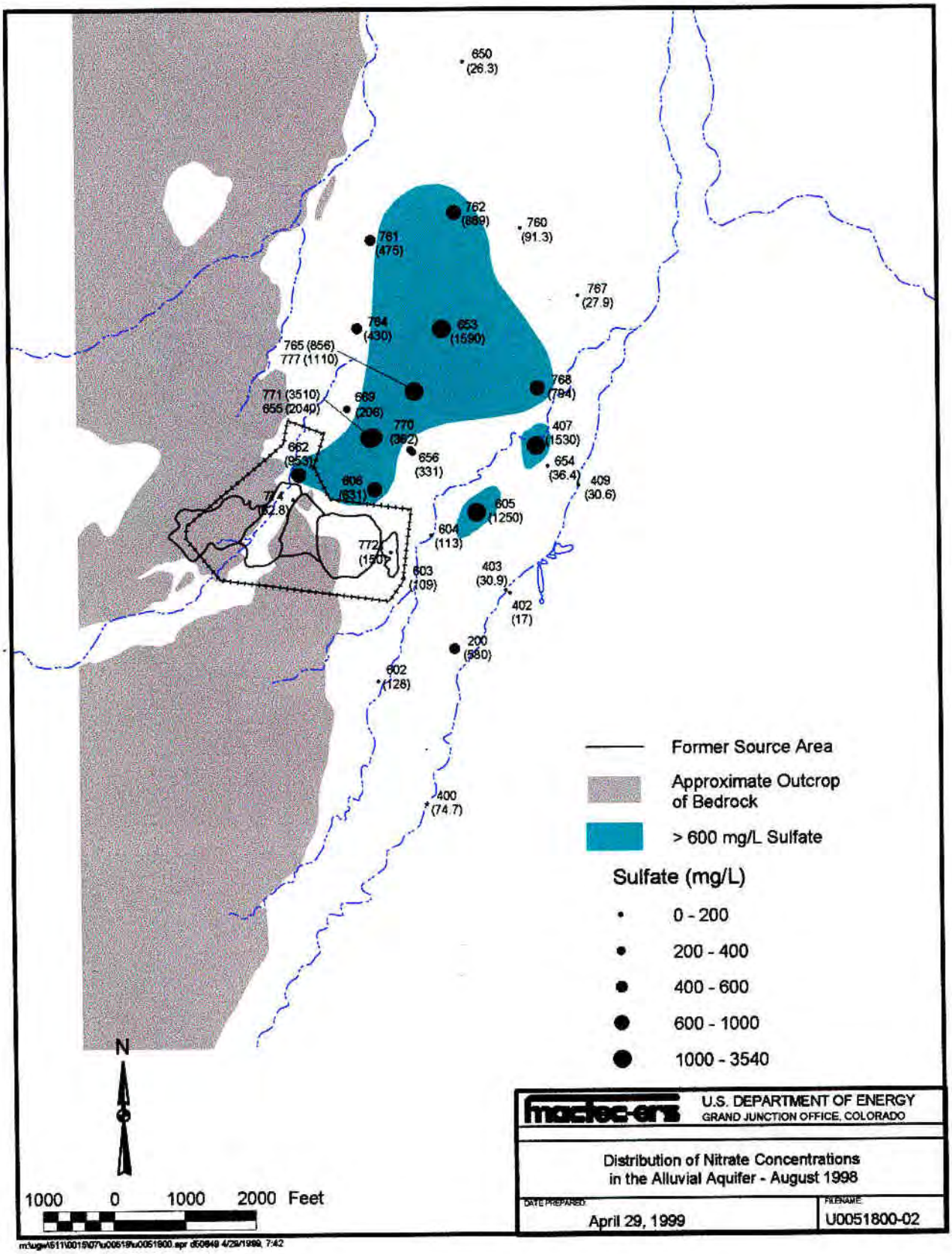

Figure 5–24b. Distribution of Sulfate Concentrations in the Alluvial Aquifer - August 1998

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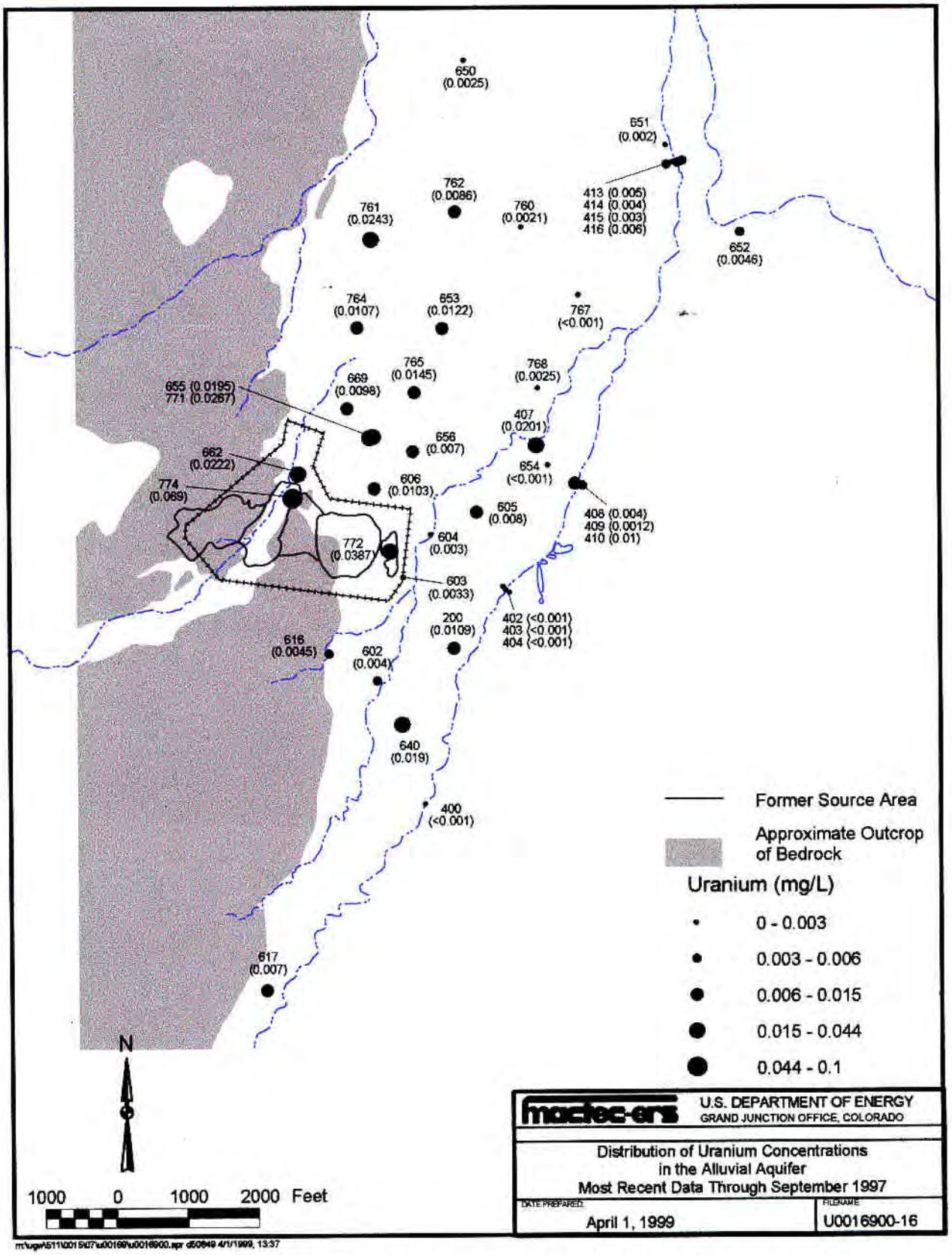

Figure 5–25. Distribution of Uranium Concentrations in the Alluvial Aquifer - Data through September 1997

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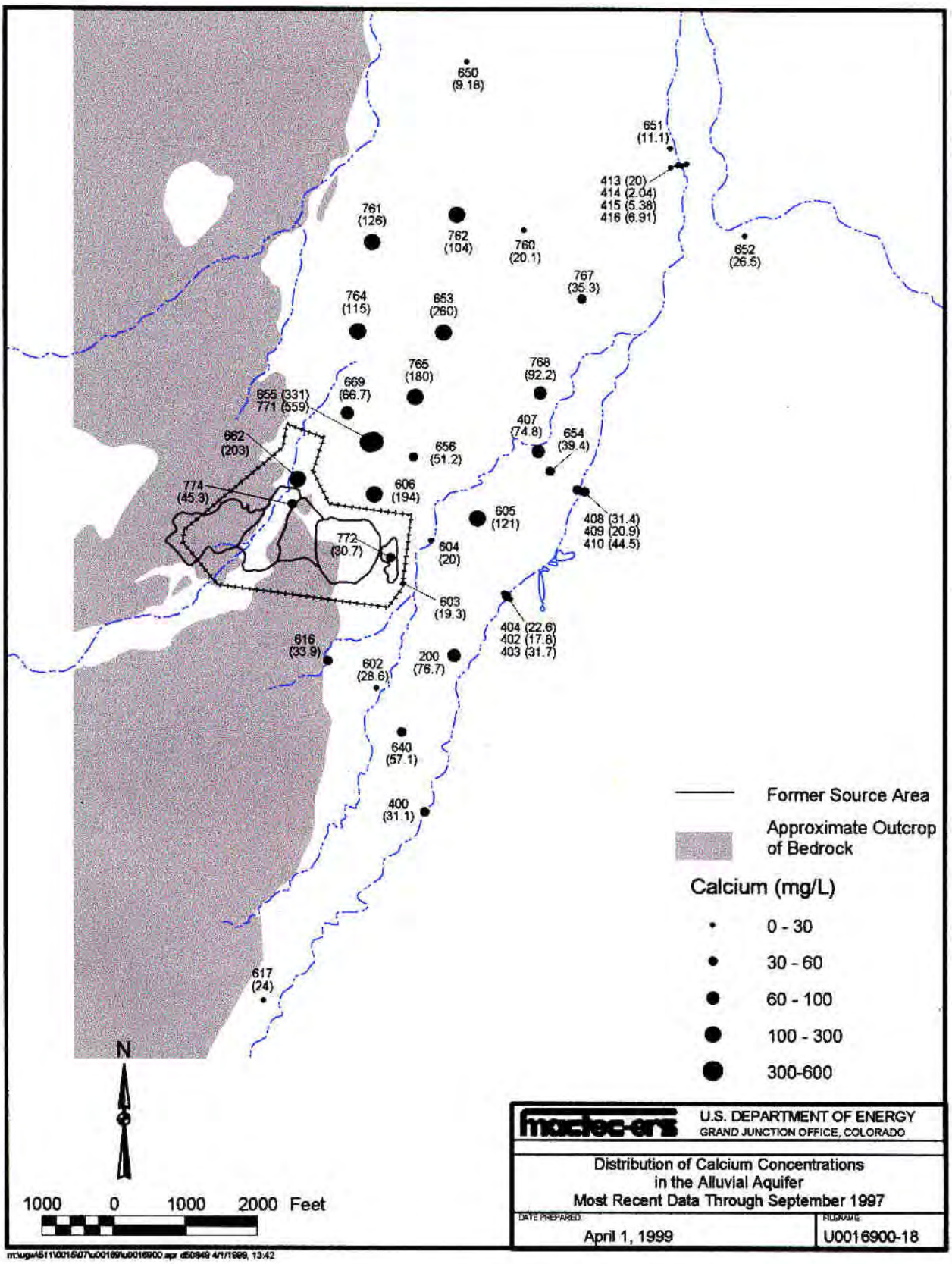

Figure 5–26. Distribution of Calcium Concentrations in the Alluvial Aquifer - Data through September 1997

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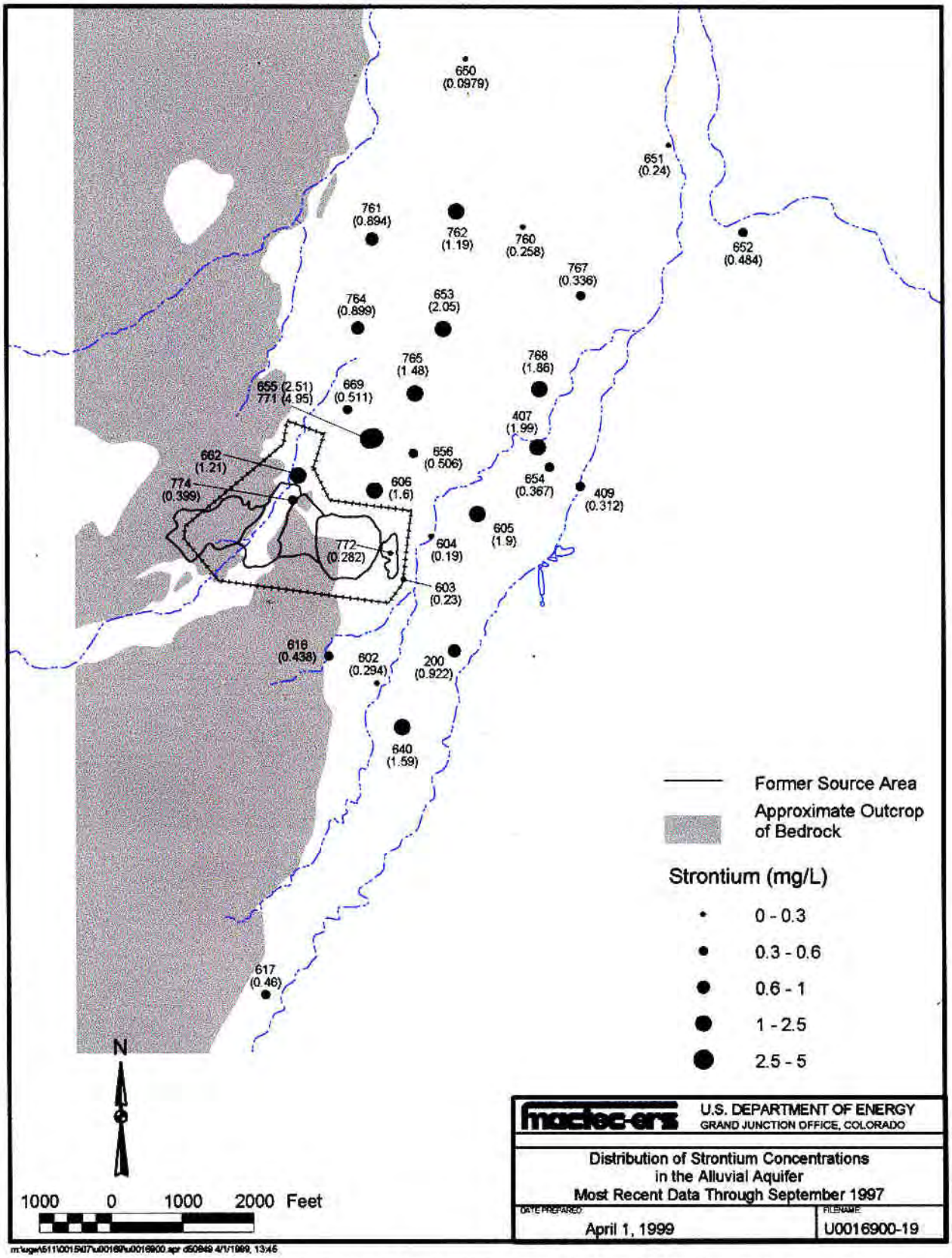

Figure 5–27. Distribution of Strontium Concentrations in the Alluvial Aquifer - Data through September 1997

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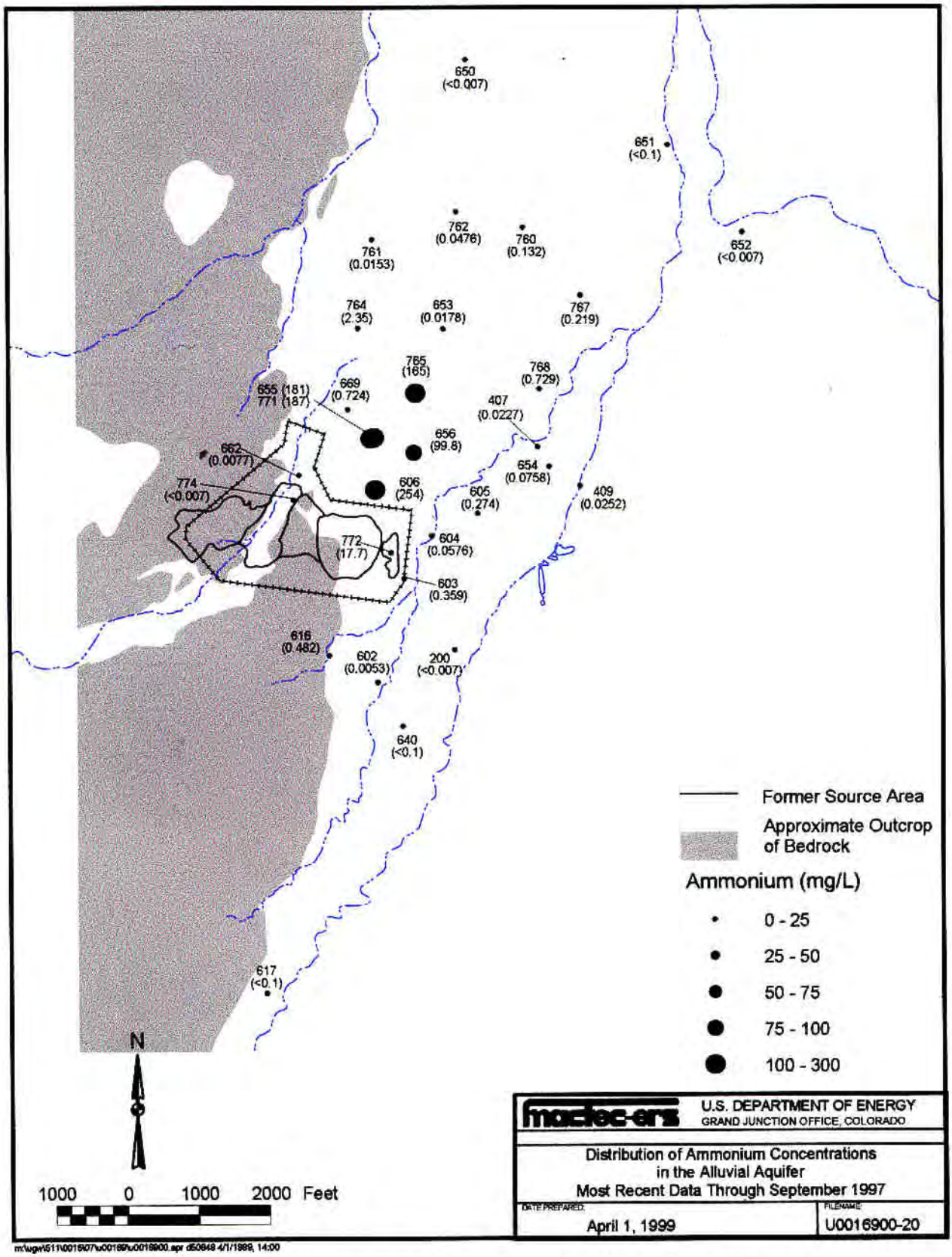

Figure 5–28. Distribution of Ammonium Concentrations in the Alluvial Aquifer - Data through September 1997

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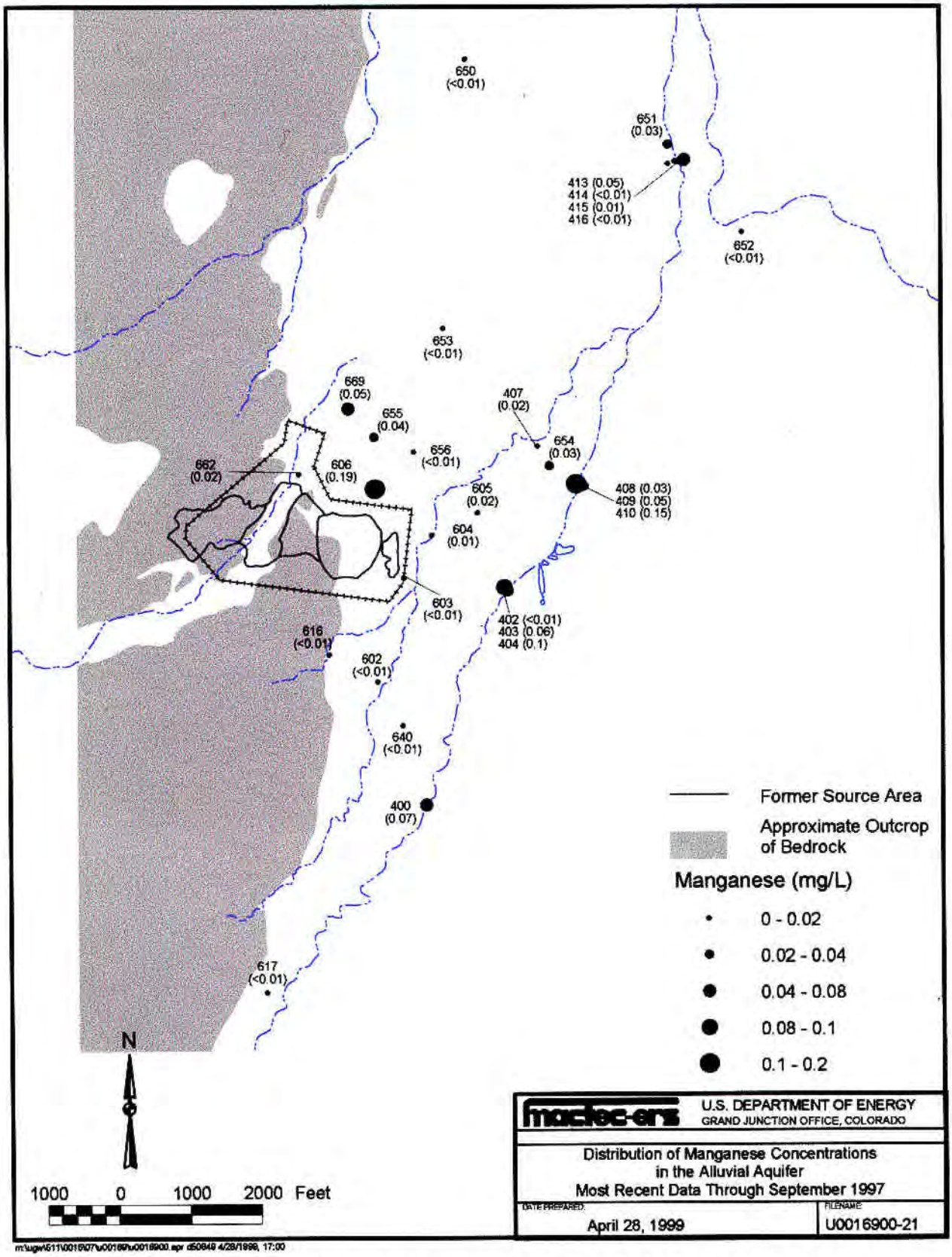

Figure 5–29. Distribution of Manganese Concentrations in the Alluvial Aquifer - Data through September 1997

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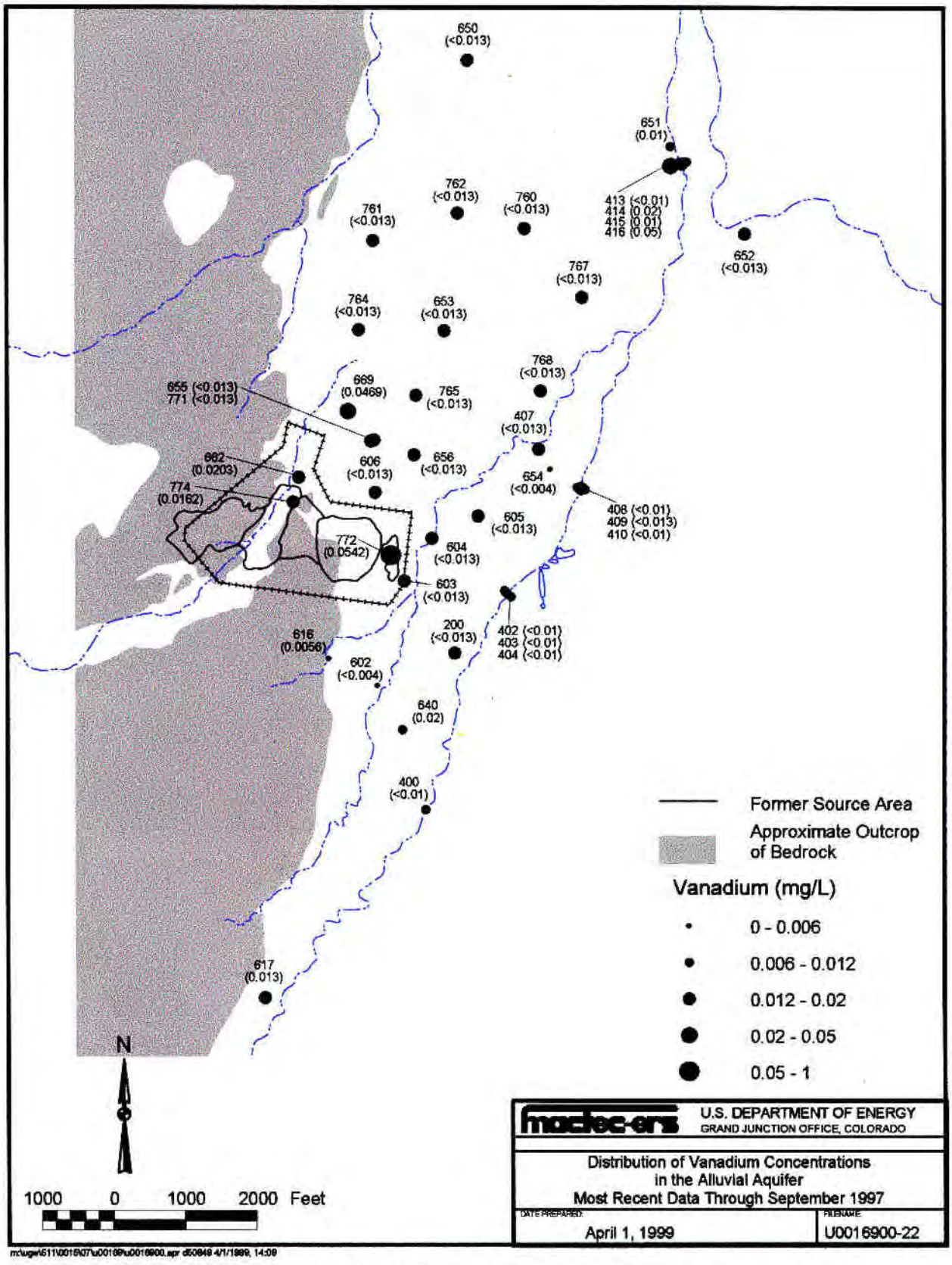

Figure 5–30. Distribution of Vanadium Concentrations in the Alluvial Aquifer - Data through September 1997

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Figure 5-31 will be provided upon request. Email lm.records@lm.doe.gov to request

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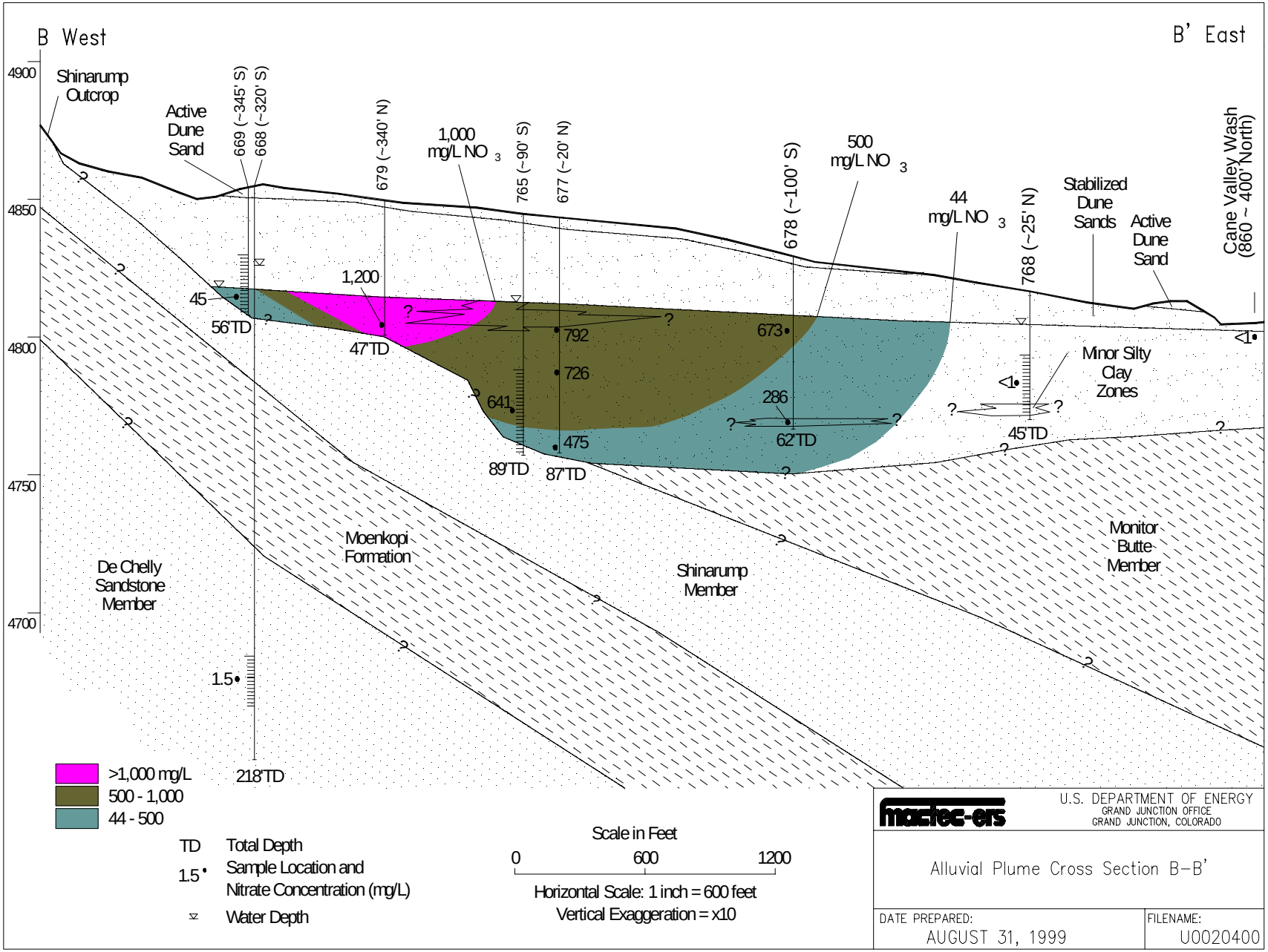

Figure 5-32. Alluvial Plume Cross Section B to B'

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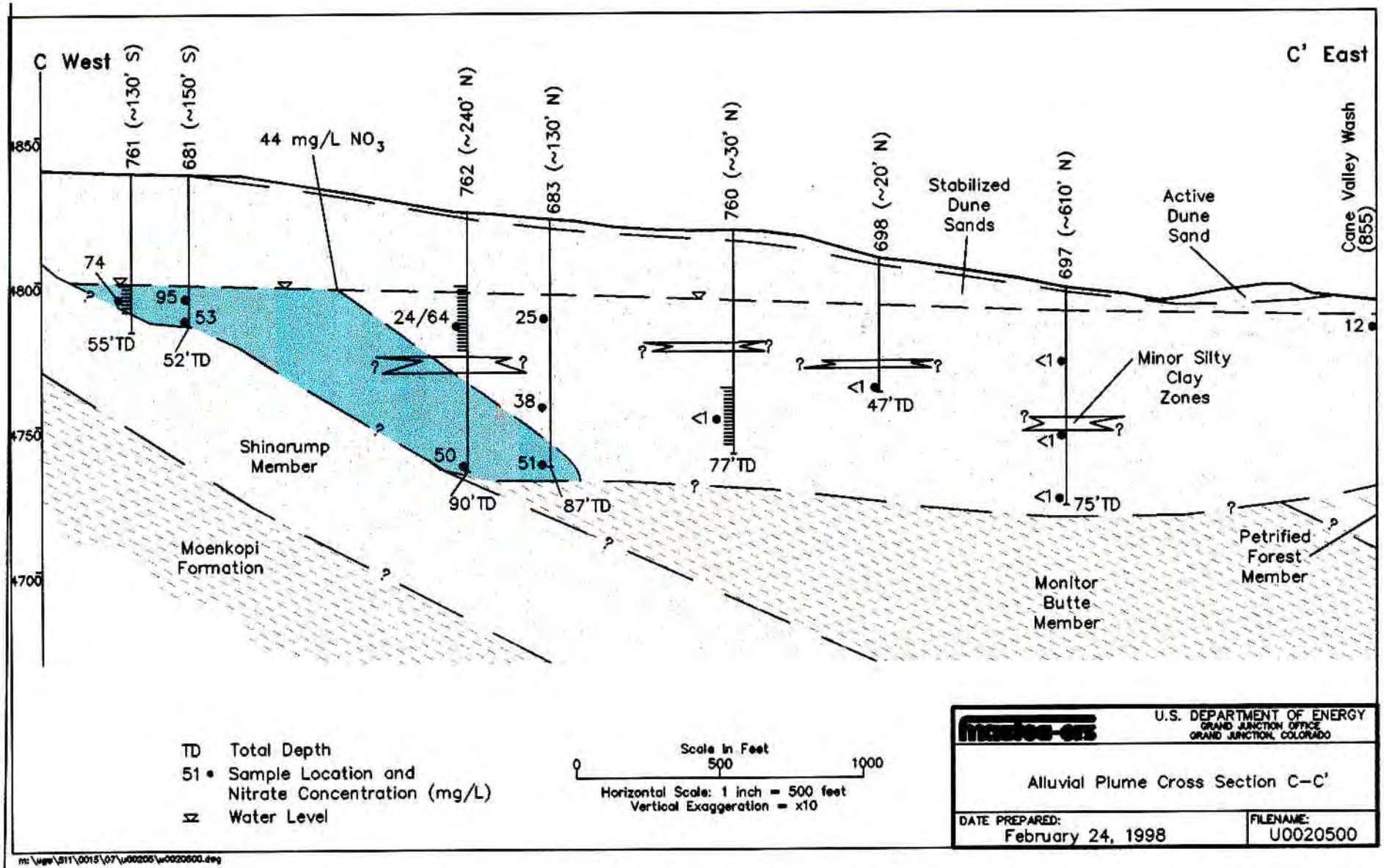

Figure 5–33. Alluvial Plume Cross Section C to C'

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The maximum width of the plume defined by the nitrate concentrations profiled in cross-section B to B' is approximately 2,500 ft, as measured from just west of monitor well MON–669 to the mid-point between MON–678 and MON–768. In addition, the concentration profiles indicate that the saturated alluvium across the entire width of the plume is contaminated above the 44 mg/L nitrate MCL. Because the aquifer matrix consists mostly of stabilized eolian dune sands with only minor amounts of fluvial material, the distribution of the contaminant plume does not appear to be controlled by a specific lithologic horizon (e.g., clay unit). Similarly, a preferential contaminant flow path does not appear to be associated with the buried paleovalley to any great extent. For example, a faint trace of the western edge of the paleovalley is suggested by the shape of the bedrock surface between MON–679 and MON–765 (cross-section B to B'). Erosion of the Shinarump sandstone by the ancestral Cane Valley Wash drainage provides an alternate explanation for this discernable bedrock slope. In any case, the presence of a clearly defined erosional paleovalley controlling contaminant migration is not evident.

Plume Migration Trends

Downgradient migration of nitrate contamination in the alluvial system is evidenced by examining nitrate concentrations versus time for ground water samples collected at selected locations along the longitudinal axis of the plume. Nitrate concentrations, measured over a 10.3-year period (April 1988 to August 1998) in water samples from monitor wells MON–606, –655, and –653, are shown in Figure 5–34. Monitor well MON–606 is located near the downgradient edge of the former new tailings pile, MON–655 is located near the centroid of the nitrate plume, and monitor well MON–653 is located at the leading edge of the 500 mg/L nitrate boundary. Results for monitor well MON–606 located near the former source area, and monitor well MON–655 located near the centroid of the high nitrate concentrations, both indicate a decreasing trend in nitrate concentrations since 1988. Conversely, concentrations at the leading edge of the 500 mg/L nitrate boundary (MON–653) indicate an increase in nitrate concentrations; 31 mg/L in 1988 to 124 mg/L in 1998. This translates to an approximate historical rate of increase of 9 mg/L nitrate per year at location MON–653 ([124 – 31 mg/L] / 10.3 years).

Similarly, the sulfate plume appears to be migrating downgradient of the former new tailings area. This is evidenced by examining sulfate concentrations versus time for ground water samples collected at selected locations along the longitudinal axis of the plume. Sulfate analyses in water samples collected over a 10.3-year period (April 1988 to August 1998) from monitor wells MON–606, –655, and –653, are shown in Figure 5–35. Measurement results for monitor wells MON–606 and MON–655, located closest to the former source area both indicate a decreasing trend in sulfate concentrations since 1988. Conversely, concentrations at the downgradient monitor well MON–653 indicate an increase in sulfate concentrations; 1,060 mg/L in 1988 to 1,590 mg/L in 1998.

Volume of Contaminated Alluvial Ground Water

Estimates of the volume of contaminated ground water in the alluvial plume are based on the areal and vertical distribution of nitrate concentrations discussed previously. Separate estimates are presented for (1) the mass of relatively high nitrate concentrations delineated by the 500 mg/L boundary which begins near the former new tailings pile and extends approximately

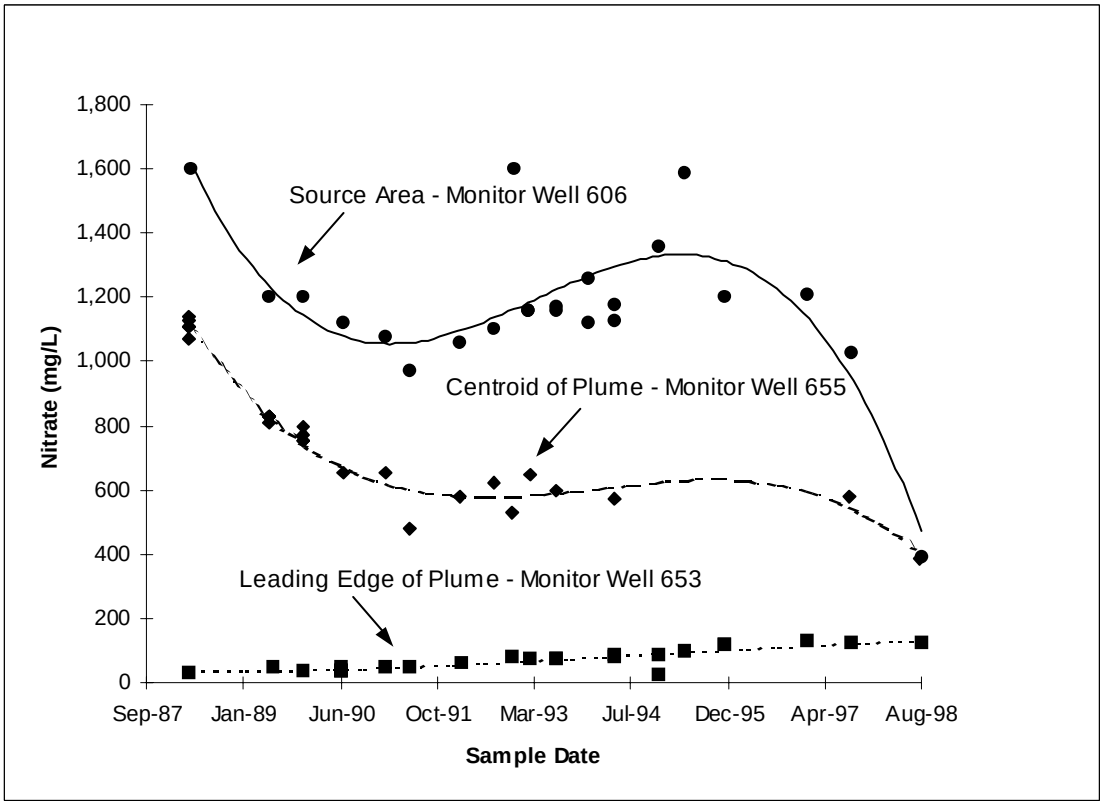

Figure 5-34. Nitrate Concentration Versus Time for Selected Monitor Wells Located Along the Longitudinal Axis of the Alluvial Plume

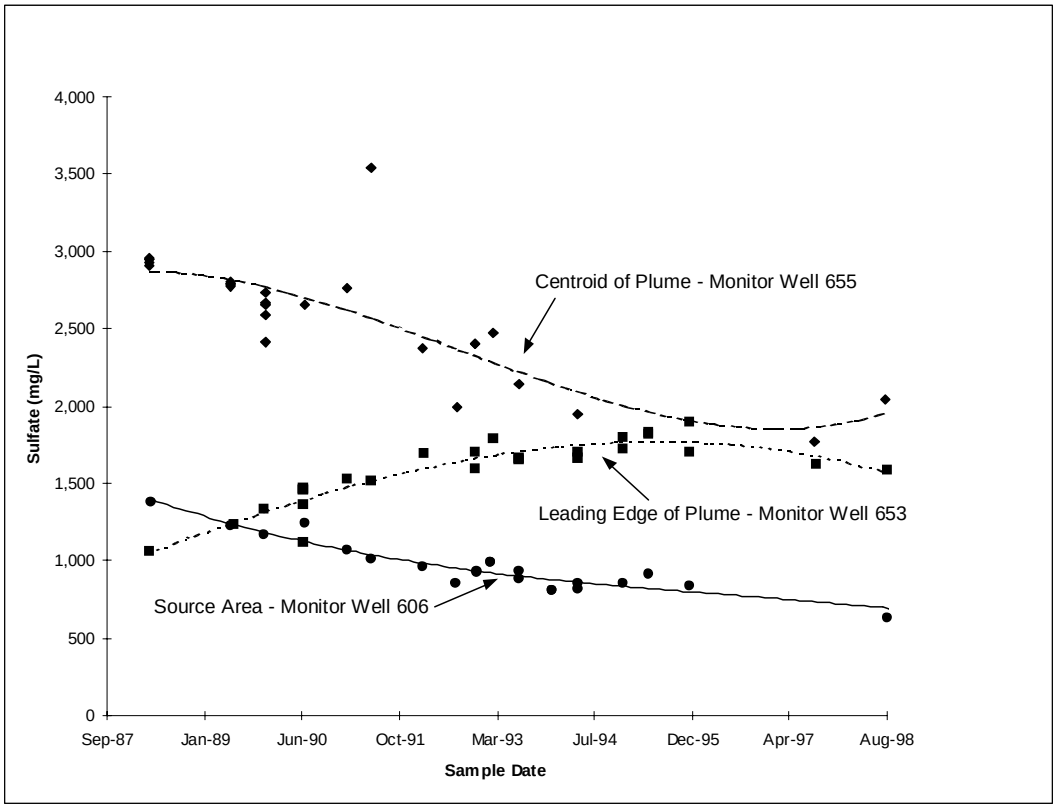

Figure 5-35. Sulfate Concentration Versus Time for Selected Monitor Wells Located Along the Longitudinal Axis of the Alluvial Plume

2,600 ft (0.5 miles) downgradient and (2) nitrate concentrations between 500 mg/L and the 44 mg/L MCL. Assumptions used in the calculations are presented with the estimated contaminant volumes in Table 5–9.

Table 5–9. Estimated Volume of Contamination in the Alluvial Aquifer

Nitrate (mg/L)	Area (ft^2)	Average saturated thickness (ft)	Contaminated saturated thickness (percent)	Estimated porosity (percent)	Estimated volume (gal)
44 to 500	8.8×10^6	34	75	25	420×10^6
500 and greater	2.4×10^6	27	100	25	120×10^6
				Total	540×10^6

5.3.3.2 Impact to the Shinarump Bedrock Aquifer

Ground water from the Shinarump aquifer is not significantly impacted by site-related contamination. Supporting evidence is provided by the 1997 ground water sampling results presented in Table 5–10 for downgradient Shinarump monitor wells MON–659 and MON–660. The range in natural background for each constituent, based on the upgradient water quality results previously presented in Section 5.3.1.4, is included in Table 5–10 for comparison. Results obtained at monitor well MON–614 are not included in Table 5–10 because the screen filter pack spans across both the alluvium and Shinarump sandstone and therefore water from this location may not be representative of the Shinarump Member. Results are not available for on-site monitor wells MON–607 and MON–609 because the wells are dry.

The 1997 sampling data presented in Table 5–10 demonstrate that concentrations of uranium and nitrate do not exceed the upper range in natural background at any location. Several other site-related constituents do exceed the upper range in natural background, however the maximum concentrations observed for these constituents are all relatively low; 0.5 mg/L ammonium, 25.9 mg/L calcium, 130 mg/L sulfate, and 5.9 pCi/L radium-226. No constituent occurs in concentrations that exceed any MCL or at concentrations that present a health risk (Section 6.0).

Further evidence supporting the unlikeliness that site-related contamination is significantly impacting the Shinarump aquifer is provided by water-level measurements and ground water sampling results obtained for monitor well pairs MON–606 (alluvium) and MON–659 (Shinarump) which are installed approximately 100-ft downgradient from the former new tailings area (Figure 5–31). A comparison of site-related contaminant concentrations and water elevations for the well pair is presented in Table 5–11. The data indicate that only ground water from the alluvial monitor well MON–606 contains site-related contamination; the highest levels of nitrate (1,030 mg/L) detected in the most contaminated portion of the plume. In addition, water-level measurements indicate a neutral to slight upward hydraulic gradient in the Shinarump Member, which would limit downward migration of contaminants from the alluvium.

Table 5–10. Comparison of Downgradient Ground Water to Natural Background Ground Water Quality in the Shinarump Aquifer

Analyte	Downgradient			Background Range[c]
	FOD[a]	Mean[b]	Range[c]	
Major (mg/L)				
Ammonia as NH_4	2/2	0.2664	.0247–.508	.0078–.1
Calcium	2/2	14.355	2.81–25.9	2.45–17.5
Chloride	2/2	8.835	7.92–9.75	7–11
Magnesium	2/2	15.635	1.57–29.7	1.61–22.8
Nitrate	2/2	1.9875	.235–3.74	<.014–8.86
Potassium	2/2	2.1735	.777–3.57	.93–2.91
Sodium	2/2	99.05	84.1–114	64.1–119
Sulfate	2/2	76	22–130	53.1–77
Metal (mg/L)				
Aluminum	0/2	0.075	<.1–<.2	<.1–.7
Arsenic	3/4	0.0049	<.005–.0062	<.005–<.01
Barium	2/4	0.046	.038–.046	<.1–.4
Cadmium	0/2	0.0005	<.001–<.001	<.001–<.001
Iron (Filtered)	½	0.0975	<.03–.18	<.03–.13
Manganese (Filtered)	½	0.0725	<.01–.14	<.01–.08
Manganese (Unfiltered)	½	0.0675	<.01–.13	<.01–<.01
Mercury	0/2	0.0001	<.0002–<.0002	<.0002–<.0002
Molybdenum	½	0.012	<.01–.019	<.01–.17
Selenium	0/2	0.0001	<.0002–<.0002	<.001–<.005
Strontium	2/2	0.1534	.0407–.266	.0471–.35
Uranium	½	0.001	<.001–.0015	<.0003–.0099
Vanadium	0/2	0.002	<.004–<.004	<.01–.7
Zinc (Filtered)	½	0.1225	<.05–.22	<.005–.085
Zinc (Unfiltered)	2/2	0.007	.005–.009	.101–.101
Other				
Alkalinity as $CaCO_3$ (mg/L)	2/2	240	196–284	168–220
Bromide (mg/L)	0/2	0.05	<.1–<.1	<2–<2
Fluoride (mg/L)	2/2	0.3	.2–.4	.3–.5
Redox Potential (mV)	2/2	–158.5	-252–65	-27–446
Silica (mg/L)	2/2	13.5	12.5–14.5	8–12
Sulfate/Chloride	2/2	7.8	2.6–13	5.8–10.4
Sulfide (mg/L)	0/2	0.275	<.1–<1	<.01–<.1
Total Dissolved Solids (mg/L)	2/2	364	308–420	301–370
Total Phosphorus as PO_4 (mg/L)	½	0.825	<.1–1.6	.2–1
pH (s. u.)	2/2	7.975	7.28–8.67	7.44–8.57
Radiologic (pCi/L)				
Gross Alpha	½	5.6525	<5.99–8.31	.5–13
Gross Alpha (Excluding Uranium)	½		7.28–7.28	
Gross Beta	½	6.835	<5.64–10.85	<2.7–8
Lead–210 (Filtered)	2/2	1.69	1.69–1.69	.1–1.3
Radium–226	2/2	3.335	.75–5.92	.2–2.17
Radium–228	½	0.8	<.4–1.4	.2–1.6
Thorium–230	2/2	0.15	0–.3	0–.3

[a]Frequency of detection; number of samples above detection/number of samples analyzed.
[b]Arithmetic mean based on most current sampling; one-half the detection limit used for values below detection.
[c]Minimum to maximum value detected; < indicates value below the detection limit.

Table 5–11. Chemical and Water Level Measurements Obtained at Alluvial and Shinarump Monitor Well Pairs

Monitor Well Pair	Completion Formation	Nitrate (mg/L)	Sulfate (mg/L)	Water Elevation (ft amsl)
MON–606	Alluvium	1,030 [a]	674 [a]	4,829.5 [b]
MON–659	Shinarump	3.7 [b]	130 [c]	4,829.9 [b]
MON–653	Alluvium	125 [a]	1,630 [a]	4,802.2 [b]
MON–660	Shinarump	<1 [b]	22 [c]	4,805.3 [b]

[a]August 1997
[b]January 1997
[c]November 1995

A similar situation exists further downgradient at monitor well pair MON–653 (alluvium) and MON–660 (Shinarump) which are installed near the center of the leading edge of the most contaminated portion of the alluvial aquifer (Figure 5–31). At this paired location, water from the alluvial monitor well MON–653 contains 1,630 mg/L sulfate which is one of the highest sulfate levels detected in the plume (Figure 5–24a) while the underlying Shinarump ground water from monitor well MON–660 contains sulfate concentrations that are consistent with natural background. Water-level measurements at this well pair location also indicate an upward hydraulic gradient from the deeper Shinarump to the alluvium, further demonstrating the unlikeliness that site-related contamination is present in the Shinarump aquifer.

5.3.3.3 Impact to the De Chelly Bedrock Aquifer

Ground water samples collected from the De Chelly aquifer do not exhibit wide spread site-related contamination. This is evidenced by the 1997 sampling data presented in Table 5–12, which summarizes on-site and downgradient water quality. The data demonstrate that concentrations of site-related constituents such as ammonium, nitrate, potassium, strontium, vanadium, radium-226, and radium-228 do not exceed the upper range in natural background at any on-site or downgradient location. Other constituents such as magnesium and sulfate occur in one instance at concentrations only slightly above the upper limit of natural background.

Calcium and uranium are the only site-related constituents that occur frequently above natural background in the De Chelly aquifer. Calcium concentrations exceed the upper limit in natural background only in water collected from on-site monitor wells (MON–657, –618, –619, and –776) located near the former old tailings pile. The maximum calcium concentration of 56.5 mg/L, which is only twice the upper limit of natural background (28.6 mg/L), was detected in monitor well MON–657.

Table 5–12. Comparison of On-Site and Downgradient Ground Water to Natural Background Ground Water Quality in the De Chelly Aquifer

Analyte	Downgradient			Background Range[c]
	FOD[a]	Mean[b]	Range[c]	
Major (mg/L)				
Ammonium as NH_4	3/8	0.0117	.0053–.0352	<.004–<.1
Calcium	8/8	27.4	11.7–56.5	6.34–28.6
Chloride	8/8	7.0475	3.23–9.78	3.11–34.7
Magnesium	8/8	20.4875	8.2–36.4	8.37–31.9
Nitrate	8/8	5.5361	.0836–18.8	.66–22
Potassium	8/8	2.6388	1.53–3.7	1.82–5.29
Sodium	8/8	43.6863	6.89–82.1	7.15–92.1
Sulfate	7/7	48.5714	13.1–103	11.6–66.3
Metal (mg/L)				
Aluminum	4/7	0.1714	<.1–.3	<.05–.8
Arsenic	2/12	0.0051	<.005–.016	<.0015–.022
Barium	8/12	0.1667	<.1–.4	.036–.5
Cadmium	0/8	0.0005	<.001–<.001	<.00013–<.005
Iron (Filtered)	4/7	0.05	<.03–.091	<.03–.25
Iron (Unfiltered)	3/5	0.05	<.03–.13	.04–2.95
Manganese (Filtered)	2/7	0.0103	<.01–.034	<.0015–.21
Manganese (Unfiltered)	3/5	0.062	<.01–.25	<.0015–.42
Mercury	1/7	0.0002	<.0002–.0005	<.0002–<.0002
Molybdenum	2/8	0.0288	<.01–.19	<.0049–.18
Selenium	6/8	0.0019	<.0002–.0038	.00033–.005
Strontium	8/8	0.3943	.183–.57	<.1–.68
Uranium	7/8	0.0222	<.001–.0673	.0004–.0096
Vanadium	4/8	0.0197	<.004–.0542	<.0019–.8
Zinc (Filtered)	2/7	0.0247	<.01–.048	<.001–.725
Zinc (Unfiltered)	5/5	0.0572	.007–.197	.004–.05
Other				
Alkalinity as $CaCO_3$ (mg/L)	8/8	168.75	106–273	77–235
Bromide (mg/L)	0/5	0.62	<.1–<2	<.1–<2
Fluoride (mg/L)	7/7	0.3286	.2–.5	.2–.6
Redox Potentia (mV)l	8/8	21.25	-212–204	108–466
Silica (mg/L)	7/7	11.0143	8–14.1	5–11.5
Sulfate/Chloride	7/7	7.1	4.1–14.6	1.7–15.4
Sulfide (mg/L)	0/7	0.095	<.01–<1	<.01–4.9
Total Dissolved Solids (mg/L)	8/8	270.625	160–373	118–370
Total Phosphorus as PO_4 (mg/L)	0/7	0.05	<.1–<.1	<.1–.3
pH (s. u.)	8/8	7.8725	7.36–8.74	7.37–9.36
Radiological (pCi/L)				
Gross Alpha	3/8	8.935	<2.57–29.02	0–10
Gross Alpha (Excluding Uranium)	3/8		0–0	
Gross Beta	7/8	6.74	<2.7–13.55	0–9
Lead–210 (Filtered)	1/8	0.51	<.61–1.31	0–1.2
Radium–226	8/8	0.1738	.09–.25	0–2.4
Radium–228	0/8	0.4063	<.3–<1.3	0–4.9
Thorium–230	7/7	0.2714	0–.7	0–1.1

[a]Frequency of detection; number of samples above detection/number of samples analyzed.
[b]Arithmetic mean based on most current sampling; one-half the detection limit used for values below detection.
[c]Minimum to maximum value detected; < indicates value below the detection limit.

Uranium is present at concentrations above the upper limit in natural background in ground water collected in 1997 at MON–664, –657, –619, and –776 (Figure 5–36). The maximum uranium concentrations are present at monitor wells MON–657 (0.067 mg/L) and –619 (0.053 mg/L), however these maximum values only slightly exceed the 0.044 mg/L uranium MCL. Monitor wells MON–657 and –619 are located approximately 400-ft apart in an area once occupied by the old tailings pile. The western portion of the former old tailings area is underlain by a buried paleovalley approximately 120 ft deep where Quaternary material rests in direct hydrologic contact with the Hoskinnini Member of the lower Moenkopi Formation, as shown on the geologic cross-section D to D' (Figure 5–37). The medium- to coarse-grained Hoskinnini sandstone provides a hydrologic connection with the underlying De Chelly aquifer. A recent aquifer test confirmed the presence of a hydrologic connection between the alluvial and De Chelly aquifers in this region of the site (Section 4.6). Ground water samples collected from MON–657 and –619 which are completed in the De Chelly aquifer, while having slightly elevated uranium concentrations, have relatively low concentrations of mill-related constituents such as nitrate and sulfate. This suggests that the water used for size separation of the ores during the mechanical processing period from 1955 to 1964 may be the source for the elevated uranium in the De Chelly aquifer. Generally, the upward hydraulic gradient in the De Chelly aquifer has prevented downward migration of process waters. However, the hydraulic gradient has locally been reversed when production well MON–619 was pumped to supply water for the milling operation. Currently, the production wells are no longer in operation and the upward gradient has reestablished.

Cross-section D to D' graphically illustrates how elevated uranium in the alluvial aquifer may have been introduced into the De Chelly as a result of pumping production well MON–619. This model is supported by the aquifer test previously discussed in Section 4.6 and by examining the uranium concentrations observed in water samples collected from the surficial aquifer adjacent to MON–619. For example, monitor well MON–774 is installed in the alluvium approximately 95 ft west of MON–619. The uranium concentration detected in water from this alluvial well is consistent with the uranium concentration detected in water from the De Chelly well MON–619, suggesting the alluvial water is the source of the elevated uranium. It is notably important that the water level in the alluvium is below the contact between the Shinarump sandstone and the Moenkopi Formation, thereby eliminating the possibility that contaminated process water was also drawn into the Shinarump when MON–619 was in production.

Although the pumping of well MON–619 to supply processing water for the milling operations may have actively drawn contaminated ground water from the alluvial aquifer into the De Chelly Sandstone the area of impact is small, isolated, and the uranium concentrations are only slightly above the MCL. The limited areal extent and isolated nature of the slightly elevated uranium concentrations occurring in 1997 at MON–619 and –657 is shown in Figure 5–36. It is evident from the limited areal extent of the plume that the uranium concentrations are less than the MCL at monitor well MON–776, located 70 ft upgradient from well MON–619. Similarly, downgradient uranium concentrations decrease to natural background levels within a short distance from MON–657. This is evidenced by the low uranium concentration of 0.003 mg/L detected in ground water sampled from monitor well MON–775, which is located approximately 400 ft downgradient from monitor well MON–657.

Monitor well MON–775 is situated along the axis of the paleovalley, at the most downgradient extent where the Shinarump Member and most of the Moenkopi Formation have been eroded away, allowing water from the contaminated alluvial aquifer to be in direct hydrologic contact with the De Chelly Sandstone Member. However, concentrations of site-related constituents observed in the De Chelly aquifer at this downgradient location are consistent with natural background concentrations, indicating that the overlying alluvial water is not impacting the lower De Chelly aquifer. An upward hydraulic gradient in the De Chelly aquifer prevents downward migration of water from the alluvium to the De Chelly.

In summary, production well MON–619 can be considered an isolated point source for the elevated uranium concentrations observed in the De Chelly aquifer. As shown in Figure 5–38a, uranium concentrations in ground water collected at MON–619 during the period 1985 through 1998 have declined significantly since the well was pumped during an aquifer test in 1993, indicating that the De Chelly aquifer is diluting the concentrations by naturally flushing the uranium downgradient. Uranium observed in downgradient well MON–657 exhibit a similar pattern of decreasing uranium concentration versus time. Uranium concentrations in the De Chelly are expected to continue to decrease with time as the aquifer dilutes and flushes the uranium downgradient.

Continued impact to the De Chelly groundwater is not expected since well 619 is no longer being pumped for production. In addition, dedicated low-flow bladder pumps were installed at the end of the 1997 field characterization to prevent contaminated alluvial water to be actively drawn into the De Chelly through the erosional window located in the adjacent paleovalley when the well is sampled for water quality. The most recent sampling results obtained in 1998 suggest that natural flushing is working (Figure 5–38b). For example, uranium concentrations at well 657 have decreased to levels below the MCL, thereby reducing the areal extent of the plume. However, additional monitoring is required to verify this trend.

5.3.3.4 Impacts to Cane Valley Sediments

Chemical results for sediment samples collected along Cane Valley Wash and a tributary downgradient of the frog ponds at locations MON–620, –624, –626, and –627 (Figure 5–39) are presented in Table 5–13. For comparison, concentrations observed in background sediments and soils collected upgradient of the frog ponds and the former mill site (see Section 5.3.1.2) are also presented in Table 5–13. The data indicate that sediments along Cane Valley Wash have not been impacted by the milling activities. For example, examination of chemical results in Table 5–13 for sediments collected at Cane Valley Wash locations MON–624, –626, and –627, and at location MON–620 tributary drainage, indicates no notable differences in sediment chemistry from natural background. Sulfate concentrations are notable in the background samples and at the downstream sample collected at location MON–624, which reflect the accumulation of natural sulfate salts in the sediments due to evaporation and transpiration.

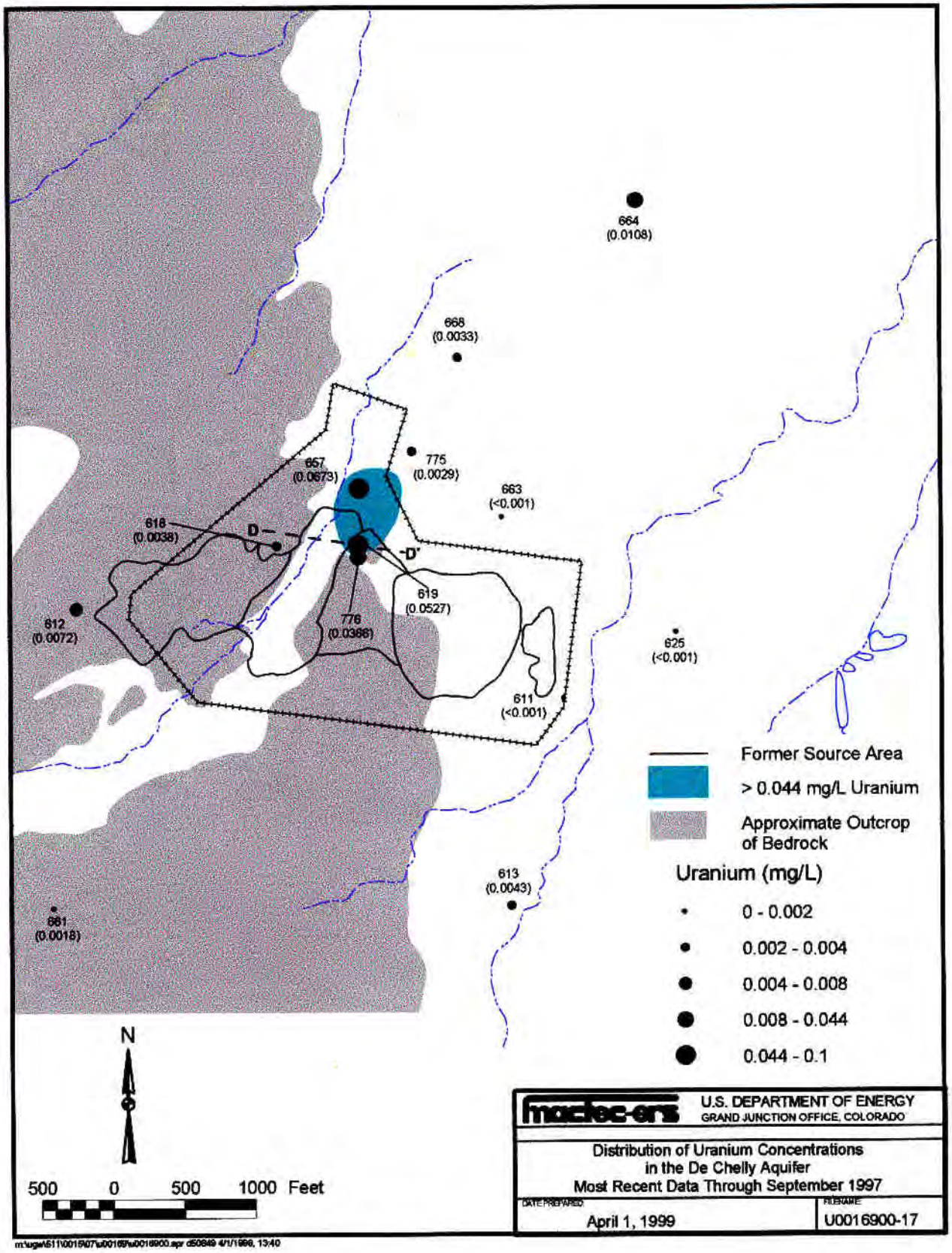

Figure 5–36. Distribution of Uranium Concentrations in the De Chelly Aquifer - September 1997

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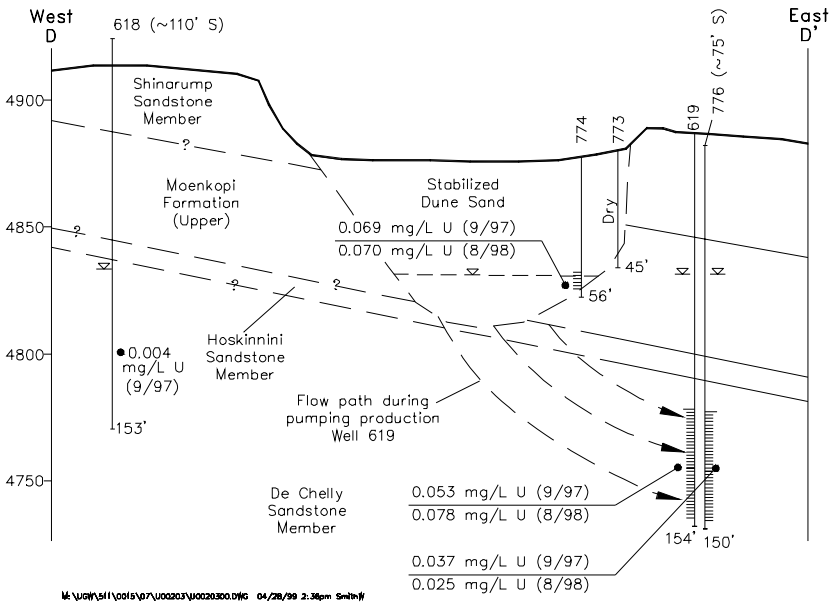

Figure 5-37. Uranium Concentration Profile D to D'

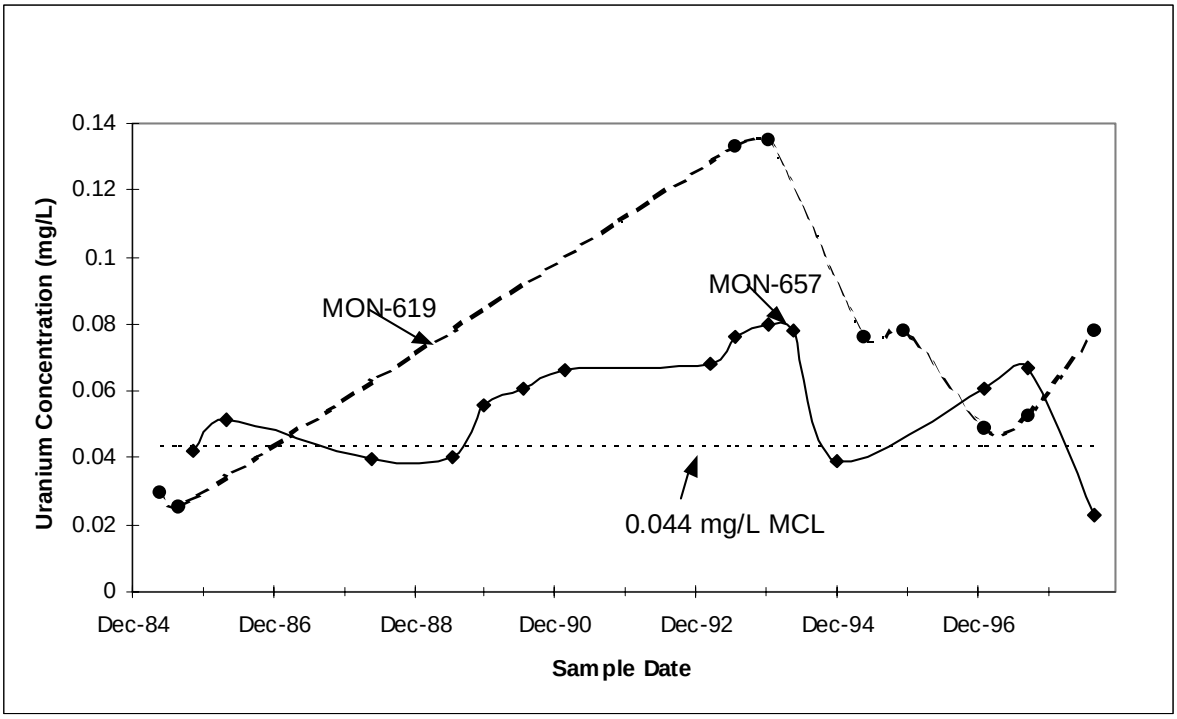

Figure 5-38a. Uranium Concentration Versus time for De Chelly Bedrock Wells Located Near the Former Old Tailings Area

Table 5–13. Comparison of Selected Constituents in Sediments along Cane Valley Wash to Background Sediments

Constituent[a]	MON–620	MON–624	MON–626	MON–627	Range in Natural Background[b]
Manganese	NA[c]	NA	NA	NA	84.8 – 225
Nitrate	0.9	3.8	0.4	0.9	1.8 – 9.4
Strontium	28	46	26	54	25.8 – 116
Sulfate	48	935	42	35	292 – 771
Uranium	1.0	<1.0	<1.0	<1.0	0.3 – <1.0
Vanadium	9.0	3	5	9	5.7 – 16.5

[a]All units in mg/kg.
[b]Background locations MON–623, –869, and –870.
[c]NA = not analyzed.

5.3.3.5 Impacts to Cane Valley Waters

Shallow Alluvial Ground Water: Nitrate and sulfate concentrations are elevated in an alluvial plume as a result of mill-related fluids that entered the ground water at the site and migrated northward. In addition to the mill-related sulfate plume, there are several wells in the alluvium in the nearby Cane Valley Wash area that have high sulfate concentrations. These include wells MON–200, –407, –605, –640, –687, and –855 with sulfate concentrations of 543; 1,210; 1,550; 686; 500; and 1,500 mg/L, respectively. The elevated sulfate concentrations in the Cane Valley Wash wells are believed to be due to effects of evaporation and are not related to the mill. Justification for this interpretation is provided below.

The ratios of sulfate-to-chloride concentrations vary depending on if the source is related to past millsite activities or if it occurs naturally. Tailings fluids were enriched in nitrate and sulfate but had relatively low chloride concentrations as exemplified by the chemistry of the tailings solutions collected from the new tailings pile and the evaporation pond (Table 5–7). New tailings pile water collected from a lysimeter contained 4,510; 530; and 20 mg/L of sulfate, nitrate, and chloride, respectively. Evaporation pond water contained 1,610; 570; and 45 mg/L of sulfate, nitrate, and chloride, respectively. Because of the high sulfate-to-chloride ratios in the tailings fluids, contamination caused sulfate concentrations to increase with little effect on the chloride concentration. This is shown in the recent characterization data. Sulfate-to-chloride ratios are distinctively different between the plume area and the Cane Valley Wash area (Figure 5–40). Sulfate-to-chloride values greater than 10 define an area similar to the sulfate and nitrate plumes. The ratios are higher in the plume area than in the Cane Valley Wash area.

The different ratios can easily be seen when sulfate concentrations are plotted against chloride concentrations for the most recent alluvial ground water samples (Figure 5–41). Plume-related ground water is distinguished on the Figure by having nitrate concentrations of over 44 mg/L. The plume-related ground water samples are clearly distinguishable from the background samples by their high sulfate and low chloride concentrations. Samples with elevated sulfate and chloride concentrations, but with nitrate concentrations of less than 44 mg/L, can be explained by an evaporation model. Samples that have the lowest concentrations of sulfate and chloride are assumed to represent water that has been relatively unaffected by either evaporation or

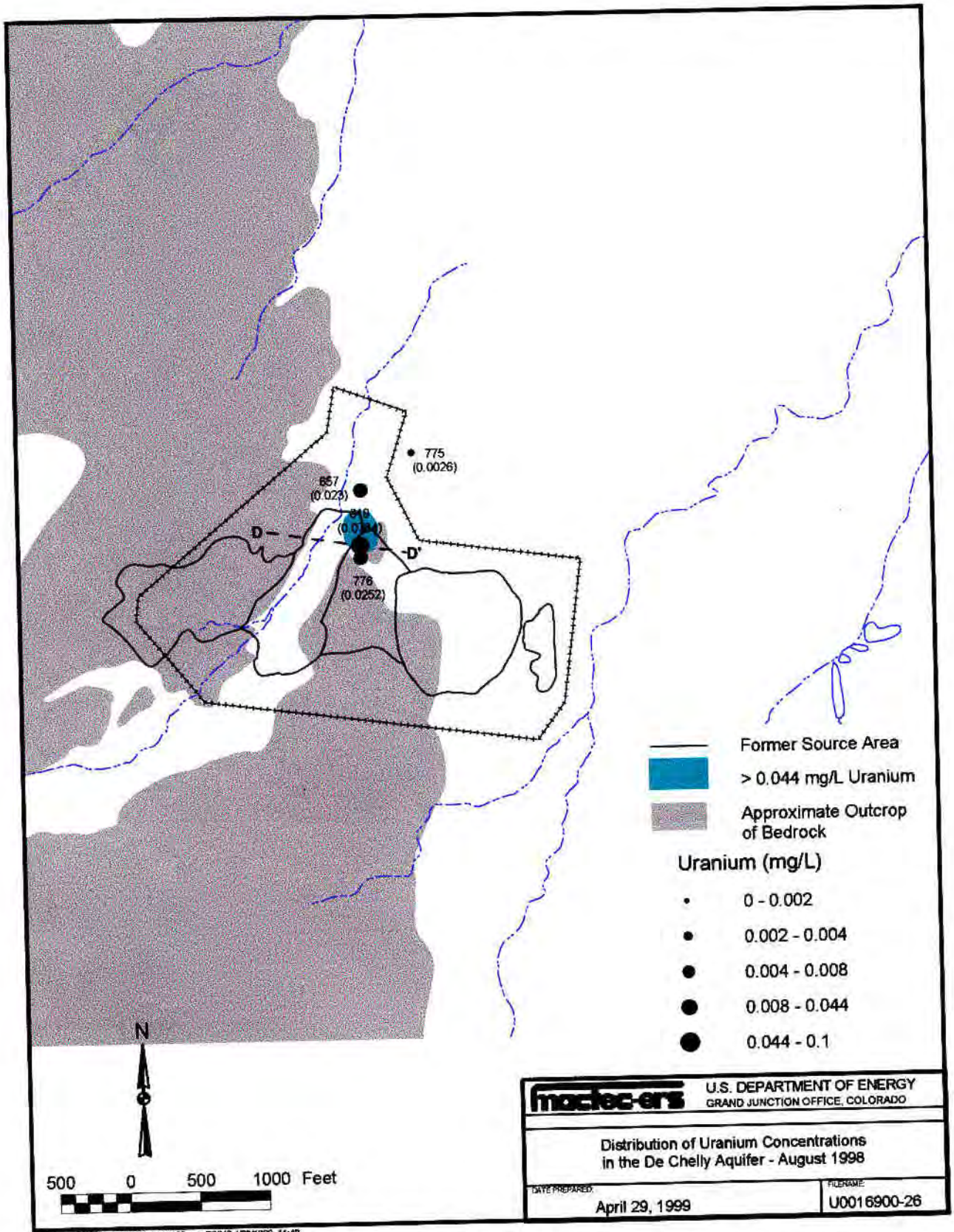

Figure 5–38b. Distributions of Uranium Concentrations in the De Chelly Aquifer - August 1998

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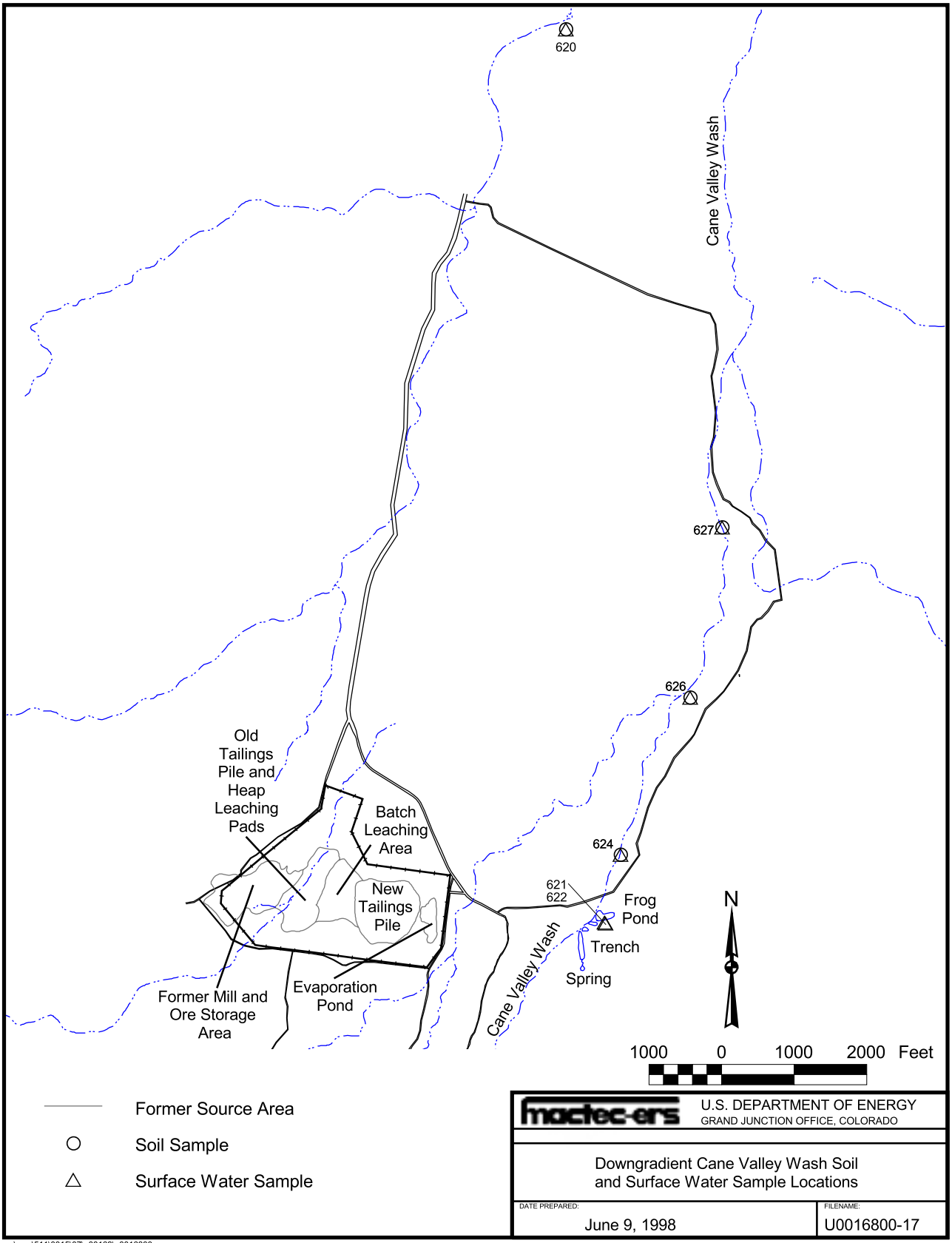

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Figure 5-39. Downgradient Cane Valley Wash Soil and Surface Water Sample Locations

contamination from the mill site. Two dilute ground water compositions covering the range of sulfate-to-chloride ratios were selected to represent this unaltered water ("dilute waters 1 and 2" on Figure 5–41). Evaporation will cause both sulfate and chloride concentrations to increase. The evaporation of the two dilute waters is bracketed by a shaded area on Figure 5–41. All of the low nitrate ground waters fall within the range modeled by the evaporation trends. These waters are indicative of background conditions along Cane Valley Wash.

Because of the high sulfate-to-chloride ratio in tailings water, ground water that is contaminated by mill-tailings fluids plot above the evaporation trends (Figure 5–41). If tailings fluid from the new tailings pile is added to the dilute background water (the analysis is shown for "dilute water 1" but would be similar if "dilute water 2" were used), sulfate concentrations increase with very little increase in chloride concentration, and thus plot along a nearly vertical line. Because the sulfate-to-chloride ratio is lower for the evaporation pond fluid, the addition of this water to "dilute water 1" produces compositions that fall along a more inclined line but still distinct from the evaporation trends. All but two of the high-nitrate ground waters fall within the range modeled by the addition of contaminated water from the new tailings pile or evaporation pond. The high sulfate-to-chloride ratios are indicative of water that has been contaminated by the milling fluids.

Relatively high sulfate concentrations exist upgradient from the site where it could not have been contaminated by the mill fluids (for example, wells MON–200 and MON–640 have sulfate concentrations of 543 and 668 mg/L, respectively). These elevated background sulfate concentrations, together with the distinct evaporation signature, indicate that the elevated sulfate concentrations in Cane Valley Wash result from evaporation of uncontaminated water.

Surface Water: The uranium MCL was exceeded in at least one surface-water sample collected from locations MON–621, –622, –624, and –627 (Figure 5–39) from the most recent sampling (Table 5–14). The highest observed concentration of uranium is 0.0647 mg/L. No other COPCs exceeded MCL concentrations. For those COPCs that do not have MCLs, concentrations are generally low, except sulfate which had a concentration of 2,060 mg/L at sampling location MON–627 in 1996. Sulfate concentrations vary considerably in Cane Valley Wash. Sulfate has had elevated concentrations at some surface sampling locations sporadically during their sampling history. The elevated concentrations are attributed to evaporation.

The effects of evaporation are seen in the sulfate-to-chloride ratios which range from 1 to 5.7 in surface water (Figure 5–40). These ratios are consistent with the values (less than 10) for shallow alluvial ground water along Cane Valley Wash (Figure 5–40). The evaporation signature of the surface waters is readily observed on a sulfate-to-chloride diagram (Figure 5–41).

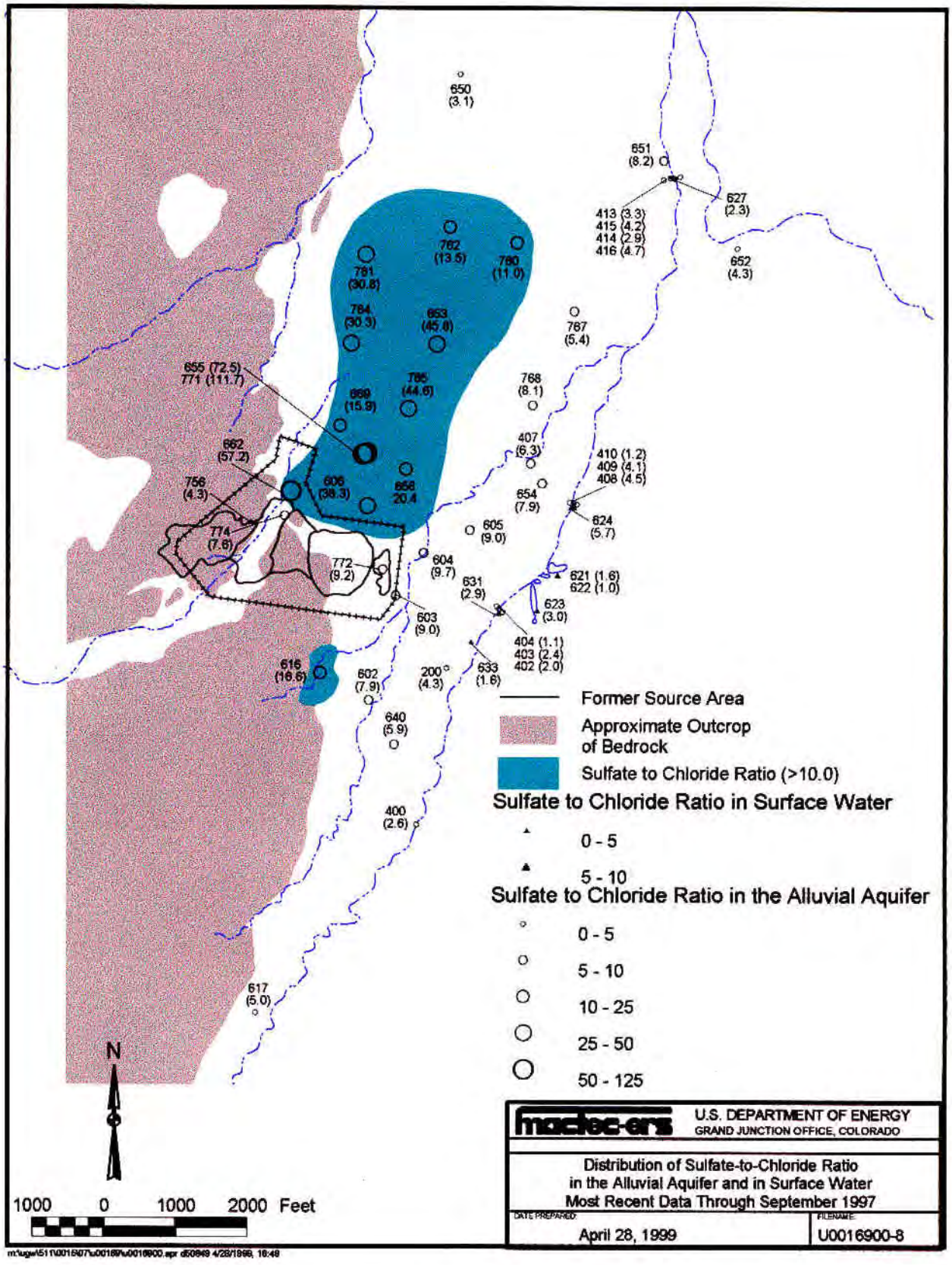

Figure 5–40. Distribution of Sulfate-to-Chloride Ratio in the Alluvial Aquifer and in Surface Water

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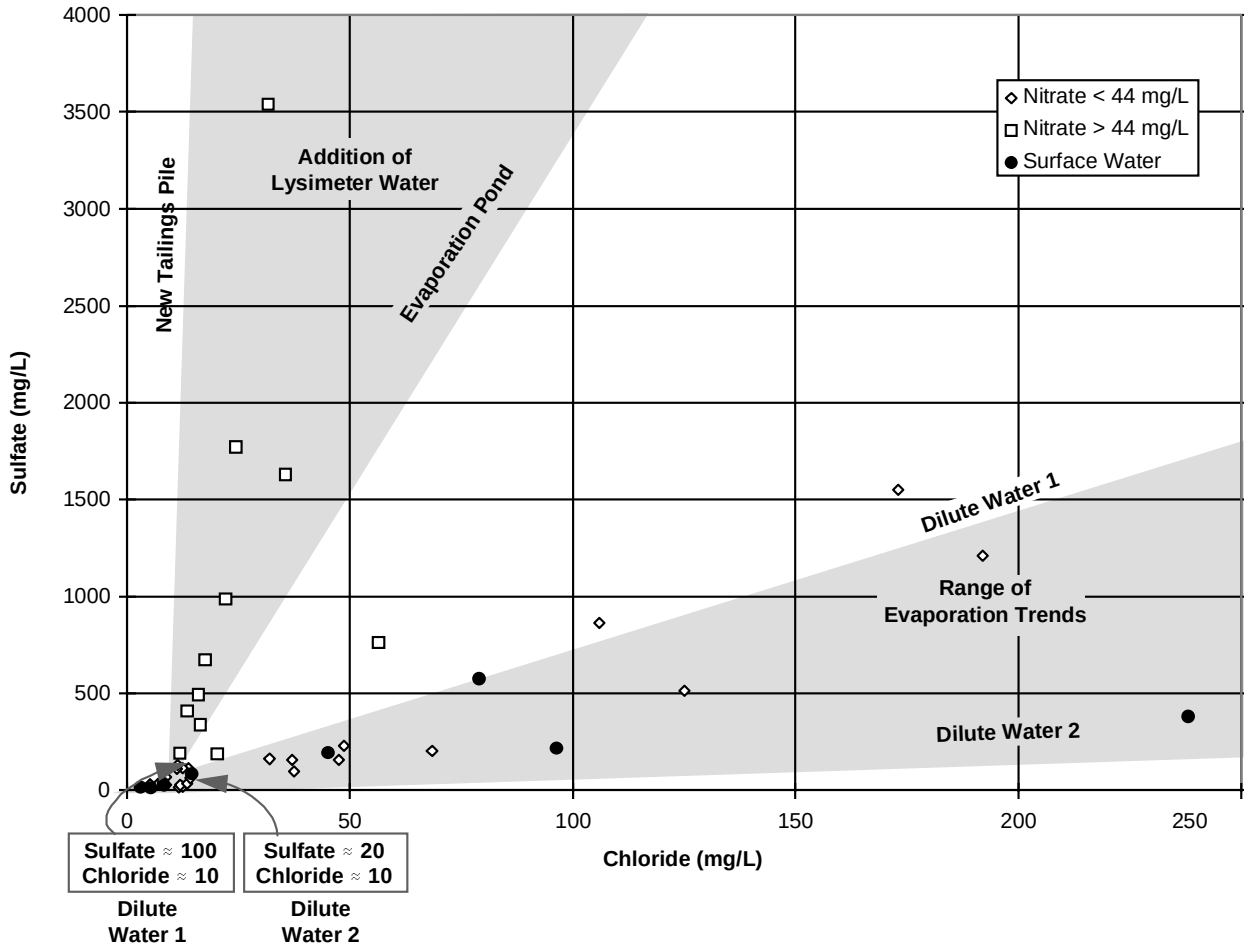

Figure 5-41. Sulfate-to-Chloride Ratio

Table 5–14. COPC Concentrations Observed in Surface Water

Sample	Date	COPC (mg/L)					
		Mn	NO_3	Sr	SO_4	U	V
0620	6/27/93	1.53	<1.0	0.42	84	0.028	0.04
	1/12/97	NA	0.667	0.575	NA	0.0252	0.0052
0621	11/22/92	0.005	0.071	0.55	50	0.014	0.003
	1/11/97	NA	0.0735	0.696	NA	0.0572	<0.0044
0622	11/24/92	0.009	0.48	0.61	37	0.001	<0.0019
	1/11/97	NA	0.0745	1.03	NA	0.0647	0.0102
0623	11/19/95	0.041	<1.0	0.49	30	0.002	<0.01
	1/11/97	NA	0.0744	0.407	NA	0.0063	<0.004
	8/27/97	NA	0.198	0.537	24.9	<0.001	<0.013
0624	1/16/96	0.02	<1.0	NA	191	0.064	0.02
	1/13/97	NA	0.624	0.851	NA	0.0546	0.0042
0627	1/15/96	0.04	1.4	NA	2,060	0.062	0.12
	8/21/97	NA	0.105	0.159	217	0.0126	0.0285
0631	1/16/96	<0.01	<1.0	NA	573	0.020	0.01
	1/14/97	NA	0.352	0.38	NA	0.0274	0.0175
0633	1/19/96	<0.01	<1.0	NA	379	0.002	0.01
MCL		None	44	None	None	0.044	None

The elevated uranium concentrations may also be due to evaporation as observed on a plot of uranium-to-chloride. With the exception of one point, the uranium-to-chloride ratio in surface location 627 parallels the sulfate-to-chloride ratios suggesting that evaporation is a cause of the uranium concentrations (Figure 5–42). However, the entire Cane Valley area was the scene of mining, milling, and uranium exploration activity for many decades and some uranium in surface water could have come from tailings or ore-related materials dispersed in the soils.

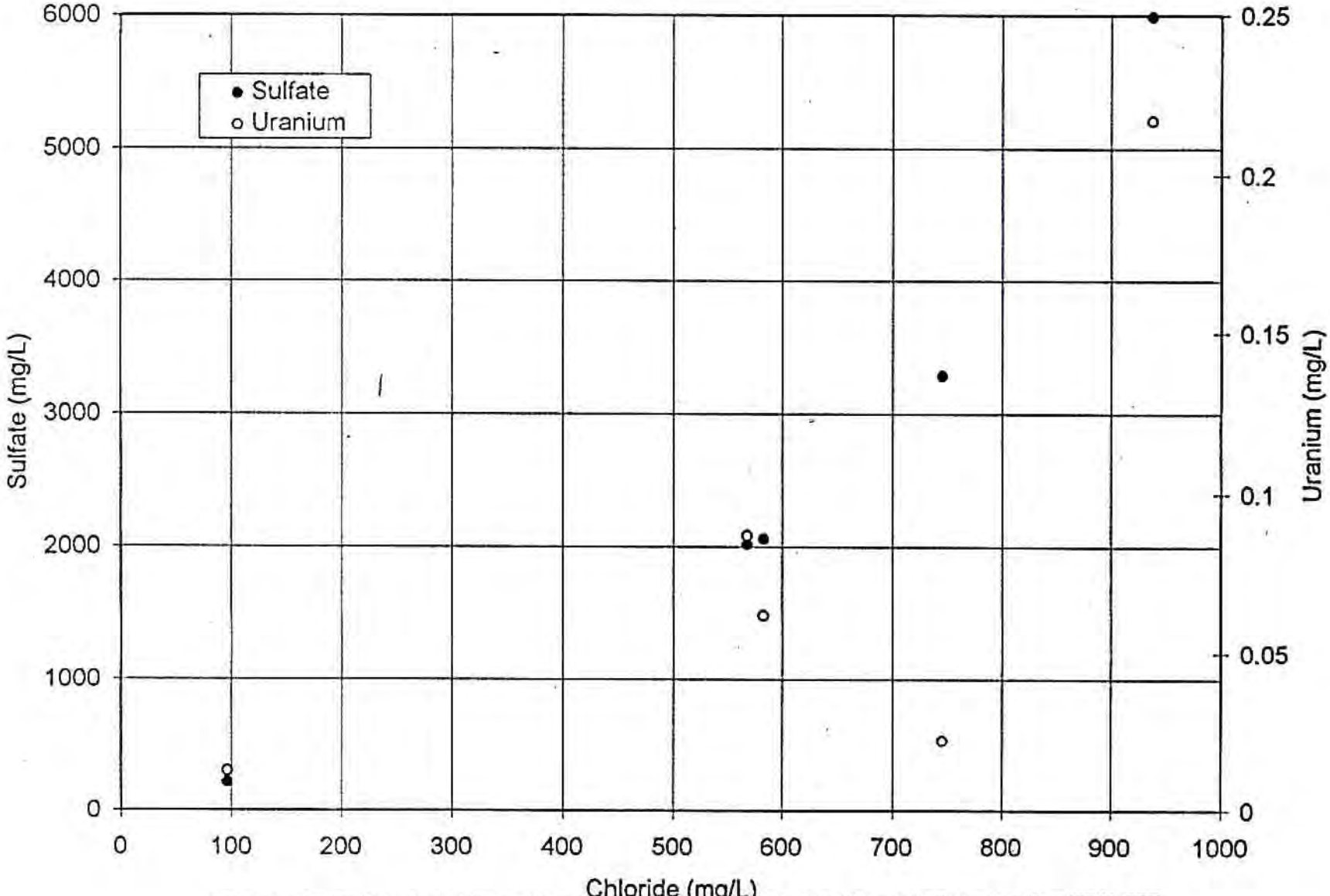

Figure 5–42. Sulfate, Chloride, and Uranium Concentration Relationships at Surface Location MON–627

5.3.4 Fate and Transport of Ground Water Contaminants

Site-related contaminants are subject to dispersion and dilution by ground water transport and attenuation by radioactive decay and various chemical reactions, including oxidation/reduction reactions, precipitation and coprecipitation, adsorption onto the aquifer mineral surfaces, cation exchange, and biologically controlled oxidation/reduction reactions. The mobility or potential for attenuation depends on the chemical species of ions in the aqueous environment. The chemical species present in natural systems are a function of pH, Eh, and the concentrations of various anions and cations. The predominant species of the site-related constituents identified as occurring in the alluvial aquifer at concentrations above natural background were predicted using the geochemical model PHREEQE (Parkhurst et al. 1980, DOE 1996d). The dominant solution species are summarized in Table 5–15.

Table 5–15. Dominant Solution Species in the Alluvial Ground Water

Constituent	Dominant Species	Common Name	Molar percent
Manganese	Mn^{2+} $MnSO_{4\ AQ}$	Manganese Manganese sulfate	64 31
Nitrate	NO_3^-	Nitrate	100
Strontium	Sr^{2+}	Strontium	100
Sulfur	SO_4^{2-}	Sulfate	100
Uranium	$UO_2(CO_3)_3^{4-}$ $UO_2(CO_3)_2^{2-}$	Uranyl tricarbonate	70 30
Vanadium	$HV_2O_7^{3-}$ $H_2VO_4^-$	Pyrovanadic acid	83 15

Ground water quality data provide insight into the fate and transport of site-related contaminants in ground water at the site. Several constituents identified in the tailings solutions are absent or occur at much lower levels in the contaminated ground water, due to reactions of these contaminants with the aquifer matrix. Contaminants present in tailings solutions, but absent (below detection) in ground water at the site, are trace metals including antimony, chromium, cobalt, copper, molybdenum, and nickel. Contaminants in tailings solutions that are greatly attenuated, but detectable above background, are ammonium, radium-226, uranium, and vanadium. Contaminants that are at levels in ground water similar to those in the tailings solutions are major cations and anions (calcium, nitrate, magnesium, chloride, potassium, strontium, and sulfate).

Dispersion and precipitation reactions control concentrations of the major cations and anions such as calcium, iron, magnesium, nitrate, potassium, silica, sodium, strontium, and sulfate. Precipitation/dissolution reactions will occur in the portion of the plumes closest to the former tailings piles. Precipitation reactions currently are active because the shallow ground water in this zone is oversaturated with gypsum. With the tailings piles removed, the ground water sulfate concentrations will decrease, allowing gypsum to redissolve. Dissolution will buffer sulfate and calcium concentrations until the gypsum is exhausted. At this point, dilution with background water will lower sulfate and calcium concentrations.

Cation exchange reactions with clays and oxidation to nitrate perhaps, mediated by bacterial action, will decrease ammonium concentrations. Dispersion and adsorption mainly will decrease manganese, uranium, vanadium, and zinc concentrations.

5.4 Ecology

This section discusses the results of the ecological investigation (Section 4.7) with respect to the feasibility of phytoremediation and native plant farming.

5.4.1 Feasibility of Phytoremediation and Native Plant Farming

DOE plans active remediation of ground water constituents that exceed EPA standards or pose a risk to human health and the environment at Monument Valley. High nitrate levels, as high as 1,200 mg/L in the alluvial aquifer, exceed the EPA standard for nitrogen (equivalent to 44 mg/L nitrate). Residual ammonium in soils and substrates where the tailings piles were removed (subpile soils) may be a continuing source of ground water nitrate (Section 5.3.2.2). DOE is implementing phytoremediation of subpile soil ammonium and is evaluating phytoremediation of the shallow portions of the alluvial aquifer in an ongoing pilot study as a component of an active ground water remediation strategy for the site. In Section 8 of this SOWP, DOE evaluates the feasibility of a native plant farming alternative; pumping and treating high-nitrate ground water by using it to irrigate and fertilize revegetation plantings (Baumgartner et al. 1996).

This section provides background information concerning plant extraction and fate of ground water and soil nitrogen, and discusses the feasibility of phytoremediation and native plant farming alternatives.

5.4.1.1 Background Information

Plant Extraction and Fate of Nitrogen in Ground Water and Soil

Nitrogen (N) is an essential macronutrient for the growth of higher plants. Nitrate (NO_3^-) and ammonium (NH_4^+) in soils and ground water are the most common plant-available forms of N in arid and semiarid ecosystems (Coyne et al. 1995). Utilization of NO_3^- by higher plants involves the uptake, storage, translocation, and incorporation of N into organic forms. Most N uptake is through roots, although foliar uptake may also occur. N taken up from soil by the roots of terrestrial plants is either in the NO_3^- form or the NH_4^+ form. NO_3^- and NH_4^+ are taken in through the epidermis of plant roots and into the symplast of cortical and endodermal cells by way of a combination of passive diffusion and active transport which requires expenditure of energy.

Once in the plant, NO_3^- is reduced to ammonia (NH_3) or NH_4^+ either in the root or after it is transported up the xylem into the leaves. NO_3^- may be stored in cell vacuoles for a period of time before it is reduced. Reduction of NO_3^- is driven by photochemical energy captured through photosynthesis. The NH_3 or NH_4^+ is converted to amides and, through reactions catalyzed by transferases, amides are converted to amino acids. The amino acids are the building blocks for complex nitrogenous compounds in the plant protoplasm including proteins, chlorophyll, growth regulators, alkaloids, nucleosides, nucleotides, and nucleic acids.

Understanding the fate of N bound in live plant protoplasm is important for an evaluation of phytoremediation. Some N is lost as NH_3 directly from plants through stomates to the atmosphere. However, most N is returned to the soil either by death and decay of plant tissue or via grazing animals. Most N in terrestrial ecosystems resides in soil organic matter. Bacteria and fungi decay dead plant protoplasm (litter) producing amino acids and other soil organic residues. This soil organic matter is eventually converted to NH_4^+ and NH_3 by amnifying bacteria. N in plant biomass ingested by grazing animals is excreted in urine or feces and then rapidly hydrolyzed to NH_4^+.

N that has been returned to the soil as NH_4^+ and NH_3 is either taken up again by higher plants, used as an energy source by nitrifying bacteria, forming NO_3^-, or lost through volatilization and leaching (Coyne et al. 1995; Barbour et al. 1987). A combination of high temperatures and dry soil, common at Monument Valley, can result in substantial volatilization of NH_3. The potential for leaching of NH_4^+, NH_3, and NO_3^- is a function of the soil water balance which depends to a great degree on vegetation condition. Without plants, up to 50 percent of precipitation may be available to move N compounds back towards the ground water in arid and semiarid ecosystems (Gee et al. 1994). However, very little if any leaching would be expected where vegetation in good condition returns precipitation to the atmosphere via ET (evaporation from leaf and soil surfaces).

Phytoremediation Feasibility

Data to evaluate the feasibility of using phytoremediation as one possible component of the cleanup strategy were collected during 1997 (Section 4.7). Greasewood (*Sarcobatus vermiculatus*) and fourwing saltbush (*Atriplex canescens*) are both deep-rooted shrubs presently growing over most of the alluvial nitrate plume (Figure 5–43). Greasewood is an obligatory phreatophyte; its presence is evidence that it is rooted in the plume. In general, the gradient from a greasewood-dominated plant community to a fourwing saltbush-dominated community from southeast to northwest across the plume (see Figure 4–11) is correlated with increasing depth to ground water. However, the greasewood and fourwing saltbush stands are currently in poor condition because of historical overgrazing. Also, part of the greasewood stand may have been sprayed with herbicides during the surface remediation activity.

Evidence from rooting depth literature, photograph comparisons, and plant succession in disturbed areas all support the premise that by (1) increasing the abundance and expanding the distribution of greasewood and other phreatophytes; and (2) protecting phreatophyte populations from grazing, phytoremediation can contribute significantly to cleanup of nitrate in the alluvial aquifer. The greasewood and fourwing saltbush populations already cover a large portion of the plume area (Figure 5–43). A review of rooting-depth literature (e.g., Nichols 1993; Branson et al. 1981) indicates that the plume is potentially within reach of greasewood roots. A comparison of recent and old photographs shows that the greasewood population may be a consequence of milling activities, that the population has spread over the past 15 years, and that plants growing in the plume area are much larger than plants growing outside the plume area, apparently a response of nitrate fertilization. Greasewood and fourwing saltbush plants have been established and have grown rapidly within the fenced area that was disturbed during surface remediation only 3 years ago. This is evidence that (1) planting greasewood and fourwing

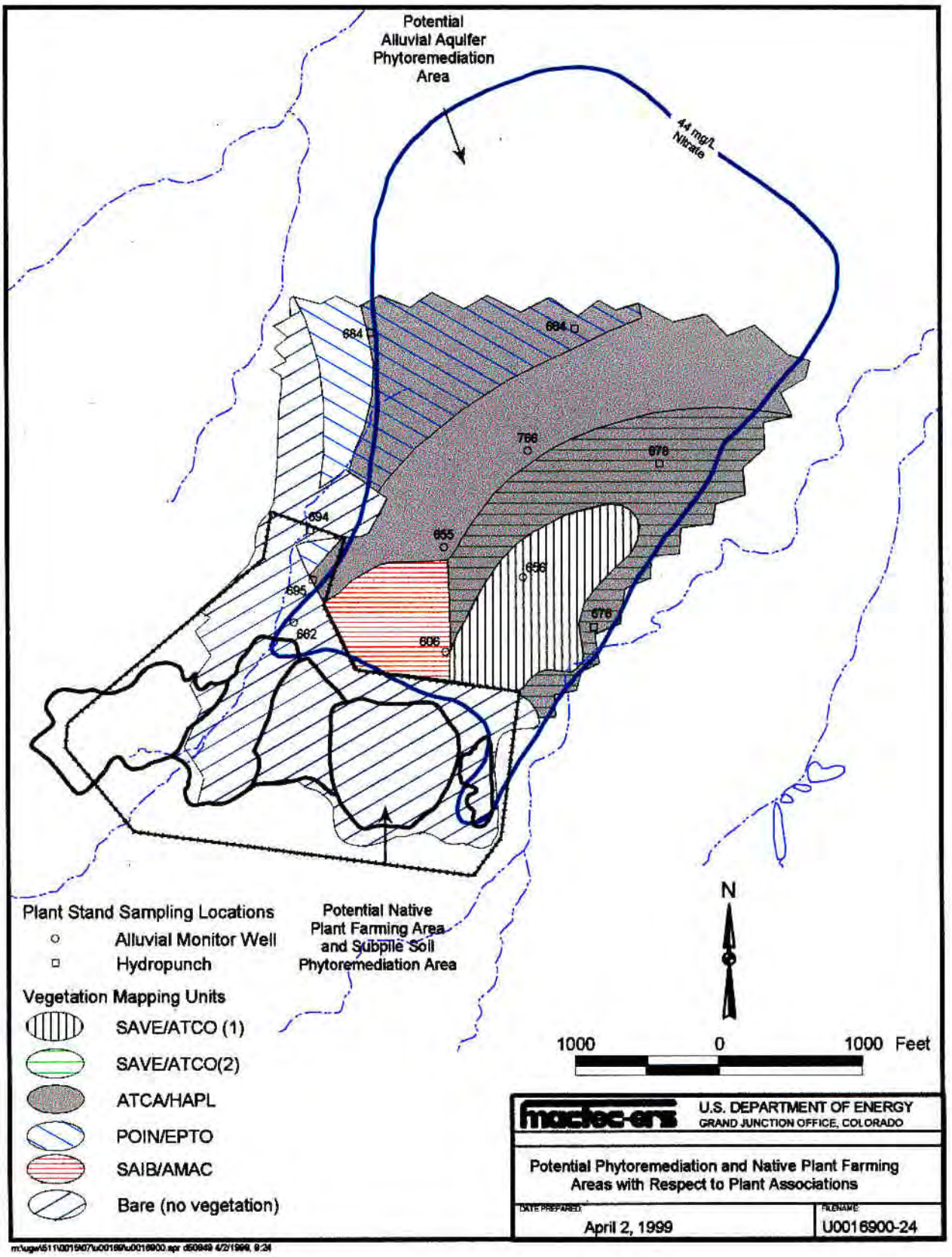

Figure 5–43. Potential Phytoremediation and Native Plant Farming Areas with Respect to Plant Associations

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saltbush may accelerate population expansion into other disturbed (bladed and overgrazed) areas of the plume; and (2) relatively high productivity (and nitrate uptake) is possible if plants are protected from grazing.

Phytoremediation of ammonium and nitrate in subpile soil may also be feasible. High ammonium and nitrate occurred in one hand-augered soil boring (boring 866, Section 5.3.2.2) in the northern portion of the new tailings pile area (Section 4.5). While ammonium is not a COPC, ammonium in subpile soil may generate nitrate that could enter the ground water system. Successful revegetation of subpile soils with a diverse mixture of deeper-rooted shrubs and shallower-rooted grasses and forbs may control this potential source of ground water nitrate (Schnoor et al. 1995). Vegetation in good condition would extract ammonium and nitrate and create a favorable soil water balance. In arid ecosystems, ET from healthy vegetation can prevent leaching of contaminated soil (Weand and Hauser 1997).

Native Plant Farming Feasibility

DOE is also considering the feasibility of a native plant farming alternative as a component of the cleanup strategy for Monument Valley. The concept involves pumping the nitrate-contaminated alluvial aquifer for a productive use, specifically, to irrigate planted areas. In this way, *nitrate the ground water contaminant* becomes *nitrate the fertilizer* as it is taken into the nitrogen biogeochemical cycle of the terrestrial ecosystem (see background information section above).

Arable land classification in arid regions is primarily concerned with soil depth, soil water retention, soil permeability, soil chemistry (salinity, sodicity, and alkalinity), percent coarse fragments, and topography. Soils overlying the plume and in revegetation areas at Monument Valley range from a loamy sand, with about 70 percent fine sand, 25 percent silt, and less than 5 percent clay, to dune sand with greater than 90 percent fine sand, less than 5 percent silt, and virtually no clay (Table 4–10). The “field capacity” of these soils should fall between about 7 and 12 percent volumetric water content (e.g., Brady 1974). The permeability of these soils averages about 1.0×10^{-4} cm/s (Table 4–9). These soils are deep, have very few coarse fragments, and slopes do not exceed 8 percent. Salinization would not be expected for these deep, coarse-textured soils under normal irrigation practices.

Overall, based on an arable land classification system used by the Navajo Nation and the BIA (Appendix E), the soils in the plume and revegetation areas do not fall into the highest class but are suitable for irrigation of a native forage crop (e.g., Glenn et al. 1998).

Before a native plant farming process could be implemented, however, a number of areas of uncertainty would have to be addressed. Uncertainties associated with well installation and production rates are addressed in Section 8.0. Ecological uncertainties which would have to be addressed before a native plant farming system could be implemented include the following:

- Soil salinization from high sulfate and other soluble salts in the plume water,
- Effects of high nitrate and sulfate levels on plant survival, species composition, species diversity, plant abundance, and productivity of planted areas, and

- Feasibility of managing the soil water balance and limiting deep leaching of nitrates and sulfates.

The economic and process feasibility of the native plant farming process are evaluated in Section 8 of this SOWP. No range improvements will be implemented without prior approval of the Navajo Nation and local residents of Cane Valley.

6.0 Risk Assessment

The 1996 BLRA evaluated the potential public health and environmental risks of ground water contamination caused by activities at the former Monument Valley processing site (DOE 1996b). Since the completion of this document, additional site characterization data, new toxicity data on human health, and results of a study on plant uptake are now available to update the BLRA. Results of the update indicate that nitrate, sulfate, and uranium should be retained as contaminants of concern (COCs) based on risks to human health due to ingestion of ground water. The update of the ecological risk assessment showed that no adverse effects to plants can be expected from use of plume water for irrigation. This section summarizes the results of the original BLRA and then provides the human health and ecological risk assessment updates based on the more recent information. More detailed discussion of the human health risk assessment update can be found in Appendix F.

6.1 Human Health Risk Assessment

As with other UMTRA ground water sites, the BLRA serves as the basis for risk information. Sections 6.1.1 and 6.1.2 summarize the results of the BLRA for Monument Valley. The BLRA was based on characterization data collected from 1988 to February 1993. More recent analytical results are available for this site, and they provide an opportunity to evaluate the effect of changing site conditions on risks. Section 6.1.3 updates the BLRA using the more recent analytical data.

Ground water is the sole source of domestic water (water used for drinking, bathing, and other household purposes) in the Monument Valley area. Domestic well users upgradient of the site and several domestic wells downgradient of the site have not been impacted by the contaminant plume. The depth to ground water is shallow in some areas near the site (i.e., Cane Valley Wash). In the areas where ground water reaches the surface, ponds form that are accessed by people and animals. However, these areas have not been impacted by the contaminated ground water; therefore, complete exposure pathways to contaminated ground water discharging to the surface water and sediment do not exist.

Currently, no one uses the contaminated ground water and, therefore, there are no complete exposure pathways or human health risks for ground water. However, future use of the contaminated ground water is possible. Evaluation of potential future use indicates certain health risks could occur if the contaminated ground water (i.e., ground water within the most contaminated portion of the plume) were ingested as drinking water.

6.1.1 Contaminants of Potential Concern

COPCs are contaminants that could cause adverse health effects if taken into the body. COPCs for human health were selected on the basis of comparison to background data, acceptable nutritional and dietary ranges, and toxicity benchmarks. A constituent was placed on the initial list if it was detected in concentrations that exceeded background levels in monitoring wells at the 0.05 level of significance and if the site is a likely source for the contaminant. Fifteen chemicals were identified in concentrations above background: ammonium, calcium, chloride, iron,

magnesium, manganese, nitrate, potassium, silica, sodium, strontium, sulfate, uranium, vanadium, and zinc.

This initial list of site-related constituents was further evaluated for toxicity to human health using the health-based criteria. Several constituents, although present above background, were screened from the list because their concentrations are within an acceptable nutritional rate or at a level of low toxicity and relatively high normal dietary intake compared to the value detected. Chloride, iron, potassium, and zinc were removed from the list because they are found at acceptable nutritional requirement levels. Constituents considered to be of low toxicity and/or high normal dietary intake that were eliminated from the list are ammonium, calcium, magnesium, silica, and sodium. The effect of these constituents on persons with diabetes was not evaluated separately, although this may be a sensitive subpopulation within the Navajo Nation.

All remaining constituents on the list were considered COPCs because of their potential toxic effects if ingested. These contaminants were evaluated quantitatively in the risk assessment: nitrate, sulfate, manganese, strontium, vanadium, and uranium. Of these contaminants, uranium is the only carcinogen.

Five exposure routes were evaluated for their potential adverse health effects: ground water ingestion and dermal contact, the ingestion of garden produce irrigated with contaminated ground water, and the ingestion of milk and meat from livestock watered with ground water from the contaminated aquifer. Toxicity due to exposure from these exposure routes was evaluated by calculating the exposure dose, using the upper 95 percent confidence limit of the mean concentration in ground water. The significant risk contributors by exposure route were identified by calculating a ratio of these exposure doses to the exposure dose from ground water ingestion (as drinking water). Ground water used as a sole source of drinking water was determined to contribute the most significant potential future risk relative to the other exposure pathways. Therefore, only the ingestion of contaminated ground water was further evaluated in the BLRA.

Probability distributions for contaminant concentrations and exposure variables were integrated to estimate the range of contaminant exposure doses people could ingest from a hypothetical well constructed in the most contaminated portion of the plume. Filtered water quality data from 1988 to 1993 from DOE monitor well MON–655 were used to evaluate on-site levels of contaminants in the alluvial aquifer. The estimated amounts of contaminants a child could ingest through drinking water were compared to toxic effects anticipated for each contaminant at these amounts. Children (1 to 10 years) were evaluated for these exposure scenarios because children consume relatively more water than adults and, consequently ingest a higher contaminant dose than adults. However, when a subpopulation was identified as more sensitive to exposure to certain contaminants, that population was evaluated. For the Monument Valley site, infants have been identified as the population most sensitive to sulfate and nitrate. Adults were evaluated for the carcinogen uranium, because carcinogens are evaluated over a lifetime of 70 years.

6.1.2 Potential Public Health Impacts

The most serious noncarcinogenic health risks from ingesting ground water at this site would result from nitrate. Some degree of methemoglobinemia (the interference of the oxygen-carrying capacity of the blood) could occur with any infant consumption of nitrate in ground water at this site. Severe diarrhea in infants could be expected from sulfate exposure. Manganese levels could cause mild neurological symptoms such as irritability and speech disturbances. More than 99 percent of the strontium and vanadium exposures fell below any levels where any noncarcinogenic toxic effects have been observed in humans. The entire range of uranium exposures fell below any level where noncarcinogenic toxic effects have been observed in humans.

6.1.3 BLRA Update

Risk is estimated by evaluating concentrations of the contaminants in ground water exposure (exposure factors such as the number of liters ingested per day of drinking water) and the toxicity of the COPCs. The purpose of this update is to evaluate changes in these areas (if any) and then evaluate if all the COPCs should be retained and the potential overall impact to human health.

Additional analytical results (contaminant concentrations) became available after the original BLRA (DOE 1996b) was completed. Table 6–1 presents a comparison of the analytical results used in the BLRA to the more recent analytical results.

Table 6–1. Comparison of New Data with Data Used in the BLRA at Downgradient Wells (mg/L)

BLRA COPCs	UMTRA MCL	BLRA Downgradient Data[a]		Most Recent Downgradient Data[b]		Ratio of Median of Recent Data to Median BLRA Data	Trend of Median
		Median	Maximum	Median	Maximum		
Manganese	NA	0.13	0.17	0.19	0.31	1.5	Increase
Nitrate	44	1,140	1,600	1,190	1,590	1.04	Increase
Strontium	NA	3.0	3.2	2.55	2.88	0.85	Decrease
Sulfate	NA	2,660	3,540	1,450	2,140	0.73	Decrease
Uranium	0.044	0.028	0.031	0.026	0.027	0.93	Decrease
Vanadium	NA	0.01	0.08	0.01	0.013	1.0	No Change

[a]The BLRA used data from well MON–655 or MON–606.
[b]Nitrate, strontium, sulfate, and uranium are from well MON–655, while nitrate and manganese are from well MON–606.The most recent data includes any data collected after February 1993 (the date of the last sampling round used in the BLRA) to the present.

Table 6–1 shows that manganese and nitrate show an increasing trend in the median concentration, vanadium has not changed, and strontium, sulfate, and uranium are decreasing. However, concentrations for all COPCs using the more recent data have remained similar to the concentrations listed in the BLRA.

Since the completion of the BLRA, several new wells have been added to better characterize the plume. In some cases, new data from these new wells have yielded higher concentrations than the data from the wells used in the BLRA. A comparison of the BLRA data and the most recent data from all wells (see Table 5–8 for a complete data summary) is presented in Table 6–2.

Table 6–2 shows higher maximum concentrations have occurred in the new wells for strontium, sulfate, uranium, and vanadium. In all cases, concentrations have increased by a factor of five or less. Maximum concentrations for manganese and nitrate are still occurring in well MON–606.

Table 6–2. Sampling Results Using All Available Wells (mg/L)

BLRA COPCs	Well used in BLRA	Maximum Concentration from Most Recent Data[a]	Well with Maximum Concentration	Maximum Concentration from Most Recent Sampling[b]
Manganese	MON–606	0.31	MON–606	0.21
Nitrate	MON–606	1,590	MON–606	1,030
Strontium	MON–655	2.88	MON–771	4.95
Sulfate	MON–655	2,140	MON–771	3,540
Uranium	MON–655	0.027	MON–774	0.069
Vanadium	MON–655	0.0130	MON–772	0.0540

[a]This is the maximum concentration from data collected between March 1993 and August 1997.
[b]These are data collected from the most recent round of sampling (August 1997).

The BLRA used standard default assumptions. These same assumptions are still valid and are commonly used to evaluate risks to human health.

Some toxicity values have been updated since the BLRA was completed. Table 6–3 lists the toxicity values used in the BLRA vs. the recent data from the Integrated Risk Information System (IRIS) (EPA 1998). IRIS, which is managed by EPA, provides the most up-to-date toxicity information. No changes have occurred for most COPCs; however, toxicity values for manganese and nitrate have been updated.

Table 6–3. Comparison of Toxicity Benchmarks for Noncarcinogens[a]

COPC	Target Organ or Effect	Reference Dose Used in Original BLRA	Most Recent Value from IRIS[b]	Impact on Risk
Manganese	Central nervous system	0.005	0.014	Decrease
Nitrate	Blood	7.0	7.0	No change
Strontium	Bone	0.6	0.6	No change
Sulfate	Gastroenteritis	NA	NA	NA
Uranium[c]	Body weight loss, moderate nephrotoxicity (kidney)	0.003	0.003	No change
Vanadium	Liver, kidney, nervous system, cardiovascular	0.007	NA[d]	Assumed no change

[a]The toxicity benchmark for these compounds is the reference does or RfD. Risk (expressed as a hazard quotient) is estimated by dividing intake (mg/kg-day) by the RfD (mg/kg-day). Therefore, as the RfD increases, the risk decreases.
[b]Obtained from IRIS on February 24, 1998.
[c]This is for elemental uranium, which is a noncarcinogen (does not cause cancer). However, the isotopes of uranium cause cancer, which is not accounted for in this comparison of RfDs.
[d]IRIS does not provide an updated value for vanadium.

6.1.4 Impacts on the COPC List

Because of the changes in contaminant concentrations and toxicity benchmarks, it is useful to evaluate if all the COPCs listed in the BLRA should be retained. After this final screening of COPCs they will be referred to as contaminants of concern (COCs). To provide a consistent update to the BLRA, only data comparisons from the same wells used in the BLRA are evaluated. However, the data from the more recent wells will also be evaluated when elimination of a COPC is being considered.

The compounds that show a decrease in downgradient concentration data from the wells used in the BLRA are sulfate, uranium, and strontium. IRIS does not list toxicity information for sulfate and it is found at relatively high concentrations; therefore, it must be retained as a COC. Uranium showed a minor decrease in median concentration, it is clearly associated with uranium milling, and causes both noncancer and cancer effects. For these reasons, uranium is retained as a COC. Strontium showed decreasing concentration and is not normally associated as a contaminant from uranium milling. Although the median concentration of vanadium did not change, the maximum concentration decreased. Therefore, this could possibly be eliminated as a COC. To evaluate if strontium and vanadium should be retained as COCs, risks associated with strontium were reevaluated using standard equations and assumptions from the BLRA and EPA (1989). Evaluations for strontium and vanadium are presented in Appendix F.

Manganese is the only COPC that had a change in toxicity that may result in it being eliminated from the final COC list. Appendix F evaluates if manganese should be retained as a COC.

Appendix F shows that the elimination of manganese, strontium, and vanadium from the COPC list will have little impact on the total site risks. Therefore, it is appropriate to eliminate those compounds from the final COC list. Therefore, the final COCs are nitrate, sulfate, and uranium.

6.1.5 Updated Impacts on Risks from Ingestion of Ground Water

Nitrate—The concentrations of nitrate in well MON–655 (defined as the plume in the original BLRA) have increased slightly, but are very similar to the concentrations listed in the BLRA.

However, EPA has revised the reference dose (RfD) to indicate that nitrate is more toxic than previously thought (at the time of the BLRA). The revised hazard quotient (HQ) for nitrate is 28.4, which is significantly greater that the other quantifiable COCs.

Sulfate—IRIS (EPA 1998) still does not list an RfD for sulfate. The Navajo Nation has proposed a standard of 250 mg/L for sulfate based on the EPA's guidelines for taste, odor, and color. This will be combined with the use of the sulfate-to-chloride ratio, to distinguish between areas of natural high sulfate levels and areas having elevated sulfate due to milling activities, as a cleanup goal for the Monument Valley project. See Section 8.1.2.1 for additional information.

Uranium—Uranium causes both noncarcinoginic and carcinogenic effects. The concentration in the plume (defined as well MON–655 in the BLRA) shows a decreasing trend. Using the recent maximum concentration, uranium has an HQ of 0.25.

Since the completion of risk assessment the carcinogenic toxicity factor for uranium has changed. Based on this change, the increased individual lifetime cancer risk for uranium in the alluvial aquifer is estimated to be 2E–05, or 2 chances in 100,000 of developing cancer. For the maximum concentration of uranium detected in the De Chelly Sandstone (0.13 mg/L), the estimated risk level is 1E–04, or 1 chance in 10,000 of developing cancer. The estimated risk levels fall into the upper end of EPA risk range for carcinogens of 1E–04 to 1E–06 (1 chance in 10,000 to 1 chance in 1,000,000 of developing cancer). EPA's risk range is used for Comprehensive Environmental Response, Compensation, and Liability Act sites. Actions to mitigate risks within the risk range are generally within the discretion of the risk managers, based on site-specific factors (40 CFR Part 300).

6.1.6 Ingestion of Contaminated Plants

The BLRA recommended that the potential contaminant uptake by plants rooted into ground water or irrigated with ground water be investigated further. This corresponds to two potential human exposure pathways:

(1) Ingestion of produce irrigated with water pumped from the contaminated aquifer, and

(2) Cultural uses of plants rooted into or irrigated with contaminated ground water.

This subsection evaluates these pathways.

6.1.6.1 Ingestion of Contaminated Produce

Residents living near the Monument Valley site use well water to irrigate vegetable gardens. Currently, no irrigation wells access contaminated ground water. Plant uptake of contaminants from irrigation water and subsequent ingestion of contaminated produce is a potential future exposure pathway. The BLRA (DOE 1996b), lacking literature on contaminant uptake by garden vegetables, did not evaluate this pathway.

The University of Arizona's Environmental Research Laboratory conducted a 2-year study to acquire plant uptake data needed for risk assessments at UMTRA sites (Baumgartner et al. 1996). The study started with a synthesis of pertinent literature concerning uptake and interactive effects of metals, nitrates, and sulfates. Overall, the literature review found that water-to-plant and soil-to-plant concentration ratios for some UMTRA constituents are highly variable and dependent on plant species and soil and water chemistry. DOE funded greenhouse studies to evaluate concentration ratios for manganese, molybdenum, selenium, and uranium, and to test metal uptake responses to a range of nitrate concentrations. The greenhouse studies progressed through two phases: tests with simulated ground water contamination, and tests using actual ground water from the Tuba City, Arizona, site (Baumgartner et al. 1996). Some of the results were applicable to the Monument Valley risk assessment.

Table 6–4 summarizes an evaluation of the produce ingestion pathway at Monument Valley. The list of human health COPCs matches the BLRA (DOE 1996b), except for vanadium. The maximum background concentration exceeded the alluvial aquifer concentration, therefore,

vanadium was removed from the list. Estimates of maximum crop levels of COPCs are used for a screening assessment of the produce ingestion pathway.

Table 6–4. Estimated Maximum COPC Concentrations in the Alluvial Aquifer and in Irrigated Produce for the Produce Ingestion Pathway at Monument Valley

COPC[a]	Max GW Level[b] (mg/L)	Max Level Well	Max Crop Level[c] (ppm)	Notes
Manganese	0.21	MON–606	<50.0	The maximum crop level is inferred from the response function for Mn uptake in a root crop (Baumgartner et al. 1996).
Nitrate	1,600	MON–606	<5,000	Baumgartner et al. (1996) indicate that crop tissue levels of NO_3 peak at about 1,000 mg/L of NO_3 in irrigation water; crop tissue levels dropped at about 2,000 mg/L NO_3 in water. These crop levels are comparable to levels in fertilized produce (Brown and Smith 1966; Peck et al. 1971).
Strontium	4.95	MON–771	<5.0	The expected plant-to-soil concentration ratio for Sr is less than 1 (Kabata-Pendias and Pendias 1992).
Sulfate	3,540	MON–771	<5,000[d]	The expected maximum crop level is for total S, not sulfate (Baumgartner et al. 1996).
Uranium	0.069	MON–774	<0.1	The maximum crop level is inferred from the literature search and the response function for U uptake in root crop derived in the greenhouse (Baumgartner et al. 1996).

[a] The list of COPCs, minus vanadium, is from the human health evaluation, Table 3.8 in DOE (1996b). On-site levels of vanadium did not exceed background level and it was removed from the COPC list.
[b] Maximum ground water level for nitrate is from 1988–1993 data (Table 3.7 in DOE 1996b) all others from 1997 data (Table 5–8).
[c] Maximum crop concentrations are estimates based on either the literature review or the plant uptake study by Baumgartner et al. (1996).
[d] Total sulfur in crop tissue.

6.1.6.2 Update on Potential Exposure to Garden Produce

The original BLRA did not evaluate ingestion of contaminated garden vegetables because information on plant uptake was not available. However, based on the recent University of Arizona study (see Section 5.4.1.1) maximum contaminant concentrations in crops are now available (see Table 6–4). Using these data and standard exposure factors from the literature, a screening-level evaluation of potential risks from this pathway was conducted (see Appendix G). The results are summarized below:

Contaminant	Maximum Crop Concentration (mg/kg)	Hazard Quotient
Manganese	<50	<0.26
Nitrate	<5,000	<2.2
Strontium	<5.0	<0.006
Uranium	<0.1	<0.02

The ingestion of contaminated produce is a minor contributor to total risks compared to ingestion of contaminated ground water (risks from ingestion of contaminated produce are 2.5 percent of

the risks from contaminated water using standard default exposure factors and assuming the same concentrations). However, the ingestion of garden produce can become important when contaminants bioconcentrate in the edible portion of produce. The only contaminant that appears to significantly bioconcentrate is manganese. This is the only contaminant that has a higher HQ from ingestion of garden produce than ingestion of ground water; however, the sum of these HQ's is still less than 1.0.

Of the contaminants that could be quantified as part of the food ingestion pathway, only nitrate exceeded 1.0. Nevertheless, the food ingestion pathway results in a HQ of less than 10 percent of the drinking water ingestion pathway (2.2 vs. 28.5). However, this serves to confirm the importance of nitrate as a dominant COC.

An HQ for sulfate could not be quantified. However, only limited bioconcentration is occurring

$$\frac{5{,}000\ \ mg/kg\ \ of\ \ plant\ \ tissue}{3{,}540\ \ mg/kg\ \ of\ \ water}\ \ or\ <1.4$$

Therefore, the ingestion of food crops (using irrigation water from the contaminated aquifer) will not add significantly (less than 4 percent) to the total risks for sulfate (1,500 to 2,000 ppm). The risk range identified by EPA Region VIII did not include ingestion of garden vegetables irrigated with contaminated ground water. However, because of the safety factors associated with the EPA estimate and the limited contribution from ingestion of garden produce, the EPA Region VIII guideline will still be protective of human health.

6.1.6.3 Cultural Uses of Native Plants

The Navajo people traditionally gather wild plants for many purposes in everyday life; for food, for medicine, for religious ceremonies, and for tools. Many plant uses are prescribed through century-old rituals. Others are adaptations to more recent changes in the landscape and vegetation. Some wild plants of the semiarid plateau country, called phreatophytes, send their roots into shallow aquifers. If used for cultural purposes at Monument Valley, phreatophytes could potentially access and bioaccumulate ground water contaminants. Bioaccumulation could also occur in shallow-rooted wild plants if contaminated ground water were pumped for irrigation, inadvertently as part of a rangeland improvement project, or as a consequence of the native plant farming alternative. Traditional uses of phreatophytes or plants irrigated with plume water could potentially cause adverse human exposure. People who use contaminated plant tissues for food, medicine, and ceremonial burning could be exposed via ingestion, inhalation, and dermal pathways. The BLRA (DOE 1996b) did not evaluate these pathways because site-specific plant ecology and plant uptake data were not available.

Table 6–5 lists and describes cultural-use plants growing over contaminated ground water at the Monument Valley site. The list includes Navajo and English common names as well as scientific nomenclature. The list is a subset of the comprehensive plant list for the site (Table 4–18). The selection of species and descriptions of cultural uses were summarized from Mayes and Lacy (1989) and from discussions with the staff of the Navajo Nation's Historic Preservation

Department in Window Rock, Arizona. Only two species are likely rooted into the contaminated alluvial aquifer: *Sarcobatus vermiculatus* and *Atriplex canescens*. The remaining species would be exposed to contaminants only if irrigated.

Table 6–5. Navajo Cultural Uses of Plants Growing on the Reclaimed Tailings Area and Plume Area at the Monument Valley Site

Plant Name[a]	Cultural Uses[b]
Azee' ntł'iní (gummy medicine) Scarlet globemallow *Sphaeralcea coccinea* (Pursh) Rydb.	Medicinal: Azee' ntł'iní is one of the Navajo Life Medicines. Roots and leaves are pounded, mixed with water to form a sticky lotion and topically applied to stop bleeding sores on skin. Azee' ntł'iní is also used to make a tonic as a cure for colds and influenza. Ceremonial: Azee' ntł'iní is the principal medicinal tonic for Coyote Way, Beauty Way and Night Way ceremonies. Other: The top of the plant has been used as a beverage and dried as a tobacco.
Azee' hááłdzid (rotten medicine) Threadleaf groundsel *Senecio douglassii* DC.	Medicinal: A medicine made from Azee' hááłdzid has been used as a tea, as a poultice, and as a steam treatment for arthritis, boils, and rheumatism.
Ch'il deeníní (sharp plant) Russian thistle *Salsola iberica* Sennen and Pau	Medicinal: This adventive from Asia has been ashed to make a lotion and tonic for influenza and smallpox. Ceremonial: The ashed plant is used as a blackening in the Enemy Way and Evil Way ceremonies. Other: A green dye can be made from fermented young plants.
Ch'il diilyésiitoh (big dodge weed) Broome snakeweed *Gutierrezia sarothrae* (Pursh) Britt. and Rusby	Medicinal: Ch'il diilyésiitoh is one of the Navajo Life Medicines. The pulp of a plant that has been chewed is placed on insect bites and cuts. A medicine derived from the plant is given during childbirth. An ointment from ashes of the plant is rubbed on the forehead to cure headaches and fever. A tonic made from the root is used to treat stomach disorders and sometimes as an antidote for taking too much of other remedies. Ceremonial: Ch'il diilyésiitoh is a Life Way pollen. Ashes of the plant are used in almost all other ceremonies as a blackening. It is also used to make prayersticks and as a ceremonial fumigant and emetic. Other: Ch'il diilyésiitoh flowers are used to make a yellow dye.
Dá'ák'óózh deeníní (sharp saltbush) Shadscale *Atriplex confertifolia* (Torr. & Frem.) Wats.	Food: Leaves of Dá'ák'óózh deeníní are used to flavor corn roasted in a pit.
Dibéhaich'iidii (gray sheep scratch) Locoweed *Astragalus* species	Medicinal: As one of the Navajo Life Medicines, Dibéhaich'iidii is used as a diuretic, for stomach disorders, and to treat venereal disease. Ceremonial: Dibéhaich'iidii is a medicine used in the Bead Way ceremony.

Table 6–5 (continued). Navajo Cultural Uses of Plants Growing on the Reclaimed Tailings Area and Plume Area at the Monument Valley Site

Plant Name[a]	Cultural Uses[b]
Díwózhiiłbeii (gray greasewood) Fourwing saltbush *Atriplex canescens* (Pursh) Nutt.	Medicinal: The leaves of Díwózhiiłbeii are chewed to produce a poultice for insect bites. It is mixed with juniper mistletoe as a tonic for stomachaches and toothaches and to increase perspiration during sweatbaths. Leaves and roots are also used to produce cough medicine and a snuff for sinus pain. Ceremonial: Díwózhiiłbeii is used as an emetic in Evil Way and Navajo Wind Way ceremonies. Other: Díwózhiiłbeii was once used for food seasoning. Leaves and twigs, when combined with other ingredients, are used to intensify color in dyes.
Díwózhiishzhiin (black bushy shrub) Black greasewood *Sarcobatus vermiculatus* (Hook.) Torr.	Medicinal: Like Dá'ák'óózh deeníní and Díwózhiiłbeii, the leaves of Díwózhiishzhiin are chewed to produce a poultice for insect bites and stings. Ceremonial: The hard wood of Díwózhiishzhiin is used to make ádístsiin, sticks used to stir mush during Navajo weddings and girls' puberty rituals. The wood is also used to make equipment for the Lightening Chant and Mountain Chant ceremonies. Other: The hard wood was ideal for planting sticks, awls, knitting needles, and traps. Because the wood burns long and hot, like oak, it is used for cooking and to burn out soft cottonwood logs to make boxes.
Dlóóbibé'ézhóó' (prairie-dog comb) Purple threeawn *Aristida purpurea* Nutt.	Ceremonial: Dlóóbibé'ézhóó' leaves are part of a medicine used in the Enemy Way ceremony.
Gad ni'eełii bílátah łichí'ígíí (juniper with red flowers) Tamarisk *Tamarix ramosissima* Ledeb.	Medicinal: Gad ni'eełii bílátah łichí'ígíí is a naturalized adventive from Eurasia that looks like juniper and, although it is only remotely related, it is sometimes used as a substitute for juniper in certain healing tonics and smoke treatments.
Haashch'éédáá (god's food) Tomatilla *Lycium pallidium* Miers	Food: Red Haashch'éédáá berries are eaten raw or cooked. Medicinal: Dried Haashch'éédáá berries and roots are a Navajo Life Medicine. The ground root is used to relieve toothache. Ceremonial: Haashch'éédáá is used to form equipment and as an emetic in the Evil Way and Female Shooting Way ceremonies.
Hosh niteelí (broad cactus) Prickly-pear *Opuntia phaeacantha* Engelm.	Food: Hosh niteelí fruit was eaten fresh, dried, or cooked in stews. Medicinal: Peeled fleshy stems of Hosh niteelí are bound over cuts to stop bleeding. Ceremonial: The Navajo origin myth includes Cactus People, consequently Hosh niteelí is used in several ceremonies.
Nididlídii (scorched) Indian ricegrass Oryzopsis hymenoides (R. & S.) Ricker.	Food: In the past, Nididlídii was used to make mush or ash cakes. Seeds were scorched to remove the chaff, ground, mixed with milk or water and cooked.
Tłéé' łigahiits'óóz (white at night) Evening primrose *Oenothera albicaulis* Pursh	Medicinal: Tłéé' łigahiits'óóz is used to produce a lotion for boils, mixed with other herbs to treat kidney ailments, and ground to produce a dusting powder on sores. It is also an ingredient of a poultice for spider bites. Ceremonial: Tłéé' łigahiits'óóz is an ingredient of various medicines in Bead Way, Big Star Way, Red Ant Way, and Blessing Way ceremonies.

Table 6–5 (continued). Navajo Cultural Uses of Plant Growing on the Reclaimed Tailing Area and Plume Area at the Monument Valley Site

Plant Name[a]	Cultural Uses[b]
Tł'ochin nilchiin (smelly onion) Wooton sandverbena *Tripterocalyx carneus* (Greene) Galloway	Medicinal: Tł'ochin nilchiin is used as part of a medicine to treat internal injuries. It is also made into a tea to treat gallbladder and gallstone problems. Other: Tł'ochin nilchiin was sometimes carried as a good luck charm for protection.
Tł'oh łigaii (grass white) White goosefoot *Chenopodium album* L.	Food: Young Tł'oh łigaii were traditionally eaten raw or cooked with other foods. Mature plants were threshed and the seed were ground and either mixed with corn meal or mixed with milk or water to make ash cakes. Ceremonial: Tł'oh łigaii is eaten during the Night Way ceremony.
Tł'oh łichí'í (red grass) Galleta *Hilaria jamesii* (Torr.) Benth.	Medicinal: A tea is made from Tł'oh łichí'í and given to babies to help them grow into strong adults.
Tł'oh azihii łibáhígíí (gray rubbing grass) Joint-fir *Ephedra torreyana* S. Wats.	Food: Branches of Tł'oh azihii łibáhígíí are roasted and boiled to produce a tea. Medicinal: Tł'oh azihii łibáhígíí is used as a diuretic to treat kidney and bladder problems, venereal disease, and afterbirth pains
Tsá'ázi'ts'óóz (narrow yucca) Narrowleaf yucca *Yucca angustissima* Engelm	Medicinal: Roots of Tsá'ázi'ts'óóz are soaked in water and the liquid is strained and given to women having a long labor. Ceremonial: Many parts of the Tsá'ázi'ts'óóz plant are used for ceremonies: a soap made from the roots is used to wash and purify ceremony participants, ceremony equipment is lashed using fiber from the leaves, and juice from heated leaves is a pigment base for paint. Other: Soap from Tsá'ázi'ts'óóz roots is also used to wash wool for weaving. Various parts of the plant are ingredients in dyes.
Zéé'iilwo'ii (god's plume) Cheatgrass brome *Bromus tectorum* L.	Medicinal: Zéé'iilwo'ii, a native of Eurasia but now circumboreal, arrived in the Four Corners area in the late Nineteenth Century. It has since been incorporated in several ceremonies: as plumage in the Night Way chant, as a blackening in the Evil Way and Handling Trembling Way, and as a medicine in the Night Way and Plume Way.

[a]The Navajo name and English interpretation, in parentheses, are given first, followed by the English common name. The scientific nomenclature for genera, species, and authorities, given third, is consistent with Voss (1983) and the choices of Welsh et al. (1987).
[b]Descriptions of cultural uses are summarized from Mayes and Lacy (1989) and discussions and correspondence with the staff of the Navajo Nation's Historic Preservation Department.

Table 6–6 summarizes estimates of maximum concentrations of COPCs in cultural-use plants. The estimates rely in part on the greenhouse study of contaminant uptake in *Atriplex canescens* (fourwing saltbush). *A. canescens*, as a halophyte (salt tolerant plant), accumulates salts to maintain favorable osmotic potentials and therefore would be expected to have elevated ion concentrations compared to other cultural-use species in the area; *A. canescens* is a conservative proxy for other cultural-use plants at the site. Estimates of maximum plant levels of COPCs given in Table 6–6 are used for a screening assessment of the cultural-use pathway (Section 6.1.6.4).

Table 6–6. Maximum Constituent Levels in the Alluvial Aquifer and Estimated Maximum Levels in Cultural-use Plants Growing over the Plume

COPC[a]	Max GW Level[b] (mg/L)	Max Level Well	Max Plant Level[c] (ppm)	Notes
Manganese	0.21	MON–606	<60.0	The maximum plant level of is inferred from the results of greenhouse tests of Mn uptake in fourwing saltbush (Baumgartner et al. 1996).
Nitrate	1,600	MON–606	<5000	The maximum plant level is inferred from the results of greenhouse tests of NO_3^- uptake in fourwing saltbush (Baumgartner et al. 1996). Tissues levels of NO_3^- were not significantly influenced by levels of NO_3^- in irrigation water.
Strontium	4.95	MON–771	<5.0	The expected plant-to-soil concentration ratio for Sr is less than 1 (Kabata-Pendias and Pendias 1992).
Sulfate	3,540	MON–771	<7000[d]	The maximum plant tissue level is total S, not sulfate, for fourwing saltbush from the greenhouse study (Baumgartner et al. 1996).
Uranium	0.069	MON–774	<0.3	The maximum crop level is inferred from the literature search and the response function for U uptake in fourwing saltbush derived in the greenhouse (Baumgartner et al. 1996).

[a] The list of COPCs, minus vanadium, is from the human health evaluation, Table 3.8 in DOE (1996b). On-site levels of vanadium did not exceed background level, so it was removed from the COPC list.
[b] Maximum ground water level for nitrate is from 1988 to 1993 data (Table 3.7 in DOE 1996b); all others from 1997 data (Table 5–8).
[c] Maximum crop concentrations are estimates based on either the literature review or the plant uptake study by Baumgartner et al. (1996).
[d] Total sulfur in crop tissue.

6.1.6.4 Update on Potential Exposure to Cultural-Use Plants

Potential exposures to cultural use plants can not be quantified without better understanding the exposure factors associated with their use (e.g., amount ingested, frequency of use). In this case, specific information from the actual users is needed because literature values or estimates are not available. A review of Table 6–6 indicates anticipated maximum concentrations in cultural-use plants are similar or slightly elevated over levels found in garden produce. Assuming the same route of exposure (ingestion) with the same quantity of ingestion (50 grams per day), this would result in similar or slightly higher risks. Other exposure routes (inhalation and dermal) would likely result in lower risks (although inhalation risk could become important depending on the circumstances [e.g., sweat baths, smoking]). Overall, it is anticipated that cultural uses will result in lower risks than ingestion of contaminated produce.

6.2 Ecological Risk Assessment

The Monument Valley BLRA (DOE 1996b) identified the following data needs to complete evaluations ecological risks associated with contaminated ground water:

- lateral extent and magnitude of ground water contamination,

- composition and structure of plant communities growing over the contaminant plume, and
- potential contaminant uptake by plants rooted into ground water or irrigated with ground water.

Data from the 1997 site characterization (Section 4.0) and a plant uptake study conducted for DOE by the University of Arizona's Environmental research Laboratory (Baumgartner et al. 1996) were used to complete evaluations of two ecological exposure pathways:

- Toxicity to native plants rooted into contaminated ground water, and
- Toxicity to crop plants irrigated with contaminated ground water.

The BLRA (DOE 1996b) evaluated the toxicity of ecological COPCs to native plants by comparing concentrations in the Monument Valley plume with published screening-level benchmarks. Of the 14 COPCs considered, benchmark values were available for only 4. Irrigation of forage and garden plants is another hypothetical future use of contaminated ground water and was also incomplete in the BLRA.

Additional analytical results (Section 4.9) and the results of the University of Arizona's plant uptake study (Baumgartner et al. 1996) were used to develop a screening-level assessment of potential crop and range plant toxicity. Results of the 1997 ground water sampling were used to remove chloride, sodium, and vanadium from the list of ecological COPCs; maximum on-site levels of these constituents were lower than maximum background levels. The results of the University of Arizona's plant uptake study, coupled with site-specific data on soils, ground water quality, and agricultural species, help complete the evaluation of possible adverse effects to crop and range plants. Table 6–7 summarizes the phytotoxicity evaluation. Phytotoxic effects are often inferred from biomass productivity responses a range of contaminant concentrations. Overall, no adverse changes are expected in biomass productivity of crop and range plants that access or are irrigated with plume water containing the maximum measured contaminant levels.

Table 6–7. Maximum Levels of Ecological COPCs in the Alluvial Aquifer, Estimated Maximum Levels in Crop and Range Plants, and Screening Benchmark Levels

COPC[a]	Max GW Level[b] (mg/L)	Max Level Well	Screening Benchmark (mg/L)[c]	Notes
Ammonium	360	MON–606		
Calcium	559	MON–771	NA	Although Ca is not phytotoxic, it inhibits or stimulates the absorption of other trace elements (Kabata-Pendias and Pendias 1992).
Iron (unfiltered)	10.2	MON–606	10	The screening benchmark is from Will and Suter (1995).
Magnesium	600	MON–771	NA	Mg is not phytotoxic but can either inhibit or stimulate the absorption of other trace elements (Kabata-Pendias and Pendias 1992).
Manganese (unfiltered)	0.21	MON–606	4.0	Will and Suter (1995) suggest 4.0 mg/L as a screening benchmark for Mn; Baumgartner et al. (1996) measured no change in crop or native plant biomass as high as 12.4 mg/L Mn.
Nitrate	1,600	MON–606	2,000	Baumgartner et al. (1996) measured a drop in crop (Sudan grass) and range plant (fourwing saltbush) biomass at GW NO_3^- levels >2,000 mg/L.
Potassium	50.1	MON–771	NA	K is not phytotoxic but it can inhibit absorption of other trace elements (Kabata-Pendias and Pendias 1992).
Strontium	4.95	MON–771	NA	Kabata-Pendias and Pendias (1992) report that no evidence exists suggesting that stable Sr in the biosphere (>1,000 mg/kg is some soils) are phytotoxic.
Sulfate	3,540	MON–771	NA	Sulfur toxicity in plants is caused by atmospheric sulfuric acid deposition and occasionally by acid sulfate soils (Pierzynski et al. 1993). Ground water pH levels indicate that acidity is not a problem.
TDS	5,800	MON–771	NA	
Uranium	0.069	MON–774	40.0	The screening benchmark is from Will and Suter (1995).

[a] The list of COPCs is for constituents that exceeded background concentrations.
[b] Maximum ground water levels for ammonium and nitrate are from 1988–1993 data (Table 3.7 in DOE 1996b) or 1997 data.
[c] NA indicates that standard benchmark concentrations are not available or not applicable.

7.0 Ground Water Compliance Strategy

Selection of the final strategy to achieve compliance with the EPA ground water protection standards is governed by the framework defined in the final PEIS for the UMTRA Ground Water Project (DOE 1996c). Stakeholders review and acceptance of the final PEIS is documented and supported by the Record of Decision (DOE 1997). Presented below is a discussion of how the selection process was used to determine the final ground water compliance strategy at the Monument Valley site and a proposed future ground water sampling and analysis plan to monitor compliance and the effectiveness of the selected remedy.

7.1 Compliance Strategy Selection Process

The PEIS framework used to determine the appropriate ground water compliance strategy for the Monument Valley site is summarized in the flow chart provided as Figure 7–1. The framework takes into consideration human health and environmental risk, stakeholder input, and cost. A step-by-step approach is followed until one, or a combination of one or more, of three available compliance strategies is selected. The three compliance strategies are:

- ***No remediation***—Compliance with the EPA ground water protection standards would be met without altering the ground water or cleaning it up in any way. This strategy could be applied at the Monument Valley site for those contaminants at or below MCLs or background levels or for those contaminants above MCLs or background levels that qualify for supplemental standards or ACLs as defined in Section 2.1.1.

- ***Natural flushing***—Allows natural ground water movement and geochemical processes to decrease contaminant concentrations to regulatory limits within a period of 100 years. The natural flushing strategy could be applied at the Monument Valley site if ground water compliance can be achieved within a 100 years or less, where effective monitoring and institutional controls can be maintained, and where the ground water is not and is not projected to be a drinking water source.

- ***Active ground water remediation***—Requires application of engineered ground water remediation methods such as gradient manipulation, ground water extraction, treatment, land application, phytoremediation, and in situ ground water treatment to achieve compliance with the standards.

7.2 Monument Valley Compliance Strategy

DOE is required by the PEIS to follow the ground water compliance selection framework summarized in Figure 7–1 in selecting the appropriate compliance strategy to clean up the ground water aquifers affected by former processing activities at the Monument Valley site. Three aquifers are known to exist at the site. Site-specific characterization data were used in combination with the PEIS flow chart presented in Figure 7–1 to select an appropriate compliance strategy for each aquifer. A discussion of the selected compliance strategy, and how

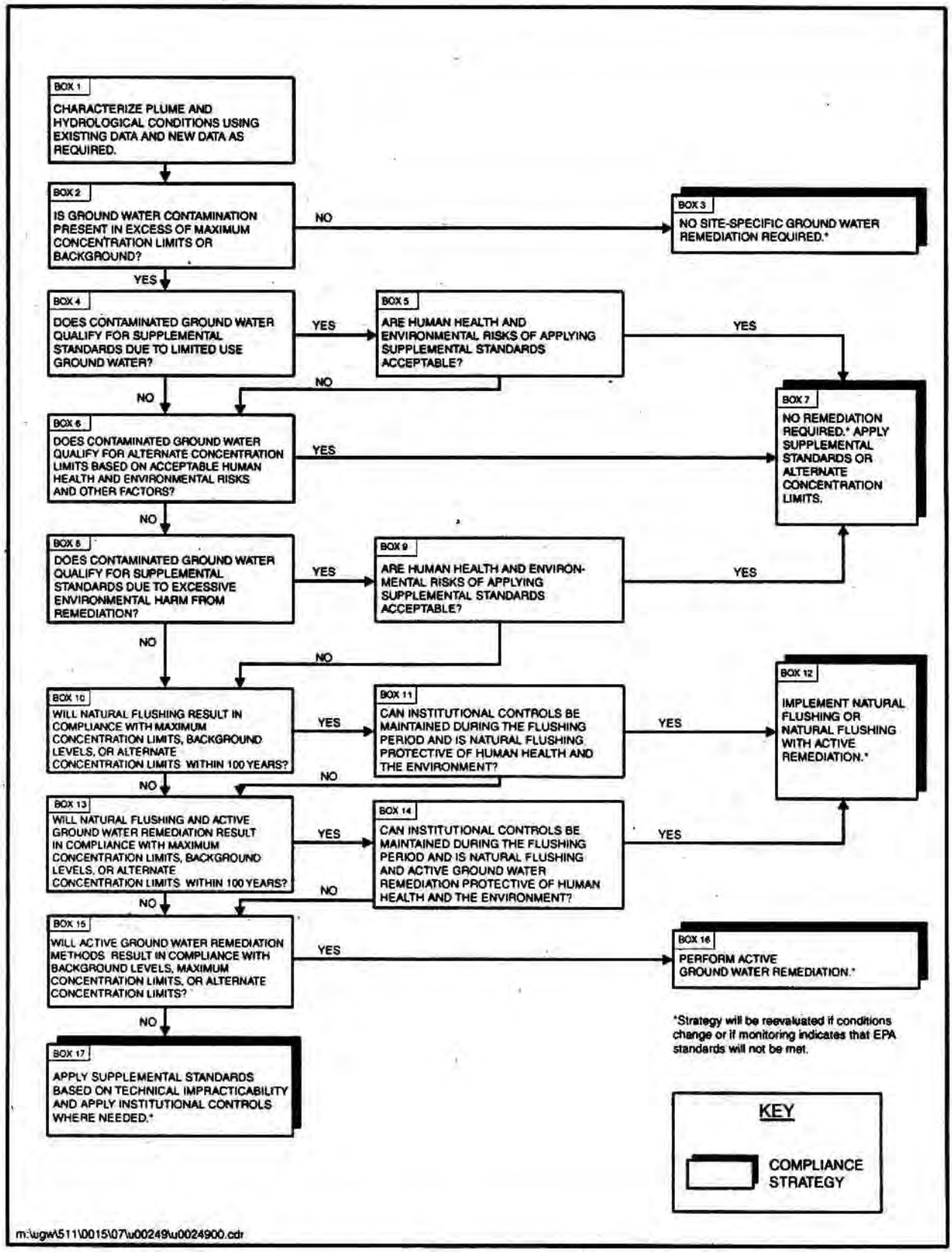

Figure 7–1. PEIS Compliance Strategy Selection Framework

the strategy was determined, is presented separately below for the alluvial, Shinarump, and De Chelly aquifer systems. Potential remediation technologies are evaluated in Section 8.0.

7.2.1 Alluvial Aquifer Compliance Strategy

DOE has determined that active remediation and natural flushing of the alluvial aquifer is the appropriate strategy for nitrate, sulfate, and uranium which are identified as COCs (Section 6.0). The proposed ground water compliance strategy is no remediation for ammonium, calcium, iron, magnesium, manganese, potassium, and strontium; constituents that do not pose a potential risk (Section 6.0) and do not exceed an EPA standard (Section 5.3.3.1). An explanation of how the targeted strategy was selected is presented in Table 7–1.

Table 7–1. Explanation of the Compliance Strategy Selection Process for the Alluvial Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See conceptual site model presented in Section 5.0. Move to Box 2.
2	Is ground water contamination present in excess of MCLs or background?	Nitrate and uranium exceed the MCL. Other site-related constituents (ammonium, calcium, iron, sulfate, magnesium, manganese, potassium, and strontium) exceed background levels. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	Alluvial ground water is not classified as limited use. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risk and other factors?	Ammonium, calcium, iron, magnesium, manganese, potassium, and strontium qualify for ACLs because they are dietary components that are present at levels that fall within nutritional ranges or because they occur at concentrations that do not pose a risk (Section 6.0). Move to Box 7 **- no remediation required -** for these constituents**.** Uranium and nitrate occur at concentrations above the MCL. Sulfate does not have an MCL, however concentrations occurs at levels that potentially can cause adverse health effects. Move to Box 8 for these constituents.
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	Although the applicability has not been formally assessed, it is unlikely that remedial action would cause excessive harm to the environment. Move to Box 10.

Table 7–1 (continued). Explanation of the Compliance Strategy Selection Process for the Alluvial Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
10	Will natural flushing result in compliance with MCLs, background, or ACLs within 100 years?	Uranium occurs at one isolated location near the former source area at a concentration that only slightly exceeds the MCL. Because the source has been removed and the uranium concentration is near the MCL, natural flushing is expected to reduce uranium to the MCL within 100 years. Move to Box 11. Given the relatively high levels and broad distribution of nitrate, and the estimated ground water velocities, active remediation appears to be the only viable compliance strategy to reduce nitrate to the MCL within the most contaminated portion of the plume. Move to Box 14. Given the broad distribution of concentrations of sulfate that exceed the proposed treatment goal (Section 6.0, and the estimated ground water velocities, active remediation appears to be the only viable compliance strategy to reduce sulfate to the treatment goal within the most contaminated portion of the plume. Move to Box 14.
11 and 14	Can institutional controls be maintained during the flushing period and is the compliance strategy protective of human health and the environment?	Although the final compliance strategy is protective of human health and the environment, DOE is also considering additional approaches to risk management at the site. One approach is to provide an alternate source of drinking water for residents living near the former mill site, even though the ground water is not currently used for a public water supply. Move to Box 12 **- implement natural flushing with active remediation.**

7.2.2 Shinarump Aquifer Compliance Strategy

DOE has determined that no remediation strategy is appropriate for the Shinarump aquifer because ground water contamination is not present in excess of MCLs and no COCs were identified (Section 6.0). An explanation of how the targeted strategy was selected is presented in Table 7–2.

Table 7–2. Explanation of the Compliance Strategy Selection Process for the Shinarump Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See conceptual site model presented in Section 5.0. Move to Box 2.
2	Is ground water contamination present in excess of MCLs or background?	No constituent exceeds an MCL (Section 5.3.3.2). Site-related constituents ammonium, calcium, iron, magnesium, manganese, potassium, sulfate, and zinc occur at concentrations that only slightly exceed background levels. Move to Box 4.

Table 7–2 (continued). Explanation of the Compliance Strategy Selection Process for the Shinarump Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	Shinarump ground water is not classified as limited use. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risk and other factors?	No COCs are present in the Shinarump aquifer (Section 6.0). Calcium, iron, magnesium, manganese, potassium, sulfate, and zinc qualify for ACLs because they are dietary components that are present at levels that fall within nutritional ranges. Ammonium qualifies because it occurs at a concentration that does not pose a risk (Section 6.0). Move to Box 7 **- no remediation required.**

7.2.3 De Chelly Aquifer Compliance Strategy

DOE has determined that natural flushing to reduce uranium concentrations below the MCL is the appropriate strategy for the De Chelly aquifer. The proposed ground water compliance strategy is no remediation for magnesium and sulfate; constituents that do not pose a potential risk (Section 6.0). An explanation of how the targeted strategy was selected is presented in Table 7–3.

Table 7–3. Explanation of the Compliance Strategy Selection Process for the De Chelly Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See conceptual site model presented in Section 5.0. Move to Box 2.
2	Is ground water contamination present in excess of MCLs or background?	Uranium slightly exceeds the MCL, however the area of impact is small, isolated, and the risk of uranium as a carcinogen is within EPA's risk range (Section 6.0). Site-related constituents magnesium and sulfate occur at concentrations that only slightly exceed background levels. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	De Chelly ground water is not classified as limited use. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risk and other factors?	Magnesium qualifies for an ACL because it is a dietary component that is present at a level that falls within the nutritional range. Sulfate qualifies because it occurs at a concentration that does not pose a risk (Section 6.0). Move to Box 7 **- no remediation required.** Uranium occurs at one small and isolated location near the former source area at a concentration that only slightly exceeds the MCL. Move to Box 8.
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	Although the applicability has not been formally assessed, it is unlikely that remedial action would cause excessive harm to the environment. Move to Box 10.

Table 7–3 (continued). Explanation of the Compliance Strategy Selection Process for the De Chelly Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
10	Will natural flushing result in compliance with MCLs, background, or ACLs within 100 years?	Uranium occurs at one isolated location near the former source area at a concentration that only slightly exceeds the MCL (Sections 5.3 and 7.4.4). Because the source has been removed and the uranium concentration is near the MCL, natural flushing is expected to reduce uranium to the MCL within 100 years. Move to Box 11.
11	Can institutional controls be maintained during the flushing period and is the compliance strategy protective of human health and the environment?	Although the areal extent of the uranium is confined within the fenced boundary at the site and the final compliance strategy is protective of human health and the environment, DOE is also considering additional approaches to risk management at the site. One approach is to provide an alternate source of drinking water for residents living near the former mill site, even though the ground water is not currently used for a public water supply. Move to Box 12 **- implement natural flushing.**

7.3 Future Ground Water Monitoring Activities

Monitor well locations, analytes, and sampling frequencies have been reviewed to ensure that data acquired from future ground water monitoring activities are appropriate and adequate to evaluate the effectiveness of the proposed compliance strategy. The proposed monitor well locations that will be sampled are shown in Figure 7–2. Ground water monitoring procedures specified in the *Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1997b) will be followed for sample collection, sample preservation and shipment, analytical procedures, and sample chain-of-custody. The selection rationale for the proposed sample locations, analytical requirements, and sampling frequency are discussed separately below for the surface water and the alluvial and bedrock aquifers.

7.3.1 Monitoring Requirements for the Alluvial Aquifer

Most of the future monitoring efforts will be concentrated on the alluvial aquifer because it is the ground-water system most affected by site-related contamination. A list of 25 proposed alluvial wells to be monitored, and the associated analytical and sampling frequency requirements, are summarized in Table 7–4. Uranium, nitrate, and sulfate are the only COCs present in the alluvial aquifer. Uranium will be monitored at the one location where an isolated uranium occurrence slightly exceeds the MCL. Nitrate, sulfate, and chloride will be monitored at all the proposed sample locations. Chloride is included as an analyte to calculate a sulfate-to-chloride ratio; relatively high values are indicators of site-related sulfate contamination. (See Section 8.1.2.1.)

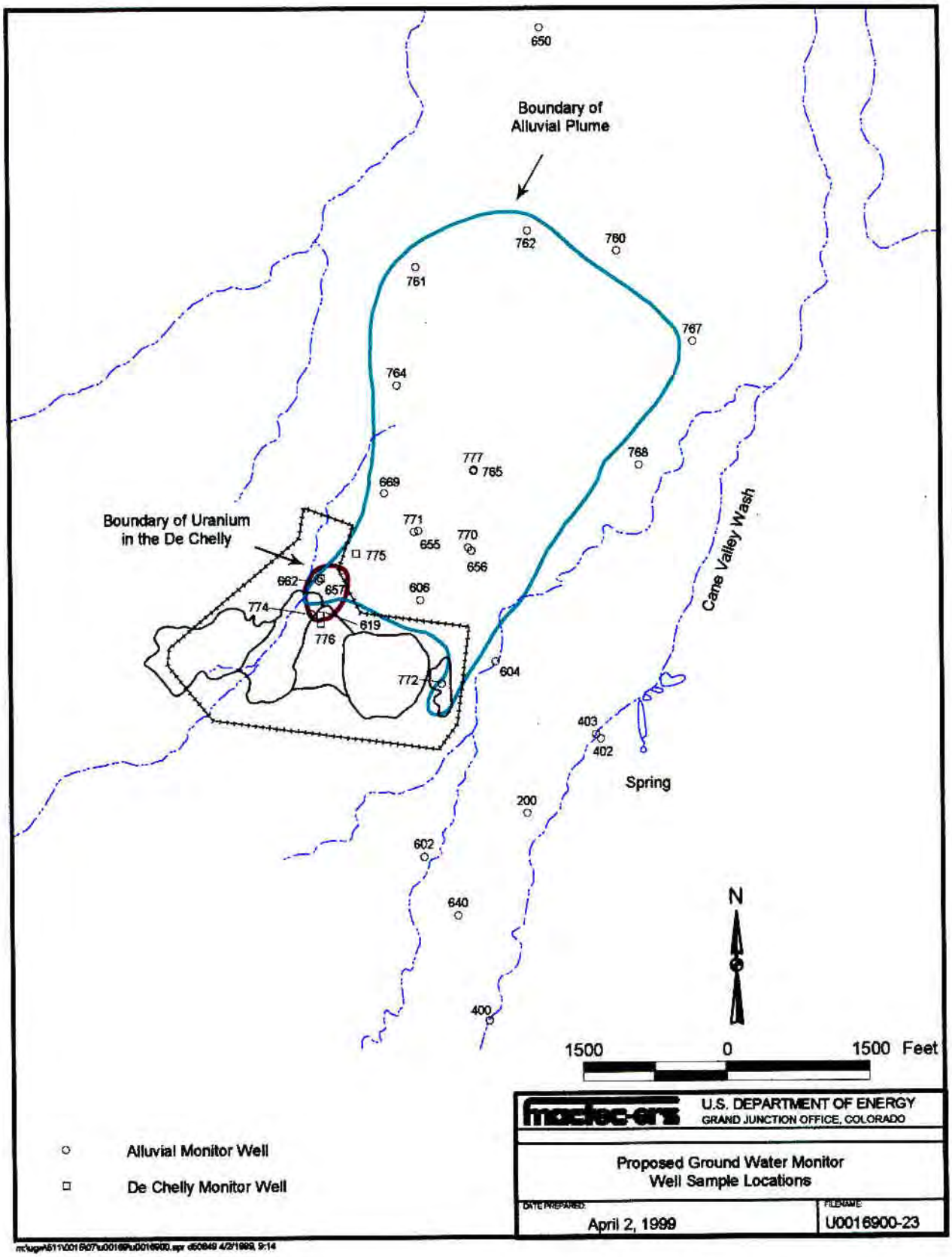

Figure 7–2. Proposed Ground Water Monitor Well Sample Locations

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Ammonium will be monitored only at selected locations close to the former source areas where relatively high ammonium concentrations are present in the ground water and where phytoremediation of the former source area is planned (Section 8.0). All of the proposed on-site and downgradient monitor wells will be sampled annually. Three background locations will also be sampled concurrent with each routine annual sampling event.

Table 7–4. Summary of Proposed Sample Locations, Analytical Requirements, and Monitoring Frequency for the Alluvial Aquifer

Monitor Well	Monitoring Purpose	Analyte	Frequency
MON–604, –662, –669, –764, –767, and –768	lateral boundary of plume	nitrate, sulfate, chloride	annual
MON–760, –761, and –762	leading edge of plume	nitrate, sulfate, chloride	annual
MON–650	most downgradient location	nitrate, sulfate, chloride	annual
MON–655, –656, –770, –771, –765, and –777	vertical contaminant profile	ammonium, nitrate, sulfate, chloride	annual
MON–606, –772, –774	on and near site	ammonium, nitrate, sulfate, and chloride plus uranium at location MON–774	annual
Group I: MON–200, –400, and –402	natural background	nitrate, sulfate, chloride	biennial[a]
Group II: MON–403, –602, and –640	natural background	nitrate, sulfate, chloride	biennial[a]

[a]Group I and II will be alternated with each annual sampling event.

Some downgradient and lateral migration of site-related contaminants may continue until the proposed phytoremediation and active treatment technologies begin to contract the plume. Further expansion and eventual contraction of the plume will be monitored at six monitor well locations established at or near the lateral boundaries of the plume and at three locations established near the leading edge of the plume. The most downgradient location will also be monitored.

Three locations where paired well installations exist will be sampled to monitor the vertical concentration profile within the most contaminated portion of the alluvial plume. Additional or alternate sampling locations to monitor concentration reductions within the plume may be proposed at a later date when extraction wells are installed as part of the active remediation.

Three geochemical types of background water known to exist at the site will be monitored at six upgradient locations. For monitoring purposes, the six locations will be divided into two groups of three upgradient wells, designated as Group I and Group II in Table 7–4. Each group contains one location that is characteristic of the sodium-carbonate type (MON–400 and –602), the sodium-sulfate type (MON–200 and –640), and the calcium-magnesium-carbonate type water (MON–402 and –403). One group of upgradient wells will be selected for inclusion in each annual sampling event.

7.3.2 Monitoring Requirements for Surface Water

Surface water located east of the site along the Cane Valley Wash has not been impacted by the former milling operations (Section 5.3). In addition, it has been demonstrated that the lateral extent of the alluvial ground water plume does not extend cross-gradient to the east of the site under Cane Valley Wash. Continued monitoring of the alluvial ground water plume, as proposed above in Table 7–4, will provide sufficient notice if the contaminant plume begins to migrate under Cane Valley Wash. For these reasons, no additional surface-water sampling will be performed unless the future alluvial ground water monitoring results indicate the alluvial plume is expanding and migrating under Cane Valley Wash.

7.3.3 Monitoring Requirements for the Shinarump Aquifer

Sufficient data have been collected to determine that site-related constituents are not significantly impacting the water quality in the Shinarump aquifer (Section 5.3). No site-related constituents occur at concentrations that exceed a MCL or at concentrations that pose a health risk (Section 6.0). For this reason, it is recommended that no additional ground water monitoring of the Shinarump Member be performed. Existing Shinarump monitor wells should be abandoned.

7.3.4 Monitoring Requirements for the De Chelly Aquifer

Wide spread site-related contamination as a result of the former uranium processing operations is not evident in the De Chelly aquifer (Section 5.3). Uranium is present in a few ground water samples at concentrations that slightly exceed the 0.044 mg/L MCL, however the area of impact is small, isolated, and the concentrations appear to be decreasing with time. The slightly elevated uranium concentrations are associated with former production well MON–619 located in the area of the former old tailings pile. This De Chelly well, which is hydrologically connected to alluvial ground water in the adjacent paleovalley, was used to supply water for the milling operation. Pumping the well actively drew uranium contamination from the alluvium into the De Chelly. Uranium concentrations have declined significantly in ground water monitoring samples collected from MON–619 since the well was pumped during an aquifer test in 1993. Further decreases are expected since well 619 is no longer used as a production well and dedicated low-flow bladder pumps have been installed for water quality sampling purposes.

Future ground water monitoring will include MON–619 and three other De Chelly monitor wells located in the vicinity of the former old tailings pile. These locations are shown in Figure 7–2. The analytical and sampling frequency requirements are listed in Table 7–5. Uranium will be monitored at all the proposed sample locations on an annual basis for a period of 5 consecutive years. If at the end of the 5-year period of natural flaushing the uranium concentrations are not trending lower, an alternate remediation strategy will be applied as per the PEIS. If, however, the uranium concentrations decrease below the UMTRA standard, then an additional 3-years of monitoring will be conducted to verify that the concentrations remain low before the wells are abandoned.

Table 7–5. Summary of Proposed Sample Locations, Analytical Requirements, and Monitoring Frequency for the De Chelly Aquifer

Monitor Well	Monitoring Purpose	Analyte	Frequency
MON–619	location of point-source of uranium in De Chelly	uranium	annual
MON–776	upgradient of point-source	uranium	annual
MON–657	leading edge of uranium >MCL	uranium	annual
MON–775	downgradient of leading edge	uranium	annual

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8.0 Development and Evaluation of Active Remediation Alternatives

As presented in Section 7.0, active remediation is the selected alluvial ground water compliance strategy for the Monument Valley site. The purpose of this section is to develop and evaluate different active remediation alternatives and recommend an alternative for remediation of the alluvial aquifer.

Section 8.1 gives an overview of the process used to evaluate and screen technologies and alternatives, including a detailed explanation of the evaluation criteria. Section 8.2 develops a list of potential technologies that could be used for remediation of the site, evaluates the technologies, and screens out technologies that are not feasible. Section 8.3 lists technologies that passed the initial screening, combines the technologies into alternatives, and evaluates the alternatives. The proposed alternative for active remediation is presented in Section 8.4 along with a discussion of how the proposed method may be deployed and uncertainties and limitations of the proposed alternative.

8.1 Process for Development and Evaluation of Technologies and Alternatives

This section gives an overview of the process used to arrive at a proposed alternative for remediation of contaminated alluvial ground water at the Monument Valley site. It also includes a description of the criteria used to evaluate technologies and alternatives.

8.1.1 Overview of the Process

The process used to select a proposed alternative for remediation of the alluvial aquifer is to:

- Develop, evaluate, and screen technologies that could be used for remediation of the site.
- Combine the technologies into alternatives and evaluate the alternatives.
- Select an alternative as a proposed method to remediate the site.

A number of technologies were considered for remediation of the site. Technologies considered could be used for extraction of ground water, disposal of ground water, or treatment of ground water. The initial screening of technologies, generally qualitative in nature, considered whether the particular technology was appropriate for use at Monument Valley, given the types, quantities, and locations of the contaminated water, and the concentrations of contaminants at the site. This initial screening did not consider cost or implementability except in the most general sense. The technologies that were considered appropriate for detailed review, based on the initial screening, were then combined into alternatives for extraction, treatment, and disposal.

The next step in the process was the evaluation of the alternatives to determine the preferred alternatives for extraction, treatment, and disposal. The evaluation of alternatives used the same criteria as the evaluation of technologies (i.e., effectiveness, implementability, and cost) but was done in more detail and included a detailed cost estimate for each alternative. The final step in the

evaluation of alternatives was to do a comparative analysis of the alternatives considering the evaluation criteria.

The last part of the process was to propose an alternative for remediation of the site. Discussion of the proposed alternative includes a potential deployment schedule describing the phases of the remediation and limitations of the proposed approach.

8.1.2 Evaluation Criteria

Each remediation alternative was evaluated for its effectiveness, implementability, and cost. The proposed alternative is the one that represents the best mix of all three criteria. The evaluation criteria were developed from standard engineering practice for assessing the feasibility of any large-scale project. A detailed discussion of each evaluation criterion is provided in the following sections.

8.1.2.1 Effectiveness

The effectiveness evaluation criterion considers a number of factors, which include

- Remediation time frame.
- Conformance with aquifer restoration standards and goals.
- Short-term effects (i.e., effects of remediation on workers, the community, and the environment).
- Disposal of treatment residuals.

Remediation Time Frame

The remediation time frame is largely dependent on how quickly contaminated ground water is removed from the alluvial aquifer. Therefore, extraction technologies and alternatives will have the most influence on the remediation time frame. Shorter remediation time frames generally correspond to higher extraction rates. DOE has established 20 years as a goal for remediation of the alluvial aquifer, and extraction and disposal alternatives were developed considering this goal.

Conformance with Aquifer-Restoration Standards and Goals

The general requirements for contaminant levels in the ground water at UMTRA sites are specified as MCL's in 40 CFR 192.04, Table 1. The only constituents in the ground water at the Monument Valley site that exceed MCL's are nitrate and uranium. The regulation does not specify ground water restoration standards for other contaminants of concern (COCs) that exceed background concentrations.

Sulfate is the only COC that exceeds background concentrations but does not have an MCL. The Navajo Nation's proposed cleanup standard for sulfate is 250 mg/L. DOE has agreed to treat sulfate in the contaminant plume to 250 mg/L, or to background concentration, whichever is greater.

The plume boundary shown in Figure 5–40 represents the area having a sulfate-to-chloride ratio of 10 or greater. There is significant similarity between the sulfate-to-chloride "plume" in Figure 5–40 and the nitrate plume shown on Figure 5–23a. Since nitrate in the Monument Valley ground water is almost exclusively due to milling activities (the median nitrate concentration in the background water is 4 mg/L), this indicates that a sulfate-to-chloride ratio greater than 10 is also a good indication of ground water contamination resulting from milling activities.

Although the sulfate-to-chloride ratio is a useful indication of contamination, it is not by itself sufficient to indicate a location of contamination requiring remediation. For example, Well 760, based on the 1997 data (Figure 5–24a) had a sulfate-to-chloride ratio of 11.0 and a sulfate concentration of 126 mg/L. The sulfate-to-chloride ratio suggests that the sulfate level is elevated due to milling activities, but the sulfate concentration is already below the Navajo Nation sulfate standard, therefore additional remediation is not warranted. Conversely, the sulfate concentration at upgradient Well 640 was 668 mg/L, but the sulfate-to-chloride ratio was 5.9, which suggests that the high sulfate level in that area is natural and not a result of milling activities. Thus, neither sulfate concentration nor sulfate-to-chloride ratio is sufficient in itself to indicate an area of contamination. The proposed sulfate treatment goal for Monument Valley will incorporate both criteria. The treatment goal will be achieved when the sulfate concentration is less than 250 mg/L or the sulfate-to-chloride ratio is less than 10.0.

The proposed area of remediation is shown in Figure 8–1 which combines the 44 mg/L nitrate plume (Figure 5–23a), the sulfate concentrations exceeding 250 mg/L (Figure 5–24a), and the distribution of sulfate-to-chloride ratios greater than 10 (Figure 5–40). The area where the 250 mg/L sulfate concentrations and the sulfate-to-chloride ratio plume overlap has both a sulfate concentration greater than 250 mg/L and a sulfate-to-chloride ratio greater than 10. This "overlap plume" represents the area in which sulfate remediation would be required. The nitrate plume is also illustrated in Figure 8–1. The similarity of the nitrate plume and the overlap plume is apparent. The nitrate plume covers the larger area overall, and the overlap plume extends beyond the boundary of the nitrate plume in only a small area on the north edge. Thus, adoption of the proposed sulfate treatment goal will not result in a significant expansion of the volume of water to be treated beyond what is required for treatment of the nitrate plume alone.

The Monument Valley aquifer-restoration standards (requirements of 40 CFR 192) and aquifer-restoration goals (cleanup standards not required by 40 CFR 192) for the COCs in the alluvial aquifer are listed below. Extraction, disposal, and treatment technologies and alternatives were evaluated on whether they could meet these standards.

Aquifer restoration standards (required by 40 CFR 192):

- Nitrate 10 mg/L as N (44 mg/L as NO_3^-)
- Uranium 30 pCi/L combined U-234 and U-238 (0.044 mg/L assuming secular equilibrium)

Aquifer restoration goals (not required by 40 CFR 192)

- Sulfate 250 mg/L

OR

Sulfate-to-chloride 10.0

Short-Term Effects

Short-term effects are a consideration of the potential effects to the community, workers, and the environment. The Monument Valley site is remote. Dinnehotso, Arizona, approximately 6.5 air miles away, with a population of 616, is the only community within 10 miles of the site, although the site and town are physically separated by the formidable barrier of Comb Ridge. Mexican Hat, Utah is 15 miles north of the site; Mexican Water, Arizona, is 14 miles to the east; and the town of Kayenta, Arizona, is about 24 air miles to the southwest. The land surrounding the site remains open and undeveloped. Thus, the community near the site is defined as the scattered farms, camps, and residences, and the temporary and permanent inhabitants of these areas and structures. The nearest residence is located within 1 mile of the site. Given the nature of the contaminants and the treatment processes being considered, it is unlikely that persons living in the area would be affected by the operation of the treatment facility.

The nearest highway is State Highway 160, which passes within about 6.5 miles to the south of the site adjacent to the hamlet of Dinnehotso. Users of the state highway cannot reasonably be considered as members of the community.

Evaluating the effects to workers entails considering the risks to persons employed to construct the treatment system and to those employed to operate and maintain the system during its operational life, as well as persons supporting the remedial action, such as samplers and equipment operators disposing of treatment residuals.

The evaluation of short-term effects also considers environmental effects. Environmental effects include potential environmental harm caused by deployment of a technology or alternative and whether the potential harm of remediation outweighs the benefits to be derived from restoration of the alluvial aquifer.

Disposal of Treatment Residuals

Active treatment processes, as well as many more passive systems such as evaporation, produce a significant amount of residual waste. This may include dissolved solids from the ground water, as well as the residuals from any other chemicals that may have been added during the treatment

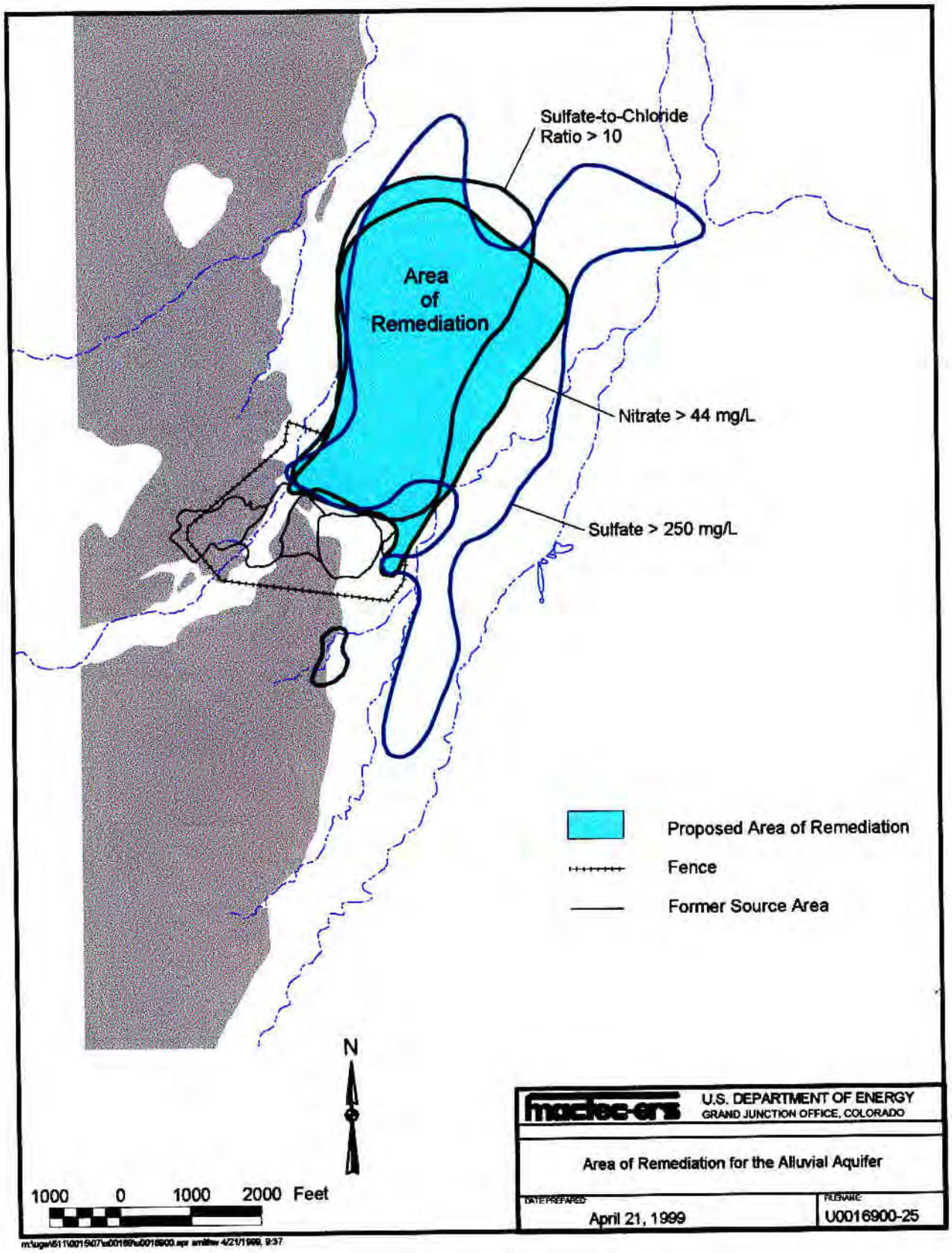

Figure 8–1. Proposed Area of Remediation

process (e.g., antiscalants or softening agents). These residuals must be contained during the remediation process, and disposed of either during or at the end of remediation. The need to keep residuals contained during remediation, to minimize their volume, and to dispose of them as needed, may become a significant cost over the lifetime of the remediation process.

8.1.2.2 Implementability

Implementability is an assessment of the feasibility of building, operating, and maintaining a remediation system.

The following aspects of feasibility will be discussed in this SOWP:

- Ease of construction.
- Ease of operation and maintenance.
- Expected reliability.
- Ability to handle changes in influent composition.
- Ability to handle increases in extraction capacity.

Construction

The Monument Valley site is remote, and skilled construction labor may be limited in the immediate vicinity. Thus, treatment systems which are easier to construct are preferred. Treatment systems which might seem to be very difficult to construct may be relatively simple due to the fact that in many cases “off-the-shelf” treatment systems are used, and construction at the site consists largely of constructing influent and effluent piping and supplying electricity. A plant-farming system, on the other hand, requires construction of an extensive and elaborate water-distribution system which will cover many acres. While the construction of such a system does not require a high degree of technical skill, it does require more labor.

Consideration of construction also requires examining the uncertainty associated with construction, such as the potential for schedule delays caused by technical problems.

Expected Reliability

Reliability is defined as the probability that a system will meet required performance standards. This includes both the physical reliability of the equipment comprising the system, and the process reliability, which considers the potential for variability in process performance both on a day-to-day and on a year-to-year basis. Evaluation of the potential reliability of a treatment system must consider the technical and operational complexity and required level of training for operators.

Ability to Handle Changes in Influent Composition

The concentrations of contaminants in the alluvial aquifer are expected to decrease gradually as remediation progresses. Particularly during the early years of remediation, the composition of the feed to the treatment system may vary significantly depending on which wells are in operation at any given time, and on whether currently unknown "hot spots" are uncovered as remediation progresses. Some technologies are better suited to handle such variations than others, and this ability will be considered in evaluating whether specific technologies are suited for use at the Monument Valley site.

Ability to Handle Increases in Extraction Capacity

The volume of the contaminated plume in this SOWP is an estimate based on sampling from a number of wells at the site. The likelihood that the actual volume of the contaminant plume will eventually be found to be significantly higher than the present estimate is not considered high, but the possibility must be allowed for. Another possibility is that additional regulatory or other drivers may also emerge, during the time between the start and the completion of remediation, that will cause the timetable for completion of remediation to be accelerated beyond the current goal of 20 years. In either event, it might become necessary to increase the extraction capacity of the system to handle more water than is currently planned. The ability of a remediation system to handle such increases, and the incremental cost involved in doing so, must be considered in evaluating whether specific technologies are suited for use at the Monument Valley site.

Cost

During the initial screening of technologies, the potential cost of individual technologies is not considered. Cost estimates for extraction, treatment, and disposal processes which pass the initial screening process have been developed in some detail. Capital costs (both direct and indirect) and operating and maintenance (O&M) costs were calculated for each process. The accuracy of the cost estimates for evaluation of the alternatives is defined to a level of accuracy of +50 percent to –30 percent.

In evaluating cost, the most important consideration is not the direct capital cost (although the realities of project funding mean that capital cost cannot be totally disregarded), but rather the total cost of treatment over the life of the project. These costs were determined by combining the initial capital cost for the treatment system with the estimated O&M costs over the project duration, using a net present worth analysis. By discounting all costs to a common base year, the costs for expenditures in different years can be compared on the basis of a single figure (i.e., the net present worth). Guidance issued by the Office of Management and Budget (OMB) was used to calculate net present worth. The guidance recommends using a real interest rate (i.e., a rate that does not consider inflation) to discount out-year costs that have not been adjusted for inflation.

Where possible, direct capital costs are developed from invoice costs of similar systems. If that information is not available, generic unit costs, vendor information, and conventional cost-estimating guides have been used. O&M costs are based on labor costs, energy costs, material and equipment costs, and maintenance costs.

8.2 Evaluation of Technologies

8.2.1 Technologies Considered for Remediation

During the alternatives evaluation process for the Tuba City site, which will be remediated in parallel with the Monument Valley site, technologies for ground water extraction, effluent discharge, and treatment were evaluated for remediation of the Tuba City site. This process is described in detail in Section 8.2.1 of the Final SOWP for the Tuba City site (DOE 1998). Where applicable, the "lessons learned" during the alternatives evaluation process for the Tuba City site were also applied to the Monument Valley site.

The chemical composition of the contaminant plume at Monument Valley is different from that at Tuba City. In most cases, the plume concentrations at Monument Valley are significantly lower than the concentrations of the same contaminants in the plume at Tuba City. Thus, the screening process for treatment technologies at Monument Valley did involve giving a "second look" at processes which were deemed unsuitable for Tuba City during the initial screening process for that site, such as native plant farming.

8.2.2 Extraction Technologies

Because of its depth, the most downgradient extent of the ground water at the Monument Valley site can only be withdrawn effectively through a well. Two types of extraction-well systems were considered: Conventional vertical wells and horizontal wells.

8.2.2.1 Conventional Vertical Wells

Vertical wells are the most commonly used ground water extraction devices, so the bulk of field experience and knowledge relates to conventional vertical wells. Installation of vertical wells is relatively straightforward in most cases, and, when combined with proper well design, construction, and development, vertical wells may provide acceptable yields. Vertical extraction wells can be readily converted to injection wells as needed, or vice versa, and can also be easily decommissioned when necessary. Finally, the theoretical performance of a vertical well can be simulated analytically or numerically during the design process using readily available and accepted mathematical formulations, while no comparable knowledge base exists for other technologies. Thus, vertical wells were recommended for detailed evaluation.

8.2.2.2 Horizontal Wells

Detailed evaluation of horizontal wells indicated that the technology could produce unprecedented difficulties due to the flowing (unconsolidated) sands encountered during the field investigations. Additional concerns with this technology arise because the long lengths of well screen that are required increases the difficulties of well completion and development. Also, as the aquifer cleanup proceeds, few options are available for sealing off the restored parts of the alluvial aquifer. Because of these risks, horizontal wells were not recommended for further evaluation.

Therefore, conventional vertical wells were the only extraction process deemed viable for the Monument Valley extraction system.

8.2.3 Effluent Discharge Technologies

This section describes the various ways in which effluent from the treatment plant can be discharged. Discharge options that do not involve injection include native plant farming, evaporation, and discharge to surface water; these options result in a loss to the aquifer of origin. The injection scenarios, in which the effluent is returned to the aquifer of origin, are also investigated.

8.2.3.1 Native Plant Farming

Native plant farming treats the extracted ground water by using the nitrate in the water as a fertilizer for a field of native plant species, which then discharge the water by ET. The details of native plant farming as a treatment option are discussed in Section 8.2.4.

Native plant farming is a no-injection option with a seasonal and cyclic demand. The maximum withdrawal rate is set by the requirements of the irrigation system (see Section 8.2.4) to a maximum rate of about 257 gpm (seasonal) for about 23 years. Preliminary estimates suggest that the aquifer can sustain this extraction rate, but this must be confirmed through detailed numerical modeling of the well field during the design phase to verify sustainable extraction rates.

8.2.3.2 Evaporation

Evaporation treats extracted ground water by allowing the water to evaporate under conditions in which the nonvolatile contaminants are contained and allowed to concentrate for later disposal. The advantages and disadvantages of evaporation as a treatment option are discussed in Section 8.2.4. The hydrologic effects of evaporation are similar to those of native plant farming (see above), except that since water may be fed to an evaporation pond year-round (even though the evaporation process will not be particularly effective during the winter), the limitation on maximum withdrawal rate depends only on the recharge ability of the alluvial aquifer. Treatment of two pore volumes of the contaminant plume in a 20-year period would require a continuous extraction rate of 103 gpm.

8.2.3.3 Discharge to Surface Water

Under this option the extracted and treated ground water would be discharged to Cane Valley Wash at a rate of about 103 gpm or about 166 acre-feet per year. Cane Valley Wash is ephemeral and intermittent in the vicinity of the site, and the discharge from the treatment system would be absorbed relatively rapidly except in periods of flash flooding. After the remediation period, natural discharge to Cane Valley Wash from the pumped region would be less than what it is today until water levels recovered to the pre-pumping condition.

8.2.3.4 Injection Wells

With this option, injection wells would be used to conduct the treated effluent directly back into the alluvial aquifer. Injection would control migration of the plume, promote rinsing of the solid matrix, preserve the ground water resource, and improve yields in the withdrawal wells. Injection wells would be designed in accordance with specifications attributed to recovery wells, and

considerable care would be required for all aspects of well completion. With injection wells, the suspended sediment concentration in particular would need to be very low to help prevent clogging. Other factors to consider with injection wells are the consequences of air entrainment and the entrance velocities for the treated effluent (Driscoll 1987). The entrance velocity for injection wells should not exceed 0.05 ft per second.

Injection wells could be deployed along the downgradient portion of the plume to control its migration, and within the body of the plume to enhance flushing. The benefit of using injection wells is that the treated ground water is returned to the same aquifer from which it was extracted and the ground water resource is conserved to the maximum extent practical.

8.2.3.5 Effluent Discharge Technologies Recommended for Detailed Evaluation

Technologies that do not rely on injection include evaporation, native plant farming, and discharge to surface water. These technologies are limited in their effectiveness because, without injection, the contaminated part of the alluvial aquifer will be limited in the amount of water it can deliver. If no injection is used, the only method to achieve greater drawdown is by adding more wells, increasing the capital and operating costs of the system. Evaporation and native plant farming are potential treatment technologies that are discussed in Section 8.2.4. Discharge of treated water to surface water, although technically possible, is not an appropriate use of an expensive resource since there is no flowing stream in the area, so much of the treated water would merely evaporate or be taken up in vadose soils.

The use of injection wells inside the plume area boosts the pumping rate that can be realized. Injection into the plume surcharges the hydraulic heads in the pumping zone and allows a higher rate of extraction. The greater extraction rates that stem from injection into the plume can therefore accelerate the ground water restoration. Injection well design incorporates many of the same considerations that apply to vertical pumping wells. These design considerations are addressed in Section 8.2.2.

8.2.4 Treatment Technologies

Many treatment processes were identified as potentially applicable for cleaning up the contaminated ground water at the Monument Valley site. The processes can be categorized as follows:

- Phytoremediation.
- Native plant farming (land/plant treatment process).
- Evaporation systems.
- Distillation systems.
- Through-medium processes such as continuous ion exchange.
- Biological processes.

- Chemical treatment processes.
- Membrane separation processes, including reverse osmosis and nanofiltration.

This Section will review the potential applicability of these treatment processes and will eliminate those which are obviously unsuitable. The processes which are not eliminated in this first screening will be evaluated in greater detail in Section 8.4.

8.2.4.1 Phytoremediation and Native Plant Farming

Phytoremediation and native plant farming are both types of phytoremediation processes, which rely on the natural affinity of plants for nitrates and other nitrogen species. Nitrate and ammonia in the water are taken up by the plant roots and assimilated into plant tissues. Nitrate is then reduced in the leaves and roots of the plant to ammonia or ammonium ion, which in turn is converted to amino acids. Amino acids are the building blocks for complex nitrogenous compounds, which are essential for maintenance and growth of plant cells.

In this document, “phytoremediation” will be used to refer to passive systems which depend solely on the action of plant root systems in the treatment of shallow contamination zones. “Native plant farming” will refer to systems which rely on extraction wells to draw water from deeper parts of the alluvial aquifer to the surface, where it is used in a slow-rate infiltration system to irrigate tolerant indigenous plants.

The phytoremediation and native plant farming systems proposed for the Monument Valley remedial action consists of a combination of passive phytoremediation and active native plant farming. The phytoremediation process will be used for treatment of the ammonia-contaminated soils in and around the former location of the tailings piles. The principal species that will be used for phytoremediation process of the subsurface ammonia-contaminated soils at Monument Valley is fourwing saltbush, *atriplex canescens,* which is a halophyte, or salt-tolerant plant.

Ammonium in subpile soils is not a ground water contaminant of concern (COC). However, DOE recognizes that subpile ammonium may constitute a continuing source of ground water nitrate, which is a COC. The Monument Valley remediation program will include a surface revegetation program to accelerate the removal of subpile soil ammonium.

Phytoremediation can also be used for treatment of the uppermost, or shallowest, portion of the aquifer. In this mode, roots of phreatophyte plants extending down into the upper portion of the water table will draw contaminated water without the need for irrigation. (An irrigation system might be required for the first one or two growing seasons to allow the plants to establish themselves. But once the roots have extended into the aquifer, irrigation would no longer be required and the system could be shut down.) The principal phreatophyte species that will be used for aquifer phytoremediation at Monument Valley is black greasewood, *sarcobatus vermiculatus*.

The native plant-farming system would be used for treatment of the deeper sections of the plume, by pumping the water from the alluvial aquifer to the surface where it can be used for irrigation. Such a system would use drip, sprinkler, or ridge-and-furrow systems for distributing the extracted ground water. Application of contaminated water is made throughout the growing

season. During the winter months when no growing takes place, the extraction system would be taken out of service. The principal species that would be used for native plant farming at Monument Valley is fourwing saltbush, *atriplex canescens.*

Fourwing saltbush has a high affinity for salts in general, so a significant portion of the sulfates and other non-nutritive dissolved solids will accumulate in the plant tissues. Dissolved solids that are not taken up by saltbush will build up in the soil. The ability of the soil to tolerate high loading of sulfate and other dissolved solids is uncertain, though preliminary data suggest that the drainage properties of the soil at the Monument Valley site are such that salts will not tend to crust on or at the soil surface to an appreciable extent. The ability to maintain sufficient control of the application rate to avoid deep percolation of contaminated water also is uncertain, and monitoring for deep percolation discharges from a land treatment system may be difficult.

The treatment residual which is produced by the native plant-farming treatment system is the plant material that is harvested for use as feedstock. As long as this material is considered to be a useful feedstock by area residents, the native plant-farming system will produce little or nothing in the way of wastes or residuals which must be disposed of as waste material. If the area residents reject any or all of the plant material, an alternative use will have to be found for it, or else it will have to be disposed of as a waste. Because of its considerable bulk, the cost of disposing of this material as waste could be significant. Cost estimates for the native plant-farming treatment system will assume that all of the plant material will be used as livestock feed throughout the duration of the project. But this important aspect of the system's viability is subject to a certain degree of uncertainty that is outside the control of the remediation program.

The nominal time frame for completion of remediation at Monument Valley is 20 years. The area available for native plant farming is 28 hectares or 69 acres. The planted area, based on plantings 3 m apart with a plant canopy of 1 m, is one-third of the total area or 9.33 hectares. The estimated annual ET for the fourwing saltbush crop is 1.97 cubic meters per year per square meter of area, for a total estimated maximum annual irrigation rate for high productivity of 48.5 million gal. Remediation of 1.08 billion gal using native plant farming, then, will require 22.2 years, which has been rounded off in this document as 23 years. This is slightly higher than the nominal goal of 20 years but is still acceptable, particularly considering that this calculation makes no allowance for the possible contribution of a passive phytoremediation system for the shallow portion of the contaminant plume, and is therefore conservative.

The maximum irrigation rate at the Monument Valley site is approximately 15 liters per square meter of area per day. For the active irrigation area of 9.33 hectares, this gives a peak irrigation rate of 616 gpm, which is clearly too high a flow rate for the aquifer to sustain. Since the irrigation system operates for only about 10 hours per day even at peak demand, the actual rate can be reduced using a holding tank which will be filled during the hours that the irrigation system is out of service. Using such a holding tank, which must have a capacity of about 300,000 gal, reduces the peak demand on the extraction system to 257 gpm.

A system combining phytoremediation for ammonia-contaminated soil and for the shallow portion of the alluvial aquifer, and native plant farming for cleanup of the deeper parts of the alluvial aquifer, will be considered as a treatment alternative. Some issues remain to be resolved, however, before the phytoremediation/native plant-farming system could confidently be

implemented as the sole treatment process at Monument Valley. These issues or areas of uncertainty include:

- The relative inefficiency of an extraction system that does not include any injection to the aquifer.

- Uncertainty as to the rate at which contaminated water can be absorbed and nitrate can be uptaken by the combination of species chosen for the phytoremediation/native plant-farming system. The estimates given in this SOWP are based on literature values for similar species.

- Uncertainty as to the effect on the soil of greatly increased concentration of non-nitrogen-bearing species (e.g., sulfate) which could build up in the soil as a result of the irrigation process. The soil in this area is sandy and free-draining, so this buildup is not expected to be a problem, but the potential certainly exists given the high concentration of sulfate in the ground water.

- Uncertainty as to the effect on the alluvial aquifer of the loss of a volume of over 1 billion gal, since the native plant-farming system will not allow for any recharge of the aquifer.

- Uncertainty as to the effect on the alluvial aquifer, and on the contamination zones, of the cyclic nature of the extraction process, since the extraction system will have to be shut off during the winter months.

- Seasonal variability in the required irrigation rate. Due to high ET rates during the peak summer months at the Monument Valley site, the maximum irrigation rate will vary considerably from month to month, which introduces additional complications for the extraction system. The actual area which can be planted for the native plant-farming system, and thus the time required for remediation using the native plant-farming system, will likely be limited by the maximum practical extraction rate.

None of these factors rules out exclusive use of a combination of phytoremediation and native plant farming as the treatment system for the Monument Valley site, although they must be considered before exclusive implementation of such a system is undertaken. Detailed consideration of all factors is the function of the alternatives evaluation process.

8.2.4.2 Evaporation Systems

Solar evaporation, which consists of putting the water into large lined or unlined outdoor ponds at influent rates that match the rate of natural evaporation, is an established method for reducing the volume of contaminated surface or ground water, that does not contain volatile hazardous compounds, in arid and semiarid regions of the United States. Nonvolatile contaminants such as nitrates, sulfates, uranium, and other components of TDS will not evaporate, and instead will concentrate as a sludge that must be periodically removed for disposal. Solar evaporation systems are constrained by climatic effects, notably temperature (solar radiation), humidity, and wind.

Pan evaporation rates at the Mexican Hat weather station, which is the station closest to the Monument Valley site, average about 84 in./year, and exceed 10 in. per month from May through

August. Precipitation at Monument Valley averages about 8 in./year. Evaporation rates exceed precipitation rates for all months except January. Thus, an evaporation system at Monument Valley would be expected to be very effective for most of the year. The surface area required to achieve complete evaporation could be considerable, however, preliminary calculations suggest that solar evaporation of a constant flow of 103 gpm would require a solar evaporation pond having an area of over 24 acres.

The effectiveness of solar evaporation systems can be enhanced by adding spray systems in which water is sprayed as a fine mist into the air above the solar pond. The fine mist droplets evaporate much more readily than does the bulk water at the pond surface. Use of a spray system can substantially reduce the size of the pond required. For instance, addition of a spray system could reduce the size of the evaporation pond for the Monument Valley site from 24 acres required for a simple solar evaporation pond to about two acres for a spray pond. However, addition of a spray system considerably increases the complexity of the system and requires more maintenance and operator attention than simple solar evaporation.

In general terms, evaporation is a very low-cost way to remediate large amounts of contaminated water in arid climates. However, an evaporation system would exacerbate any deleterious effects associated with treatment systems which do not recharge the alluvial aquifer. Also, evaporation results in a loss to the aquifer of all the contaminated ground water, without returning any value (such as the crops which would be produced in a native plant-farming system) in exchange. Therefore, evaporation was not selected for detailed evaluation as a treatment alternative.

8.2.4.3 Distillation Systems

In a simple distillation process, water is vaporized by heating it to its boiling point. The water vapors are then condensed and recovered as clean water. Nonvolatile contaminants such as nitrates, sulfates, uranium, and other components of TDS will not evaporate and will be left behind in the evaporation chamber, where they will concentrate and must be bled off (removed at a slow rate) periodically. The condensed water can be injected into the alluvial aquifer, as described in Section 8.2.3.4. The concentrate, or brine, may be taken off site for disposal; alternately, it may be evaporated to dryness in a small solar pond or in another process (such as a brine crystallizer), and the residue can then be disposed of as a solid.

Distillation is one of the most expensive treatment technologies to implement, because of the significant capital costs of distillation systems. Historically, distillation has also been relatively expensive to operate because of the high energy requirement to boil large quantities of water. However, distillation does recover almost all of the water for injection into the aquifer and the product water is of very high quality. The treated water produced by treatment of the ground water at Monument Valley using a simple distillation process will contain virtually no dissolved or suspended solids. Since the Monument Valley ground water does not contain volatile contaminants, the condensate from a distillation system will exceed the project standards and goals by orders of magnitude.

Energy requirements for distillation units can be greatly reduced by the use of “vapor recompression,” in which the heat that is given off by the condensation of the water vapor is recovered in a fan or compressor and used to preheat the feed water.. Evaporation of water using

a standard boiler with no energy recovery requires almost 2,400 kW-hr of electricity per 1,000 gal of water evaporated. Commercial vapor recompression distillation systems can process 1,000 gal of water while consuming as little as 35 kW-hr. This low energy consumption makes distillation more nearly economically competitive with other treatment processes.

Distillation has already been chosen as the primary treatment technology at the Tuba City site and has been demonstrated at that site in a pilot study conducted in the fall of 1998. The knowledge gained during that study, which confirmed the applicability of distillation to cleanup of UMTRA ground waters, could be applied directly to the Monument Valley site. For this reason, and because of the high quality of the treated water produced by the distillation process, distillation was selected for detailed evaluation as a treatment alternative.

8.2.4.4 Through-Medium Processes

In a through-medium process, a flow stream is passed through a column or reactor containing an insoluble adsorptive or exchange medium. A through-medium process can be used to remove uranium before biological treatment or native plant farming. Synthetic ion exchange resins, which are manufactured to have high affinities for certain types of ions, are widely used in through-medium processes for removal of uranium and many other dissolved ionic contaminants.

Conventional ion exchange processes are generally impractical for liquids having dissolved solids loadings higher than about 1,500 mg/L, due to high elutriation rates at higher solids levels. The TDS level in the Monument Valley alluvial aquifer will average about 1,500 mg/L and may be several times this amount in “hot spots.” A conventional ion exchange unit treating the Monument Valley ground water would require regeneration approximately every 30 to 50 bed volumes, and the regeneration would produce approximately 5 bed volumes of waste liquid with high salt content. Thus, the on-stream time would be poor, due to the need for frequent regeneration; chemical consumption would be high; and the volume of regenerant liquid would be at least 10 percent of the total feed. Thus, conventional ion exchange processes appear to be a poor choice as a remediation technology for Monument Valley.

State-of-the-art continuous ion exchange processes could be applicable to Monument Valley. The best of these continuous systems offer greatly reduced waste streams, averaging as low as 2 percent of the total water treated, with very low utility requirements and minimal chemical consumption. Bench-scale testing conducted during the summer and fall of 1998 indicated that a continuous ion exchange process could be competitive with other processes such as distillation and reverse osmosis (RO) for treatment of nitrate-contaminated water. However, DOE’s agreement to treat the Monument Valley ground water for sulfate significantly impacts the viability of any ion-exchange process. Vendor information suggests that treating for sulfate would require additional treatment modules and a significant increase in operating costs and waste generation over what would be required for nitrate removal alone. The advantage that such a system might offer over other processes, e.g., distillation, for treatment of nitrate-contaminated waters does not appear to remain viable for treatment of sulfate. The technology proposed, while technically intriguing, is unproven, and would probably provide treated water which would be of marginal quality at best, meeting or slightly exceeding the aquifer restoration standards, at a cost at least comparable to that of distillation which produces a much better quality treated water. Thus, continuous ion exchange was not chosen for detailed evaluation as a treatment alternative.

8.2.4.5 Biological Processes

Biological processes use bacteria to convert hazardous compounds to other forms which are less hazardous or more amenable to disposal. This may either be done in situ by injecting the bacteria and/or the carbon nutrient source into the aquifer, or ex situ by pumping the water into an above-ground treatment pond or reactor. This section will deal with ex-situ processes. In-situ biological processes were reviewed as part of the ITRD process and were rejected for further consideration in the UMTRA Ground Water Project

Nitrate, the principal regulated COC in the Monument Valley alluvial aquifer, is amenable to treatment with biological processes. Biological denitrification can eventually reduce nitrate levels in water to less than the MCL or to background levels. The primary byproduct of denitrification is nitrogen gas (N_2), along with small amounts of nitrous oxide (N_2O). Because nitrogen gas is relatively inert, denitrification generates a treatment residual that does not require handling and disposal, and it has no significant effect on the environment.

Denitrification may be done either in a pond, or in a biological reactor or series of reactors. Use of a pond will not be practical at the Monument Valley site, because the denitrification reaction loses effectiveness when the water temperature drops below about 50 °F. A pond-based denitrification process at Monument Valley could operate only seasonally, since it would be impractical to maintain the temperature of a large outdoor pond at 50 °F during the winter months. The treated water would be suitable for injection, but would only be available seasonally and would require pretreatment to remove residual organics before it would be suitable for injection. Therefore, at Monument Valley the biological denitrification process is best suited for indoor reactors, rather than an outdoor pond.

The average sulfate concentration in the Monument Valley ground water is about 755 mg/L, which is above the proposed treatment goal of 250 mg/L. Bacteria which have an affinity for nitrate also have an affinity for sulfate, and desulfurization will tend to take place in parallel with denitrification. While biological denitrification generates nitrogen gas which does not require handling and disposal and has no significant effect on the environment, biological desulfurization produces hydrogen sulfide (H_2S) as a byproduct. Hydrogen sulfide is malodorous, explosive, and extremely toxic. Nitrogen gas can be freely discharged to the atmosphere, while the control, handling, and ultimate disposal of H_2S will require other unit processes, such as a scrubber or a flare stack, that are ancillary to the primary sulfate-reducing reactor.

Desulfurization, then, is undesirable and should be avoided if possible. From the bacteriological standpoint, denitrification is the preferred reaction path, but, given the relatively high sulfate levels present in the Monument Valley ground water, it is uncertain whether denitrification can proceed to the extent required to reduce nitrate levels to below 44 mg/L without inducing at least some level of desulfurization.

Biological processes cannot be regarded as a feasible “stand-alone” treatment technology for Monument Valley since they do not address uranium, and because concerns regarding desulfurization suggest that sulfur should be removed using some other process. However, biological denitrification is an attractive process which is in wide use for remediation of nitrate-contaminated waters. Therefore, biological denitrification will be retained for detailed evaluation

as part of a treatment process which utilizes one or more additional processes for removal of sulfate and uranium.

8.2.4.6 Chemical Treatment

Chemical treatment is typically defined as a system using precipitation, coagulation and flocculation, gravity settling, and filtration processes, generally including addition of chemicals for pH adjustment, formation of precipitates, and the like. Such systems are widely used for treatment of contaminated waters produced during remediation of former uranium mill sites. They are very effective for removal of COC's such as uranium, radium, and sulfate. However, conventional chemical treatment processes are not effective for removal of nitrate, which would have to be addressed by some other technology.

Nitrate could be removed using an ex-situ biological denitrification process downstream of the chemical process. The removal of sulfates in the chemical process by precipitation of barium sulfate obviates the need for a biological desulfurization step and thus also eliminates the need to dispose of hydrogen sulfide formed as a by-product of biological desulfurization.

Nitrate can also be removed using a native plant-farming process. However, coupling chemical treatment with native plant farming does not produce an improvement in the overall treatment process over what can be achieved with native plant farming alone, because the native plant-farming process is not expected to require removal of sulfate, or any other constituents which the chemical treatment process is designed to address, prior to being introduced into the irrigation system. Also, the native plant-farming process is seasonal, as described above, and will be shut down during the winter. So the treatment process would have to be shut down also during the time that the native plant-farming process is out of service, or else the chemical treatment process would require some other denitrification process while the native plant-farming system is down.

The Alternatives Analysis performed during the preparation of the SOWP for the Tuba City remediation project included a detailed analysis of a combined process utilizing biological denitrification along with a chemical process for removal of sulfate and uranium. One potential advantage of such a process is that DOE owns a 100-gpm chemical treatment facility which is presently in operation at the Monticello, Utah Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) site. This equipment would be available for reuse by the time that the Monument Valley remediation begins. The cost analysis for this system at Tuba City concluded that it was a poor choice for that site due largely to high operating costs, which were driven by the high sulfate levels at Tuba City. The amount of sulfate which must be removed at Monument Valley is about one-fourth the amount that must be removed at Tuba City. Thus the use of such a process at Monument Valley appears at least feasible. This process would also permit reuse of existing government-owned equipment. Therefore, chemical treatment is retained for detailed evaluation as part of a treatment process combined with the biological process for removal of nitrate.

8.2.4.7 Membrane Separation Processes

Membrane separation includes all processes in which extremely fine or molecular-level filters are employed. The fine filter, operated under pressure, allows clean water to pass through the element

as a clean stream, or permeate, on the downstream side of the element, while the contaminants collect as a concentrate, or brine, stream on the upstream side. The most commonly-employed membrane separation processes, in increasing order of effectiveness in removal of dissolved ionic species, are ultra-filtration, nanofiltration, and reverse osmosis (RO). As a general rule, the more completely a membrane separation process removes contaminants from an aqueous stream, the more brine is produced.

The most promising membrane separation process for the Monument Valley ground water is RO, which can remove sulfate ions at 98 to 99 percent efficiency, and nitrate ions at 70 to 90 percent efficiency. Nanofiltration and ultrafiltration, on the other hand, are effective for removal of sulfate ions, but are much less effective against nitrate. The nitrate removal efficiency of the RO process appears to be adequate for the requirements of the Monument Valley project. The cost differential between RO and the other processes which are not effective against nitrate is minimal compared with the additional cost that would be required for an auxiliary denitrification system for either of those processes. Therefore, neither nanofiltration nor ultrafiltration need to be considered further for Monument Valley.

The primary disadvantages of RO units are the relatively high capital costs and the large volume of brine that will be generated due to the high concentrations of dissolved solids in the Monument Valley ground water. In order for the concentration of nitrate in the permeate (treated water product) to remain below the treatment standard of 44 mg/L, the brine stream from a RO unit operated at Monument Valley must be about 20 percent of the influent. Disposal of this quantity of brine would be a significant logistical problem. The disposal costs for the brine in its liquid state would be prohibitive, and therefore construction of a solar evaporation pond would be required. No such pond exists at the Monument Valley site at the present time, so the cost of building one or more evaporation ponds must be included in the project capital costs. However, this is also true - if to a lesser extent - for other active treatment processes which produce a brine stream that must be concentrated prior to disposal.

DOE owns a state-of-the-art, highly automated RO system, with a maximum permeate flow of 150 gpm, which was purchased new in February, 1998 for installation at the Monticello, Utah, CERCLA site. Operation of this RO unit at Monticello will be complete by the time ground water remediation at Monument Valley gets underway, and the RO unit from Monticello will then be available for use at another site. Use of an existing RO system would significantly reduce the capital costs of the treatment system, as well as making possible reutilization of a significant government resource. RO is otherwise technically acceptable as a treatment process. Accordingly, the RO process was selected for detailed evaluation as a treatment alternative.

8.3 Evaluation of Alternatives

This section combines technologies evaluated in the previous section into pumping alternatives and treatment alternatives. The pumping alternatives make use of the extraction and disposal technologies retained for detailed evaluation. The treatment alternatives make use of the treatment technologies retained for detailed evaluation.

8.3.1 Pumping Alternatives

Pumping alternatives are a combination of extraction and injection technologies that are part of a

comprehensive strategy. Two pumping alternatives are presented. The objective of each pumping alternative is to meet aquifer-restoration standards and goals within a specified time period. Each pumping alternative is evaluated on the basis of implementability, effectiveness, and cost. To optimize the likelihood of a successful remediation, the remediation program will be designed to treat two pore volumes of nitrate-contaminated water within a period of approximately 20 years. The volume of nitrate-contaminated water, as shown in Table 5–9, is 540 million gallons. Therefore, the system will be designed to treat twice this amount, or 1.08 billion gallons. Administrative issues associated with implementing either pumping alternative would be minimal, although a permit from the Navajo Nation will be required to extract the ground water.

8.3.1.1 Pumping Alternative 1—Plume-Focused Extraction Wells without Injection

Pumping Alternative 1 consists of a recovery-well field inside the plume area, with no injection wells. This pumping alternative would be used to supply irrigation water for the native plant-farming system. The best guide to the maximum sustainable extraction rates for wells when injection is not incorporated is the maximum sustainable flow rates for the existing wells at the site. The following table presents the data that has been developed on the maximum sustainable flows for a number of the existing wells. Sustainable flow rates for wells 765 and 655 are based on aquifer tests. Sustainable flow rates for the remaining wells are based on field information recorded when the wells were sampled for water quality.

Well No.	Maximum Sustainable Flow Rate, gpm
MON–604	1
MON–605	2
MON–650	3
MON–653	3
MON–655	0.6
MON–760	1
MON–761	2
MON–762	1
MON–765	3
MON–767	1
MON–768	1.5
MON–770	1
MON–772	1
Average	1.6

Thus, an average flow rate of about 1.5 gpm per well appears reasonable for the Monument Valley site. Sustaining the required peak extraction rate of 257 gpm (see the discussion under “Phytoremediation and Native Plant Farming” in Section 8.2.4 for the derivation of this quantity), at an average flow rate of 1.5 gpm per well, will require 172 ground water recovery wells. This pumping alternative will operate seasonally, drawing an average of 48.5 million gal of water per year. Thus, removing two pore volumes of nitrate-contaminated water will require approximately 23 years using this pumping alternative.

Effectiveness

An average of 257 gpm must be extracted over a 24-hour period during the peak demand period. Estimated pumping rates indicate that the proposed well field could supply this quantity at an average extraction rate of 3 gpm per well. Actual pumping rates will be determined in the field after the wells are emplaced.

Implementability

The complete extraction system includes not only the extraction wells and associated piping, but also a holding tank, with a capacity of approximately 300,000 gal, and appropriate controls to operate the extraction pumps as needed. The function of the holding tank is to act as surge capacity to enable the maximum irrigation demand of 616 gpm to be met with an extraction system having a capacity of 257 gpm. Since irrigation is only required for 10 hours per day, the irrigation system will draw from the holding tank, which will be replenished during the night while the irrigation system is out of service. Construction of the tank and control system is straightforward.

Construction of the well field would be relatively straightforward and could be accomplished using readily available technology. The technical obstacles to constructing a remediation well field are relatively few. However, the fine-grained eolian sands present their own set of technical demands, including how to obtain the maximum possible ground water withdrawal rate from each well, how to control sand pumping, and how to control the pumping rates in a large well field. These obstacles can probably be overcome through careful well-design, construction, and development techniques.

Cost

The total capital cost for this pumping alternative, including all 172 wells, pumps, holding tank, controls, and piping to direct the water from the well field to the treatment system, is estimated at $4.57 million. Annual O&M costs are estimated at $0.42 million. The net present value for this pumping alternative, calculated over the 23-year project lifetime, is estimated at $9.05 million.

8.3.1.2 Pumping Alternative 2—Extraction and Injection Wells

Pumping Alternative 2 consists of vertical extraction and injection wells. The objective of this pumping alternative is to achieve aquifer restoration without depleting the ground-water resource beyond treatment-plant losses. This pumping alternative would be used to supply water to the active treatment alternatives that employ treatment plants generating a clean water stream that would be used for injection.

The injected ground water would be pumped into the periphery of the plume to control its migration, similar to the “line-drive” approach used conventionally in the solution mining industry (Driscoll 1987, Roberts 1980). Returning the treated ground water to the plume would control drawdown, accelerate flushing within the plume, and accelerate aquifer restoration. Assuming that the extraction rate can be increased to 2.5 gpm per extraction well through the use of injection wells results in an estimate of 40 extraction wells for Monument Valley. The cost estimate will

assume the same number of injection wells; the slightly lower efficiency of the injection wells will compensate for losses in the treatment plant.

Effectiveness

The combination of extraction and injection within the plume is the most expedient method to move water through the contaminated part of the alluvial aquifer. A system consisting of extraction and injection wells yields a balanced flow system that limits drawdown within the plume and expedites flushing. The pumping rate increases as the density of wells increases. However, as the number of wells increases, so does the cost of operation and maintenance.

Implementability

There are no technical or administrative issues that would preclude implementation of the extraction and injection wells associated with this pumping alternative. These are conventional technologies that would be relatively straightforward to implement and would use readily available technology. The fine-grained eolian sands in the alluvial aquifer will present some technical difficulties when installing the extraction and injection wells. Well design, construction, and development techniques to specifically control sand pumping would be required. Also, operation of the system of 40 extraction wells and 40 injection wells will require oversight.

Cost

The total capital cost for this pumping alternative, including all extraction and injection wells, pumps, controls, and piping to direct the water from the well field to the treatment system and from the treatment system back to the injection wells, is estimated at $2.31 million. Annual O&M costs are estimated at $0.32 million. The net present value for this pumping alternative, calculated over the 20-year project lifetime, is estimated at $5.56 million.

8.3.1.3 Recommended Pumping Alternative

Pumping Alternative 2—Extraction and Injection Wells, preserves the best technical options and combines them into one comprehensive pumping alternative, at a substantially lower cost than Pumping Alternative 1. Considering the extraction system in isolation, Pumping Alternative 2 would be the preferred pumping alternative. However, the extraction system does not stand alone and cannot be considered except as part of the entire treatment system. Pumping Alternative 1 is suitable for treatment processes which do not return treated water for injection, while Pumping Alternative 2 is suitable for processes that do; pumping alternatives cannot be “mixed and matched” freely with treatment alternatives for which they are inappropriate. Since both pumping alternatives are technically feasible, the “preferred” pumping alternative would be the alternative that serves the preferred complete treatment system.

8.3.2 Treatment Alternatives

The treatment alternatives evaluated in this section are:

- Treatment Alternative 1—Native plant farming

- Treatment Alternative 2—Chemical Treatment with Biological Denitrification
- Treatment Alternative 3 —Distillation
- Treatment Alternative 4 —Reverse Osmosis

All these treatment alternatives incorporate phytoremediation of the subsoil ammonia in the vicinity of the former tailings piles as well as phytoremediation of shallow portions of the aquifer using plantings of black greasewood.

There are two distinct types of alternative treatment processes. The native plant farming process will operate on a seasonal basis, with flow rates varying from month to month with the seasonal ET cycle and shutting off during the winter when no irrigation is possible. This process will use Pumping Alternative 1—Plume-Focused Extraction Wells without Injection, described in Section 8.3.1. The other three treatment alternatives operate continuously at a steady flow rate, and will use Pumping Alternative 2—Extraction and Injection Wells.

The treatment system shall be designed to treat two pore volumes over an active life of approximately 20 years. As shown in Section 5.3.3.1, the estimated volume of the nitrate plume is 120,000,000 gal (120 million gal) of highly contaminated water containing more than 500 mg/L of nitrate, and 420,000,000 gal (420 million gal) of less contaminated water containing between 44 and 500 mg/L of nitrate. This gives a total volume of 540,000,000 gal (540 million gal). The total amount of water to be treated is two pore volumes, or a total volume of 1.08 billion gal.

Treatment of 1.08 billion gal in 20 years, assuming operation of the extraction and treatment system for 365 days per year, 24 hours per day, will require a flow capacity of 103 gpm. Cost estimates for the extraction, treatment, and injection systems for the processes utilizing distillation, chemical treatment with biological denitrification, and RO shall be designed for this capacity.

The native plant farming process will operate on a seasonal basis. As described in Section 8.2.4, the native plant-farming system will treat a total of 48.5 million gal per year. Thus, treatment of 1.08 billion gal will require a remediation period of 23 years. The peak extraction rate is 257 gpm. The cost estimate for the extraction systems for the native plant farming process shall be based on this capacity.

Cost estimates for all treatment processes will be compared based on a net present worth, calculated over the total project life (either 20 or 23 years), using the OMB standard discount rate of 7 percent.

8.3.2.1 Treatment Alternative 1—Native Plant Farming

Effectiveness

Phytoremediation will be used to treat soils contaminated with ammonium. Ammonium, which is readily converted to amino acids, is an ideal nutrient source for plant growth. Phytoremediation may also be used to treat shallow portions of the alluvial aquifer. In both cases, an initial period of

irrigation, taking no longer than one or two growing seasons, will be required for the plants to establish their root systems. Once the plants are established, no further irrigation is required. Conversion of ammonium is expected to be essentially complete within the remediation time frame of 23 years.

The native plant-farming system will be used to treat nitrate-contaminated water. Nitrate is also an excellent nutrient source. The average nitrate concentration in the alluvial aquifer is nearly ideal for growth of the fourwing saltbush used in the native plant-farming system. The irrigation system will be controlled so that there is no migration of water from the vadose zone into the alluvial aquifer. Nitrate levels in the soil are expected to be at or below the treatment standard of 44 mg/L at the conclusion of the 23-year treatment duration.

Fourwing saltbush has a very high affinity for sulfate as well. The salt content of a mature fourwing saltbush can be as high as 12 percent, and since sulfate comprises almost exactly half of the TDS in the alluvial aquifer, the saltbush may reach a final sulfate concentration of as much as 6 percent. The nitrogen content of the plant biomass will be about 1.6 percent, equivalent to a nitrate level of about 7.3 percent. Average sulfate levels are nearly four times as high as average nitrate levels (755 parts per million [ppm] versus 217 ppm). Therefore, the fourwing saltbush can be expected to take up nearly one-fourth of the sulfate in the water. The remainder will accumulate in the vadose zone.

One important concern regarding the effectiveness of the native plant-farming system is how well the plants will grow with less than optimal nitrate levels. The ideal nitrate concentration for growth of fourwing saltbush is about 230 mg/L, while the average nitrate concentration in the alluvial aquifer is 217 mg/L. Since production wells are placed in all areas of the alluvial aquifer and all are expected to be in operation simultaneously, the extraction system will deliver a mixture having very nearly average concentrations. At the outset of the project, the nitrate concentration will be essentially ideal for optimum growth of the saltbush. However, the concentration of nitrate and other TDS in the alluvial aquifer will decline over time as contaminated water is removed and replaced by natural dilution from background water. Under these conditions, the plant productivity may decline, which may cause a drop in system effectiveness as treatment progresses.

Implementability

There are a number of uncertainties associated with the implementability of the native plant-farming system. These were covered as part of the discussion of “Phytoremediation and Native Plant Farming” in Section 8.2.4. Another significant area of uncertainty regarding the native plant-farming system concerns the availability of water to supply the system. The extraction system must be able to sustain a flow rate of some 257 gpm continuously for several months. Preliminary estimates suggest that the aquifer will be capable of sustaining this flow rate, but this has not been confirmed by modeling or extensive testing.

Only two alluvial aquifer pump tests were done during the current investigations at the Monument Valley site (see Section 4.6.3.2). One of the wells (well MON–765) was able to sustain a flow rate of 3 gpm over about 20 hours before the test ended due to equipment failure, while the other (well MON–655) was able to sustain a flow rate of only 0.6 gpm even after significant development. However, neither of these tests was a long-term test. So the question of long-term

sustainability of the required flow rate once the entire extraction system is in operation remains open. Furthermore, the preliminary modeling done so far is based on the very limited data currently available on the hydraulic properties of the alluvial aquifer. Thus, the number of wells provided in the design used for estimating the capital and installation costs of the extraction system is an approximation at best, and there is significant uncertainty regarding the actual count and number of extraction wells.

Cost

The cost estimate assumes that the entire available area of 28 hectares (69 acres) would be planted in fourwing saltbush. The capital cost for planting this acreage, and the irrigation system required to sustain it, is estimated at $0.98 million. Annual operating costs assume that the labor requirement to maintain the fields will average 1.5 full-time equivalents (FTEs), although this will be distributed unevenly since the field work is seasonal rather than year-round. Total O&M costs then are estimated at about $0.25 million. The present worth cost of this treatment alternative, projected over the total estimated time of 23 years, is $3.69 million.

8.3.2.2 Treatment Alternative 2—Chemical Treatment with Biological Denitrification

Effectiveness

The average alkalinity of contaminated ground water at the Monument Valley site is 217 mg/L as $CaCO_3$, and the TDS of the influent water averages 1,506 mg/L. The practical limit of hardness removal using the lime-soda process is approximately 50 mg/L as $CaCO_3$ (15 mg/L of calcium and 3 mg/L of magnesium). For uranium, the removal efficiency of this process usually exceeds 95 percent. The average sulfate level in the Monument Valley contaminant plume is 755 mg/L, and the barium level is less than 1 mg/L. Addition of barium to remove sulfate will not significantly increase the barium level in the effluent since barium will not be added in stoichiometric excess.

Barium addition is an established method for removing sulfate from water by precipitation of barium sulfate. Typically, barium is added in the form of barium chloride, barium nitrate, or barium acetate. All these barium compounds are highly soluble, and although barium added to the contaminated water forms the insoluble barium sulfate precipitate, the dissociation of the barium compounds will add chloride, nitrate, or acetate ions to the water.

The average chloride level in the contaminant plume is 25 mg/L. The addition of barium chloride in the quantities required would increase the chloride levels in the effluent to at least 397 mg/L. Although chloride is not a regulated constituent, and there is no treatment goal for chloride for the Monument Valley remediation project, the chloride treatment goal for the Tuba City project is 250 mg/L. A chloride level of 400 mg/L or higher would undoubtedly be considered objectionable for the Monument Valley project. Also, the sulfate treatment goal for Monument Valley incorporates both the sulfate concentration and the sulfate-to-chloride ratio, and injection of treated water containing high chloride levels would greatly complicate the progress of the remediation.

The addition of barium nitrate to a water with existing high levels of nitrate will increase the nitrate loading for the biological denitrification system by at least 650 mg/L, or over threefold.

Use of barium acetate avoids these problems and will also supply a potentially useful feed source for the bacteria used in the biological denitrification process. However, preliminary cost information suggests that barium acetate will be considerably more expensive than barium chloride or barium nitrate. Also, a supply source for barium acetate may be a problem because it is not currently manufactured in bulk quantities.

A possible treatment alternative is the use of barium hydroxide rather than one of the other barium compounds. Barium hydroxide appears to be comparable in price to barium chloride and less expensive than either barium nitrate or barium acetate. Also, the use of barium hydroxide does not add objectionable chloride, nitrate, or organics to the ground water. Barium hydroxide is a fairly strong base, and some data suggest that it could be useful as a supplement to, or a replacement for, lime soda in the uranium-removal process. Hydroxide in excess of what is required for the uranium-removal process, could be removed by bubbling carbon dioxide into the solution. This would generate carbonates, which would be an operating concern, because of TDS and alkalinity, but not a regulatory concern.

Extensive data have been gathered on the efficacy of the biological denitrification process at DOE's Weldon Springs facility near St. Louis, Missouri. The treatment cycle implemented at Weldon Springs produces an effluent containing less than 10 mg/L of nitrate as nitrogen (NO_3-N) from a feed containing about 500 mg/L NO_3-N, which is about ten times the average nitrate concentration in the Monument Valley contaminant plume. Biological denitrification is an anoxic process, but the Weldon Springs denitrification pond is open to the atmosphere. Oxygen penetration does not appear to be significant below the top few inches of the pond surface, and natural convection creates circulation within the pond that is adequate to allow complete conversion of nitrate.

This treatment alternative produces an effluent that meets or exceeds the requirements of 40 CFR 192 and is protective of human health and the environment. Chemical treatment and microfiltration can achieve nearly complete removal of uranium, sulfate, and other dissolved solids from the raw water. Biological denitrification can achieve removal of nitrate from the treatment plant effluent sufficient to meet or exceed the regulatory treatment standard.

Implementability

For typical applications, the stability, reliability, and process efficiency of the chemical treatment systems can be predicted with a high degree of certainty. Chemical treatment is an appropriate and typical approach for cleaning up a high-TDS water. Operational parameters for chemical addition systems, mixing systems, settlers, sludge removal equipment, and filters under a wide variety of conditions are well established.

Chemical treatment is an established method for treating water containing inorganic and radionuclide contaminants. The chemical treatment equipment that would be used for this system is DOE property and was used at the Monticello, Utah CERCLA site for treating ground water contaminated with high levels of uranium, radium, and selenium. After chemical treatment, the flow stream will undergo microfiltration (included with the system) to remove solids formed during the chemical reactions. The chemical treatment equipment is currently being operated as

part of the waste-water treatment facility at Monticello, and is expected to be available for service before the Monument Valley remediation project begins.

A chemical system with chemical reactors and appurtenant processes will need constant maintenance and management. The level of maintenance is tied directly to the severity of the operating condition within the system. For example, very high or very low pH in the flow stream or use of corrosive chemicals such as iron coagulants can deteriorate equipment. Under adverse conditions, tanks, mixers, chemical feed systems, valves, instruments, piping, and pumps require continuous maintenance and frequent replacement.

An especially critical element of chemical treatment plant operation is managing, handling, and disposing of chemical sludge. Chemical treatment produces much greater quantities of sludge than do the other treatment alternatives. Lime-soda softening produces a sludge consisting of calcium carbonate and magnesium oxide contaminated with uranium. Sulfate precipitation with barium produces a sludge of insoluble, chemically inert barium sulfate. The process described here does not attempt to segregate these sludges, so although the barium sulfate sludge will have little or no radioactive contamination, it will be combined with the contaminated lime-softening sludge. A moderate degree of uncertainty is associated with predicting the activity of the sludge; thus, identifying a suitable method and location for disposing of the sludge is moderately uncertain.

The proposed chemical treatment plant is a 100-gpm water treatment plant that has been in service for several years. It is modular and trailer-mounted for ease of movement and setup. Specialists will be needed to oversee setup of the system, but this is not expected to take a great deal of time.

The denitrification system consists of a pair of “sequencing batched reactors” (SBRs) in which the denitrification reaction will take place. The reactors will be operated in a “fill and draw” system in which one reactor is filling while the other is anoxically mixing for the denitrification process and preparing for discharge at the end of the treatment cycle. The system will require significant design work but will not be particularly difficult to construct.

Operation of the denitrification facility will take close operator attention. Denitrification is a batch process with a number of process steps that must be carefully controlled. For instance, the pH will drop rapidly once the denitrification process is underway and acidic ions are liberated. The pH of the ground water is around 6.5. If the pH in the ponds drops below about 6, the denitrification will stop, and once it has stopped, it cannot be restarted easily. Also, at the end of the nitrate treatment cycle, it may be necessary to aerate the treated water to get the pH into a neutral (7 to 8) range and to strip residual organics that contribute to chemical oxygen demand.

There is another potentially serious implementability concern with the biological denitrification process. The design presented in this SOWP is based on information from a system vendor who estimated that the denitrification process would require about 16 hours to reach completion. Based on this residence time, the SBRs must have a capacity of around 200,000 gallons each. However, sources at the Weldon Springs facility indicate that the ponds there require three to five days to complete denitrification. Such a residence time would require a capacity of over a million gallons at the flow rate of the Monument Valley site.

Further, the denitrification reaction loses effectiveness when the water temperature drops below about 50 °F. The ambient temperature at the Monument Valley site will be below 50 °F for extended periods, so some means will have to be provided for maintaining the temperature of the denitrification reactors.

The design upon which the cost estimate is based assumes that SBRs can be used. However, the treatment system should not be designed and installed without first testing this assumption on a laboratory or pilot scale. If biological denitrification were chosen as part of the remediation technology at the Monument Valley site, a testing program should be completed before the final design is begun.

As with construction, specially trained persons will be needed to operate the system. Operators and managers are not available in the local area or on the reservation. An extensive training program will be needed if reservation residents are to operate this treatment alternative without extensive oversight by DOE contractors. The cost estimate assumes that two operators per shift will be required for continuous operation. One operator will work primarily on the chemical treatment process and the other will concentrate on the SBRs. A high degree of management oversight will be required to ensure that the plant operates safely and efficiently. The chemicals necessary for operation of the chemical treatment plant are not available near the site. The most probable source of commercial quantities of chemicals is Phoenix, Arizona—360 road miles from the Monument Valley site.

In addition to the large amount of chemical sludge produced by the chemical treatment process, the biological process generates a significant amount of biological sludge. The combination process generates approximately twice as much total sludge as does spray evaporation. A small amount of residual methanol will remain in the sludge from the SBRs after denitrification is complete. Although this methanol should evaporate during the sludge evaporation step, it should be kept in mind when permitting issues for the facility are discussed.

Improving the reliability of the chemical treatment system will require adding redundant reaction tanks, settlers, and membrane modules. Increasing the capacity of the biological denitrification system will require building additional treatment reactors.

Chemical treatment with biological denitrification will require tank capacity for feed, equalization and holding between the chemical treatment facility and the denitrification reactors, and evaporation ponds for sludge dewatering. Since there are no ponds in existence at the Monument Valley site, two double-lined ponds will be constructed for sludge dewatering. The denitrification process requires a holding tank of approximately the same capacity as the denitrification reactors for the treated effluent, and the reinjection system will draw from this tank.

Cost

The capital cost of this system is estimated at $0.97 million, making it comparable to the native plant farming system as by far the least expensive systems to install. The low capital cost is due largely to the use of an existing chemical treatment facility. However, the annual O&M cost for the system is estimated at $1.38 million. The major components of the high O&M cost are $0.49 million for treatment chemicals, particularly barium, and $0.46 million for operating labor for this

manpower-intensive system. Due to the high O&M costs for this system, the estimated 20-year present worth cost of this treatment alternative is estimated at $15.6 million.

8.3.2.3 Treatment Alternative 3 —Distillation

Effectiveness

Evaporation and water recovery using simple distillation is an established and proven technology for treatment of contaminated water. A distillation unit will consistently produce a product effluent containing less than 50 mg/L of dissolved solids, and will often meet or exceed drinking water standards with no further treatment required. The concentrated "brine," which contains essentially all of the dissolved solids, radionuclides, and other nonvolatile contaminants from the original feed, typically averages 5 percent or less of the total feed, depending on the concentration of contaminants in the feed.

The following data was developed during pilot testing, at the Tuba City site, of a distillation system similar to that which would be used at Monument Valley. All concentrations are given in mg/L.

Parameter	Influent	Effluent
Sulfate	2,440	0.824
Nitrate	819	2.48
Ammonium	61.9	2.09
TDS	4,900	37
Uranium	146	<1.1

Based on these data, the likelihood that the treated effluent from the distillation system will be able to meet or exceed the applicable treatment standards is extremely high.

Pretreatment for the feed water is expected to consist of addition of sulfuric acid for removal of carbonate, and an antiscalant to minimize fouling of the heat-transfer surfaces.

The distillation process will incorporate phytoremediation of subsoil ammonia in the vicinity of the former tailings pile. For a discussion of the effectiveness of this process, see the "Effectiveness" discussion under Section 8.3.2.1 "Treatment Alternative 1—Native Plant Farming."

Distillation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The treated effluent is of high quality, while the volume of the concentrated brine is less than that produced by the other active processes.

Implementability

Commercial distillation units are self-contained and include all instrumentation required for

monitoring and controlling the operation. The units are designed for outdoor operation with no building required. The operation of the unit can be monitored at a remote location using the instrumentation and computer software provided as part of the package. The electricity demand of the distillation unit is low. However, since no electric power is currently available at the Monument Valley site, additional electrical power equipment will be required at the site for the distillation system (or for any other treatment system, for that matter).

Commercial distillation systems are reliable and generally require a low level of oversight and only scheduled maintenance during their operating life. Installation of the distillation unit will be straightforward, and can be done by project construction personnel. Operation of the distillation system will require a minimum of managerial and technical supervision. The acid pretreatment system can operate unattended, although periodic replenishing of the acid will be required, as well as occasional maintenance. The cost estimate for the operation of the distillation system includes two full-time employees for operation and maintenance.

For optimal operation, the distillation system should be operated as nearly continuously as possible. However, it is expected that the flow rate produced by the extraction system will have a fair amount of variability. To dampen out variations in the extraction rate and produce a constant flow rate of feed to the distillation unit, a feed tank of approximately 20,000 gal capacity will be erected at the site immediately adjacent to the treatment unit. Water from the extraction system will flow into the feed tank, and the distillation unit will take its feed from the tank, whose level will be allowed to vary as needed.

Concentrated brine is continuously generated by the distillation process. The concentration of solids in the brine discharged from the distillation unit is low enough that disposal is impractical without further concentration. The brine must be evaporated further, perhaps to dryness, by dewatering via solar evaporation. Since the solar evaporation rate is relatively slow compared to the rate of brine production, a relatively large double-lined solar evaporation pond will be constructed for this purpose. For a discussion of the implementability of solar evaporation ponds, see the “Implementability” section under Section 8.3.2.4 “Treatment Alternative 4—Reverse Osmosis.”

Commercial distillation units are modular in design. Increasing the capacity of the overall system above the current design capacity will require addition of more distillation units unless additional capacity is specified as a design requirement.

Cost

The capital cost of the distillation system, including the evaporation pond and required ancillary equipment, is estimated at $3.01 million, and annual operating costs will be about $0.55 million. The present worth cost of this treatment alternative, projected over the total estimated time of 20 years, is $8.62 million. The most expensive capital item is the distillation unit itself, which will cost about $2 million. The most expensive O&M line items are electricity to operate the unit, estimated at $0.14 million per year, and operating manpower, which will average about $0.125 million.

8.3.2.4 Treatment Alternative 4—Reverse Osmosis

Effectiveness

The RO system proposed for Monument Valley has been in service at a former uranium mill in Monticello, Utah since April, 1998. The following data were taken from two pilot testing runs of the RO process using Monticello treatment pond water. The concentrations shown are in micrograms per liter (µg/L) or parts per billion (ppb) for the feed, permeate, and concentrate, respectively, while "Reduction" is the percent by which each component was reduced in the permeate compared to its concentration in the feed.

First Run

Component	Feed	Permeate	Concentrate	Reduction
Uranium	585	4.3	2,443	99.4%
Chloride	169	3.74	669	98.3%
Nitrate (as N)	2.84	0.28	9.9	92.8%
Sulfate	762	25.2	3,010	97.5%
	Average			97.3%

Second Run

Component	Feed	Permeate	Concentrate	Reduction
Uranium	551	3.6	4,370	99.4%
Chloride	162	2.72	1,270	98.5%
Nitrate (as N)	2.62	0.679	19.3	76.8%
Sulfate	750	9.36	5,910	98.9%
	Average			95.9%

The data above suggests that nitrate removal and reject water generation are interrelated. In the first run, the nitrate removal was almost 93%, and reject water generation was 25%. In the second run, the nitrate removal dropped to 77%, but the reject water generation was cut in half, to about 12.5%. The minimum nitrate removal required for the Monument Valley remediation is 79.7%, since the feed contains 217 mg/L of nitrate and the treated water must meet the standard of 44 mg/L. Assuming that this performance is representative of the performance of a full-scale system, will require about 14% reject water generation. The reject water will be sent to a separate pond, with an estimated surface area of 3.5 acres, for solar evaporation.

The RO process will incorporate phytoremediation of subsoil ammonia in the vicinity of the former tailings pile. For a discussion of the effectiveness of this process, see the "Effectiveness" discussion under Section 8.3.2.1 "Treatment Alternative 1—Native Plant Farming."

Implementability

As mentioned above, the RO process will utilize an existing DOE-owned facility currently in operation at Monticello, Utah. This unit will become available as soon as the Monticello repository is closed in the summer of 1999. The RO equipment will be relatively easy to install and operate. The system is very well instrumented although continuous operator attention is required. There is a low potential for schedule delays in the construction of the system at the Monument Valley site. However, specialists will be needed to oversee construction of the system.

The RO process can be modified and improved by replacing the filter elements with elements offering greater removal efficiency. The existing system has a capacity of 150 gpm of permeate, equivalent to about 173 gpm of influent at the predicted rate of reject water generation. The system consists of three parallel trains. Operation at the required Monument Valley flow rate will require operating two of these trains while the third is left in a stand-by mode.

As with construction, specially trained persons will be needed to operate the system. Operators and managers are not available in the local area. An extensive training program will be needed if local residents are to operate this treatment alternative without extensive oversight by DOE technical contractors. The cost estimate assumes that one operator per shift will be required for continuous operation of the complete treatment system. A moderate degree of management oversight will be required to ensure the plant operates safely and efficiently.

The RO process will generate little, if any, additional sludge compared with the distillation process. However, as noted above, it does generate a very large quantity of reject water. The large solar evaporation pond required for concentration of this quantity of reject water is a major operational consideration in itself.

Operating the evaporation pond will require the following principal functions: Embankment inspection and maintenance, liner inspection and repair, monitoring water levels, and monitoring for leaks. Given the high degree of automation in the RO system, it is anticipated that all of the pond operation functions can be performed by the reverse-osmosis system operator. The first three functions can be performed with periodic inspections by the operator working the day shift. The need for inspections can be minimized by installing and maintaining adequate fencing to keep livestock and wildlife away from the pond.

Monitoring for leaks will consist primarily of monitoring the water levels in the sump(s) of the leak detection system. This can be done remotely using a telemetry system. Leak detection pump status can also be monitored remotely using telemetry.

The principal environmental compliance issue associated with maintaining large, lined ponds is uncontrolled release through overflow, or leaks. Use of double-lined ponds and an interliner leak detection system will control subsurface releases. Such engineering controls are highly reliable. Overflow of the pond is unlikely because the levels change relatively slowly due to their size, and will be monitored on a regular basis by operating personnel.

A large, open body of water in an arid region attracts birds and insects, creating a potential exposure pathway for contamination. Over time, the concentration of uranium, metals, and

metalloids (e.g., selenium) in the pond water will increase. Birds and insects may be attracted to the ponds and exposed to high levels of contaminants. The risk increases with a spray system in which contaminants become airborne. Thus, the ability to control waterfowl and insect access to heavily contaminated water will be a concern for the system's operator.

Cost

The capital cost of the RO system is approximately $1.19 million. The RO unit itself is surplus DOE-owned equipment from another site. The single largest direct capital cost item is the construction of the large solar evaporation pond for the reject water, which accounts for almost half of the total capital cost. The estimated annual O&M cost is $1.0 million, of which the single largest item is unit operators, since it is assumed that 24-hour coverage will be required. Thus the 20-year present worth value for this process is $11.7 million.

8.4 Comparative Evaluation of Alternatives

The following section compares the four treatment alternatives and recommends a proposed treatment alternative for implementation at the Monument Valley site. The treatment alternatives are compared with one another on the basis of each of the evaluation criteria presented in the introduction to this section. For purposes of this discussion, the treatment alternatives utilizing injection of treated effluent with either the distillation, RO, or chemical treatment/biological denitrification processes will be referred to as "active" systems. The extraction and irrigation aspects of the native plant farming system are active too, but the treatment process itself is passive.

8.4.1 Comparative Effectiveness

8.4.1.1 Conformance with Project Treatment Standards (40 CFR 192) and Goals

Of the active treatment alternative systems, distillation produces the highest-quality effluent (treated water), with a composition that will exceed the project standards and goals by one to two orders of magnitude and almost total removal of sulfate, nitrate, and radionuclides.

The RO process also will be able to meet or exceed the project standards and goals. However, since nitrate removal and reject water generation are interrelated, and there will naturally be a tendency to try to minimize reject water generation, the level of uncertainty regarding the nitrate removal efficiency of the RO process is higher than that for the distillation process. RO will also produce an effluent that will exceed the project treatment goal for sulfate.

The chemical treatment with biological denitrification process is also capable meeting or exceeding the aquifer restoration standards and goals. In view of the relatively low nitrate levels in the ground water, the denitrification process should easily meet or exceed the regulatory requirements. The system is also capable of easily meeting or exceeding the required levels of sulfate removal. However, the barium chemical used to remove sulfate will be the highest single cost line item in the operating budget for the treatment facility, so there will naturally be pressure to minimize consumption. In view of this consideration, the ability of the facility to consistently

meet or exceed the required sulfate removal levels must be considered as less certain than that of the other two active processes.

The native plant-farming system will consume nitrate efficiently and essentially completely as long as irrigation rates are kept low enough that no recharge to the aquifer takes place. The saltbush will also remove about 25 percent of the sulfate. The residual sulfate will concentrate in the vadose zone, and is not expected to pose a contamination concern for the soils in the remediation area.

Ultimately, however, the success or failure of the remediation process will be determined not by the quality of the effluent water from the treatment process, but by the quality of the ground water in the alluvial aquifer. Distillation, RO, and chemical treatment with biological denitrification all utilize injection of treated water back into the aquifer as an integral part of the remediation process. From the standpoint of ultimate aquifer cleanup, the injection process serves two useful functions. First, it provides a pressure gradient within the plume which will help to direct contaminated water towards the extraction wells; and second, it provides a pressure gradient at the perimeter of the plume which will contain the spread of contaminants beyond their present limits. Whether, in the long run, treatment standards and goal can be met by any pump-and-treat system is problematic (see “Limitations of the Proposed Treatment System” in Section 8.5.3), but if it is possible to meet the cleanup goals within the specified time frame, one of these three treatment systems provides the likeliest route.

The potential long-term effectiveness of the treatment system utilizing native plant farming is much more difficult to assess. Whereas the active systems operate on a continuous basis with steady flow rates of both extracted and injected water, the native plant treatment alternative does not use injection at all, and the extraction rates vary from month to month, and are shut down completely for several months each year. The effect of such cyclic extraction on the behavior of the contaminants which make up the plume is difficult to predict.

All of the treatment processes can be designed to provide optimal protection of health for the plant operators and persons living or working in the vicinity, as well as those who depend on the alluvial aquifer for part or all of their water supply.

8.4.1.2 Effect on the Aquifer

The native plant farming treatment alternative operates on a seasonal basis and does not utilize injection of treated water, so over the course of the remediation process, approximately 1.08 billion gal of water will be removed from the alluvial aquifer.

If one of the active remediation alternatives is employed, some loss of ground water will occur during the remedial action. Loss of ground water will be minimized by distillation, because the waste water stream from the distillation process is small. Chemical treatment with biological denitrification will generate somewhat more waste water than the distillation system. Reverse osmosis will have much higher water loss, because of the large amount of reject water generated by the RO process.

8.4.1.3 Ease of Residual Disposal

As noted in the discussion above, as long as the local population regards the plant material produced by the native plant-farming treatment alternative as a useful livestock feed, this alternative does not produce a treatment process residual. In addition, the extraction system will have to be maintained during the remediation operation and dismantled at its end. (This is also true of the active processes, but since the extraction and reinjection systems for those processes have less than half the number of wells required for the native plant-farming system, generation of this type of waste will be proportionately less.) These materials should be classified for free release and disposal at a commercial landfill operation. For this reason, estimates of the volume of such materials have not been made.

The principal treatment residual produced by the active treatment processes is the concentrated sludge that contains the dissolved and suspended solids which were removed from the ground water during treatment. As described in the detailed evaluation of the treatment alternatives, the three processes produce somewhat different amounts of this sludge. Sludge production will also vary over the lifetime of the project. The initial rate of sludge production will be relatively high because the concentrations of contaminants will be highest at the beginning of the remediation project, and it will decline towards the end of the remediation cycle as the concentration of contaminants in the plume declines. The ground water contains the equivalent of 339 tons of sludge per year, based on the average TDS concentration in the ground water. The following are average figures for sludge generation which can be used to compare the two treatment alternatives.

- Distillation will generate about 424 tons of sludge per year. The distillation process requires the addition of sulfuric acid, which will increase the sulfate concentration in the brine, as well as a small amount of antiscalant.

- RO will generate about 505 tons per year. The RO process requires a lime softening step prior to the RO step, which will add somewhat more chemicals than the chemical pretreatment of the distillation process.

- Chemical treatment with biological denitrification will generate about 552 tons of sludge per year; the major component which is added is the barium used to remove sulfate.

The other major treatment residual will be the pond liners, which will be disposed of at the end of the remediation program. This is a comparatively small quantity compared with 8,000 to 16,000 tons of chemical sludge. Treatment Alternative 4, RO, produces the greatest amount of this waste, because of the large solar evaporation pond required to handle the reject water from the RO process. Treatment Alternatives 2 (chemical treatment with biological denitrification) and 3 (distillation) require much smaller ponds and will generate proportionately much smaller quantities of this waste.

Used piping, process equipment, filter elements, etc. which are discarded during treatment or are left over from the treatment systems at the end of the remediation, should be able to be free-released and disposed of at any commercial landfill operation, or reused elsewhere if the need exists. For this reason, estimates of the volume of such materials have not been made.

8.4.2 Comparative Implementability

8.4.2.1 Constructability

The distillation treatment system is a self-contained unit and will be relatively simple to construct. The RO system or the chemical treatment system will be dismantled and shipped from Monticello to Monument Valley, and should be relatively simple to reconstruct also. The SBRs used for the biological denitrification process must be constructed at the site. The distillation system can be installed outdoors, and will require a concrete slab or slabs as a foundation, as well as piping and electrical connections. The RO unit or the SBRs, on the other hand, must be installed indoors in order to guard against freezing. Therefore, a permanent building will be required for the systems using these processes. The RO system may also require a feed heater in order to reduce the amount of reject water produced. The feed preheater is part of the existing installation, having already been installed at Monticello.

The solar evaporation pond for brine is not expected to be difficult to construct. The larger size of the pond required for the RO system will add cost but is not expected to add significantly to the difficulty of installation.

The native plant-farming system~~s~~ utilizes extensive irrigation systems with many thousands of feet of piping. Installation of these systems will not require highly skilled labor, but will require a considerable amount of less-skilled labor, as will the seeding and/or planting necessary to establish the system.

The extraction and injection system for the active treatment alternatives will be easier to construct than the extraction system for the native plant-farming treatment alternative~~s~~ because it requires many fewer wells (40 vs 86), does not require a holding tank with controls, and because the injection wells will not require pumps.

8.4.2.2 Ease of Operation and Maintenance

The distillation and RO systems are expected to be relatively easy to operate, because they are packaged systems designed to require minimal operator interface beyond routine monitoring. Both of these treatment systems will shut off automatically in the event of problems, and will relay the required information to the system monitor. The cost estimate for the distillation system assumes only a single day-shift operator for operations and maintenance, though the operator for the extraction and injection systems will be available to supplement this operator on the rare occasions that additional labor is expected to be needed. Based on experience at Monticello, it is considered unwise to allow the RO system to operate unattended for extended periods of time; among other things, cleaning of the osmosis elements, which must be done frequently, is a manual operation. So the cost estimate for the RO system includes 24-hour operator coverage, with a single operator on-site during the daytime, and two operators at night when the extraction/injection system operator will not be available. Operation of the chemical treatment plant and the biological denitrification facility is expected to require the same level of manpower support as the RO unit. These positions are specialty jobs, and persons filling them will require extensive training.

The operation/maintenance personnel for the active treatment systems will require a relatively high degree of technical and mechanical competence. Maintenance of the distillation system is expected to be infrequent, but will not be inexpensive, since special parts and services which may only be available from the vendor or manufacturer will be required to repair and maintain these units. Maintenance of the RO system will be more frequent, as described above. The most onerous maintenance task on the RO system will be element change out and replacement, which it is hoped will be required relatively infrequently. Maintenance of the chemical treatment and biological denitrification systems is expected to be somewhat more frequent than that of the RO system, though not dramatically more so.

The native plant farming system is expected to be very labor-intensive. The extensive irrigation system will require continuous maintenance during the irrigation season, and there is expected to be a regular need for "gardening" duties such as harvesting excess saltbush growth and weeding. During the winter months, and at night during the summer, the system will probably be left unattended. The cost estimate assumes two operators working 12 hours per day for 240 days.

The type and skill level needed for operation and maintenance of the extraction and injection systems used for the active treatment alternatives is expected to be comparable to that for the simple extraction system required for the native plant-farming system. While the extraction system for the native plant-farming treatment alternatives is more extensive than that required for the active treatment alternatives, the simple extraction system is only operated seasonally, while the active treatment alternatives require year-round maintenance and operation. The cost estimates for the extraction and injection system assumes one employee full-time on day shift year-round, while the cost estimate for the seasonal extraction system assumes two employees on day shift for 240 days a year.

8.4.2.3 Expected Reliability

A less complex system is generally more reliable than a complex design. The distillation system is expected to require less than 10 percent down-time for routine maintenance. An estimated down-time of 15 to 20 percent or greater will be required to properly maintain the RO and chemical treatment/biological denitrification systems; the RO system is expected to be the more reliable of the two. The vast irrigation system required for the native plant-farming treatment alternative is expected to be relatively unreliable, but most problems are expected to be local in nature; failures affecting all or large portions of the irrigation system will be relatively uncommon. The most troublesome aspect of reliability of the irrigation system is expected to be the initial startup at the beginning of each irrigation season.

The extraction/injection system required for the active treatment alternatives is expected to be significantly more reliable than the extraction system required for the native plant-farming systems, because of its smaller size and because it will be operated continuously, which is generally easier on equipment such as pumps than frequent and extensive shutdown.

8.4.2.4 Ability to Handle Changes in Influent Composition

The native plant farming system is not expected to be significantly impacted by minor changes in influent composition. An increase in nitrate concentration will promote greater growth of vegetation, while a decrease will cause less vigorous growth. Major increases in nitrate concentration could exceed the nitrate uptake capacity of the existing plants. Changes in the concentration of other, inert constituents will not affect the quantity or the quality of the vegetation produced.

Changes in influent concentration will affect the rate of brine generation in the distillation system, but since the contaminants in the ground water are not volatile, this system is expected to be reasonably tolerant of changes in influent.

Under most circumstances, a change in influent composition would affect only the quantity of reject water produced by the RO system. However, since the nitrate concentration in the influent is already high, a significant increase in nitrate concentration could cause “breakthrough” of nitrate into the treated water at levels above the nitrate treatment standard.

In the case of the chemical treatment with biological denitrification process, changes in feed composition will require changes in chemical addition rates, so the system will have to be reviewed to insure that it provides adequate flexibility.

If it becomes necessary to meet treatment standards for additional constituents, the effectiveness of an active process will depend on the nature of the contaminant to be treated. Distillation removes a very high percentage of all nonvolatile contaminants, while RO removes a very high percentage of all contaminants having ion sizes larger than nitrate. RO would be ineffective for treatment of species having very small ions, while distillation would be ineffective against volatile species such as light organics. The chemical treatment system may lack the flexibility to be able to treat additional contaminants effectively, since the treatment required may be different than what is provided in the present design.

8.4.2.5 Ability to Handle Increases in Extraction Capacity

The native plant farming system is limited in capacity by the area available for planting, and by the maximum water available from the alluvial aquifer. The system proposed here stretches both of these factors approximately to their limits. It will probably not be possible to accelerate the remediation timetable using these processes beyond the 23 years currently projected. If it becomes necessary to treat more water than currently projected, additional treatment time will be required.

The RO system is designed for a permeate rate of up to 150 gpm if all three trains are in operation. Assuming a brine rate of 14 percent, two trains can handle a total feed flow of up to 115 gpm. Since the design extraction rate is 103 gpm, normal operation will require operating two of the trains while the third remains in stand-by mode, which is the arrangement recommended by the manufacturer. However, all three trains can be operated in parallel if required. Also, the maximum feed rate for a RO system is to an extent a function of the feed composition. If an increase in feed rate were accompanied by a decrease in contaminant

concentration, such as if wells from less contaminated areas of the plume were put on-line, it would have little effect on the RO system.

The distillation and chemical treatment and biological denitrification units will be specified for a maximum capacity of about 100 gpm. Increasing the capacity beyond 100 gpm will require purchase and installation of additional unit(s). This will be expensive but, in the case of the distillation system, fairly easy to implement. Adding capacity to the chemical treatment unit will be more complicated because the existing system was custom-built, and a new unit would also have to be custom-built. Selection of the design and construction firms would require competitive bidding and monitoring, including some capabilities (e.g. electrical review) which the GJO site would have to subcontract.

8.4.3 Comparative Cost

The estimated capital cost, annual O&M cost, and present worth value for each of the individual treatment processes, as well as for the extraction and injection treatment alternatives, have been given in their respective articles in Section 8.3, and are summarized below. All costs are in millions of dollars. ("Phytoremediation of Subsoil Ammonia" refers to the planting of native species in the area of the former tailings pile to remediate ammonia-contaminated soil. "Phreatophyte Phytoremediation" refers to the planting of black greasewood over the area of the nitrate plume to clean the shallow portion of the alluvial aquifer.)

	------ Cost in Millions of $ ------		
Process	**Capital**	**O&M (annual)**	**Present Worth**
Phytoremediation of Subsoil Ammonia	0.20	n/a	0.20
Phreatophyte Phytoremediation	0.25	n/a	0.25
Native Plant Farming	0.98	0.25	3.69
Chemical Treatment with Biological Denitrification	0.97	1.38	15.58
Distillation	3.01	0.55	8.62
Reverse Osmosis	1.19	1.00	11.72
Vertical Extraction Wells w/o Injection	4.57	0.42	9.05
Vertical Extraction Wells with Injection	2.31	0.32	5.56

The costs for the treatment alternatives are calculated by totaling the costs of the various processes which comprise them. All treatment alternatives include phytoremediation of subsoil ammonia and phreatophyte phytoremediation. The native plant farming alternative includes vertical extraction wells without injection, while the other three alternatives include vertical extraction wells with injection.

The costs of the complete treatment alternatives then are as follows.

Treatment Alternative No.	Treatment Alternative Name	------ Cost in Millions of $ ------ Capital	O&M (annual)	Present Worth
1	Native Plant Farming	6.00	0.67	13.17
2	Chemical Treatment with Biological Denitrification	3.73	1.71	21.56
3	Distillation	5.77	0.87	14.61
4	Reverse Osmosis	3.94	1.32	17.71

8.4.4 Comparative Summary

The preceding discussion has presented ten evaluation criteria, and has compared the four treatment alternatives with regard to each of these criteria. The following table lists the treatment alternatives, in order of preference with 1 being the highest, for each of these ten evaluation criteria.

Criterion	Relative Ranking of Treatment Alternatives			
	1	2	3	4
Conformance with Project Treatment Standards (40 CFR 192) and Goals	Dist	RO	Chem/Bio	NPF
Impact on the Aquifer	Dist	Chem/Bio	RO	NPF
Residual Disposal	NPF	Dist	RO	Chem/Bio
Constructability	Dist	RO	Chem/Bio	NPF
Ease of Operation and Maintenance	Dist	RO	Chem/Bio	NPF
Expected Reliability	Dist	RO	Chem/Bio	NPF
Ability to Handle Changes in Influent Composition	NPF	Dist	RO	Chem/Bio
Ability to Handle Increases in Extraction Capacity	RO	Dist	Chem/Bio	NPF
Comparative Cost—Initial Capital Outlay	Chem/Bio	RO	NPF	Dist
Comparative Cost—Present Worth	NPF	Dist	RO	Chem/Bio

Key: Dist = Distillation
Chem/Bio = Chemical Treatment with Biological Denitrification
RO = Reverse Osmosis
NPF = Native Plant Farming

8.4.4.1 Determination of Proposed Treatment Process

Treatment Alternative 1—Native Plant Farming offers the lowest total project cost by a significant margin. It also is the preferred technology from the standpoint of residual disposal - assuming, as noted above, that the market for the plant material remains viable - and ability to handle changes in influent composition. On the other hand, it is the least desirable treatment alternative for six of the ten criteria listed: Conformance with treatment standards and goals,

impact on the alluvial aquifer, constructability, ease of operation and maintenance, expected reliability, and ability to handle increases in extraction capacity.

Treatment Alternative 2—Chemical Treatment with Biological Denitrification is the preferred technology only from the standpoint of lowest initial capital cost, because it utilizes existing DOE-owned equipment. This advantage evaporates when overall project cost is considered, the very high operating costs of this system giving it by far the highest total cost over the lifetime of the project. It is also the least-preferred technology for residual disposal and ability to handle changes in influent composition, and next-to-least for another five criteria.

Treatment Alternative 3—Distillation is the first choice in five criteria—conformance with treatment standards and goals, impact on the aquifer, constructability, ease of operation and maintenance, and expected reliability. It finished second in four of the other five criteria, by far the best showing of any technology in the process-related criteria. The only serious drawback to distillation is that it has the highest capital cost of any treatment alternative.

Treatment Alternative 4—Reverse Osmosis, is rated highest in ability to handle increases in extraction capacity; it is the only technology, in fact, that can absorb a significant increase in extraction capacity without either requiring additional costly treatment modules, or being overdesigned in the first place. (This is not an inherent feature of the RO technology, but instead is due to the fact that this treatment alternative utilizes an existing DOE-owned facility with a capacity of nearly 175 gpm.) RO is second choice in five criteria - constructability and initial capital outlay (because it utilizes existing equipment), conformance to treatment standards and goals, ease of operation and maintenance, and expected reliability. It did not rate last in any category - the only treatment alternative for which this was true.

Of the “active” processes, chemical treatment with biological denitrification is the poorest choice, rating third or fourth in eight of the ten criteria. The only attractive aspect of chemical treatment with biological denitrification is its low initial capital cost, but this is more than outweighed by its present-worth value, which is the highest of any of the treatment alternatives. Thus, there is no reason to consider this alternative further.

Reverse osmosis is considerably more attractive than chemical treatment with biological denitrification. It also offers a relatively low capital cost, while rating higher than chemical treatment with biological denitrification in almost all the process-related criteria. Compared to distillation, however, RO is definitely an inferior alternative. Although RO is less expensive to implement, since it utilizes existing DOE-owned equipment, it is more expensive to operate (primarily because it requires more operating manpower), which results in a total project cost which is 25% higher than the total cost for distillation. And while RO offers numerous process advantages compared with chemical treatment with biological denitrification, it trailed distillation in almost all of these same criteria. Since RO is less attractive from a process standpoint than distillation, and has a significantly higher total project cost, distillation is the preferred “active” alternative.

Although distillation is the least expensive “active” treatment alternative, its total project costs exceed those of the native plant farming alternative by almost $1.5 million. However, many

uncertainties are associated with implementation of the native plant farming option. These have been covered in detail elsewhere and will be briefly reiterated here.

- The inefficiency and possible ineffectiveness of an extraction system that does not include injection of treated water to the aquifer.
- Uncertainty as to the rate at which contaminated water can be absorbed and nitrate can be uptaken by the species chosen.
- Uncertainty as to the effect on the soil of sulfate build up as a result of the irrigation process.
- Uncertainty as to the effect on the alluvial aquifer of the loss of over 1 billion gallons of water.
- Uncertainty as to the effect on the alluvial aquifer, and on the contamination zones, of the cyclic nature of the extraction process.
- Uncertainty as to whether the plant material produced will be accepted as a viable feed. Even if initial acceptance is good, if there should be unexplained incidents of livestock death after eating this material, at any time during the twenty-plus year lifetime of the remediation project, it could immediately become impossible to give the material away, and what is now an asset would instantly become a major liability.

A pilot test of the native plant farming process had been proposed by the DOE which could have answered at least some of these questions. It could not, however, have addressed several of them, including the issue of long-term stakeholder acceptance of the plant material. So, even though native plant farming process it offers several attractive features, including low total project cost and beneficial reuse of a contaminant, the unresolved questions regarding the native plant farming process make its selection as the treatment of choice for the Monument Valley site very problematic.

No comparable uncertainties exist with the distillation process. Distillation is currently being implemented at the Tuba City site, so the DOE will have actual experience operating this process at an UMTRA Ground Water site by the time remediation begins at Monument Valley. Distillation more than satisfies the regulatory requirements of 40 CFR 192 and will produce a high-quality effluent that will recharge the aquifer with a minimum of loss while containing and preventing the spread of the contaminant plume. Whether any pump-and-treat process can successfully remediate the contaminated ground water at the Monument Valley site cannot be known at this time. What can be said with confidence, however, is that the distillation process will give a successful remediation if any pump-and-treat process can. DOE must also consider the stakeholders' desires to preserve the integrity of the aquifer if possible. Distillation preserves the water resource to the greatest extent of any of the treatment processes considered, while the native plant farming system does not preserve it at all. In consideration of all the above, then, Treatment Alternative 3—Distillation, is the preferred treatment technology for the Monument Valley ground water remediation program.

8.5 Proposed Remediation Process~~es~~

The proposed remediation process incorporates phytoremediation of subpile ammonia using fourwing saltbush, phreatophyte phytoremediation of shallow portions of the aquifer using black greasewood, and active remediation of the deeper portions of the aquifer using distillation. The phytoremediation aspects of the remediations are discussed in Section 8.2.4.1. The active portion of the remediation program consists of three systems. This section discusses each of those systems.

8.5.1 Description of Proposed Remediation Process

8.5.1.1 Proposed Extraction System

The extraction system consists of a total of 40 extraction wells, varying in depth up to a maximum of approximately 90 ft., depending on the depth of the alluvial aquifer at the particular location. The expected flow rate per well is 3 gpm, giving the extraction system a peak capacity of 120 gpm once all wells are in service.

A typical extraction-well design for the Monument Valley site would consist of a 10-in. diameter borehole completed with 6-inch diameter wire-wrapped well screen, or pre-packed screen, and blank PVC. The section of the well containing the well screen will be completed with an appropriately-sized sand pack. The final design of the well and the size of the pump will be optimized based on field conditions.

The extraction wells will be installed across the nitrate plume, an area measuring approximately 11.2 million square feet or about 260 acres. The water pumped from these wells must be collected from across this substantial area and delivered to the treatment facility. Each pump will discharge into a 6 in. PVC outlet pipe. These outlet pipes will be directed into a series of headers, which in turn will connect to the main 6 in. PVC extraction system discharge pipe which is routed to the feed pond.

8.5.1.2 Proposed Distillation Treatment System

The water from the extraction system will be collected in a 20,000-gal feed tank. This tank will be equipped with a level control system with full instrumentation and controls. From the feed tank, contaminated water is pumped directly to the distillation system, which will consist of one self-contained unit with a feed capacity of between 100 and 120 gpm. The unit will be instrumented to permit continuous operation with remote monitoring capability.

The concentrated brine from the distillation unit, which is expected to average less than 2 percent of the total feed, will be pumped to a solar evaporation pond for final concentration. This pond will be sized to hold all of the sludge produced during the lifetime of the remediation project. The dry sludge from this pond will be removed at the end of the project.

The treated water from the distillation system, expected to average 98 percent or more of the total feed, will be pumped to a distillate tank having a capacity of approximately 10,000 gal. This

will provide holding capacity so that the injection system can continue to operate during minor upsets in the distillation system.

8.5.1.3 Proposed Injection System

Water from the distillate tank is pumped continuously to the injection system. This will consist of approximately 40 injection wells, varying in depth to a maximum of approximately 90 ft. The expected flow rate per well is 3 gpm, giving the injection system a peak capacity of 120 gpm once all wells are in service. The construction of the injection wells is similar to that of the extraction wells, except that no pump or discharge piping are used.

8.5.2 Summary

The proposed system meets or exceeds the requirements of 40 CFR 192, and is protective of human health and the environment. The products of the treatment system are a high-quality treated water, constituting about 98 percent of the mass of the water extracted from the alluvial aquifer, which will be injected into the alluvial aquifer; and a concentrated brine, containing essentially all the dissolved and suspended solids present in the untreated ground water, which will be concentrated on-site in a solar evaporation pond and removed for disposal at a remote location at the conclusion of the remediation project.

8.5.3 Limitations of Proposed Alternative

Although ground water extraction and ex-situ treatment, also known as pump and treat, was found to be the best method to meet cleanup goals in the aquifer, the effectiveness of pump-and-treat systems has been limited. Few sites with contaminated ground water have ever been restored to drinking water standards (Travis 1990; EPA 1996); however, the vast majority of sites where pump and treat is now being used are dealing with sources composed of non-aqueous-phase liquids. Nevertheless, although the constituents at the Monument Valley site are dissolved and expected to behave conservatively, the cleanup standard for nitrate has been set at the drinking water standard. Consequently, the effectiveness of the ground water extraction system is the primary factor that determines whether aquifer cleanup goals are met.

Technical criteria will need to be established to evaluate the success of the remediation. These criteria will be developed in the GCAP after discussion with stakeholders. The GCAP will define the logic that will be used to evaluate the success or failure of the remedial action. It will also propose the steps that might be taken if the concentrations indicate significant “tailing,” that is, an absence of continued improvement in the ground water quality with time.

The main factors that influence the effectiveness of ground water extraction systems are hydraulic inefficiencies, heterogeneity of the aquifer, and sorption of contaminants to the aquifer material. Hydraulic inefficiencies account for the diffusion of contaminants into low-permeability sediments and hydrodynamic isolation (stagnation points) within a well field. Heterogeneities of the aquifer (e.g., changes in the hydraulic conductivity and effective porosity) will affect the ability to extract ground water from all areas of the aquifer. The sorption of contaminants to the aquifer material retards the movement of the contaminants in the ground water. The more a contaminant sorbs to the aquifer matrix the more ground water must be extracted to remove the contaminant.

If active remediation cannot achieve the cleanup levels, other methods of protecting human health might be pursued. A provision in 40 CFR 192 allows the use of ACLs that would be set at a higher concentration than the current cleanup goals but that would still be protective of human health. The use of ACLs may require that the area within the fence surrounding the formal site be extended to incorporate areas of the plume that could not be remediated to the cleanup levels. Use of ACLs and extending the fenced area would only be considered if active remediation could no longer effectively reduce contaminant levels in the aquifer.

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