Final Site Observational Work Plan for the UMTRA Project Site near Tuba City, Arizona

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Prepared by
MACTEC Environmental Restoration Services, LLC
Grand Junction, Colorado

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Plate 1. Selected Geologic Features at the Tuba City, Arizona, UMTRA Project Site

Plate 2. Geologic and Geohydrologic Cross Sections through the Tuba City, Arizona, UMTRA Project Site

Acronyms and Abbreviations

ACL	alternate concentration limit
BLRA	baseline risk assessment
Btu	British thermal units
CeRaM	Center for Radioactive Waste Management
CFR	*U.S. Code of Federal Regulations*
Cl	chloride
cm	centimeter(s)
cm/s	centimeter(s) per second
cm/yr	centimeters per year
COCs	contaminants of concern
COPCs	contaminants of potential concern
DOE	U.S. Department of Energy
EA	environmental assessment
EPA	U.S. Environmental Protection Agency
ET	evapotranspiration
ft	foot (feet)
ft^2	square foot (feet)
ft/day	feet per day
ft^2/day	square foot (feet) per day
ft^3/day	cubic feet per day
GCAP	Ground Water Compliance Action Plan
GJO	Grand Junction Office
gpm	gallons per minute
GWPP	Ground Water Program Plan
ITRD	Innovative Treatment Remediation Demonstration
L	liter
Mg	magnesium
MCL	maximum concentration limit
mg/L	milligrams per liter
mi^2	square mile
Mn	manganese
Mo	molybdenum
N	nitrogen
Na	sodium
NEPA	National Environmental Policy Act
NH_4	ammonium
NRC	U.S. Nuclear Regulatory Commission
O&M	operating and maintenance
OMB	Office of Management and Budget
pCi/g	picocuries per gram
pCi/L	picocuries per liter
PEIS	Programmatic Environmental Impact Statement
PNNL	Pacific Northwest National Laboratory
ppm	parts per million

Acronyms and Abbreviations (continued)

PVC	polyvinyl chloride
RAP	remedial action plan
RO	reverse osmosis
RRM	residual radioactive material
S	sulfur
SBR	sequencing-batched reactor
Se	selenium
SI	saturation index
SO_4	sulfate
SOP	standard operating procedure
SOWP	site observational work plan
Sr	strontium
TAC	Technical Assistance Contractor
TAGR	Technical Approach to Groundwater Restoration
TDS	total dissolved solids
TEM	Transient Electromagnetic
U	uranium
UMTRA	Uranium Mill Tailings Remedial Action (Project)
UMTRCA	Uranium Mill Tailings Radiation Control Act
USC	*United States Code*
U.S.G.S.	United States Geological Survey
yd^3	cubic yards

1.0 Introduction

The Tuba City, Arizona, Uranium Mill Tailings Remedial Action (UMTRA) Project site observational work plan (SOWP) Revision 1 is the basis for compliance with the U.S. Environmental Protection Agency (EPA) ground-water standards at this UMTRA Project site. The purpose of the SOWP Revision 1 is (1) to summarize data collection efforts to date, (2) to review the site-specific ground-water compliance strategy at the Tuba City UMTRA Project site, (3) to describe a set of remediation alternatives that, if implemented, would attain compliance with EPA ground-water standards, and (4) to propose a recommended remediation alternative for the site.

A site conceptual model is developed from data collected. The model describes sources of existing contamination at the site and defines the current conditions and potential environmental and health risks. The site conceptual model is also a basis for predictive modeling simulations of the proposed remediation alternatives. Remediation alternatives for the Tuba City site were developed by the U.S. Department of Energy (DOE) Grand Junction Office (GJO), in conjunction with project stakeholders, through the Innovative Treatment Remediation Demonstration (ITRD) Program administered by Sandia National Laboratories. The ITRD process examined new and innovative as well as conventional ground-water extraction and treatment methods. On the basis of lessons learned during the ITRD process, DOE is proposing a recommended alternative for the site. This recommendation will provide a basis for technical and policy discussions with stakeholders and lead to consensus regarding site remediation.

1.1 UMTRA Project Programmatic Documents

Programmatic documents that provide guidance for the SOWP include the *UMTRA Ground Water Program Plan* (GWPP) (DOE 1992), the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (PEIS) (DOE 1996), and the *Technical Approach to Groundwater Restoration* (TAGR) (DOE 1993b). The GWPP states the mission need and objectives for the UMTRA Ground Water Project and provides an overall technical plan and management approach for conducting the project. The PEIS provides an objective programmatic decision-making framework for conducting the UMTRA Ground Water Project, assesses the potential programmatic effects of conducting the project, provides a method for determining the site-specific ground-water compliance strategies, and provides data and information that can be used to effectively evaluate site-specific environmental effects. The TAGR provides general technical guidance for conducting the UMTRA Ground Water Project.

The surface remedial action plan (RAP) provides site characterization information (DOE 1989). This information is updated in the SOWP to formulate the site conceptual model. Because a ground-water compliance strategy requiring active remediation was selected for this site, it is necessary to prepare a ground-water RAP or surface RAP modification.

In 1994, a baseline risk assessment (BLRA) identified the potential public health and environmental risks at the site (DOE 1994a). Potential risks identified in the BLRA are considered in this SOWP to ensure that the proposed compliance strategy is protective of human health and the environment.

1.2 Relationship to Site-Specific Documents

DOE has used numerous contractors to support characterization of the Tuba City site during the project history. Adjunct technical support has come from the University of Arizona in conjunction with Navajo Community College in Tuba City, the University of New Mexico in combination with the Center for Radioactive Waste Management (CeRaM), and the ITRD Program in conjunction with Sandia National Laboratories and its contractors. These institutions have completed work on a number of tasks and have authored several site-specific reports that describe various technical engineering aspects of remediating the Tuba City site. Interactions among these institutions, DOE, and other project stakeholders combined with engineering assessments performed by the Technical Assistance Contractor (TAC) in Albuquerque, New Mexico, and the Technical Assistance and Remediation contractor in Grand Junction, Colorado, form the basis of current knowledge of site conditions and for the remedial alternatives described in this document.

1.3 SOWP Revisions

The UMTRA program has traditionally been conducted according to the observational method of site characterization. Therefore, the initial version of the SOWP was relatively general in its conclusions and recommended further study using geophysical, hydrological, geochemical and pilot-scale remediation tests. Results from those investigations and adjunct studies are now completed and can be found in separate reports. The compilation of results from those studies, updates to the site conceptual model, and presentation of engineering alternatives for the site are included in this document.

2.0 Regulatory Framework

A ground-water compliance strategy is proposed for the Tuba City site to achieve compliance with the EPA ground-water standards applicable to Title I UMTRA Project sites. This section identifies the requirements of the Uranium Mill Tailings Radiation Control Act (UMTRCA), the EPA ground-water protection standards (40 CFR Part 192), National Environmental Policy Act (NEPA), and other regulations that are applicable to the UMTRA Ground Water Project.

2.1 Federal Regulations

2.1.1 Uranium Mill Tailings Radiation Control Act

The U.S. Congress passed UMTRCA (42 USC 7901 *et seq.*) in 1978 in response to public concerns about the 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 uranium mill processing sites.

UMTRCA has three titles that apply to uranium processing sites. Title I designates 24 inactive processing sites to undergo remediation, directs the EPA to promulgate standards, mandates remedial action in accordance with standards prescribed by the EPA, directs remedial action to be selected and performed with the concurrence of the U.S. Nuclear Regulatory Commission (NRC) and in consultation with states and Indian tribes, directs the 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, and Title III applies to certain uranium mills in New Mexico. The UMTRA Ground Water Project has responsibility for administering only Title I of UMTRCA.

In 1988, Congress passed the Uranium Mill Tailings Remedial Action Amendments Act (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.

EPA Ground-Water Standards

UMTRCA requires that EPA promulgate standards for protecting public health and the environment from residual radioactive materials (RRM) associated with the processing of uranium. On January 5, 1983, EPA published standards in 40 CFR Part 192 for the disposal and cleanup of RRM. The standards for ground-water compliance were revised, and a final rule was published on January 11, 1995, and codified in 40 CFR 192.

Subpart B, of 40 CFR 192 "Standards for Cleanup of Land and Buildings Contaminated with Residual Radioactive Materials from Inactive Uranium Processing Sites," requires that contamination as a result of RRM meet one of the following standards:

- Background level—concentrations of constituents in the uppermost aquifer that was not contaminated by processing activities.

- Maximum concentration limit (MCL)—the EPA's maximum limits for concentrations of certain constituents in ground water, as proposed for the UMTRA Project. The MCLs for inorganic constituents that apply to the UMTRA Project sites are given in Table 2–1.

- Alternate concentration limit (ACL)—an alternate limit for a constituent that does not pose a substantial present or potential hazard to human health or the environment, as long as the limit is not exceeded. An ACL may be applied after it is determined that an MCL does not exist for a particular constituent.

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

Constituent	Maximum Concentration Limit[a,b]
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[c]
Selenium	0.01
Silver	0.05
Combined radium-226 and radium-228	5 pCi/L
Combined uranium-234 and uranium-238	30 pCi/L[d]
Gross alpha activity (excluding radon and uranium)	15 pCi/L

[a]40 CFR 192.
[b]Milligrams per liter (mg/L) unless otherwise noted.
[c]Equivalent to 44 mg/L nitrate as nitrate (NO_3^-).
[d]Equivalent to 0.044 mg/L.

pCi/L—picocuries per liter.

In lieu of the three standards, DOE may, with NRC concurrence, apply supplemental standards to contaminated ground water. Supplemental standards may be applied if any of the following conditions is met:

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

- Remedial actions to meet the standards would directly produce environmental harm that is clearly excessive compared to the health and environmental benefits, 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 impracticable from an engineering standpoint.

- The ground water is considered limited-use (Class III) ground water. Subpart B of the standards 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 (i.e., contamination that is not related to millsite activities) that cannot be cleaned up using treatment methods reasonably employed in public water supply systems; or the quantity of water available for continuous use is less than 150 gallons (570 liters [L] per day) (40 CFR 192.11(e)). When limited-use ground water applies, "supplemental standards shall ensure that current and reasonably projected uses of the ground water are preserved" (40 CFR 192.22(d)).

- 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.

The regulations also require DOE to inform any private owners or occupants of the location affected by hazardous constituents and to solicit their comments. DOE has implemented a public participation program to meet the requirements of Subpart C.

2.1.2 National Environmental Policy Act

Implementation of UMTRCA meets the criteria for assessment subject to the requirements of NEPA (42 USC 4321 *et seq.*). DOE NEPA regulations are contained in "National Environmental Policy Act Implementing Procedures" (10 CFR Part 1021).

Pursuant to NEPA, DOE finalized a PEIS for the UMTRA Ground Water Project to analyze potential effects of implementing four programmatic alternatives for conducting ground-water compliance at the UMTRA Project processing sites.

A Record of Decision published in April 1997 presented DOE’s preferred alternative. The decision gave DOE the option of implementing one or a combination of the following compliance strategies:

- Active ground-water remediation
- Natural flushing
- No ground-water remediation

2.1.3 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 and 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.

2.2 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 or Hopi Tribe exercises the right of sovereignty. Examples include the right of the Navajo Nation to require water-use permits and permits to drill wells.

2.3 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).

2.4 Agreements

UMTRCA requires that compliance with the ground-water standards be accomplished with the full participation of the states and 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 will negotiate a cooperative agreement with the Navajo Nation and Hopi Tribe.

The Navajo Nation, in a letter dated September 18, 1997, proposed secondary cleanup levels for restoration of ground water at the Tuba City site and a level of sodium in the final treated water. They were guided in requesting these cleanup levels by the relatively high quality of the background water in the Navajo sandstone aquifer. DOE will consider the cleanup levels proposed by the Navajo Nation as targets for restoration of the aquifer and will try to meet the cleanup levels to the extent practicable. The Navajo Nation's proposed secondary cleanup levels for restoration of the groundwater are:

- TDS 500 mg/L
- Sulfate 250 mg/L
- Chloride 250 mg/L
- pH 6.5 - 8.5
- Corrosivity Non-corrosive
- Sodium 20 mg/L

3.0 Site Background

The Tuba City UMTRA site is located in Coconino County, Arizona, just south of U.S. Highway 160, in Section 20, T32N, R12E. The site is on the Navajo Indian Reservation and close to the Hopi Reservation, approximately 5 miles east of Tuba City and 85 miles northeast of Flagstaff (Figure 3–1). Moenkopi Village is just southeast of Tuba City along Moenkopi Wash.

The Tuba City UMTRA site lies at an elevation of approximately 5,100 feet (ft) above sea level on a terrace that slopes gently to the southwest. The terrace surface is a thin veneer of unconsolidated dune sand and gravels that mantle an underlying pediment on the Navajo Sandstone. The site is approximately 6,000 ft northwest of and 300 to 400 ft in elevation above Moenkopi Wash, an intermittent stream that drains to the southwest into the Little Colorado River.

Land use in the immediate vicinity of the Tuba City site is limited to occasional grazing. Adjacent land is used for dry and irrigated farming and for residences. Water from Moenkopi Wash is used for agricultural and religious applications by the Navajos and Hopis in the vicinity of the site. The limited and highly variable supply of surface water makes ground water an important resource in the area. Two points of shallow ground-water withdrawal exist within a 2-mile radius of the site, including a low-yield domestic well and a spring. Four deeper water-supply wells are located north of U.S. Highway 160 and were used by the Tuba City millsite for processing water (Figure 3–2).

3.1 Site History

The uranium mill at the Tuba City site was operated by Rare Metals Corporation of America from start-up in 1956 until 1962. In 1962, Rare Metals merged with El Paso Natural Gas Company, which ran the mill until it closed in 1966. The mill processed approximately 800,000 tons of ore during the 10-year period; tailings were placed as a slurry (a mixture of water, chemicals, and solids) in three contiguous piles at the site. Between 1956 and 1962, the mill processed an average of 300 tons of ore per day using a sulfuric-acid leach. The plant was reconfigured in 1962 to use sodium carbonate in an alkaline process, and from 1963 to 1966 an average of 200 tons of ore per day was processed. All tailings from mill processing were placed as a slurry in evaporation ponds at the site; these ponds covered an area of about 33.5 acres (MACTEC–ERS calculation U0000201). Figure 3–3 shows the locations of the former processing facilities relative to the present location of the disposal cell.

DOE began surface remedial action at the Tuba City site in 1988. The uranium mill tailings and associated materials were moved and stabilized in an engineered disposal cell. The tailings were stabilized in place, which means that all contaminated materials from around the pile were consolidated with the tailings and the pile was covered with compacted earth to inhibit radon emanation and water infiltration. Surrounding windblown contaminated soils were also placed in the disposal cell. The tailings pile was structured to allow the placement of progressively less contaminated materials into the final pile configuration.

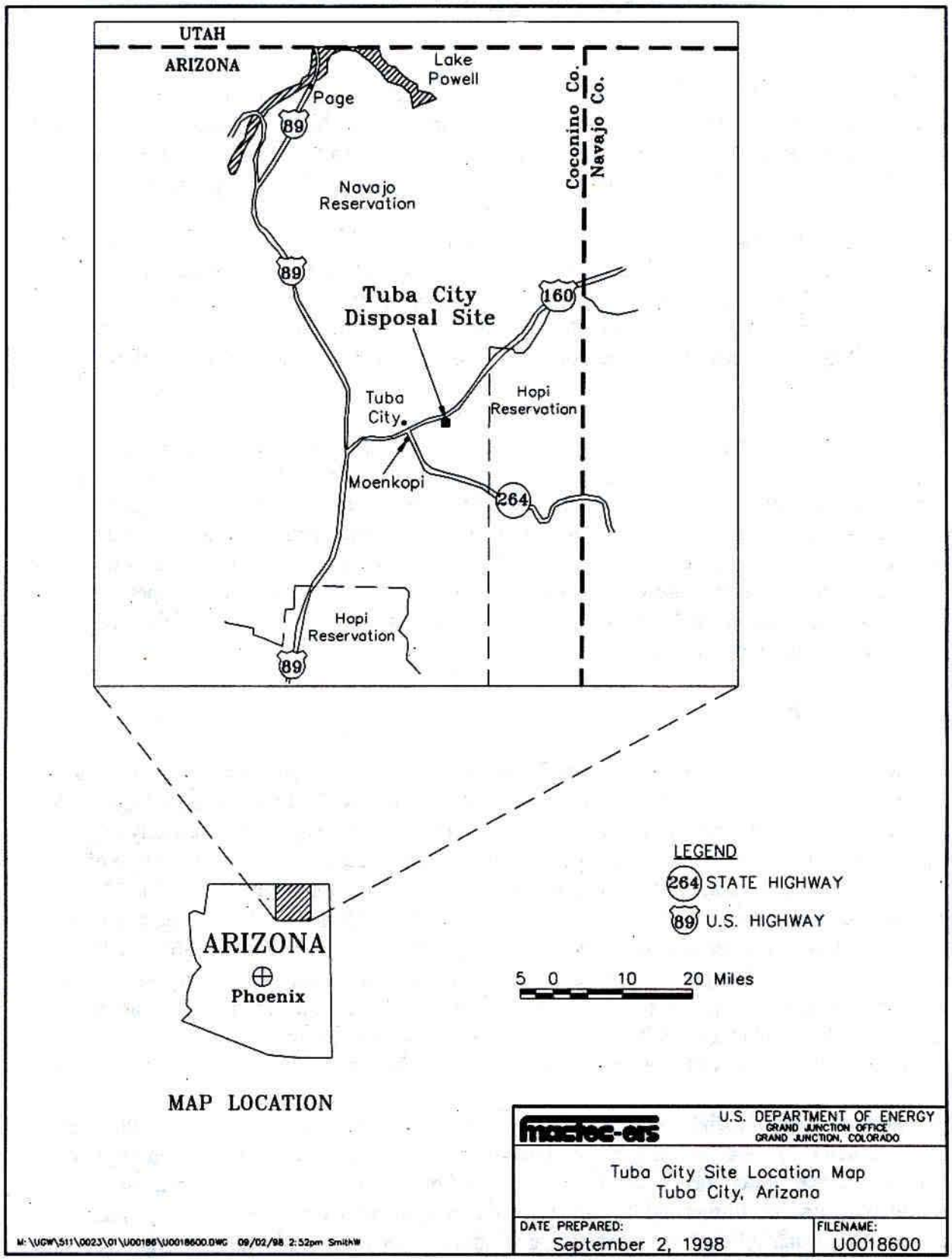


Figure 3–1. Site Location Map

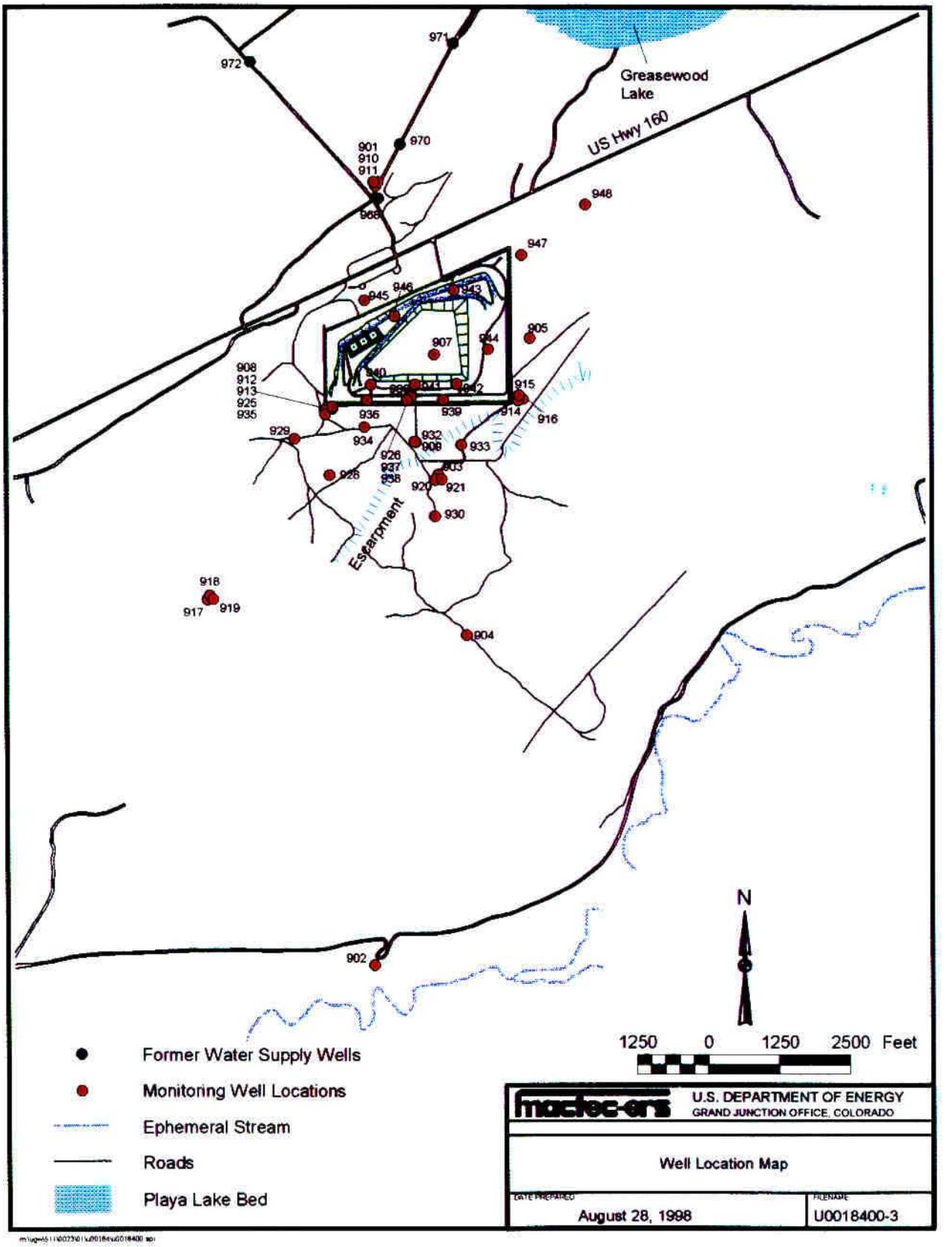


Figure 3–2. Well Location Map

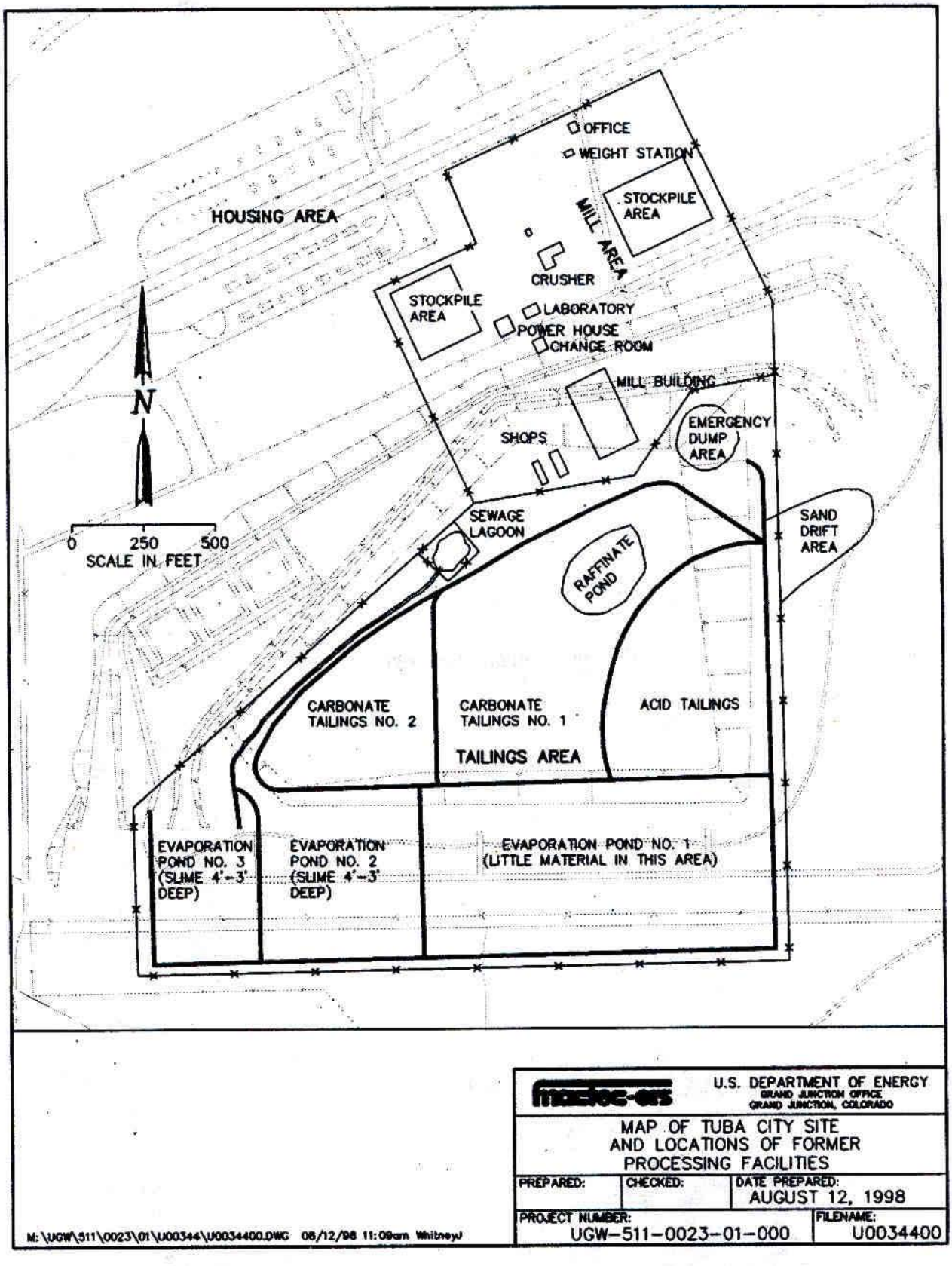


Figure 3–3. Map of Tuba City Site and Locations of Former Processing Facilities

After all the contaminated materials were placed into a pile, a layer of clayey sand material obtained from the dry lake bed of Greasewood Lake (Figure 3–2) was placed and compacted over the entire pile to reduce radon emissions and water infiltration. The Greasewood Lake fill contained no bentonite or other additives; it was compacted to have a saturated hydraulic conductivity on the order of 1×10^{-8} centimeters per second (cm/s). The geometric mean of engineering tests during construction indicate that the cover was constructed with an actual saturated hydraulic conductivity of 1.3×10^{-8} cm/s (MACTEC–ERS calculation U0000201, Adrian Brown Consultants 1988). The highly compacted radon/infiltration barrier was designed so that a minimum of 20-percent of the material would pass through a No. 200 sieve (DOE 1995a, Appendix B, Section 02200–8). The composition of the earthen cover material controls the infiltration of water and helps to protect ground water from further contamination. The remedial action was completed in April 1990. A total of 1,400,000 cubic yards (yd^3) of contaminated material was stabilized in a disposal cell covering 50 acres within the 145-acre disposal site.

The Tuba City site was licensed by the NRC in 1997. As a condition upon which the license was issued, the ground-water cleanup was deferred to the UMTRA Ground Water Project. It is the opinion of DOE, as described in Section 3.2, that drainage from the cell is a relatively minor component of the total volume of contaminated ground water (MACTEC–ERS calculation U0000201).

3.2 Sources of Ground-Water Contamination

The processing of uranium ore required from as little as 200 gallons to as much as 1,000 gallons of water per ton of ore; acid leaching generally required more water than carbonate leaching (Merritt 1971). The mill used four deep water-supply wells, that tapped the Navajo Sandstone north of U.S. Highway 160.

Based on the milling history at the Tuba City site (Merritt 1971), average water consumption is estimated to have been about 800 gallons per ton from 1956 to 1962, and about 500 gallons per ton from 1962 to 1966. This water was discharged with the milled tailings as a slurry (a mixture of solids and water) to evaporation ponds. On the basis of an estimated average water requirement of 800 gallons per ton of ore for the acid leach process, the average water use was about 240,000 gallons per day (269 acre-feet per year); similarly, an average water requirement of 500 gallons of water per ton of ore for the alkaline leach process yields an average use of 100,000 gallons per day (112 acre-feet per year).

It is assumed that the entire surface area of the ponds was available for evaporation during the 10 years of mill operation. These ponds were unlined; consequently, the unevaporated water flowed vertically downward as recharge to the ground water. The ground-water recharge component contained high concentrations of dissolved constituents derived from the milling process. These constituents included primarily nitrate, sulfate, sodium, calcium, and considerably lesser amounts of uranium. The cumulative volume of the recharge is estimated below. The actual volume of recharge during mill operation could have been more or less, depending upon the actual amount of water pumped from supply wells, the evaporation rate, the area of the pond available to evaporation, and other similar considerations.

On the basis of evaporation records obtained from the U.S. Water Conservation Laboratory, the University of Arizona Mesa Experiment Farm, the U.S. Weather Bureau, and Cooley (1970), determined the monthly incremental pan evaporation rates for Tuba City to be:

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
0.147	0.207	0.333	0.440	0.600	0.660	0.660	0.600	0.460	0.353	0.220	0.147	4.83 ft

Since the amount of water used during 1956 to 1962 was estimated to be 240,000 gallons per day, and the surface area of the pond was 33.5 acres, the monthly contribution of water to the evaporation pond is estimated to have been:

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
0.682	0.616	0.682	0.660	0.682	0.660	0.682	0.682	0.660	0.682	0.660	0.682	8.03 ft

The difference between the amount of water used and the amount of water evaporated is the water that would have recharged the ground water:

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
0.535	0.409	0.349	0.220	0.082	0.000	0.022	0.082	0.200	0.329	0.440	0.535	3.20 ft

The recharge table shows that very little, if any, net recharge occurred during June; therefore, only the positive recharge values are summed for the annual recharge.

Thus, recharge through the bottom of the tailings repository from 1956 to 1962 is estimated to have been:

$R_{1956–1962}$ = (3.20 ft per year) (43,560 ft^2 per acre) (33.5 acre) (1 year/365 days) = 12,805 ft^3/day (107 acre-feet per year).

From 1962 to 1966, average water use is estimated to have dropped to about 100,000 gallons per day (Merritt 1971). This results in a monthly addition of water to the tailings pond of:

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
0.284	0.257	0.284	0.275	0.284	0.275	0.284	0.284	0.275	0.284	0.275	0.284	3.34 ft

After accounting for the volume of water lost to evaporation, the recharge component is estimated to be:

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
0.137	0.050	–0.049	–0.165	–0.316	–0.385	–0.376	–0.316	–0.185	–0.069	0.055	0.137	0.379 ft

The recharge table shows that recharge would not have occurred during March through October; therefore, only the positive recharge values are summed for the estimated annual recharge.

Recharge through the bottom of the tailings repository during 1962 through 1966 is estimated to have been:

$R_{1962–1966}$ = (0.379 ft per year) (43,560 ft^2 per acre (33.5 acre) (1 year/365 days) = 1,515 ft^3 per day (12.7 acre-feet per year).

The recharge of process water into the aquifer is thought to be an important source of the ground-water contamination at the Tuba City site. On the basis of the estimated recharge values computed above, it is possible to estimate the total volume of process water that recharged into the ground below the tailings pile. The total amount is computed as follows:

R_{total} = $R_{1956–1962}$ + $R_{1962–1966}$
= 12,805 ft^3 per day (365 days per year) (6 years) (7.48 gallons per ft^3) + 1,515 ft^3 per day (365 days per year) (4 years) (7.48 gallons per ft^3)
= 2.26×10^8 gallons
= 693 acre-feet.

The water that drained from the tailings piles from 1956 through 1966 was the principal driving force for migration of contaminants into the aquifer. The rate of drainage from the evaporation ponds is thought to have decreased exponentially after 1966 and is now essentially complete. However, surface remediation at the site may have triggered the release of water from consolidation of the slime tailings and from slow drainage of water used for engineered compaction. These two potential sources of ground-water recharge are known collectively as transient drainage. The transient drainage component from consolidation of the slime tailings may be about 1 percent (about 260,000 gallons) of the total transient drainage; however, it is relatively short lived and may now be complete (MACTEC–ERS calculation U0000201). The drainage of water used for engineered compaction is a relatively long-lived, unsaturated-flow process that may exist for 1,000 years or more and contribute up to 36.8 million gallons (MACTEC–ERS calculation U0000201). The incremental rate at which drainage occurs from the disposal cell is relatively slow; also, in comparison to the total volume of contaminated ground water at the site, up to 1.7 billion gallons, DOE considers drainage from the cell to be insignificant. However, on the basis of pore-fluid concentrations discussed in Section 5.3.2, the contaminant concentrations in the pore fluids may exceed the maximum concentration in ground water.

Contaminants dissolved in the ground-water recharge from the tailings include primarily nitrate and sulfate as well as lesser amounts of uranium. Nitrate and uranium are the two constituents present in concentrations that exceed the UMTRA ground-water protection standards at the site. Site-related contamination in ground water has been detected at least 1,500 ft downgradient from the processing site and to a depth of 75 ft below the water table. This contamination in the ground water currently poses no risk to human health or the environment (Section 7) but may pose a potential risk if the ground water is used in the future.

3.3 Previous Investigations

Regional studies and hydrogeologic and ground-water-quality characterization have been conducted in the Tuba City area since the mid-1950s. Various DOE (and preceding agency) subcontractors have performed investigations at the processing site since 1976 (FBDU 1983; FBDU 1981). Regional reports include the published work of Harshbarger et al. (1957) and Jobin (1955), a summary of the hydrogeology of the region by Cooley et al. (1969), water-level mapping near Tuba City by Farrar (1978), and a regional ground-water flow model by Eychaner (1983). UMTRA Project reports include the *Site Observational Work Plan for the UMTRA Project Site at Tuba City, Arizona* (DOE 1995b), the *Baseline Risk Assessment for Ground Water Contamination at the Uranium Mill Tailings Site Near Tuba City, Arizona* (DOE 1994a), and the *Remedial Action Plan and Site Design for Stabilization of the Inactive Uranium Mill Site at Tuba City, Arizona* (DOE 1989).

UMTRA Project field activities have included (1) drilling and installing monitoring wells in November and December 1984, August 1985, and October and November 1995; (2) installation and testing of pilot boreholes in conjunction with DOE's ITRD program in August and September 1997; (3) aquifer testing (slug tests, short-term and long-term aquifer pumping tests) in selected areas; and (4) collecting ground-water samples and water-surface elevations from monitoring wells. Over twenty rounds of ground-water samples have been collected from monitor wells between 1986 and 1997 and have been analyzed for selected constituents. The data are on file in the SEE_UMTRA database in Grand Junction, Colorado. During the first decade, these data were collected for the UMTRA Project Surface Remediation Program; more recent data were collected for the UMTRA Ground-Water Compliance Program.

4.0 Summary of Field Investigations

Field characterization work that was recommended in the SOWP Revision 0 (DOE 1995b) has been completed. This work consisted of geologic, geophysical, hydrologic, geochemical, and ecological studies. Initially, the work was performed by DOE contractors under the direction of the DOE Albuquerque Operations Office; later work was performed under the direction of the DOE–GJO after programmatic responsibilities for the UMTRA Ground Water Project were transferred to the Grand Junction Office in 1996.

4.1 Geologic Characterization

The geologic characterization of the Tuba City site consisted of (1) a review of published and unpublished geologic reports and maps of the region surrounding the UMTRA site, (2) discussions with former UMTRA project geologists in Albuquerque, New Mexico, (3) inspection of lithologic core from several of the boreholes from the Tuba City site, (4) a field inspection of the surface of the site and nearby area to observe geologic and hydrogeologic features, and (5) preparation of a map, cross sections, and a brief report to document the site conditions (MACTEC–ERS calculation U00005AA 1996).

In the summer of 1996, the GJO received lithologic core samples for the Tuba City project from the Albuquerque UMTRA office for the following 14 boreholes: 901, 902, 903, 905, 906, 907, 911, 921, 930, 932, 937, 938, 940, and 941. Subsurface data from these boreholes in combination with surface geologic mapping and lithologic and structural information presented in published and unpublished reports (Cooley et al. 1969; Haynes and Hackman 1978; Middleton and Blakey 1983; Sergent, Hauskins, and Beckwith 1985) were used to construct the geologic map and cross sections (Plates 1 and 2, respectively). The objectives of the field work at the Tuba City site were to characterize the Navajo Sandstone, the nature of the contact between the Navajo Sandstone and the underlying Kayenta Formation, and the Quaternary alluvial terrace material.

Surface geologic and hydrogeologic features were observed at the site during a two-day field inspection in June 1996. Photographs were taken of significant geologic features and broad areal geomorphic relationships. The field data and photographs were plotted on the base map (Plate 1) at a scale of approximately 1:3,600. The base map consisted of the enlarged four adjoining 7.5-minute topographic maps with an inset of a detailed (2-ft contour interval) topographic map of the site area prepared by UMTRA contractors from recent aerial photographs.

Structural features of the Navajo Sandstone were measured using a Brunton compass. The relative abundance of calcareous cement in the Navajo Sandstone and the alluvial terrace material was determined by applying drops of a 10-percent solution of hydrochloric acid. In addition to the geologic and hydrogeologic data, monitoring wells are also plotted on Plate 1. Surface and subsurface information were used as the basis for preparing the two hydrogeologic cross sections of the site presented in Plate 2. Information presented in Plates 1 and 2 forms the basis for subsequent interpretations of the site and the site conceptual model that is presented in Section 5.

4.2 Geophysical Surveys

Surface and subsurface geophysical surveys were carried out in 1994 to assist in defining the extent of the ground-water contaminant plume at the site. Three electrical-geophysics methods were used concurrently: terrain conductivity measurements, transient electromagnetic soundings, and conductivity soundings in the monitoring wells. Lateral variations in ground conductivity were measured using the Geonics EM34 terrain conductivity meter. Vertical variations in ground conductivity were measured using the Transient Electromagnetic (TEM) method. Well logging also was performed using the Geonics EM39 logging tool. Each method measures the conductivity of the formation in a different spatial orientation. This allows the ground-water plume to be defined in terms of a three dimensional model.

Ground-water monitoring records show that ground water within the plume contains up to about 7,000 mg/L TDS. Background TDS values in this area are less than 200 mg/L. Although the wells clearly established the existence of a plume, incomplete spacing of wells left the boundaries of the plume undefined.

4.2.1 EM34 Terrain Conductivity Survey and Data

The EM34 instrument was used to outline the approximate lateral extent of ground-water contamination. Since the ground water flows generally to the south, the main area of interest is south of the mill tailings cell.

Seven arrays or “lines” of EM34 data were recorded. Figure 4–1 shows the locations of these lines along with the location of the TEM soundings. The lines were oriented either north–south or east–west, with the exception of Line 7, which was oriented along the northwestern and northern boundaries of the disposal cell. The lines were extended beyond the monitoring wells to evaluate the extent of the contaminant plume. Readings were taken at 20-meter (66-ft) intervals along the lines.

The EM34 data were viewed in the field as they were being recorded and were then used to guide the locations of subsequent TEM soundings. The significant anomalies discovered during the EM34 survey are 2 to 3 times greater than local background values of 15 to 25 mS/m. Results are plotted on Figure 4–1. The selection of anomalies was based on deviations from local background values for each line.

4.2.2 TEM Vertical Conductivity Survey and Data

The TEM survey was used to define the boundaries of the plume more accurately. Sounding locations were evaluated from the results of the EM34 survey and samples from monitoring wells. A total of 63 TEM soundings were recorded; the locations of these soundings are shown in Figure 4–1. The full set of raw data and sounding curves, along with an interpretation of these data is available in *Geophysical Surveys at Tuba City* (DOE 1994b).

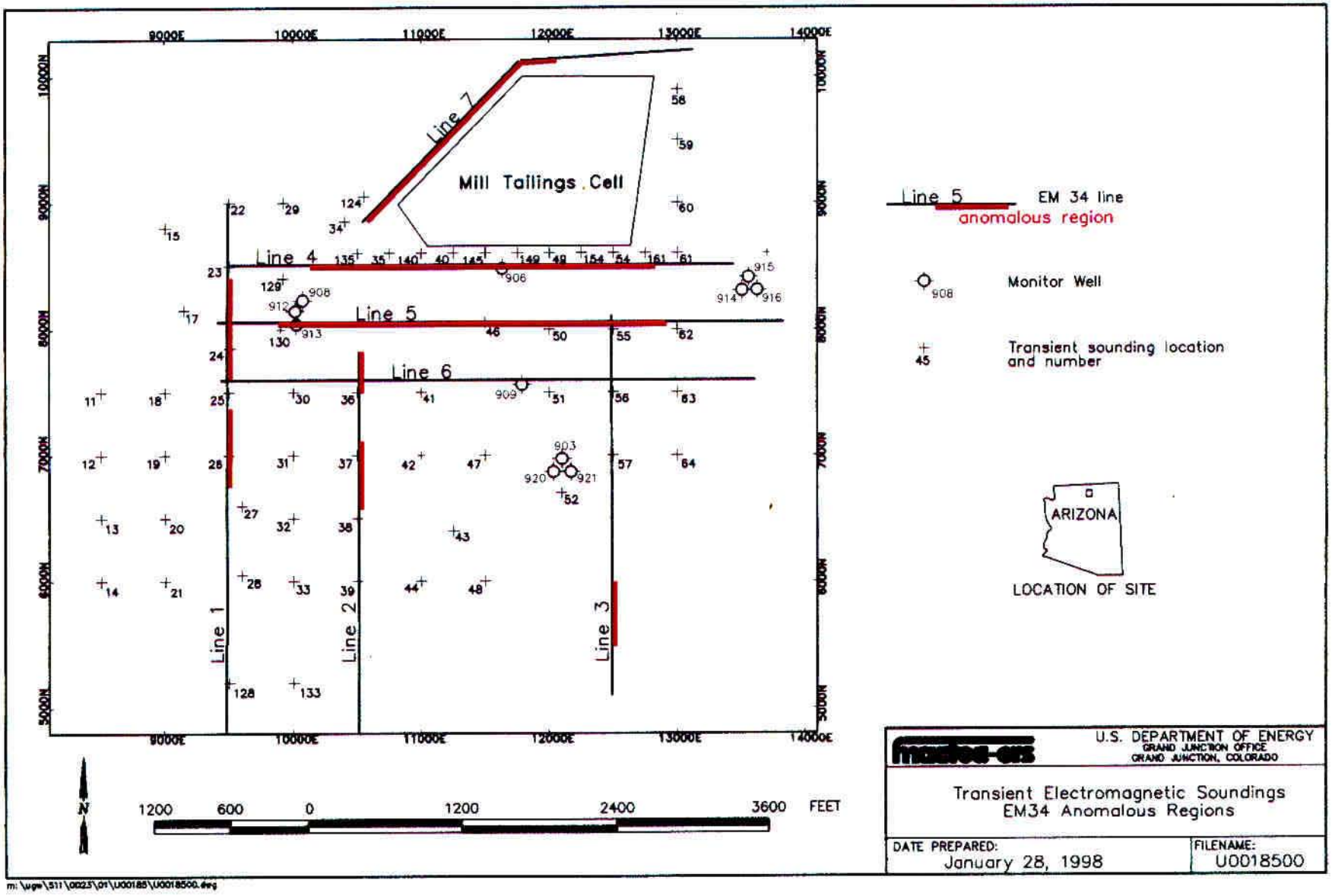


Figure 4–1. Transient Electromagnetic Soundings—EM 34 Anomalous Regions

In 1996, Rust Geotech (Rust) reinterpreted the data using existing data points from the 1994 survey and extracted enhanced information using a finer grid (Hasbrouck 1996). Rust used Interpex Ltd's TEMIX PLUS modeling software and the results were contoured using Golden Software's Surfer package. A plot of the recontoured plume data is presented as Figure 4–2.

An arbitrary value of 1.0 siemens was selected to be the lower boundary of the reinterpreted data. Figure 4–2 highlights the assumed contaminant values over 1.0 siemens. The contaminant plume appears bifurcated with a zone of zero conductance in an area approximately centered at 10250 east and 7500 north (local coordinates). The data also indicate that the ground-water plume, as defined by high TDS values, is migrating toward, but has not reached, monitoring well 930. The contaminant pathway interpreted from these data is confirmed by the presence of increased vegetation near well 930, which indicates that water is present. The interpretation of the anomaly near 9000 east, 6000 north (local coordinates) is difficult to explain, but may indicate a variability in the subsurface or within the plume. The anomaly near 14000 east 8500 north (local coordinates) at the edge of the survey area appears in the reinterpreted data but was filtered from the original data. This anomaly is not considered significant at this time. However, additional monitoring wells will be installed to the east and southeast of well 944 to obtain closure for the plume definition in this region.

4.2.3 Borehole Conductivity Logging and Data

Borehole logging was conducted to evaluate the formation conductivities and to assist in determining the depth of the contaminated zone. These data were also compared to the conductivities measured during the electrical surveys to help assess their reliability. Since several well clusters were used, it was possible to evaluate the lateral continuity of the borehole-measured conductivities and assess geologic variability. A list of wells logged, the conductivity logs, and the accompanying interpretations are available in *Geophysical Surveys at Tuba City* (DOE 1994b). Figure 4–1 presents the location of the boreholes that were logged.

In 1996, the geophysical logs were reevaluated and compared to lithologic logs (MACTEC–ERS calculation U00006AA). The results indicated a modest correlation between the geophysical and lithologic logs. The geophysical logs showed a definite response to limestone lenses. At borehole 921, for example, a thin limestone bed was penetrated at a depth of 262 ft (verified from the core log). Increases in gamma, neutron, spontaneous potential, resistance, and 16- and 64-inch resistivity logs characterize this depth, as well as a decrease in density-log response. Because of this response, other limestone lenses are inferred to exist on the basis of higher gamma and neutron log responses in borehole 903 at a depth of about 44 ft, and in borehole 913 at a depth of about 276 ft. The EM39 conductivity logging also showed an apparent correlation with calcareous nodules and calcareous cement in the rock.

Geophysical logs of deep boreholes 970, 971, and 972 provide good evidence to place the top of the Kayenta Formation at an elevation of about 4,500 ft above mean sea level. High gamma and neutron values indicate the first occurrence of thick silts and shales, which correspond to the upper Kayenta at that elevation. Above the Kayenta, the intertonguing interval of the Navajo Sandstone and the Kayenta Formation (Middleton and Blakey 1983) has no distinct geophysical signature to distinguish it from overlying "classic" Navajo Sandstone; therefore, the contact between the two units is not discernible in the logs.

4.2.4 Conclusions

Data from TEM soundings, EM34 conductivity, and EM39 conductivity logging were used to outline the contaminant plume both spatially and with depth. The EM34 data clearly show a conductive high around the southern side of the mill tailings cell. These data also identify a conductive high some 3,000 ft south of the southern boundary of the tailings cell. Large bushes in this area may indicate the presence of shallow ground water and elevated salinity from evapotranspiration. Conductivity values are also high along the entire northwestern boundary of the tailings cell (Figure 4–1). This area is upgradient of the general ground-water flow and no TEM soundings or conductivity logging were conducted in this area. Therefore, the source of these highs is unknown, but may be attributable to the remains of abandoned septic systems used in the former housing area, or other unknown sources.

The TEM data also show the outlines of a high conductivity zone south and southwest of the mill tailings cell. These data also indicate a lobe of high conductivity that runs south of the main high conductivity zone. The source of the high conductivity lobe is not known, but may be due to heterogeneity in the hydraulic conductivity of the rock, which causes the plume to bifurcate into separate lobes (Figure 4–2).

4.3 Ground-Water Monitoring Wells

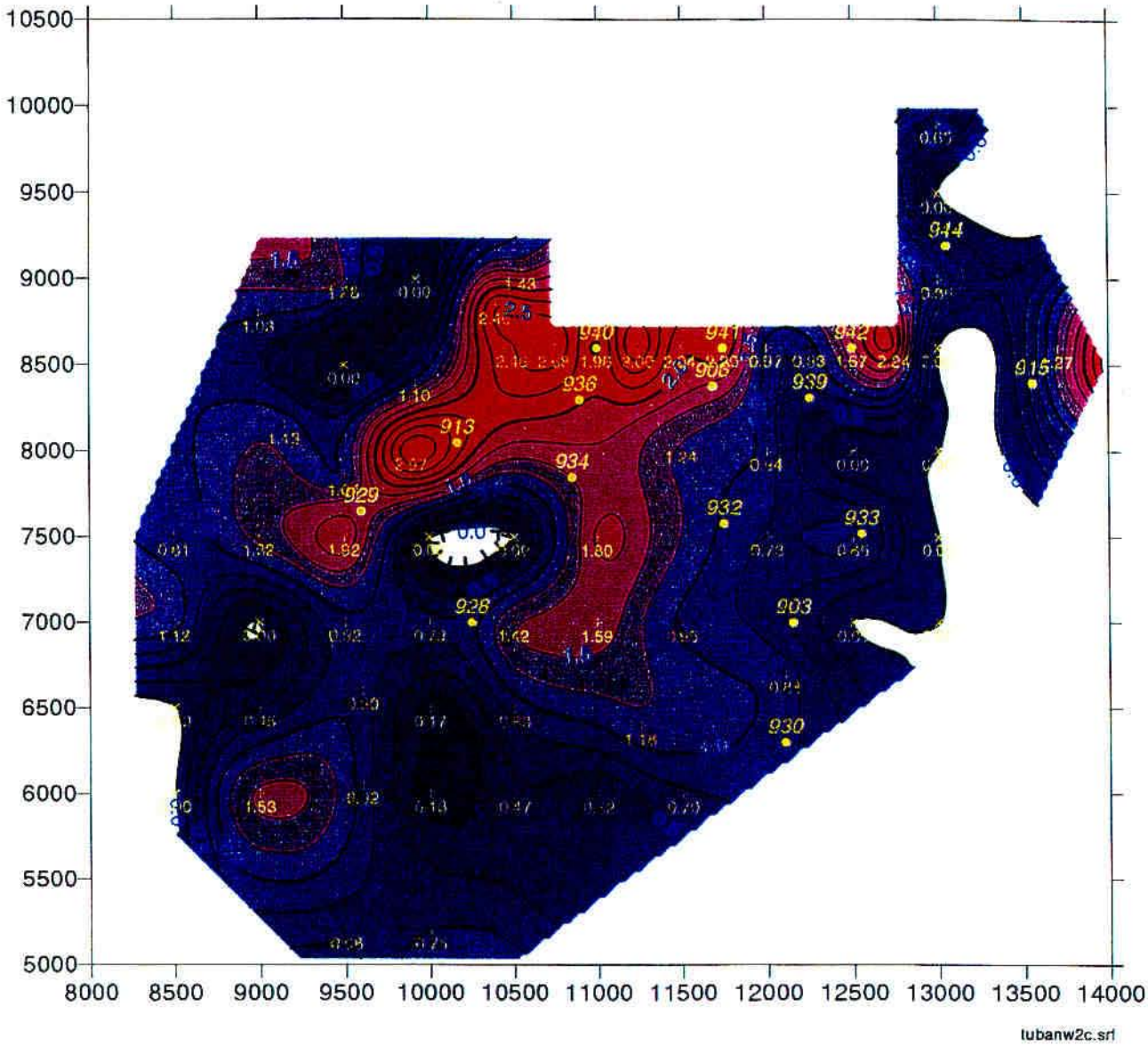
Water wells at the Tuba City UMTRA site were installed for water supply, testing, and monitoring. Figure 3–2 presents the locations of the water wells. The earliest wells were installed north of the site to supply water to the mill. These 700-ft deep wells penetrate the entire saturated thickness of the N-aquifer. The production rate from wells 968, 970, 971, and 972, which were drilled in the 1950s, was about 200 gallons per minute (gpm).

After milling operations ceased and before the surface stabilization began, characterization at the Tuba City site resulted in the installation of numerous wells. Monitoring wells 901 through 909 were installed in 1984, wells 910 through 921 were installed in 1985, and wells 925 through 948 were installed in 1995. With the exception of well 948, which was installed as a supply well, all the wells installed since 1984 are used primarily for hydrologic and chemical monitoring.

4.3.1 Installation Procedures

Monitoring wells 901 through 909 were installed using hollow stem augers to initiate the holes, and coring was used on wells 901, 902, 903, 905, 906, and 907. Wells were installed during the 1985 drilling program using mud rotary equipment; coring was performed on holes 911 and 921.

During the 1995 drilling program, hollow stem auguring was used to initiate the boreholes and was followed by fluid-based rotary methods to advance the boreholes into bedrock. Coring was used on holes 930 and 938 to evaluate microbiological properties of the rock material and to obtain lithologic samples for archiving.



Transient Electromagnetic Soundings
Conductance (conductivity x thickness) of First Conductive Layer

+ Station location and valid conductance value

× Station location with no conductance value (i.e., conductive layer not present)

● Monitoring well

Contour interval: 0.25 Siemens

Reinterpretation of data acquired for November 1994 Jacobs Engineering report

Figure 4–2. Transient Electromagnetic Soundings Conductance of First Conductive Layer

4.3.2 Monitoring Well Network

Figure 3–2 illustrates the locations of monitoring wells near the Tuba City disposal cell. Table 4–1 presents the spatial coordinates and well completion data for the wells. The distribution of monitoring wells is comprehensive from the standpoint that ground-water flow directions can be assessed from the hydraulic head data and the spatial distribution of the ground-water contamination can be assessed.

Appendix A presents a comprehensive summary of well-completion diagrams for wells installed near the site. Appendix B presents a summary of water-level measurements collected during periodic water-sampling events. Distribution of hydraulic head near the disposal cell is described in Section 5.2.2.

4.4 Hydrologic Tests

Hydrologic properties of the N-aquifer beneath the Tuba City UMTRA site have been studied by various investigators since 1985. A number of techniques have been used to investigate the hydrologic properties of the aquifer, including bail tests, laboratory tests, aquifer pumping tests, and tracer tests. Procedures and results of the hydrologic testing are presented in this section. The quality of the data improved significantly since the original hydrologic testing was performed. Average hydraulic conductivity values for the aquifer are derived from the preponderance of conductivity results from all the test results.

4.4.1 Bail Tests

Bail tests are conducted in situ within a single piezometer by instantaneously lowering the water level in the piezometer and observing its recovery in the well (Freeze and Cherry 1979). Bail tests were performed in 1985 on wells 903, 904, 906, 907, 908, and 909. Results of these tests are summarized in this section; documentation of the bail test results is presented in MACTEC–ERS calculation U00032.

Bail Test Procedure

The technology for conducting bail tests has improved considerably over the past decade through the use of electronic data loggers connected to sensitive pressure transducers. The 1985 data show that water levels were recorded at 30-second intervals; the hydraulic conductivity of the formation is sufficiently low that a 30-second recording frequency was adequate to collect useful data. The procedure for lowering the water level in the well and recording the water levels was not documented, but a record of the recovery rate of the water level in the well was available.

Table 4–1. Monitoring Well Survey and Construction Data for the Tuba City UMTRA Site

Location Code	North State Plane Coord. (ft)	East State Plane Coord. (ft)	Ground Elevation (ft)	Borehole Depth (ft below land surface)	Borehole Diameter (inches)	Top of Casing Elev. (ft)	Casing Length (ft)	Casing Diameter (inches)	Screen Depth (ft bls)	Screen Length (ft)
0901	1875919	730187	5105.1	80.0	6.6	5106.8	82.0	2.0	58.0	20.0
0902	1862289	730169	4732.9	75.0	6.5	4734.9	77.0	2.0	63.0	10.0
0903	1870827	731311	4980.4	50.0	6.5	4982.4	50.0	2.0	28.0	20.0
0904	1868033	731804	4899.4	44.0	6.6	4901.4	42.0	2.0	28.0	10.0
0905	1873198	732934	5070.1	82.0	6.6	5072.1	79.0	2.0	60.0	15.0
0906	1872180	730836	5060.4	70.5	6.6	5061.4	67.0	2.0	44.0	20.0
0907	1872920	731252	5077.2	90.5	6.0	5079.2	92.5	2.0	68.5	20.0
0908	1872000	729364	5055.9	80.0	6.6	5056.9	69.0	2.0	52.0	15.0
0909	1871392	730925	5054.2	85.0	6.6	5055.5	79.0	2.0	65.0	15.0
0910	1875842	730221	5105.9	200.0	8.5	5107.8	199.0	4.0	95.0	100.0
0911	1875922	730268	5106.2	351.4	8.5	5108.2	353.4	4.0	309.4	40.0
0912	1871943	729322	5057.9	165.0	8.5	5059.9	167.0	4.0	123.0	40.0
0913	1871872	729324	5057.9	380.0	8.5	5060.1	372.7	4.0	328.7	40.0
0914	1872116	732721	5068.4	156.2	8.5	5070.2	158.2	4.0	144.0	10.0
0915	1872206	732738	5068.6	182.0	8.5	5070.6	184.0	4.0	170.0	10.0
0916	1872143	732809	5068.1	357.7	8.5	5069.9	359.7	4.0	346.0	10.0
0917	1868646	727248	5046.1	150.0	8.5	5048.1	152.0	4.0	128.0	20.0
0918	1868727	727287	5047.4	70.0	8.5	5049.4	70.0	4.0	61.0	5.0
0919	1868657	727346	5046.2	355.0	8.5	5048.2	351.7	4.0	337.7	10.0
0920	1870736	731258	4980.6	170.0	8.5	4982.6	158.4	4.0	116.0	40.0
0921	1870740	731375	4976.8	360.0	8.5	4978.8	358.8	4.0	315.2	40.0
0925	1872005	729451	5057.8	95.0	9.9	5060.1	93.8	6.0	51.0	40.0
0926	1872125	730790	5059.4	100.0	9.9	5062.1	100.7	6.0	45.0	50.0
0928	1870813	729401	5051.3	69.0	7.9	5053.2	55.4	4.0	30.0	25.0
0929	1871453	728780	5057.9	89.5	7.9	5060.1	91.2	4.0	48.2	40.0
0930	1870098	731257	4952.1	80.0	7.9	4954.3	55.2	4.0	20.0	30.0
0932	1871400	730900	5054.0	136.0	7.9	5056.6	138.1	4.0	110.0	20.0
0933	1871340	731727	5015.4	64.0	7.9	5017.3	51.9	4.0	25.0	25.0
0934	1871648	730018	5057.2	96.0	7.9	5059.0	94.8	4.0	45.0	45.0
0935	1871977	729461	5057.4	95.0	4.0	5060.7	96.3	4.0	50.0	40.0
0936	1872121	730055	5059.0	89.5	9.9	5061.6	87.6	6.0	45.0	40.0
0937	1872115	730790	5059.5	100.0	7.9	5062.0	98.5	4.0	38.0	55.0
0938	1872124	730769	5059.4	99.0	7.9	5062.9	98.8	4.0	40.0	55.0

Table 4–1 (continued). Monitoring Well Survey and Construction Data for the Tuba City UMTRA Site

Location Code	North State Plane Coord. (ft)	East State Plane Coord. (ft)	Ground Elevation (ft)	Borehole Depth (ft below land surface)	Borehole Diameter (inches)	Top of Casing Elev. (ft)	Casing Length (ft)	Casing Diameter (inches)	Screen Depth (ft bls)	Screen Length (ft)
0939	1872131	731404	5059.7	100.0	9.9	5062.2	92.4	6.0	40.0	55.0
0940	1872391	730130	5062.2	70.0	7.9	5064.0	69.8	4.0	45.0	20.0
0941	1872398	730908	5062.3	70.0	7.9	5065.2	70.9	4.0	45.0	20.0
0942	1872409	731642	5062.5	79.0	7.9	5065.7	80.2	4.0	54.0	20.0
0943	1874034	731596	5094.2	125.0	7.9	5097.2	127.0	4.0	101.0	20.0
0944	1873006	732199	5064.1	145.0	7.9	5066.2	110.1	4.0	85.0	20.0
0945	1873857	730020	5137.3	135.0	7.9	5139.7	135.4	4.0	110.0	20.0
0946	1873582	730547	5096.7	70.0	7.9	5099.7	66.3	4.0	40.0	20.0
0947	1874642	732787	5094.3	130.0	7.9	5096.2	130.2	4.0	105.0	20.0
0948	1875515	733916	5114.5	408.0	9.9	5116.9	408.9	4.0	221.5	180.0
0968	1875624	730275	5110.0	707.0	13.0	5110.8	707.0	10.0	106.0	601.0
0970	1876568	730656	5106.0	705.0	13.0	5106.8	705.0	10.0	100.0	605.0
0971	1878307	731595	5104.9	900.0	12.0	5105.7	900.0	10.0	117.0	583.0
0972	1877990	728035	5137.3	730.0	12.0	5138.0	730.0	10.0	100.0	630.0

Bail Test Results

On the basis of water-level data in the project file, it was possible to estimate the hydraulic conductivity in the area surrounding each well. Table 4–2 presents the results of the bail testing for the project site.

Table 4–2. Summary of Hydraulic Conductivity Values as Determined with Bail Tests

Well	Hydraulic Conductivity (cm/s)
903	3.8×10^{-4}
904	7.1×10^{-4}
906	1.9×10^{-4}
907	1.6×10^{-4}
908	7.8×10^{-5}
909	7.0×10^{-5}
Average	2.6×10^{-4}

4.4.2 Laboratory Permeability Tests

Laboratory permeameter tests were run on selected core samples using the triaxial permeability methods. This method provides estimates of vertical permeability because the permeant (water) passes through the core sample in a direction that parallels the long axis of the core. The advantage of this test method is that it provides a direct measurement of the vertical permeability of the rock material; the disadvantage is that the sample is very small compared to the scale of the aquifer. Therefore, a number of measurements are required to obtain a representative estimate of the range in vertical hydraulic conductivity.

Laboratory Permeability Procedure

Laboratory permeability testing was performed on several samples obtained from recent core holes, including monitoring wells 930, 937, 938, 940, and 941. Samples were collected from a variety of depth intervals and analyzed by Keantan Laboratories. The samples were submitted for analysis in 1995 by Jacobs Engineering Group, the former TAC.

Laboratory Permeability Test Results

Copies of the original testing results are presented in MACTEC–ERS calculation U0005600. Table 4–3 presents a summary of the original test results. The test results indicate that the average vertical hydraulic conductivity is comparable to the average horizontal hydraulic conductivity; however, the minimum vertical hydraulic conductivity is two orders of magnitude lower than the mean vertical hydraulic conductivity. The minimum value of vertical hydraulic conductivity may indicate an interdune deposit that formed during a relatively quiescent period, possibly accompanied by fluvial and lacustrine deposition.

Table 4–3. Summary of Laboratory Permeability Testing

Sample ID	Depth (ft)	Vertical Hydraulic Conductivity (cm/s)
MW930	31.0–31.6	2.7×10^{-4}
MW930	38.2–39.1	3.3×10^{-4}
MW930	44.4–45.2	7.3×10^{-5}
MW930	45.8–46.4	1.6×10^{-4}
MW930	54.0–54.8	2.0×10^{-4}
MW930	55.7–56.2	3.3×10^{-4}
MW937	38.0–38.8	5.0×10^{-4}
MW937	43.1–43.9	9.8×10^{-5}
MW937	43.9–44.5	6.0×10^{-4}
MW937	52.6–53.0	8.3×10^{-4}
MW937	54.3–54.8	7.2×10^{-4}
MW937	54.8–55.2	1.2×10^{-4}
MW938	38.6–39.0	6.1×10^{-4}
MW940	51.4–51.8	1.0×10^{-4}
MW941	45.2–46.0	1.6×10^{-5}
MW941	54.6–55.0	3.8×10^{-5}
MW941	56.5–57.2	4.0×10^{-5}
MW941	61.6–62.2	2.3×10^{-4}
MW941	68.5–9.1	2.2×10^{-7}
Average		(2.8×10^{-4})

4.4.3 Aquifer Pumping Tests

Pumping tests at the Tuba City site were performed in two series during the characterization. The first series was performed in 1996 by Jacobs Engineering Group and was of relatively short duration. The purpose of the short-term test was to obtain preliminary transmissivity data for numerical modeling studies, to update the site conceptual model, and to plan the long-term aquifer pumping test. The second series of tests was performed in 1997 and was of relatively long duration. The purpose of the long-term aquifer test was to update the site conceptual model, obtain geochemical data to assess contaminant response to the sustained pumping, and accomplish contaminant mass removal from the highly contaminated zones of the plume. Results of the short-term and long-term tests are documented in MACTEC–ERS calculations U00030 and U00178AA, respectively, and summarized in the following sections.

Short-Term Aquifer Tests

Three short-term pumping tests were performed at the site from March 11 through March 22, 1996. Figure 3–2 presents the locations of control wells 939, 925, and 926 and the associated observation wells. Table 4–4 presents a summary of the pertinent pumping test variables for each aquifer test.

Table 4–4. Summary of the Pertinent Pumping Test Variables for Each Aquifer Test

Well ID		Radial Distance Between Wells (ft)	Pumping Rate (gpm)	Test Duration (minutes)
Control Well	Observation Well			
939	none		1.9	56
925	935	29.52	2.9	4,200
	908	87.56		
	912	143.9		
	913	184.0		
926	937	10.19	6.6	1,700
	938	20.36		
	906	72.32		

The test at well 939 consisted of pumping an average of 1.9 gpm for a duration of about 56 minutes, at which point the well went dry. Recovery was then monitored in the pumping well for approximately 15 hours. Since no observation wells were near well 939, only the recovery data were used to compute transmissivity, and neither storage nor specific yield values were determined from this test. Data presented in Table 4–4 illustrate that the yield from this well could not be sustained. Step testing was used during the long-term pumping test program to develop a sustainable pumping rate from this well. The sustainable pumping rate in this well is about 0.5 gpm. (MACTEC–ERS calculation U00178AA). On the basis of the recovery test data alone from the short-term test, the transmissivity in the area of this well is about 1 square foot per day (ft^2/day).

Another short-term test was performed using well 925 as the control well and wells 935, 908, 912, and 913 as observation wells. Well 925 was pumped at an average rate of 2.90 gpm for 4,200 minutes. Table 4–4 presents the radial distances between the control well and the observation wells. Data obtained during the test show that drawdown increased throughout the entire stress period, but as illustrated in Figure 4–3, the drawdown curves show some oscillatory behavior. Recovery cycle data were also collected during this series of tests. These data, together with the withdrawal data, were evaluated in MACTEC calculation U00030. Table 4–5 presents a summary of the test results for this test.

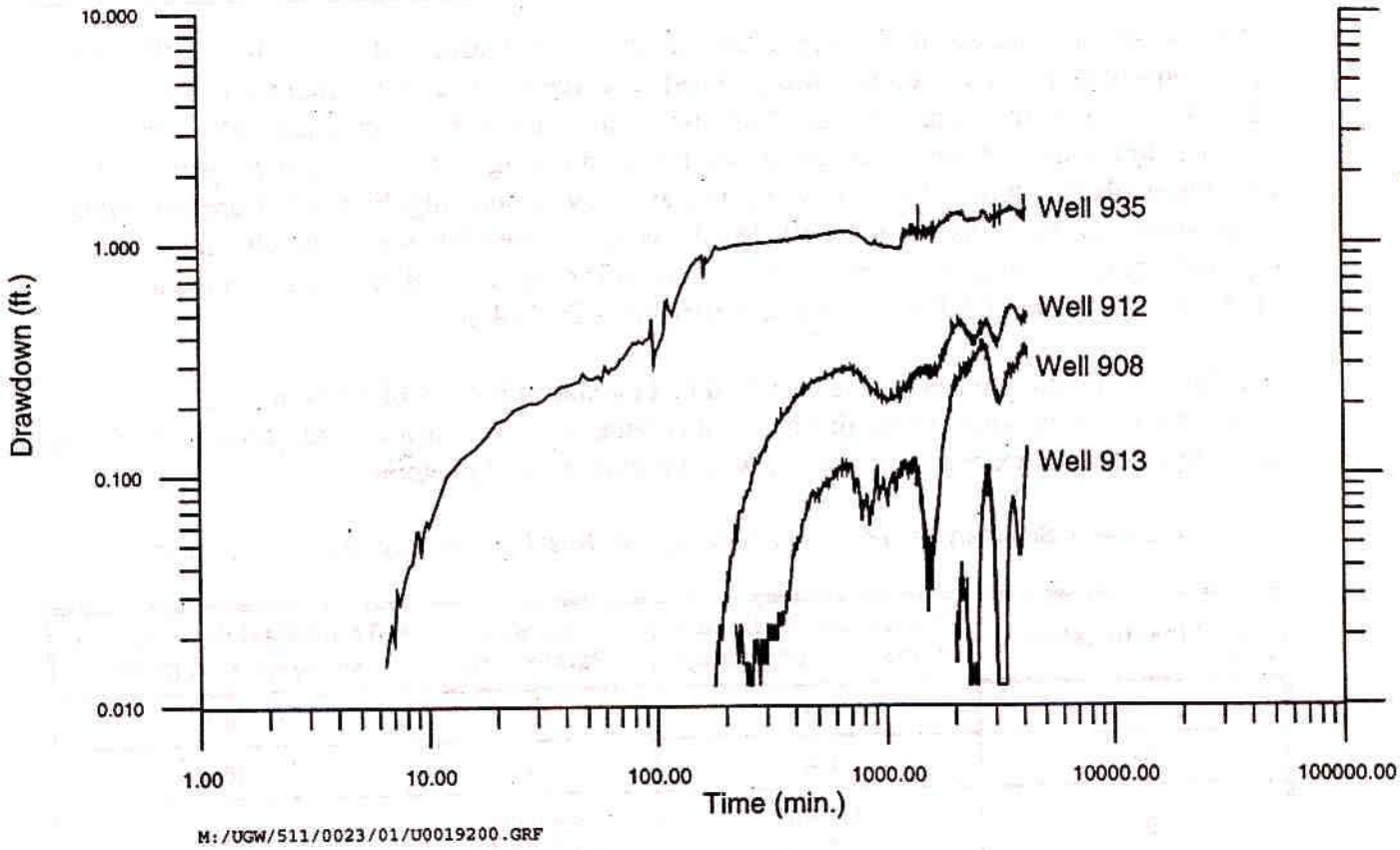


Figure 4–3. Well 925 Drawdown Results

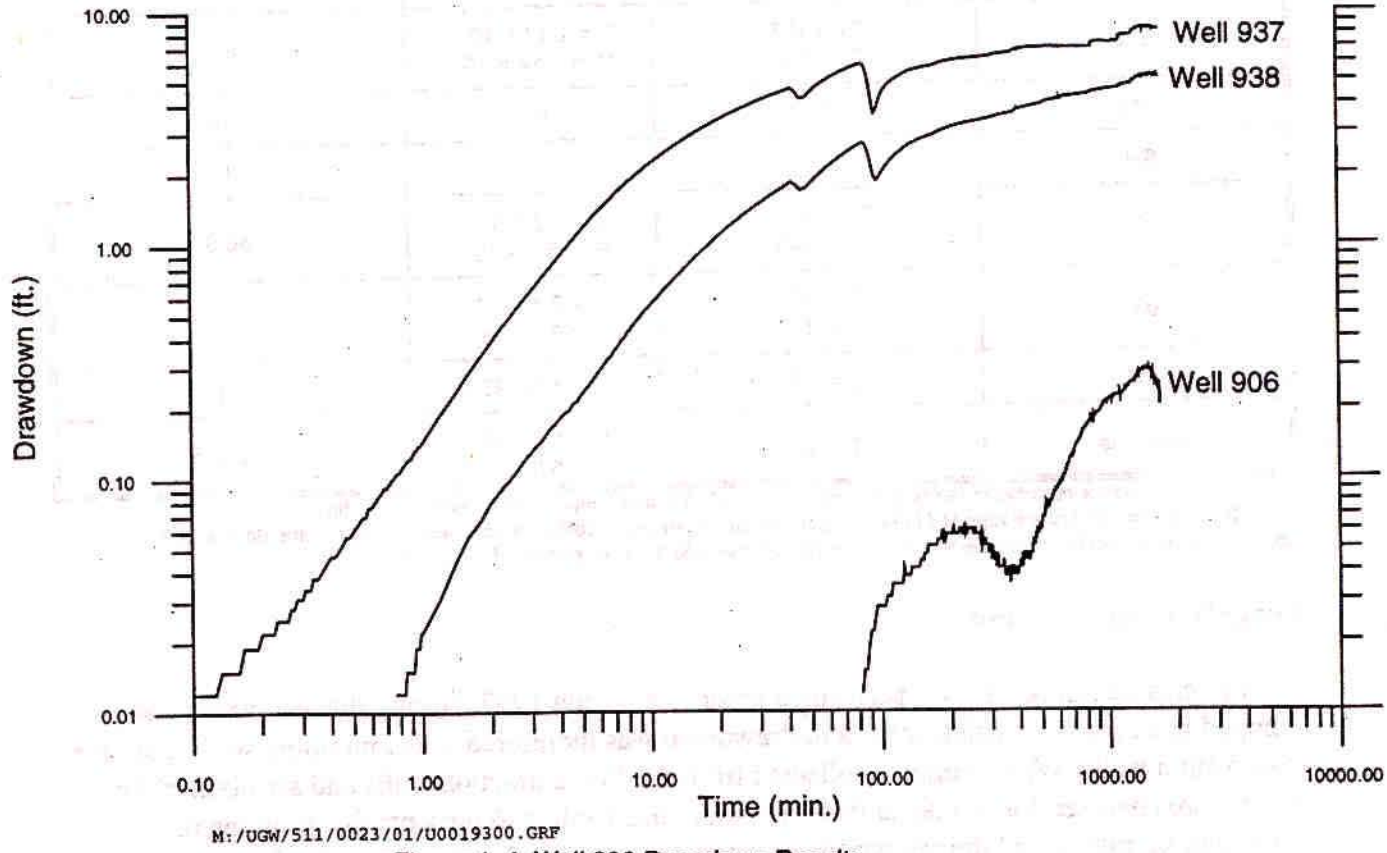


Figure 4–4. Well 926 Drawdown Results

A third short-term test was performed with well 926 as the control well and wells 937, 938, and 906 as observation wells. Well 926 was pumped at an average rate of 6.6 gpm for 1,700 minutes. Table 4–4 presents the radial distances from the control well to the observation wells. Data obtained during the test show that drawdown increased throughout the entire stress period. Data were also collected during the recovery cycle that followed pumping. Figure 4–4 presents plots of drawdown for each observation well. The drawdown in well 906 was relatively minor and required significant scale-up before evaluation. Details of the evaluations are presented in MACTEC calculation U00030 and are summarized in Table 4–5.

The short-term pumping test results indicated that the transmissivity of the N-aquifer immediately downgradient of the disposal cell is rather low in some areas, and consequently, any active ground-water extraction at the site would require careful planning.

Table 4–5. Summary of Short-Term Pumping Test Results, Tuba City UMTRA Repository

Well Identification	Transmissivity Result of Stress Test[a] (ft^2/day)	Storage Parameters[b]	Transmissivity Result of Recovery Test (ft^2/day)
939	NA	NA	0.912
925	NA	NA	165
935	T_E = 465 T_L = 365	S = 0.003 $S_y = 3.4 \times 10^{-5}$	276
908	T_L = 76.9	S_y = 0.0227	181
912	T_E = 157 T_L = 190	$S = 2.17 \times 10^{-4}$ $S_y = 7.14 \times 10^{-3}$	609
913	NA	NA	474
926	NA	NA	83.2
937	T_E = 60.9 T_L = 59.7	$S = 2.5 \times 10^{-3}$ $S_y = 5.0 \times 10^{-3}$	80.9
938	T_E = 101 T_L = 101	$S = 2.6 \times 10^{-3}$ $S_y = 3.8 \times 10^{-3}$	132
906	T_L = 2680	S_y = 0.299	1480
Average	T = 426	$S = 1.7 \times 10^{-3}$ $S_y = 5.6 \times 10^{-2}$	T = 348

[a] Transmissivity is reported in terms of T_E, the solution at early time, and T_L, the solution at late time.
[b] Storage parameters are reported in terms of S, the elastic storage coefficient obtained from early-time data, and S_y, the specific yield obtained from the late-time data that follows the delayed-yield portion of the curve.

Long-Term Aquifer Tests

A long-term pumping test was performed at the site during 1997. During the test, water was pumped from four extraction wells and drawdown was monitored in the pumping wells and nine observation wells. Water samples collected from the four extraction wells and six observation wells were analyzed for nitrate, sulfate, and uranium. Table 4–6 presents the grouping of pumping, sampling, and drawdown-observation wells. Ground-water extraction began on January 24, 1997, at well 926 and on January 27, 1997, at wells 925, 936, and 939.

Table 4–6. Pumping, Sampling, and Observation Wells for Long-Term Pumping Test

Well ID		Radial Distance Between Wells (ft)	Average Pumping Rate (gpm)	Test Duration (minutes)
Control Well	Drawdown Observation Well			
925	925*		2.3	93,700
	935*	29.52		
	908*	87.56		
	913	184.0		
926	926*		4.4	95,200
	906*	72.32		
	941*	297.4		
	937	10.19		
	938	20.36		
936	936*		0.3	94,000
	940*	280.2		
939	939*		0.5	94,200
	942*	366.0		

*Indicates a well that was also used for chemical sampling.

Water chemistry samples were collected daily for the first 7 days of pumping at the well 926 group and for the first 4 days of pumping at the other pumping-well groups. Sampling continued once per week for the next 4 weeks, then twice per month for the next month. A final sample suite was collected 1 week after the end of pumping. Pumping terminated on March 31, 1997, when lightning struck electrical lines at the site and knocked out power to the pumps. Water discharged from the wells during the tests was pumped into Treatment Pond 1 and is being allowed to evaporate. About 700,000 gallons of water was extracted from the four wells and discharged to Treatment Pond 1 during the 2-month duration of the test.

Ground-water samples were collected during the long-term pumping test for geochemical analysis using dedicated bladder pumps and standard ground-water sampling protocols. Observation wells were purged at 100 milliliters per minute until indicator parameters stabilized, and then samples were collected. Uranium analysis was performed with a UA–3 uranium analyzer. Nitrate and sulfate were analyzed with a spectrophotometer using Hach wet-chemical techniques. Extraction wells were sampled from a sampling port on the discharge line.

Pumping test analysis methods included Neuman's delayed yield method (Neuman 1975) and Moench's partial penetration method (Moench 1995) for drawdown analysis and the straight line semilog method (Neuman 1975) for recovery analysis. Hydraulic conductivities were calculated by dividing transmissivities obtained from type curve matching and recovery curve analysis by the aquifer thickness. Aquifer thickness was calculated as the water-table elevation minus 4,650 ft, where 4,650 ft is taken as the base of the aquifer (MACTEC–ERS calculation U00030).

An additional series of single-well recovery tests were also performed on wells 930, 943, 932, 920, 944, 941, 915, 929, 934, and 947. These were short-duration tests designed to evaluate the natural variability in the aquifer and to study whether geochemical conditions in the plume are contributing to changes in hydraulic conductivity.

Results

A summary of the long-term pumping test results is presented in this section; a formal analysis of the long-term pumping test data is presented in MACTEC–ERS calculation U00178AA. As presented in Table 4–7, two populations of pumping test results are apparent: the high conductivity/transmissivity and low conductivity/transmissivity wells. The high conductivity/transmissivity wells include wells 930, 943, 926, and 925, which showed transmissivities exceeding 100 ft^2/day. This group, which will be referred to as the high conductivity wells, includes wells inside and outside the highly contaminated portion of the plume. The low transmissivity/conductivity wells (transmissivities less than 100 ft^2/day), include wells 932, 939, 920, 944, 941, 936, 915, 929, 934, and 947; these wells are also located both inside and outside the highly contaminated sections of the plume. This observation suggests that aquifer and well fouling from supersaturated concentrations of contaminants is not the reason for the dramatically low yields and transmissivities observed in some of the Tuba City wells. Rather, the results suggest that the differences are simply a reflection of natural heterogeneity within the Navajo Sandstone.

Table 4–8 presents a summary of the contaminant mass removal for the wells sampled during the long-term pumping test. The mass removal summary table shows that meaningful reductions in chemical mass are achieved through conventional pumping at the Tuba City site in spite of the low production capacity of the upper portions of the N-aquifer. This is due to the high concentrations of contaminants observed south of the disposal cell. If the masses of nitrate and sulfate to be removed are 7.6 million pounds and 11 million pounds (MACTEC–ERS calculation U00033AA), respectively (based on target concentration of 500 mg/L), pumping at the observed mass removal rates would need to continue for 200 years to achieve nitrate cleanup and for 150 years to achieve sulfate cleanup. Installation of just twelve additional wells with the same average mass removal efficiency as the four test wells would reduce cleanup times to 50 and 38 years, respectively. Improvements in mass removal efficiency through recharge-based gradient control, improved well siting and design, and well-field optimization would further reduce cleanup times.

4.4.4 Aquifer Tracer Tests

A tracer test was designed and conducted in August 1997 as a limited feasibility study to examine field injectibility of the aquifer. The test was performed by a team consisting of MSE Technology Applications and Rio Algom Environmental Services, Inc. (MSE) under contract to DOE. The purpose of the project was to investigate the application of commercial in situ mining technologies for remediation of contaminated ground water.

Table 4–7. Summary of Results from Long-Term Pumping Tests

Pumping Well (rate in gpm)	Observation Well	Transmissivity (ft^2/day)		Hydraulic Conductivity (ft/day)		Storativity (S)	Specific Yield (S_y)
		Drawdown	Recovery	Drawdown	Recovery		
930 (20.1)	930	194 – 3070	1419	0.68 – 10.8	5.0	N/A	N/A
943 (3.0)	943	51.5 – 205	137 – 662	0.1 – 0.5	0.4 – 1.7	N/A	N/A
926 (4.97)	926	13.5	103.2	0.0	0.3		0.20
	937	38.0 – 427.7	103.2	0.10 – 1.2	0.3	0.0030	0.10
	938	67.7 – 427.7	103.2	0.2	0.3	0.002	0.004
	906	537.1	Inconclusive	1.5	Inconclusive	Inconclusive	Inconclusive
	941	45.2 – 452.2	282.7	0.12 – 1.2	0.8	Inconclusive	0.020
925 (2.00)	925	10.25 – 54.4	176.4	0.029 – 0.015	0.5	Inconclusive	Inconclusive
	935	96.8 – 769.0	176.4	0.27 – 2.1	0.5	Inconclusive	Inconclusive
	908	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
	913	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
932 (3.0)	932	10.8 – 217	28	0.034 – 0.69	0.1	N/A	N/A
939 (0.59)	939	1.3 – 10.0	14	0.004 – 0.03	0.04	Inconclusive	0.1 – 0.3
930 (20.1)	429	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
920 (1.5)	920	5.1	6.4	0.2	0.0	N/A	N/A
944 (1.0)	944	6.1 – 97	4.0	0.02 – .3	0.011	N/A	N/A
941 Trial 1	941	65	2.9	0.18	0.0078	N/A	N/A
941 Trial 2	941	4.8	2.5	0.01	0.0068	N/A	N/A
936 (0.30)	936	1.4 – 12.9	1.1	0.004 – 0.04	0.0	Inconclusive	Inconclusive
	940	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
915 Trial 1 (3.0)	915	2.0	Inconclusive	5.7	Inconclusive	N/A	N/A
915 Trial 2 (1.0)	915	0.3	0.29	0.0	0.0	N/A	N/A
929 Trial 1 (2.0)	929	0.9	Inconclusive	0.0	Inconclusive	N/A	N/A
929 Trial 2	929	0.7	0.09	0.0	0.0	N/A	N/A
934 Trial 1 (5.6)	934	Inconclusive	Inconclusive	Inconclusive	Inconclusive	N/A	N/A
934 Trial 2	934	0.36 – 1.6	Inconclusive	0.001 – 0.004	Inconclusive	N/A	N/A
947 Trial 1 (2.0)	947	Inconclusive	Inconclusive	Inconclusive	Inconclusive	N/A	N/A
947 Trial	947	Inconclusive	0.43	Inconclusive	0.0012	N/A	N/A

Table 4–8. Extraction Well Contaminant Mass Removal Summary

Well	Volume Discharged (liters)	Average Nitrate Concentration (mg/L)	Nitrate Mass Removed (kg)	Average Sulfate Concentratio n (mg/L)	Sulfate Mass Removed (kg)	Average Uranium Concentration (mg/L)	Uranium Mass Removed (kg)
925	801,455	850	682	2921	2341	0.172	0.138
926	1,582,373	1073	1698	1635	2587	0.413	0.654
936	96,094	3140	302	3140	302	0.252	0.024
939	167,558	766	128	1322	222	0.770	0.129
TOTALS			**2810 (kg)**		**5452 (kg)**		**0.945 (kg)**
			6,194 (pounds)		**12,018 (pounds)**		**2.08 (pounds)**

Tracer Test Procedure

The tracer test was performed by injecting water into well 926 at a rate of 8 gpm while simultaneously withdrawing water from wells 938 and 937 at a rate of 4 gpm. The withdrawal wells 938 and 937 are 10.2 ft and 20.4 ft, respectively, from well 926. A bromide tracer solution was injected into well 926 and the arrival of the tracer was monitored in the withdrawal wells. Bromide tracer began arriving in well 937 after 4 hours and in well 938 after 36 hours. Water extracted from wells 937 and 938 was piped to the lined evaporation ponds located north of the disposal cell. Field measurements of discharge rates, water levels, and bromide concentrations, together with laboratory measurements of uranium, nitrate, sulfate, and bromide concentrations in water samples were tabulated into time series plots to facilitate interpretation of the test results. Effective porosity was estimated for the test region by using the tracer-arrival times from the field test in combination with hydraulic conductivity values from modeling simulations.

Tracer Test Results

The tracer tests provided drawdown and tracer-arrival data in the tested region. The hydraulic conductivity for the region was estimated by using it as an input variable to modeling simulations of the pumping test until predicted drawdowns matched the simulated drawdowns as closely as possible. A perfect match could not be established between the predicted and simulated drawdowns because field conditions are not homogeneous and isotropic. Nevertheless, an acceptable value of hydraulic conductivity (0.2 feet per day [ft/day]) was achieved. This hydraulic conductivity value compares closely to the value of 0.3 ft/day obtained in that region from the long-term pumping test.

The bromide tracer arrival-time data were used to estimate the effective porosity of the formation. The investigators only reported effective porosity for the arrival-time data on well 937 because complete breakthrough was not achieved in well 938. Effective porosity for well 937 was obtained using the measured average linear velocity from the breakthrough curve, the computed hydraulic conductivity, and the hydraulic gradient between the injection well and the withdrawal well. On the basis of these data, an effective-porosity value of 0.03 was obtained. This value compares well with results of the long-term pumping test, in which the specific yield

for this region was reported to be 0.05; however, these estimates are significantly lower than the porosity values reported in Cooley and others (1969), in which the porosity values based on 24 laboratory measurements ranged from 25 to 35 percent. Unrealistically low specific yield estimates as compared with controlled laboratory experiments have been described for unconsolidated, granular aquifers, and have been subject to much debate in the technical literature (i.e., Nwankwor et al. [1984]; Neuman [1987]; Nwankwor et al. [1992]; and Moench [1994]). These effects are most pronounced in situations involving partially penetrating wells. The wells which were tested near the disposal cell only tap the upper 10 percent of the aquifer. Details of this tracer test investigation are presented in MSE (1997).

4.5 Monitoring and Managing Enhanced Recharge

Recharge attributable to runoff from the disposal cell may be an important influence on plume movement. This section describes ongoing work to quantify and manage this component of the site water balance.

4.5.1 Disposal Cell Runoff

Flow modeling studies (MACTEC–ERS calculation U00151AA) indicate that recharge might be enhanced in the south apron of the disposal cell. This is based in part on the premise that the compacted cover of the disposal cell acts as a catchment surface. Precipitation falling on the cover rapidly infiltrates the rock riprap, flows laterally in the underlying gravel layer to the south apron of the disposal cell, and concentrates there causing elevated recharge rates. The calibrated flow model suggests that enhanced recharge in a 6.6 acre (2.7-hectare) area south of the disposal cell might be as high 2.7×10^{-6} cm/s (836.6 millimeters per year) or 23,000 cubic meters per year (6 million gallons per year) (MACTEC-ERS calculation U00151AA).

A recharge monitoring study began in April 1997 to evaluate if runoff from the cover of the disposal cell concentrates near the south apron. Three blocks of neutron hydroprobe access ports were installed along the south apron of the disposal cell (Figure 4–5). Each block consists of six ports arranged in two parallel lines, three ports to a line. One block lies just west of well 942, another just west of well 941, and a third just east of well 940. Each port consists of a polyvinyl chloride (PVC) pipe, 5 centimeters (cm) in diameter and about 450 cm deep. The PVC pipe has a removable PVC cap. Holes for access ports were excavated with a bobcat-mounted soil auger; some holes were cleaned with hand-operated bucket augers, and the annulus was backfilled and tamped using a PVC pipe with an inside diameter slightly larger than the outside diameter of the access port.

Field calibration data (soil water content and neutron counts) were obtained as access ports were installed. Hydroprobe monitoring began in late April 1997 and continued monthly through September 1997. Monitoring data consist of shielded counts and counts taken at incremental depths to the bottom of each port. Monthly monitoring followed extended dry periods and intense late-summer storms. Preliminary count ratio data suggest that the cell is acting as a catchment surface and that recharge pulses pass below the root zone following some storm events. The recharge rate and volume remain unknown at this time; however, recharge monitoring will continue during 1998.

Activities planned for 1998 include (1) hydroprobe monitoring to evaluate the specific meteorological conditions that may contribute to recharge, (2) hydroprobe calibration, using gravimetric soil-water content data from samples taken during the installation of access ports, to derive correlations between neutron count ratios and the soil moisture content, and (3) recharge estimation in collaboration with Pacific Northwest National Laboratory (PNNL). PNNL's activity would involve screening different recharge-estimation approaches and then selecting and applying the most cost-effective approach for Tuba City conditions. If recharge attributable to runoff from the disposal cell is a primary ground-water source, then ground-water management may include recharge management. Management could involve either recharge enhancement or recharge control depending on the selected compliance strategy. At arid and semiarid sites, recharge management is essentially evapotranspiration management. Evapotranspiration is managed by manipulating plant communities. Vegetation currently growing south of the disposal cell could be removed to enhance recharge, or adapted plant species with high transpiration rates could be planted to control recharge. Pilot studies will begin in 1998 to evaluate the effectiveness of both strategies.

4.6 Ground-Water Sampling and Analysis

The objective of the ground-water sampling and analysis program is to construct a database of ground-water quality results and ground-water elevation measurements that can be used to make decisions concerning the site. Historically, samples have been collected from 42 monitoring wells, and ground water elevations have been measured at 43 monitoring wells. The only monitoring well that has not had any samples collected from it is well 933.

Currently, 13 wells are sampled quarterly, 8 wells are sampled semiannually, and 7 wells are sampled annually, for a total of 28 wells sampled per year. Table 4–9 presents a list of the wells that are presently being sampled and the sampling frequency. Sampling locations and analytical requirements for Tuba City are reviewed and updated annually.

To ensure that sample results produce valid and technically defensible data, all sample collection activities (well purging, sample collection, sample filtration and preservation, field quality assurance, field measurements and analyses, sample handling and identification, equipment decontamination, instrument calibration and operation, and documentation of field activities) follow the *Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1997). The purpose of this plan is to incorporate DOE–GJO standard operating procedures (SOPs) into ground-water and surface-water sampling conducted for the UMTRA Ground Water Project. The procedures contained in that document incorporate DOE, ASTM, and EPA guidance. The document has also incorporated certain aspects of Jacobs Engineering SOPs to maintain consistency and continuity in sample collection.

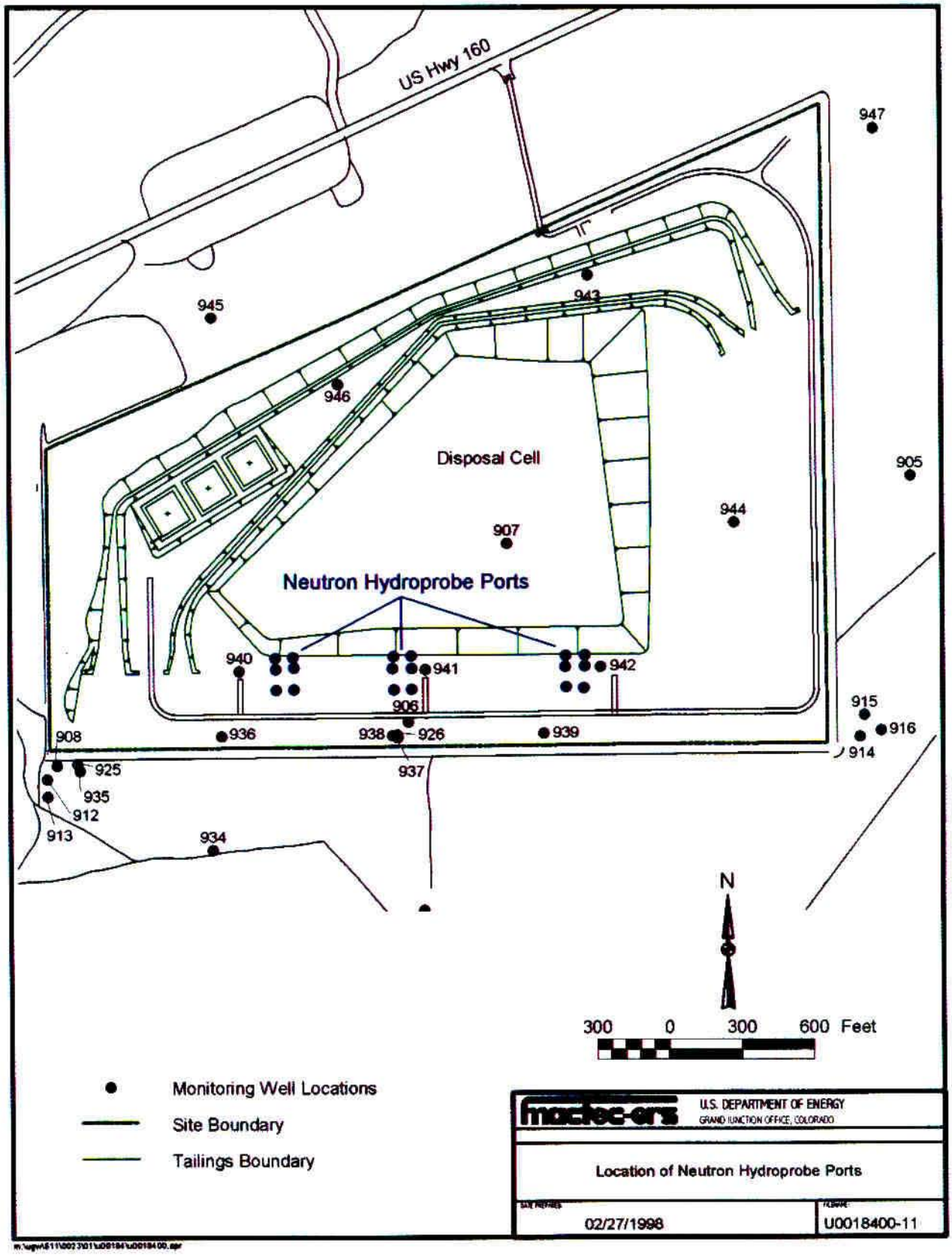


Figure 4–5. Location of Neutron Hydroprobe Ports

Table 4–9. Wells and Surface-Water Locations Currently Being Sampled and their Sampling Frequency

Wells	Sampling Frequency			
	Quarterly	Semiannuall	Annually	Not Sampled
Ground Water Project Monitoring Wells				
901		X		
903		X		
904			X	
906	X			
908	X			
909	X			
912		X		
913		X		
914		X		
915			X	
916			X	
917			X	
920			X	
921			X	
928		X		
929		X		
930			X	
932	X			
933	X			
934		X		
935	X			
937	X			
938	X			
944	X			
945	X			
Surface Project Monitoring Wells				
925				X
926				X
939				X
940	X			
941	X			
942	X			
943				X
946				X
947				X
948				X
Surface-Water Locations				
759			X	
778			X	
965			X	
969			X	

A Sampling and Analysis Work Order is generated before each sampling event. This work order identifies the locations to be sampled, the analyte list for each sample, and various other requirements, which may include certain field measurement and analyses, sitewide water-level measurements, downloading of data logger files, and well inspections and repair.

Ground-water samples are collected for analysis of metals, major cations, radionuclides, and other inorganics. Once collected, the samples are packaged and shipped to the GJO Analytical Laboratory for analysis. After the analytical results are received from the laboratory, the site sampling leader verifies and validates all the results acquired in the field and received from the laboratory.

Analytical constituents for the project include parameters measured in the field and those measured in the laboratory. Field parameters include alkalinity, pH, specific conductance, temperature, dissolved oxygen, and redox potential. These constituents provide early indications of changes in ground water quality. Laboratory measured constituents include the contaminant species cadmium, molybdenum, nitrate, selenium and uranium. Each of these constituents exceed, or have exceeded at one time, the background concentration and they are UMTRA listed constituents. In addition, the major cation species (sodium, potassium, sodium, magnesium, strontium, iron, and manganese), major anion species (chloride and sulfate), and total dissolved solids are analyzed, although they aren't UMTRA listed constituents, because they provide an indication of plume behavior and are useful for evaluating mineral saturation indices. Some of these constituents (sodium, sulfate, chloride, pH, and total dissolved solids), are also given as the Navajo Nation's proposed secondary cleanup levels for restoration of ground water. Other constituents routinely monitored at the site include: ammonia, aluminum, antimony, arsenic, barium, beryllium, boron, calcium, cobalt, copper, dissolved organic carbon, flouride, gross alpha, chromium, lead, gross beta, nickel, nitrate + nitrite, nitrite, mercury, radium-226, radium-228, redox, silica, silver, thallium, thorium-230, tin, total cyanide, sulfide, vanadium, zinc, kjeldahl nitrogen, total organic carbon, phosphorus, and turbidity.

4.6.1 Ground-Water Sampling Procedures

In accordance with the procedures in the *Sampling and Analysis Plan for the UMTRA Ground Water Project*, well purging and sampling at the Tuba City site are collected in the following manner:

- To ensure collection of a representative sample, wells are purged for a minimum of 3 casing volumes and until measurements of pH, temperature, conductivity, and turbidity are stable. The method for withdrawing water from the well is determined in the field on the basis of site-specific conditions. Field measurements are considered stable when the three most recent temperature and conductivity readings vary less than 10 percent and the three most recent pH readings vary less than 0.2 pH units. Turbidity measurements are normally performed during purging of wells to be sampled for metals or radionuclide analysis.

- Sample bottles are precleaned to guidelines established by EPA (1992). Samples requiring filtration, such as dissolved metals, are filtered with a 0.45-micrometer filter, and samples requiring refrigeration in the field are stored between 0 EC and 4 EC. For samples preserved

with acid, the pH is periodically checked to ensure the proper acidity. Only commercially supplied, certified solutions are used for sample preservation.

- Duplicate ground-water and surface-water samples are collected in the field at a frequency of one duplicate sample per 20 water samples or one per sampling event, whichever is less.

- One equipment blank sample is prepared in the field for every 20 water samples collected with nondedicated equipment, or one per sampling event, whichever is less. Equipment blanks provide a check for cross-contamination of samples from ineffective equipment decontamination.

- Each sample is assigned a unique alphanumeric sample number and a site identification number corresponding to each sampling location. Samples collected for quality assurance are assigned a fictitious site identification number and submitted blind to the laboratory.

- Chain-of-custody records are used to identify the sample custodian while the samples are transported to the laboratory.

- Decontamination of nondedicated sampling equipment includes rinsing all equipment surfaces with diluted detergent followed by deionized water. Decontaminated equipment is stored in protective containers or plastic bags between sampling.

4.6.2 Analytical Results

Appendix C presents a complete list of analytical results for wells sampled during this program. DOE (1995b) describes the technical rationale for screening the analytes to derive the contaminants of potential concern (COPCs). The COPCs for the site are molybdenum, nitrate, selenium, strontium, sulfate, and uranium. Table 4–10 presents a summary of analytical results for each COPC. Results are used in Section 6 of this report to evaluate the human and ecological risk.

Table 4–10. Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0901	Molybdenum	0.036	0.001	0.2	35
TUB01	0901	Nitrate	12.0	3	34	30
TUB01	0901	Selenium	0.0047	0.0016	0.005	35
TUB01	0901	Strontium	0.325	0.22	0.59	21
TUB01	0901	Sulfate	21.2	15	70	30
TUB01	0901	Uranium	0.0032	0.001	0.01	32
TUB01	0902	Molybdenum	0.0093	0.0057	0.01	6
TUB01	0902	Nitrate	6.5	1	11	7
TUB01	0902	Selenium	0.0052	0.0035	0.008	7
TUB01	0902	Strontium	0.083	0.01	0.2	7
TUB01	0902	Sulfate	18.4	6	70	7
TUB01	0902	Uranium	0.0036	0.0007	0.01	7
TUB01	0903	Molybdenum	0.0122	0.001	0.1	32
TUB01	0903	Nitrate	32.4	5	49	26

Table 4–10 (continued). Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0903	Selenium	0.0049	0.0016	0.01	32
TUB01	0903	Strontium	0.779	0.6	1.01	19
TUB01	0903	Sulfate	33.2	15	53.8	26
TUB01	0903	Uranium	0.0027	0.001	0.007	31
TUB01	0904	Molybdenum	0.0091	0.001	0.01	20
TUB01	0904	Nitrate	7.54	5	12.4	20
TUB01	0904	Selenium	0.0082	0.0033	0.018	21
TUB01	0904	Strontium	1.01	0.68	1.32	15
TUB01	0904	Sulfate	70.7	48	134	20
TUB01	0904	Uranium	0.0042	0.0021	0.012	20
TUB01	0906	Molybdenum	0.167	0.0048	0.8	41
TUB01	0906	Nitrate	1184	580	2330	39
TUB01	0906	Selenium	0.071	0.005	0.469	41
TUB01	0906	Strontium	7.99	4.43	9.59	24
TUB01	0906	Sulfate	2130	1120	7590	39
TUB01	0906	Uranium	0.877	0.495	2.4	39
TUB01	0907	Molybdenum	0.013	0.01	0.02	3
TUB01	0907	Nitrate	1280	630	1800	3
TUB01	0907	Selenium	0.019	0.012	0.024	3
TUB01	0907	Strontium	4.83	2.6	6.4	3
TUB01	0907	Sulfate	2130	1300	2600	3
TUB01	0907	Uranium	0.235	0.21	0.26	2
TUB01	0908	Molybdenum	0.020	0.001	0.14	30
TUB01	0908	Nitrate	909	290	1500	29
TUB01	0908	Selenium	0.031	0.002	0.066	30
TUB01	0908	Strontium	3.91	2.72	5.6	22
TUB01	0908	Sulfate	3460	2620	4010	29
TUB01	0908	Uranium	0.128	0.082	0.21	29
TUB01	0909	Molybdenum	0.0112	0.001	0.04	27
TUB01	0909	Nitrate	900	654	1100	25
TUB01	0909	Selenium	0.0124	0.004	0.03	27
TUB01	0909	Strontium	6.54	0.744	8.8	20
TUB01	0909	Sulfate	1350	804	1690	25
TUB01	0909	Uranium	0.057	0.003	0.092	26
TUB01	0910	Molybdenum	0.032	0.0048	0.21	17
TUB01	0910	Nitrate	14.0	3	21.7	16
TUB01	0910	Selenium	0.0045	0.0016	0.005	17
TUB01	0910	Strontium	0.252	0.19	0.3	10
TUB01	0910	Sulfate	14.34	7.4	20.1	16
TUB01	0910	Uranium	0.0028	0.0007	0.012	17
TUB01	0911	Molybdenum	0.037	0.0048	0.18	9
TUB01	0911	Nitrate	15.5	3	27	9
TUB01	0911	Selenium	0.0040	0.0016	0.005	9
TUB01	0911	Strontium	0.468	0.1	0.602	8
TUB01	0911	Sulfate	17.7	5.3	84.9	9
TUB01	0911	Uranium	0.0017	0.0004	0.0062	9
TUB01	0912	Molybdenum	0.046	0.001	0.23	30
TUB01	0912	Nitrate	284	64	620	29

Table 4–10 (continued). Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0912	Selenium	0.012	0.004	0.0979	30
TUB01	0912	Strontium	3.64	0.1	4.31	14
TUB01	0912	Sulfate	641	407	830	29
TUB01	0912	Uranium	0.028	0.018	0.046	30
TUB01	0913	Molybdenum	0.059	0.001	0.21	18
TUB01	0913	Nitrate	26.6	3	71	17
TUB01	0913	Selenium	0.0043	0.001	0.005	18
TUB01	0913	Strontium	1.33	0.69	2.6	16
TUB01	0913	Sulfate	17.4	2.9	38.3	17
TUB01	0913	Uranium	0.0030	0.0003	0.0133	18
TUB01	0914	Molybdenum	0.022	0.001	0.18	24
TUB01	0914	Nitrate	14.3	3	35	23
TUB01	0914	Selenium	0.0045	0.0016	0.005	24
TUB01	0914	Strontium	0.462	0.1	0.55	13
TUB01	0914	Sulfate	19.3	15.2	37.4	23
TUB01	0914	Uranium	0.0026	0.0006	0.0186	24
TUB01	0915	Molybdenum	0.031	0.001	0.18	15
TUB01	0915	Nitrate	14	4	19	14
TUB01	0915	Selenium	0.0040	0.0016	0.005	15
TUB01	0915	Strontium	0.510	0.1	0.72	13
TUB01	0915	Sulfate	26.2	15.2	78.4	14
TUB01	0915	Uranium	0.0011	0.0003	0.0018	15
TUB01	0916	Molybdenum	0.032	0.001	0.17	11
TUB01	0916	Nitrate	10.364	3	18	11
TUB01	0916	Selenium	0.0038	0.0015	0.005	11
TUB01	0916	Strontium	1.60	0.35	8.2	10
TUB01	0916	Sulfate	17	11	38.1	11
TUB01	0916	Uranium	0.0012	0.0001	0.0059	11
TUB01	0917	Molybdenum	0.029	0.001	0.24	15
TUB01	0917	Nitrate	15.1	5	20.7	14
TUB01	0917	Selenium	0.0042	0.0016	0.005	15
TUB01	0917	Strontium	0.324	0.2	0.38	13
TUB01	0917	Sulfate	16.0	9.1	32	14
TUB01	0917	Uranium	0.0013	0.0006	0.0039	14
TUB01	0919	Molybdenum	0.049	0.0048	0.23	9
TUB01	0919	Nitrate	8.99	3	12	9
TUB01	0919	Selenium	0.0040	0.0016	0.005	9
TUB01	0919	Strontium	2.66	1.61	6.6	8
TUB01	0919	Sulfate	9.18	1.6	28.4	9
TUB01	0919	Uranium	0.00099	0.0001	0.004	9
TUB01	0920	Molybdenum	0.024	0.001	0.18	18
TUB01	0920	Nitrate	13.9	3	18	17
TUB01	0920	Selenium	0.0043	0.0015	0.005	18
TUB01	0920	Strontium	0.92	0.1	1.1	12
TUB01	0920	Sulfate	13.5	9.1	26	17
TUB01	0920	Uranium	0.0020	0.001	0.007	18
TUB01	0921	Molybdenum	0.026	0.001	0.22	19
TUB01	0921	Nitrate	12.1	2	25.7	18

Table 4–10 (continued). Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0921	Selenium	0.0045	0.0011	0.0081	19
TUB01	0921	Strontium	1.14	0.4	5.91	12
TUB01	0921	Sulfate	8.64	2.6	15.4	18
TUB01	0921	Uranium	0.0069	0.003	0.049	19
TUB01	0925	Molybdenum	0.01	0.01	0.01	3
TUB01	0925	Nitrate	699	648	750	2
TUB01	0925	Selenium	0.025	0.02	0.03	3
TUB01	0925	Strontium	3.36	3.09	3.64	2
TUB01	0925	Sulfate	3200	3160	3230	2
TUB01	0925	Uranium	0.135	0.131	0.139	2
TUB01	0926	Molybdenum	0.0090	0.0062	0.01	4
TUB01	0926	Nitrate	1170	1040	1260	3
TUB01	0926	Selenium	0.060	0.055	0.065	4
TUB01	0926	Strontium	7.76	7.04	8.8	3
TUB01	0926	Sulfate	1990	1830	2100	3
TUB01	0926	Uranium	0.282	0.24	0.357	3
TUB01	0929	Molybdenum	0.0055	0.001	0.01	2
TUB01	0929	Nitrate	40.6	35	46.2	2
TUB01	0929	Selenium	0.0038	0.0026	0.005	2
TUB01	0929	Strontium	0.333	0.31	0.357	2
TUB01	0929	Sulfate	17.7	17	18.4	2
TUB01	0929	Uranium	0.0019	0.0018	0.002	2
TUB01	0930	Molybdenum	0.0055	0.001	0.01	2
TUB01	0930	Nitrate	44.2	43	45.4	2
TUB01	0930	Selenium	0.0036	0.0021	0.005	2
TUB01	0930	Strontium	0.984	0.98	0.987	2
TUB01	0930	Sulfate	50.0	47.1	53	2
TUB01	0930	Uranium	0.0024	0.002	0.0028	2
TUB01	0932	Molybdenum	0.004	0.001	0.01	3
TUB01	0932	Nitrate	48.1	40	57.2	3
TUB01	0932	Selenium	0.0029	0.0019	0.005	3
TUB01	0932	Strontium	0.869	0.81	0.921	3
TUB01	0932	Sulfate	47.2	43.1	50.5	3
TUB01	0932	Uranium	0.0032	0.003	0.0036	3
TUB01	0934	Molybdenum	0.0055	0.001	0.01	2
TUB01	0934	Nitrate	1590	796	2380	2
TUB01	0934	Selenium	0.0083	0.005	0.0116	2
TUB01	0934	Strontium	8.81	6.12	11.5	2
TUB01	0934	Sulfate	4840	2150	7540	2
TUB01	0934	Uranium	0.215	0.102	0.328	2
TUB01	0935	Molybdenum	0.0049	0.001	0.01	7
TUB01	0935	Nitrate	797	735	839	6
TUB01	0935	Selenium	0.032	0.026	0.0396	7
TUB01	0935	Strontium	4.12	3.69	4.87	6
TUB01	0935	Sulfate	3220	3120	3360	6
TUB01	0935	Uranium	0.177	0.133	0.225	6
TUB01	0936	Molybdenum	0.0079	0.0016	0.01	4
TUB01	0936	Nitrate	3590	3010	4050	3

Table 4–10 (continued). Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0936	Selenium	0.11	0.025	0.188	4
TUB01	0936	Strontium	8.98	8.07	10.3	3
TUB01	0936	Sulfate	5080	4630	5400	3
TUB01	0936	Uranium	0.258	0.246	0.272	3
TUB01	0937	Molybdenum	0.023	0.016	0.0344	3
TUB01	0937	Nitrate	881	162	1280	3
TUB01	0937	Selenium	0.079	0.0127	0.126	3
TUB01	0937	Strontium	5.82	1.04	8.57	3
TUB01	0937	Sulfate	1680	326	2420	3
TUB01	0937	Uranium	0.415	0.0731	0.6	3
TUB01	0938	Molybdenum	0.004	0.001	0.01	3
TUB01	0938	Nitrate	1040	855	1340	3
TUB01	0938	Selenium	0.055	0.0487	0.066	3
TUB01	0938	Strontium	7.54	7.24	8.08	3
TUB01	0938	Sulfate	1800	1610	2110	3
TUB01	0938	Uranium	0.191	0.133	0.244	3
TUB01	0939	Molybdenum	0.586	0.52	0.71	4
TUB01	0939	Nitrate	672	598	733	3
TUB01	0939	Selenium	0.065	0.057	0.077	4
TUB01	0939	Strontium	4.72	4.21	5.2	3
TUB01	0939	Sulfate	1310	1240	1430	3
TUB01	0939	Uranium	0.502	0.405	0.603	3
TUB01	0940	Molybdenum	0.0053	0.0013	0.01	7
TUB01	0940	Nitrate	1990	888	2580	6
TUB01	0940	Selenium	0.106	0.036	0.15	7
TUB01	0940	Strontium	9.41	7.26	10.4	6
TUB01	0940	Sulfate	7080	4560	8400	6
TUB01	0940	Uranium	0.584	0.269	0.745	6
TUB01	0941	Molybdenum	0.051	0.0343	0.0647	8
TUB01	0941	Nitrate	262	165	322	7
TUB01	0941	Selenium	0.031	0.0212	0.0458	8
TUB01	0941	Strontium	1.65	1.16	1.84	7
TUB01	0941	Sulfate	411	225	555	7
TUB01	0941	Uranium	0.195	0.145	0.237	7
TUB01	0942	Molybdenum	0.031	0.02	0.0794	8
TUB01	0942	Nitrate	1630	1520	1750	7
TUB01	0942	Selenium	0.039	0.025	0.053	8
TUB01	0942	Strontium	6.48	5.65	6.79	7
TUB01	0942	Sulfate	3150	2940	3810	7
TUB01	0942	Uranium	0.350	0.304	0.428	7
TUB01	0943	Molybdenum	0.01	0.01	0.01	3
TUB01	0943	Nitrate	34.5	34	35	2
TUB01	0943	Selenium	0.005	0.005	0.005	3
TUB01	0943	Strontium	0.445	0.44	0.45	2
TUB01	0943	Sulfate	57	54	60	2
TUB01	0943	Uranium	0.0065	0.006	0.007	2
TUB01	0944	Molybdenum	0.0046	0.001	0.01	5
TUB01	0944	Nitrate	974	924	1030	4

Table 4–10 (continued). Summary of Analytical Results for COPCs

Site Code	Location Code	Analyte	Avg of Result	Min of Result	Max of Result	No. of Samples
TUB01	0944	Selenium	0.038	0.024	0.0466	5
TUB01	0944	Strontium	5.99	5.26	6.44	4
TUB01	0944	Sulfate	1680	1630	1760	4
TUB01	0944	Uranium	1.09	0.971	1.33	4
TUB01	0945	Molybdenum	0.012	0.0084	0.017	4
TUB01	0945	Nitrate	4.42	0.856	7	4
TUB01	0945	Selenium	0.0022	0.001	0.005	4
TUB01	0945	Strontium	0.414	0.36	0.463	4
TUB01	0945	Sulfate	20.5	16	31	4
TUB01	0945	Uranium	0.0028	0.002	0.0034	4
TUB01	0947	Molybdenum	0.01	0.01	0.01	1
TUB01	0947	Nitrate	7	7	7	1
TUB01	0947	Selenium	0.005	0.005	0.005	1
TUB01	0947	Strontium	0.31	0.31	0.31	1
TUB01	0947	Sulfate	17	17	17	1
TUB01	0947	Uranium	0.001	0.001	0.001	1
TUB01	0948	Molybdenum	0.01	0.01	0.01	1
TUB01	0948	Nitrate	9	9	9	1
TUB01	0948	Selenium	0.005	0.005	0.005	1
TUB01	0948	Strontium	0.65	0.65	0.65	1
TUB01	0948	Sulfate	10	10	10	1
TUB01	0948	Uranium	0.001	0.001	0.001	1
TUB01	0968	Molybdenum	0.025	0.025	0.025	1
TUB01	0968	Nitrate	3	3	3	1
TUB01	0968	Selenium	0.005	0.005	0.005	1
TUB01	0968	Sulfate	8	8	8	1
TUB01	0968	Uranium	0.002	0.002	0.002	1
TUB01	0970	Molybdenum	0.018	0.01	0.025	2
TUB01	0970	Nitrate	9.5	2	17	2
TUB01	0970	Selenium	0.005	0.005	0.005	2
TUB01	0970	Strontium	0.7	0.7	0.7	1
TUB01	0970	Sulfate	8.5	8	9	2
TUB01	0970	Uranium	0.003	0.002	0.004	2
TUB01	0971	Molybdenum	0.018	0.01	0.025	2
TUB01	0971	Nitrate	18	1	35	2
TUB01	0971	Selenium	0.005	0.005	0.005	2
TUB01	0971	Strontium	0.2	0.2	0.2	1
TUB01	0971	Sulfate	150	150	150	2
TUB01	0971	Uranium	0.0025	0.001	0.004	2
TUB01	0972	Molybdenum	0.018	0.01	0.025	2
TUB01	0972	Nitrate	2	1	3	2
TUB01	0972	Selenium	0.005	0.005	0.005	2
TUB01	0972	Strontium	0.4	0.4	0.4	1
TUB01	0972	Sulfate	15	11	19	2
TUB01	0972	Uranium	0.003	0.002	0.004	2

4.6.3 Vegetation Characterization

In upland arid ecosystems, if soils are of fine to medium texture and vegetation is mature and healthy, removal of soil water via evapotranspiration (ET) can, over a growing season, balance precipitation entering the soil. Under these conditions, recharge is essentially zero (Gee et al. 1994). At the Tuba City site, recharge in the vicinity of contaminated ground water may be elevated because soils are coarse-textured and degradation of plant communities, as a result of remediation activities and overgrazing of adjacent rangelands, may significantly limit ET. The coppice dune soil surrounding the Tuba City site is classified as a *torriorthent* consisting of moderately fine sand (Hendricks 1985).

4.7 Plant Ecology

This section presents the results of field and laboratory studies designed to evaluate (1) potential effects of ecological disturbances on diffuse groundwater recharge, and (2) risks associated with the use of plume water for irrigation of plants. The field study characterized the condition and relative abundance of vegetation in the vicinity of the plume. A greenhouse study conducted for DOE by the University of Arizona evaluated plant uptake and phytotoxicity of plume water.

4.7.1 Vegetation Characterization

The plant communities overlying contaminated ground water can be classified into three different stages of disturbance: grazed rangeland, protected rangeland, and regraded seeded land. The occurrence of three disturbance conditions (treatments) are a consequence of surface remediation activities. Most of the area overlying contaminated ground water is either heavily grazed rangeland or regraded areas that have been reseeded. A barbed wire fence erected around the disposal site in 1986 protects the reseeded areas and also protects a parcel of rangeland vegetation from grazing.

Plant species composition, density, and cover were sampled in three 30 × 30 m plots in each of the three treatments. Table 4–11 is a comprehensive list of plant species for all plots. Plant density was estimated in 30 1^{-} m^2 quadrants systematically placed within each plot (Table 4–12). Plant cover was estimated using a line intercept method (Table 4–12). The distance d of plant canopy intercepted by a randomly placed 30–m transect was measured and percent cover for that line was estimated (Bonham 1989):

$$Percent\ Cover\ '\ \frac{Sd}{30\ m} \times 100$$

Table 4–11. Plants Growing in the Plume Area of the Tuba City Site

Scientific Name[a]	Acronym[b]	Common Names[c]
Shrubs		
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	torrey joint fir, Mormon tea

Table 4–11 (continued). Plants Growing in the Plume Area of the Tuba City Site

Scientific Name[a]	Acronym[b]	Common Names[c]
Ephedra viridis Cov.	EPVI	green joint fir, Mormon tea
Gutierrezia sarothrae (Pursh) Britt. & Rusby	GUSA	broom snakeweed, turpentine weed
Opuntia polyacantha Haw.	OPPO	plains prickly pear
Sarcobatus vermiculatus (Hook.) Torr.	SAVE	black greasewood, chico, chicobush
Yucca baccata Torr.	YUBA	datil yucca, banana yucca
Grasses		
Agropyron cristatum (L.) Gaertner	AGCR	crested wheatgrass
Bouteloua barbata Lag.	BOBA	sixweeks grama
Bouteloua eripoda (Torr.) Torr.	BOER	black grama
Bromus rubens L.	BRRU	red brome
Festuca octoflora Walter.	FEOC	sixweeks fescue
Hilaria jamesii (Torr.) Benth.	HIJA	galleta, curly grass
Muhlenburgia pungens Thurber in Gray	MUPU	sandhill muhly
Munroa squarrosa (Nutt.) Torr.	MUSQ	false buffalograss
Oryzopsis hymenoides (R. & S.) Ricker	ORHY	Indian ricegrass, sand bunchgrass
Panicum capillare L.	PACA	witchgrass
Sporabolis airoides (Torr.) Torr.	SPAI	alkali saccaton
Sporabolis cryptandrus (Torr.) Gray	SPCR	sand dropseed
Sporabolus contractus A.S. Hitchc.	SPCO	spike dropseed
Sporabolus flexousus (Thurber) Rydb.	SPFL	mesa dropseed
Forbs		
Amaranthus blitoides Wats.	AMBL	prostrate pigweed
Ambrosia confertifilia DC.	AMCO	slimleaf bursage
Astragalus wingatanus Wats.	ASWI	Fort Wingate milkvetch
Conyza Less.	CO sp.	horseweed
Cryptantha crassisepala (T. & G.) Greene	CRCR	plains cryptanth
Eriogonum subreniformi Wats.	ERSU	Stokes buckwheat
Eriogonum wetherilli Eastw.	ERWE	Wetherill buckwheat
Eriogonum wrightii Torr. in DC.	ERWR	Wright buckwheat
Euphorbia fendleri T. & G.	EUFE	fendler euphorb
Lupinus L. species	LU sp.	lupine
Lygodesmia grandiflora (Nutt.) T. & G.	LYGR	showy rushpink
Mentzelia sp. L.	ME sp.	stickleaf
Pectis angustifolia Torr.	PEAN	pectis
Phacelia ivesiana Torr. in Ives	PHIV	phacelia
Plantago patagonica Jacq.	PLPA	wooly plantain
Salsola iberica Sennen & Pau	SAIB	Russian thistle, tumbleweed
Solanum sarachoides Sendt. ex Martius	SOSA	solanum
Sphaeralcea rusbyi Gray	SPRU	globemallow, falsemallow
Stephanomeria exigua Nutt.	STEX	annual wirelettuce

[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.
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).

Table 4–12. Mean Plant Density and Cover in Grazed, Protected, and Reseeded Areas Overlying Contaminated Ground Water at Tuba City

Plant Names (*Genus species*)	Plant Density (per square meter)			Plant Cover (%)		
	Grazed	Protected	Reseeded	Grazed	Protected	Reseeded
Grasses						
Bouteloua eripoda	5.7 a	4.2 b	-	0.70 a	2.7 b	-
Bromus rubens	0.3 a	0.6 b	-	-	-	-
Hilaria jamesii	5.8 a	4.9 a	0.2 b	0.82 a	2.4 b	0.08 a
Oryzopsis hymenoides	0.8 a	0.7 a	2.2 b	0.02 a	0.32 a	6.30 b
Forbs and Shrubs						
Salsola iberica	5.0 a	3.7 a	7.7 b	0.60 a	1.90 a	2.30 a
Ambrosia confertiflora	6.7 a	0.5 b	1.4 c	0.70 a	0.10 b	0.16 b
Gutierrezia sarothrae	0.2 a	-	0.02 b	0.14 a	0.09 a	0.15 a
Lygodesmia grandiflora	0.3 a	0.04 a	-	0.06 a	0.03 a	-
Eriogonum wrightii	-	0.2	-	0.04 a	0.60 a	-
Lupinus sp.	-	-	-	-	0.02	-
Stephanomeria exigua	0.3 a	0.09 b	-	-	-	-
Plantago patagonica	0.02 a	0.02 a	-	-	-	-
Opuntia polyacantha	0.04 a	0.03 a	-	0.04	-	-
Eriogonum wetherilli	-	0.01	-	-	-	-
Eriogonum subreniforme	0.23 a	0.01 b	-	-	-	-
Astragalus wingatanus	0.12 a	0.01 b	-	0.01	-	-
Yuccaa baccata	0.01 a	0.01 a	-	-	0.14 a	0.13 a
Ericameria cervinea	0.51 a	0.11 a	3.3 b	0.11 a	0.05 a	0.60 a
Atriplex canescens	-	0.01 a	0.01 a	-	-	-
Sphaeralcea rusbyi	-	-	0.09	-	-	0.13
Ephedra viridis	0.40 a	0.30 a	0.01 b	3.43 a	8.70 b	-
Total Live Vegetation	26.4 a	15.4 b	14.9 b	6.70 a	16.90 b	9.70 c
Dead Vegetation	-	-	-	4.20 a	7.50 b	5.60 ab
Total	26.4 a	15.4 b	14.9 b	10.9 a	24.4 b	15.3 a

Note: Mean values for a given species followed by the same letter are not significantly different at P<0.05. The statistical analysis consisted of two-way analysis of variance (disturbance stage and location were variables) and Duncan's Multiple Range test of differences among means.

Species diversity, density, and cover varied among plant communities. Species diversity is lowest in reseeded areas where the vegetation is dominated by a seeded grass, *Oryzopsis hymenoides*, and a weedy annual forb, *Salsola iberica*. Species composition and diversity of grazed and protected rangeland is similar; *Ephedra viridis* is the dominant species in both areas. The primary

difference between the two is the greater percent cover of perennial grasses in protected rangeland. Overall, density is highest in the grazed rangeland because of the abundance of weedy annuals, and cover is highest in the protected rangeland because of the greater leaf area of perennial grasses and *Ephedra*. Greater than 2 times the percent cover of live vegetation in protected areas than in grazed areas indicates that transpiration rates may be twice as high or higher in protected areas. Therefore, diffuse recharge is likely highest in grazed rangeland overlying the plume.

4.7.2 Plant Uptake Study

The Tuba City BLRA identified several potential exposure pathways that involve uptake of contaminants in plant tissues (DOE 1994a), but data were unavailable to adequately evaluate these pathways. With the assistance of the University of Arizona Environmental Research Laboratory, DOE conducted a 2-year study to acquire the plant uptake data needed to complete the risk assessments. The study consisted of three parts: a literature review, a greenhouse experiment involving uptake of simulated contaminated ground water in garden and crop plants, and a greenhouse experiment involving crop and native plant uptake of actual contaminated ground water from well 906 at Tuba City.

Numerous citations were found in recent literature dealing with uptake and effects of metals on plants (Baumgartner et al. 1996). A synthesis of relevant literature suggests that water-to-plant and soil-to-plant concentration ratios for metals are highly variable and dependent on plant species and the soil or water chemistry of the site. Very little relevant literature was found concerning irrigation of crops with water with the contaminants and concentrations present at Tuba City and other UMTRA sites. The greenhouse studies were designed to acquire needed site-specific data.

The greenhouse studies were designed primarily to test hypotheses concerning plant uptake of various heavy metals. The results were input to assessments of human health and ecological risks associated with irrigation of rangeland and cropland vegetation using contaminated ground water. A secondary purpose of the experiments was to test hypotheses concerning the surface application alternative for ground-water remediation. The surface application alternative would exploit physiological reduction and assimilation of nitrate by halophytes (salt-tolerant plants). Nitrate-contaminated ground water would be treated by irrigating fields planted with the native halophyte *Atriplex canescens* (four-wing saltbush).

The greenhouse studies of metal uptake by plants progressed through two phases. The results of both phases, summarized below, are contained in a final report (Baumgartner et al. 1996).

Phase I

Phase I was a generic study using metals that frequently occur at UMTRA sites (U, Se, Mo, and Mn). The effects of single-metal doses on plant tissue concentrations and plant biomass were tested using three crop types: carrots, a root crop; squash, a fleshy vegetable; and Sudan grass, a forage grass. Influences of nitrates on metal uptake and plant growth were also tested.

A summary of the results of this dose-response study follows.

- The response of plant-to-water concentration (dose) of Mn was linear with the highest concentration ratios for carrots and the lowest for squash.
- The responses of Mo and Se tissue concentrations to dose level were logarithmic and similar for all three crops.
- Responses of U tissues concentrations varied among crop plants; significantly logarithmic in carrots and weakly linear in Sudan grass. U concentrations in squash were negligible at all dose levels.
- Uptake of metals in Sudan grass varied in response to nitrate concentration. U uptake increased significantly in response to nitrate for dose levels above 0.14 parts per million (ppm). The response of other metals to nitrate concentration was more subtle: Mn increased slightly; Mo and Se showed slight decreases.
- Except for Se, biomass production remained uniform over the range of metal doses applied. Biomass production dropped precipitously at higher Se doses.
- Biomass production was highest for NO_3 concentrations between 400 and 600 ppm but dropped rapidly above 1,000 ppm. The lowest biomass levels occurred in response to a combination of high NO_3 and high Se doses.

These results suggest that at a few UMTRA sites, use of contaminated ground water for irrigation could elevate crop tissues concentrations of Mn, Mo, and Se above maximum tolerable levels for dietary minerals. The results also show that high NO_3 concentrations or combinations of high NO_3 and Se could significantly reduce crop production at some sites. Conversely, the results show that at most UMTRA sites, use of contaminated ground water for irrigation will not elevate crop tissue concentrations of metals above maximum tolerable levels and will have little or no influence on crop production.

Phase II

A second greenhouse study used soils and ground water from the Tuba City, Arizona, site for risk assessment and phytoremediation tests to support ground-water remediation and surface reclamation planning. The experiment compared plant uptake of water from the most contaminated monitoring well at the site with water from an upgradient well and with a 50/50 mix of contaminated and upgradient water. Sudan grass and four-wing saltbush were grown in the greenhouse. Sudan grass was used to determine if contaminated ground water from the site could be used to produce an acceptable forage grass crop. Four-wing saltbush was used because it provides important winter range forage for livestock and wildlife, it is a traditional food source for Native Americans, and it is a good candidate for phytoremediation of nitrates. Four-wing saltbush was also considered a good candidate for soil stabilization surrounding the disposal cell at the Tuba City site.

The results are summarized below.

- Overall, irrigation with contaminated Tuba City site well water compared to upgradient well water may raise levels of Se and U in four-wing saltbush, but may have no significant effect

on concentrations of Mn, Mo, NO_3, total N, or total S. Irrigation with contaminated well water may increase levels of Mo, Se, U, NO_3, and total S in Sudan grass.

- Plant tissues concentrations of Mo, Se, and U were significantly higher in Sudan grass than in four-wing saltbush. Only S concentrations were significantly higher in four-wing saltbush than in Sudan grass.

- Biomass of four-wing saltbush irrigated with contaminated well water and of plants irrigated with upgradient well water was not significantly different. However, biomass of Sudan grass was significantly lower for plants irrigated with contaminated well water.

These results suggest that, except for selenium, irrigation of four-wing saltbush and Sudan grass with contaminated Tuba City well water should not result in adverse tissue accumulations of metals. Irrigation with a mixture of contaminated and upgradient well water should maintain tissue accumulations of metals below maximum tolerance levels and have no significant effect on plant productivity. These results apply only to tissue accumulation of metals and plant productivity after a single growing season and, therefore, should be interpreted with caution. Prolonged irrigation over repeated crop cycles could result in soil accumulation of metals.

5.0 Site Conceptual Model

This section contains a synthesis of all the known physical aspects of the Tuba City UMTRA site. The site conceptual model was developed with information from four major disciplines: geology, hydrology, geochemistry, and ecology; however, these disciplines are dynamically interdependent. The technical feasibility of various alternative management actions is evaluated against the backdrop of the site conceptual model.

5.1 Geology

The Tuba City UMTRA site is located along the northern rim of a broad valley formed by Moenkopi Wash, which is a tributary to the Little Colorado River. In the area of the site, three subhorizontal pediment surfaces were cut into the Navajo Sandstone during Pleistocene time by ancestral flows within the Moenkopi Wash drainage. For discussion, the three pediment surfaces, from oldest to youngest, are referred to as the upper, middle, and lower terraces. Each pediment surface is mantled by a veneer of alluvial terrace material. In the upper two terraces the alluvial deposits are cemented with calcareous cement. Cooley and others (1969) (Plate 3) mapped the terraces and classified the upper two terraces as late Black Point surfaces of early Pleistocene age; the lower terrace was classified as an early Wupatki surface of middle to late Pleistocene age.

The UMTRA disposal cell is on the middle terrace at an elevation of about 5,060 ft (Plate 1). The upper terrace is north of the cell and about 100 ft higher in elevation than the middle terrace; U.S. Highway 160 is on a narrow remnant of the upper terrace. The lower terrace is about 2,500 ft south of the cell and about 150 ft lower in elevation than the middle terrace. The lower and middle terraces are largely covered by loose dune sand deposited in Recent time by eolian processes and derived from nearby deeply weathered and easily eroded Navajo Sandstone. The loose sand is thicker on the lower terrace; thickness is about 20 ft.

The alluvial terrace deposits below the eolian material consist mainly of sand and gravel and minor amounts of silty clay. The sand and gravel deposits consist of poorly sorted, angular to well-rounded sandstone fragments and minor fragments of limestone in a silty matrix. Upper-terrace sand and gravel deposits are as much as 10 ft thick, and the matrix is well cemented by calcareous cement. The middle terrace sand and gravel deposit is as much as 15 ft thick; its calcareous-cemented matrix is nearly as well cemented as that of the upper-terrace sand and gravel. Sand and gravel in the lower terrace is of similar composition to the middle- and upper-terrace sand and gravel; however, the lower-terrace sand and gravel is uncemented.

Thin lenticular beds of limestone, within the Navajo Sandstone, are present in places immediately below the base of the alluvial material at all three terrace levels (Plate 1). These thin, resistant limestone beds probably controlled, in part, the elevation and location of the pediment surfaces formed by the ancestral Moenkopi Wash. Locations of several representative limestone beds just below the alluvial terrace material are inferred in the geologic cross sections (Plate 2).

A lens of silty, inorganic, gray clay is present at the base of the alluvial sequence along the northwest part of the middle terrace. Up to 15 ft of clay was indicated on the lithologic logs from

several boreholes (from 913 at the southwest to 907, about 0.5 miles to the northeast (Plate 2). The dimensions of the inferred subcrop are about 500 to 800 ft wide and 1 mile long. This relatively impermeable clay layer may cause perched water conditions, and ground-water recharge may have flowed laterally to the edges of the lens, especially during the time of ore processing.

Navajo Sandstone bedrock immediately underlies the middle terrace deposit. In the region of the site, the classic section of the Navajo Sandstone thins rapidly to the south because of erosion, where it is underlain by the intertonguing interval between the Navajo Sandstone and the Kayenta Formation (Middleton and Blakey 1983). For simplicity, the classic section of Navajo Sandstone will be referred to as Navajo Sandstone. The intertonguing interval comprises the lower Navajo Sandstone and contains six sedimentary facies that can be identified on the basis of lithology, sedimentary structures, and unit geometries. The remnant section of Navajo Sandstone is about 100 ft thick near the disposal cell, and the intertonguing interval is approximately 350 to 400 ft thick.

In the vicinity of the site, the contact between the two sedimentary units is transitional. The top of the intertonguing interval near the site is located (somewhat arbitrarily) about 2,500 ft south of the disposal cell at an elevation of about 4,920 ft (Plates 1 and 2). The placement of the contact at that location is based on three factors: (1) an area of enhanced growth of vegetation supported by a shallow ground-water surface referred to as the “greasewood area,” (2) monitoring-well data that suggest that contamination near the site does not penetrate below an elevation of about 4,920 ft, and (3) the abrupt thickening of the Navajo deposits north and northwest of the site, reaching 1,000 ft within a distance of 30 miles (Middleton and Blakey 1983). This transitional contact is assumed to dip gently at about 2 degrees toward the north northeast and the axis of the Tuba City Syncline (Cooley et al. 1969, Haynes and Hackman 1978).

The Navajo Sandstone, in general, comprises the uppermost part of the Jurassic/Triassic Glen Canyon Group and the chief aquifer of the N-multiple aquifer system. It was deposited mainly under eolian conditions. The sandstone is conspicuously crossbedded in most places and is composed of medium to very fine, subrounded, quartz grains that are held together by weak siliceous and calcareous cement. Outcrops of Navajo Sandstone are typically light brown to tan to white and in places contain subtle joints. One joint set 0.5 miles north of the disposal cell strikes north 10 degrees west and dips 80 degrees east. Another subvertical joint set occurs in an outcrop 1,500 ft southeast of the disposal cell; its strike ranges from north 65 to 75 degrees west. The spacing between individual joints is about 4 ft. In outcrops, these joints appear open and uncemented.

Thin subhorizontal to dish-shaped (concave upward) beds consisting of very fine sand and silt exist throughout the Navajo Sandstone section near the disposal cell. Plate 1 shows such a contact east of the disposal cell where crossbedded sandstone overlies massive sandstone. Middleton and Blakey (1983) suggested that cut and fill structures such as these represent a hiatus in the eolian deposition process. More recently, Blakey (1988) proposed the term “superscoops” to describe the dish-shaped structures formed by wind deflation that are filled with overlying eolian or aqueous sediments.

Resistant, thin, lenticular limestone beds or pans occur at numerous stratigraphic levels within the Navajo Sandstone. Plate 1 presents the locations where these beds have been mapped in the field, and Plate 2 indicates where they might exist beneath the surface. The gray, dense limestone beds are from 3 inches to as much as 2 ft thick and extend laterally from 500 ft to as much as 1,500 ft. Thin lenses, or pods, of dark gray chert and red jasper occur sporadically in the limestone beds. The limestone lenses were formed in ephemeral lakes among the dunes of the Navajo erg, or sand sea, in superscoop settings as described by Blakey (1988). The shapes of the individual limestone layers in plan view are not known, but it is assumed that they are roughly circular as they are elsewhere in the Navajo Sandstone (Pipiringos and O'Sullivan 1975). The limestone beds are highly resistant and influence topography; they are usually found at the crest of low benches or terraces in the site area. Occurrence of the limestone beds does not appear to be restricted to any particular stratigraphic interval within the Navajo Sandstone. The beds occur from just above the floodplain of Moenkopi Wash northward to the upper terrace on which U.S. Highway 160 is situated. The site conceptual model of the Tuba City UMTRA site proposes that the limestone pans are one factor responsible for the low vertical hydraulic conductivity.

5.2 Hydrology

The chief aquifer near the Tuba City site is the N-multiple aquifer system of Cooley and others (1969). On a regional scale, the N-aquifer is extremely vast. It encompasses all of the Navajo Sandstone deposits, including the intertonguing interval. The volume of water contained in it has been estimated to be at least 180 million acre-feet. On the basis of more than 40 short-term well and aquifer tests, Eychaner (1983) estimates the transmissivity to range from 20 to 800 ft^2/day. The estimated horizontal hydraulic conductivity, calculated using the total saturated thickness, ranges from 0.05 to 2.1 ft/day and averages 0.65 ft/day. The average unconfined storage coefficient is estimated to be 0.10 to 0.15. From the unconfined portions of the aquifer, the production of 10 acre-feet of water from a 1 square mile (mi^2) area would result in a water-level decline of less than 0.16 ft (Eychaner 1983). Regionally, the aquifer obtains its recharge from rainfall and snowmelt throughout the 1,400 mi^2 area where the Navajo Sandstone is exposed. A small amount of recharge occurs as leakage from overlying confining beds. Estimated annual recharge to the aquifer is 13,000 acre-feet (Eychaner 1983).

The valley of Moenkopi Wash is an important regional-discharge location for the N-aquifer. The discharge occurs as evapotranspiration to vegetation, direct discharge to springs and seeps along the axis of Moenkopi Wash, and discharge to Moenkopi Wash alluvium. On the basis of baseflow measurements made in winter months during 1926 to 1941 in Moenkopi Wash near Tuba City, the equilibrium discharge to surface water from the N-aquifer is estimated to be about 3,800 acre-feet for a full year (Eychaner 1983).

Ground-water pumping is another important stress on the N-aquifer. The largest amount of pumping occurs on Black Mesa (about 50 miles northeast of the Tuba City site) in conjunction with the Peabody Coal Mine. The pumping at the mine is expected to increase from a past average of 3,700 acre-feet per year to about 5,100 acre-feet per year or more by the year 2001. Pumping occurs where the aquifer is largely confined. The effect of this pumping on water levels in the unconfined portions of the aquifer is not discernable at the present time. However, municipal-water supply pumping near Tuba City is expected to increase during the coming decades. The combination of pumping at the mine and municipal pumping near Tuba City is

projected to result in declining water levels in the unconfined parts of the aquifer during the early half of the coming century; however, the declines are projected to be small (Eychaner 1983).

5.2.1 Site Hydrology

The saturated thickness of the N-aquifer near the disposal cell is about 400 ft, but within 2,000 ft south of the disposal cell the aquifer thins rapidly towards the south. Depth to ground water also decreases rapidly over this reach, and phreatic vegetation becomes more abundant between the middle and lower terraces. The increase in phreatophytes is evidenced by a band of relatively robust vegetation referred to as the "greasewood area" between the middle and lower terraces. Approximately 4,000 ft south of the site, seeps issue forth along cliff bands that border the incised valley of Moenkopi Wash (Plate 1). On the basis of the observed vegetation and depth to ground water, evapotranspiration appears to be the mechanism whereby most water exits the N-aquifer.

In and around the Tuba City site, a total of 43 monitoring wells tap the N-aquifer. Figure 3–2 shows the locations of the monitoring wells. Table 4–1 summarizes the construction details for these wells. Figure 5–1 shows the phreatic surface as it exists near the site and indicates that there is a ridge of elevated hydraulic heads near the south central portion of the disposal cell. The axis of the ridge appears to be oriented subparallel to the local hydraulic gradient. The conceptual model attributes this ridge of elevated hydraulic heads to the combined effects of harvested runoff from the cover of the disposal cell and areas with low hydraulic conductivity south of the disposal cell.

An alternative hypothesis is that the mound is created by drainage from the cell. Table 5–1 shows the rate of drainage from the cell and how it is expected to change with time (MACTEC–ERS calculation U0000202). Table 5–1 shows that drainage may contribute a large volume of water to the saturated zone, especially during the first 5 years after cell completion. The source of the initial drainage is attributed to expelled pore water from consolidation of the slime tailings. Figure 5–2 presents the spatial distribution of trends in water levels near the site (a more comprehensive trend analysis is presented in Appendix B). The trend distribution indicates that after reaching a local maximum in hydraulic heads during 1993 to 1994, a period of 3 to 4 years after cell completion, water levels in most of the wells around the downgradient portion of the cell have been gradually declining. An apparent correlation exists between the water-level trend distribution and the predicted drainage from the cell. Continued monitoring of water levels near the disposal cell would be required to evaluate whether the correlation is real or apparent. After the initial pulse of moisture is released from consolidation, the predicted drainage rate by itself would likely be incapable of sustaining the observed ridge of ground water.

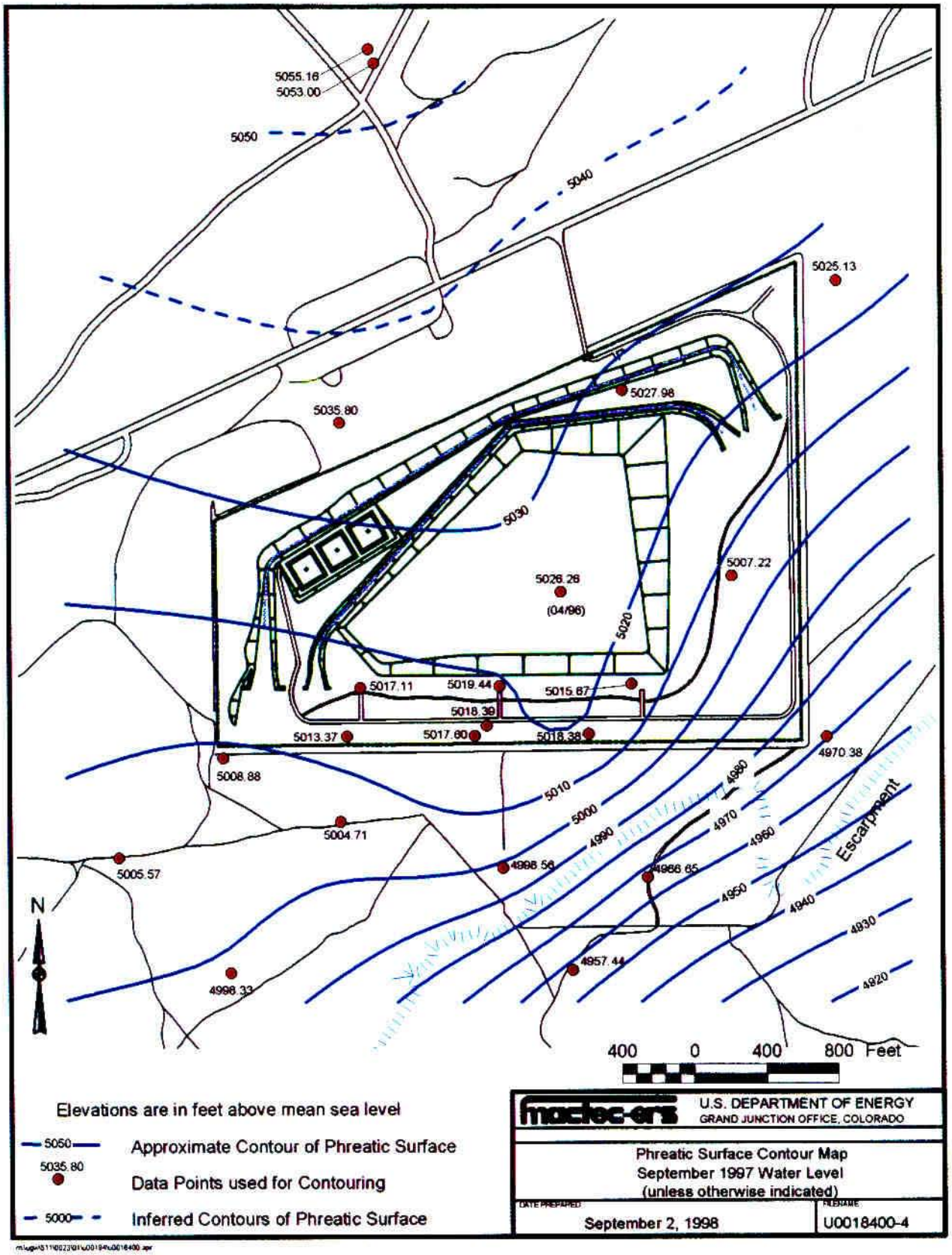


Figure 5–1. Phreatic Surface Contour Map

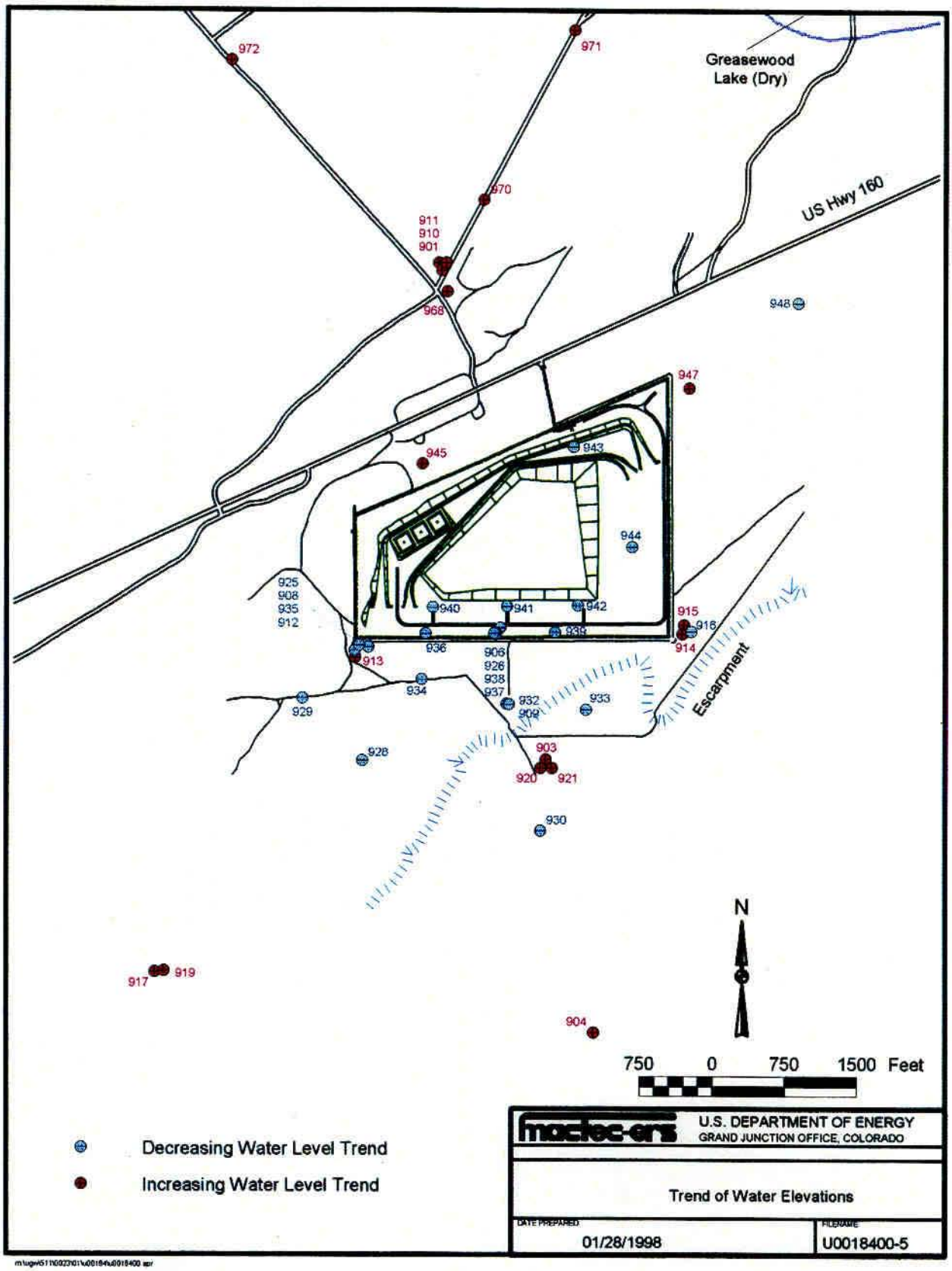


Figure 5–2. Trend of Water Elevations

Table 5–1. Estimated Drainage Volume and Rate

Time After Cell Closure (years)	Predicted Drainage		Total Cumulative Drainage Volume (gallons)	Drainage Rate (gallons per year)	Drainage Rate, Percent of Estimated Total
	Thick Slimes (cm)	Thin Slimes (cm)			
1	4.134	3.332	1,292,436	1,292,436	4
5	7.139	5.721	2,224,132	232,924	6
10	9.718	7.661	2,997,783	154,730	8
15	11.710	9.112	3,584,180	117,279	10
20	13.420	10.290	4,071,655	97,495	11
40	18.520	13.630	5,484,736	70,654	15
60	22.150	15.900	6,465,266	49,026	18
80	25.020	17.650	7,229,976	38,236	20
100	27.410	19.070	7,858,008	31,402	22
200	35.650	23.800	9,984,267	21,263	27
300	40.960	26.760	11,333,733	13,495	31
400	44.890	28.950	12,332,317	9,986	34
500	48.050	30.700	13,132,680	8,004	36
600	50.610	32.160	13,791,026	6,583	38
700	52.830	33.440	14,365,209	5,742	40
800	54.750	34.570	14,867,206	5,020	41
900	56.460	35.600	15,319,849	4,526	42
1000	57.990	36.540	15,729,182	4,093	43

cm = centimeters

5.2.2 Site Water Balance

A water balance was developed for the area near the disposal cell as a precursor to developing a numerical flow model of the site. The water balance accounts for all of the major sources and sinks of ground water and is a synthesis of everything known about the site hydrology. Table 5–2, modified from calculation U00151AA, contains a summary of the site water balance information. An earlier version of the site water balance appears in MACTEC–ERS calculation U0005600; however, the surface areas used for multiplying the fluxes were superseded by those in MACTEC–ERS calculation U00151AA. Figure 5–3 depicts how the water balance components might be distributed spatially near the site.

The total discharge for the entire thickness of the N-aquifer and the flux rate near the site may be estimated from observed hydraulic head values in well 901 (located north of the repository) and an assumed hydraulic head value of 4,700 ft assigned to Moenkopi Wash. The length of the flow path for this problem is about 10,000 ft and the average hydraulic conductivity, based on local aquifer tests and slug tests, is 0.5 ft/day. The D–F discharge formula (Bear 1979) was used to estimate the aquifer discharge per unit width as being about 4 ft^2/day, and the corresponding flux rate is estimated to be about 110 centimeters per year (cm/yr) (MACTEC–ERS 1997; calculation U0005600). The surface area perpendicular to the flow, computed as the 12,000-ft width × 400-ft depth, equals 4.8×10^6 square feet (ft^2). Consequently, the ground-water flow into the site domain is 4.7×10^4 cubic feet per day (ft^3/day) (Table 5–2 and Figure 5–3).

Table 5–2. Water Balance Based on Field Data for the UMTRA Repository Near Tuba City

Flow Component	Estimated Flux (cm/yr)	Surface Area (ft^2)	Inflow[a] (ft^3/day)	Outflow[a] (ft^3/day)
North–northwestern boundary	110	4.8×10^6	4.74×10^4	0
West–southwestern boundary	NA	NA	0	0
East–northeastern boundary	NA	NA	0	0
Evapotranspiration between middle terrace and cliff bands (ET_1)	5.0	5.3×10^7	0	2.4×10^4
Evapotranspiration at Greasewood area (ET_2)	16.	1.2×10^6	0	1.7×10^3
Evapotranspiration along cliff bands (ET_3)	110	1.7×10^6	0	1.7×10^4
Discharge to Moenkopi Wash	110	1.3×10^6	0	1.3×10^4
Diffuse recharge	1.6	5.6×10^7	8.05×10^3	0
Maximum transient drainage at t = 1 year after cell closure	4.6	1.5×10^5	4.73×10^2	0
Totals			**5.6×10^4**	**5.6×10^4**

[a] Inflows and outflows are obtained by multiplying the flux (cm/yr) by surface area (ft^2) by 8.99×10^{-5} (ft/yr)/(cm/day)

The natural recharge component at the Tuba City site is assumed to consist principally of diffuse areal recharge. No attempts have been made thus far in the program to quantify the amount of diffuse recharge at the site. Consequently, this parameter must be estimated from literature values and used subject to professional judgement. Based mostly on Stephens' (1994) field studies completed in the western United States, natural recharge at the Tuba City site is assumed to be approximately 0.65 inches per year (1.65 cm/yr) (MACTEC–ERS calculation U0005600). The diffuse recharge component functions over a surface area of 5.6×10^7 ft^2. Therefore, the calculated flow due to diffuse recharge is about 8.0×10^3 ft^3/day. This recharge rate functions over much of the area surrounding the UMTRA repository at Tuba City; however, it excludes the cell and apron areas because the compacted cover enhances runoff, which produces augmented recharge in the apron areas where the runoff collects (Table 5–2 and Figure 5–3).

From a regional perspective, the entire area between Moenkopi Wash and the middle terrace is essentially a ground-water discharge area via ET. The ET is dominated by transpiration across the lower terrace and the slope between the middle and lower terraces. Depth to ground water across this region is probably less than 20 ft, and the surface is composed of dune sands that are incapable of wicking soil moisture to the land surface. This keeps the ground water protected from direct evaporation. Transpiration is manifested through the plant community, which is dominated by greasewood, a phreatophyte whose deep roots tap directly into the ground water. Recent studies suggest that transpiration from healthy greasewood populations can range

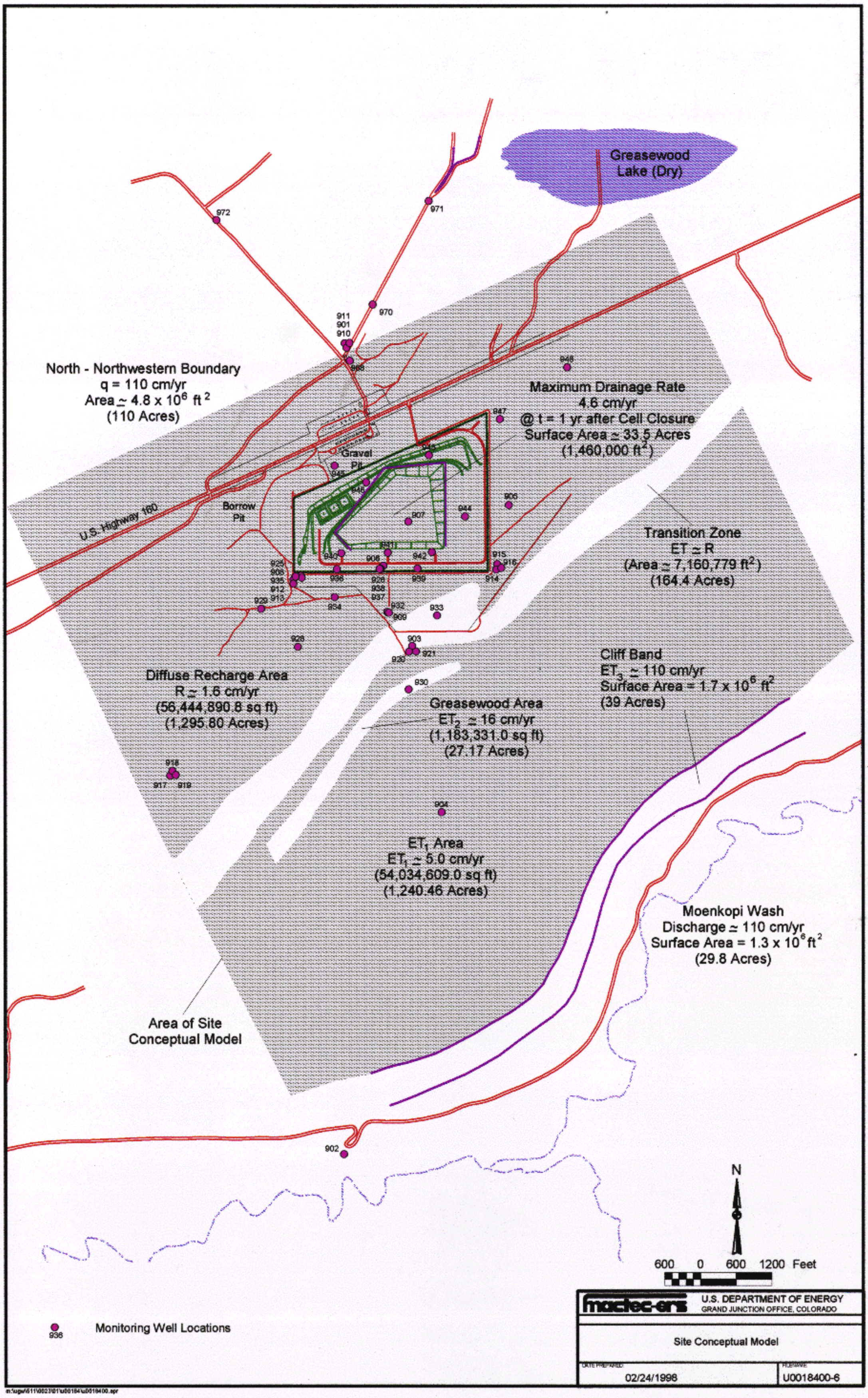


Figure 5–3. Site Conceptual Model

between 24 and 160 cm/yr (Nichols 1993, Branson et al. 1981). The region south of the UMTRA repository near Tuba City has been overgrazed; consequently, the greasewood community is probably less efficient at transpiring ground water.

The ET components south of the site are divided into two main groups on the basis of vegetation density. One group (ET_1) comprises most of the surface area between the middle terrace and the steep cliffs above Moenkopi Wash. On the basis of plant density, the ET flux in this region is assumed to be about 5 cm/yr, and the surface area over which it functions is about 5.3×10^7 ft^2. Therefore, the volumetric evapotranspiration rate for the ET_1 region is about 2.4×10^4 ft^3/day (Table 5–2 and Figure 5–3).

The second ET component, referred to as ET_2, is located on the hillslope between the middle and lower terraces in an area dominated by a relatively lush band of greasewood. The ET flux through the greasewood area is estimated to be about 16 cm/yr. The surface area over which it functions is about 1.2×10^6 ft^2 . Consequently, the ET rate through the greasewood area is about 1.7×10^3 ft^3/day (Table 5–2 and Figure 5–3).

Below the lower terrace, evaporation becomes more important. Seeps are evident along the cliff bands between the lower terrace and the bottom of Moenkopi Wash. Since parts of the cliff-band areas are perennially wet, the ET flux approaches 4.83 ft/year (147 cm/yr), the published evaporation rate for Tuba City (Cooley 1970). As before, the flux rate from the aquifer (110 cm/yr) limits the ET rate. Therefore, the ET rate along the cliff band area is ET_3 = 110 cm/yr. The cross-sectional area of the cliff band perpendicular to flow is about 1.7×10^6 ft^2. The flow of water across the cliff face is computed to be about 1.7×10^4 ft^3/day (Table 5–2 and Figure 5–3).

Discharge to the alluvium and surface water of Moenkopi Wash constitutes the remaining component of flux from the N-aquifer. The flux of the ground water is assumed to be equal to the flux computed earlier, 110 cm/yr. The surface area perpendicular to the ground-water discharge into the wash is estimated to be about 1.3×10^6 ft^2. Consequently, the flow into the wash is computed to be about 1.3×10^4 ft^3/day (Table 5–2 and Figure 5–3).

5.2.3 Surface-Water Hydrology

Moenkopi Wash is the dominant natural surface-water feature in the area. Near the village of Moenkopi the drainage area of Moenkopi Wash is 1,629 square miles. A stream gauge (U.S.G.S. station 09401260) is 1.3 miles southeast of Moenkopi Village, 100 ft upstream from the bridge on State Highway 264. Streamflow measurements have been made there since July 1976. The records are described as poor; nevertheless, the mean discharge at this location is approximately 9 cubic feet per second. Surface-water sampling, mainly of seeps tributary to Moenkopi Wash, was performed under the UMTRA program during the mid-1980s. Figure 5–4 shows the sampling locations. Table 5–3 presents a summary of the concentrations detected for the contaminants of potential concern. There has never been any indication that contamination from the disposal cell has migrated to the wash. Table 4–9 indicates that four surface-water samples are presently planned to be collected annually from locations 759, 778, 965, and 969.

Table 5–3. Summary of COPC Concentrations

Site Code	Well No.	Analyte	Average Concentration (mg/L)	Minimum Concentration	Maximum Concentration	Number of Samples
TUB01	0759	Molybdenum	0.055	0.0027	0.23	5
TUB01	0759	Nitrate	0.996	0.982	1	4
TUB01	0759	Selenium	0.0041	0.0014	0.005	4
TUB01	0759	Strontium	1.37	0.994	1.6	4
TUB01	0759	Sulfate	352	199	490	4
TUB01	0759	Uranium	0.0066	0.003	0.012	4
TUB01	0763	Molybdenum	0.015	0.01	0.025	3
TUB01	0763	Nitrate	15.5	1	21	4
TUB01	0763	Selenium	0.0062	0.005	0.009	4
TUB01	0763	Strontium	0.717	0.7	0.75	3
TUB01	0763	Sulfate	88.8	22	160	4
TUB01	0763	Uranium	0.0047	0.003	0.006	3
TUB01	0778	Molybdenum	0.066	0.0025	0.3	5
TUB01	0778	Nitrate	0.982	0.929	1	4
TUB01	0778	Selenium	0.0039	0.00071	0.005	4
TUB01	0778	Strontium	1.32	1.02	1.5	4
TUB01	0778	Sulfate	306	192	380	4
TUB01	0778	Uranium	0.0058	0.003	0.011	4
TUB01	0779	Molybdenum	0.01	0.01	0.01	2
TUB01	0779	Nitrate	7	7	7	2
TUB01	0779	Selenium	0.005	0.005	0.005	2
TUB01	0779	Strontium	0.205	0.2	0.21	2
TUB01	0779	Sulfate	12	11	13	2
TUB01	0779	Uranium	0.007	0.007	0.007	1
TUB01	0780	Molybdenum	0.01	0.01	0.01	1
TUB01	0780	Selenium	0.005	0.005	0.005	1
TUB01	0780	Strontium	0.49	0.49	0.49	1
TUB01	0780	Uranium	0.002	0.002	0.002	1
TUB01	0960	Molybdenum	0.025	0.025	0.025	1
TUB01	0960	Nitrate	18	18	18	1
TUB01	0960	Selenium	0.026	0.026	0.026	1
TUB01	0960	Sulfate	244	244	244	1
TUB01	0960	Uranium	0.039	0.039	0.039	1
TUB01	0962	Molybdenum	0.025	0.025	0.025	1
TUB01	0962	Nitrate	4	4	4	1
TUB01	0962	Selenium	0.005	0.005	0.005	1
TUB01	0962	Sulfate	90	90	90	1
TUB01	0962	Uranium	0.008	0.008	0.008	1
TUB01	0963	Molybdenum	0.025	0.025	0.025	1
TUB01	0963	Nitrate	1	1	1	1
TUB01	0963	Selenium	0.005	0.005	0.005	1
TUB01	0963	Sulfate	69	69	69	1

Table 5–3 (continued). Summary of COPC Concentrations

Site Code	Well No.	Analyte	Average Concentration (mg/L)	Minimum Concentration	Maximum Concentration	Number of Samples
TUB01	0963	Uranium	0.006	0.006	0.006	1
TUB01	0964	Molybdenum	0.025	0.025	0.025	1
TUB01	0964	Nitrate	2	2	2	1
TUB01	0964	Selenium	0.01	0.01	0.01	1
TUB01	0964	Sulfate	337	337	337	1
TUB01	0964	Uranium	0.029	0.029	0.029	1
TUB01	0965	Molybdenum	0.066	0.003	0.17	3
TUB01	0965	Nitrate	1.02	1	1.06	3
TUB01	0965	Selenium	0.0028	0.00059	0.005	2
TUB01	0965	Strontium	1.12	1.12	1.12	1
TUB01	0965	Sulfate	225	207	258	3
TUB01	0965	Uranium	0.0034	0.0031	0.004	3
TUB01	0966	Molybdenum	0.025	0.025	0.025	1
TUB01	0966	Nitrate	1	1	1	1
TUB01	0966	Selenium	0.005	0.005	0.005	1
TUB01	0966	Sulfate	9	9	9	1
TUB01	0966	Uranium	0.001	0.001	0.001	1
TUB01	0967	Molybdenum	0.025	0.025	0.025	1
TUB01	0967	Nitrate	2	2	2	1
TUB01	0967	Selenium	0.005	0.005	0.005	1
TUB01	0967	Sulfate	7	7	7	1
TUB01	0967	Uranium	0.001	0.001	0.001	1
TUB01	0969	Molybdenum	0.0427	0.0028	0.14	4
TUB01	0969	Nitrate	1.08	1	1.16	4
TUB01	0969	Selenium	0.0021	0.00053	0.005	3
TUB01	0969	Strontium	1.08	1.07	1.08	2
TUB01	0969	Sulfate	331	200	580	4
TUB01	0969	Uranium	0.0055	0.0033	0.011	4

5.3 Geochemistry of the N-Aquifer Near the Tuba City UMTRA Site

5.3.1 Background Ground-Water Quality

Well 901 is about 2,000 ft upgradient of the disposal cell and is completed in the upper portion of the Navajo Sandstone. Analyte concentrations in well 901 have remained relatively constant over the last 10 years. Contaminant concentrations in this well for the January 1997 sampling are provided in Table 5–4 and are used to represent background concentrations.

Table 5–4. Background Concentrations (January 1997 sampling of well 901)

COPC	Concentration (mg/L)	COPC	Concentration (mg/L)
Cd	< 0.001	Sr	0.313
Mo	< 0.001	SO_4	19.0
NO_3	12.8	U	0.0020
Se	0.0021		

Background water is low in dissolved solids and COPCs. The high quality is suitable for all domestic uses.

5.3.2 Tailings Pore Fluids (Lysimeter Data)

In April 1986, samples of tailings pore fluids were collected from lysimeters. COPC concentrations and pH values from that data set are listed in Table 5–5. Figure 5–5 presents the locations where lysimeter samples were collected. These data provide an indication of the chemical nature of contaminated water that migrated downward into the underlying Navajo Sandstone before remediation of the disposal cell in 1990. The fluids were acidic (pH as low as 2.43) and high in COPCs. Concentrations of uranium ranged up to 54.5 mg/L, over 50 times the highest value currently detected in the ground water. Sulfate concentrations ranged up to 32,800 mg/L, over four times the highest concentration currently detected in the ground water. Molybdenum concentrations ranged up to 5.8 mg/L, over ten times the highest concentration currently detected in the ground water. Selenium concentrations were below detection limits, although some selenium contamination is currently present in ground water. No cadmium or strontium concentrations were measured, and only one value of nitrate is available.

Table 5–5. COPC[a] Concentrations in Tailings Pore Fluids, April 1986

Lysimeter	pH	Mo (mg/L)	NO_3 (mg/L)	Se (mg/L)	SO_4 (mg/L)	U (mg/L)	NH_3(mg/L)
0661	6.00	0.19	NA	< 0.005	NA	0.925	220
0664	3.96	0.10	NA	< 0.005	15,100	43.3	0.1
0666	6.50	4.46	NA	< 0.005	4,200	30.7	0.1
0673	7.17	NA	NA	< 0.005	NA	2.53	NA
0676	6.10	NA	NA	< 0.005	NA	0.711	NA
0677	3.63	NA	NA	< 0.005	15,800	4.85	NA
0679	3.07	NA	NA	< 0.005	NA	2.17	NA
0680	2.90	NA	NA	< 0.005	15,600	7.41	0.1
0682	2.43	5.80	5.8	NA	32,800	54.5	110

[a]No lysimeter data are available for Cd or Sr.
NA = not analyzed.

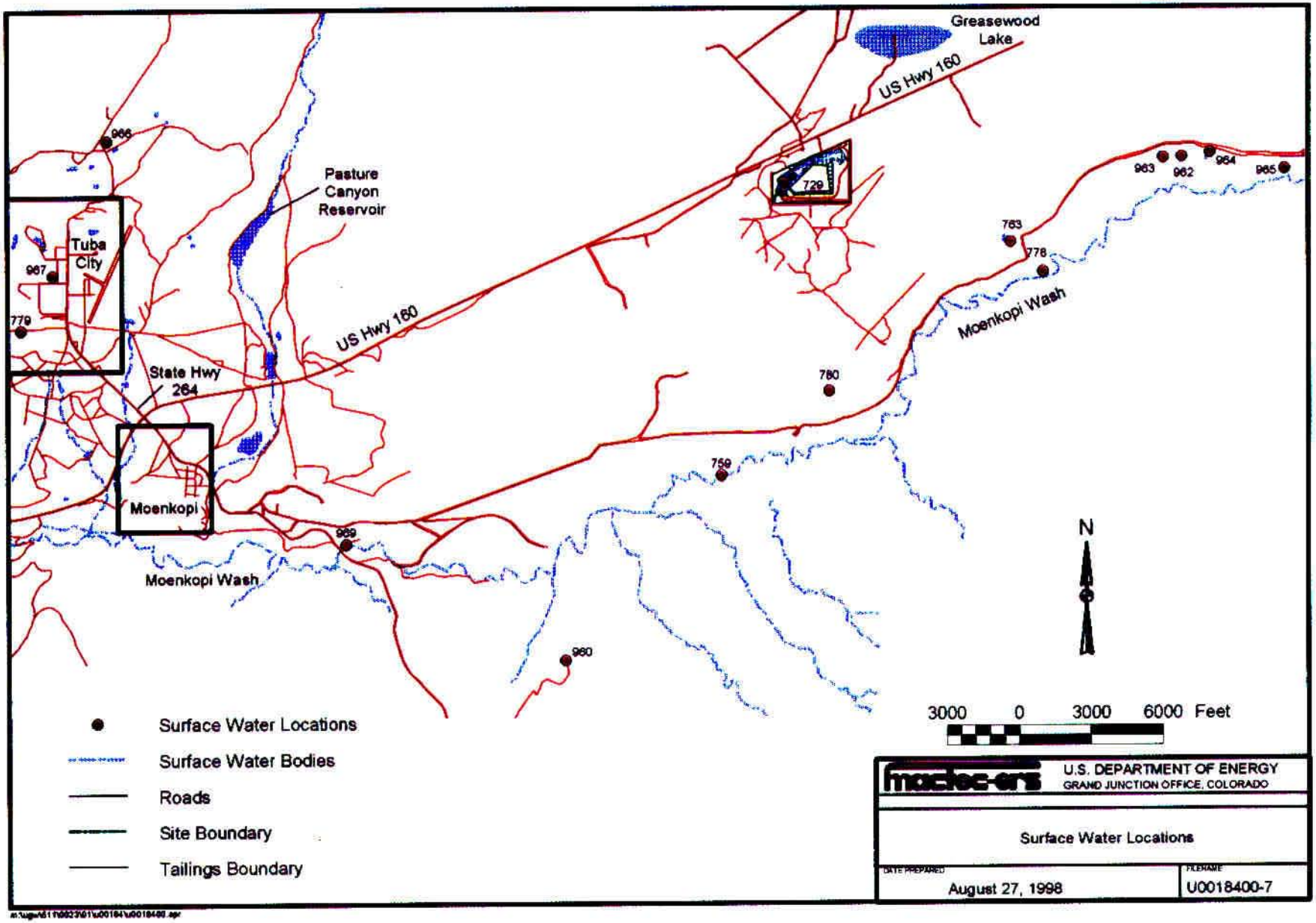


Figure 5–4. Surface Water Locations

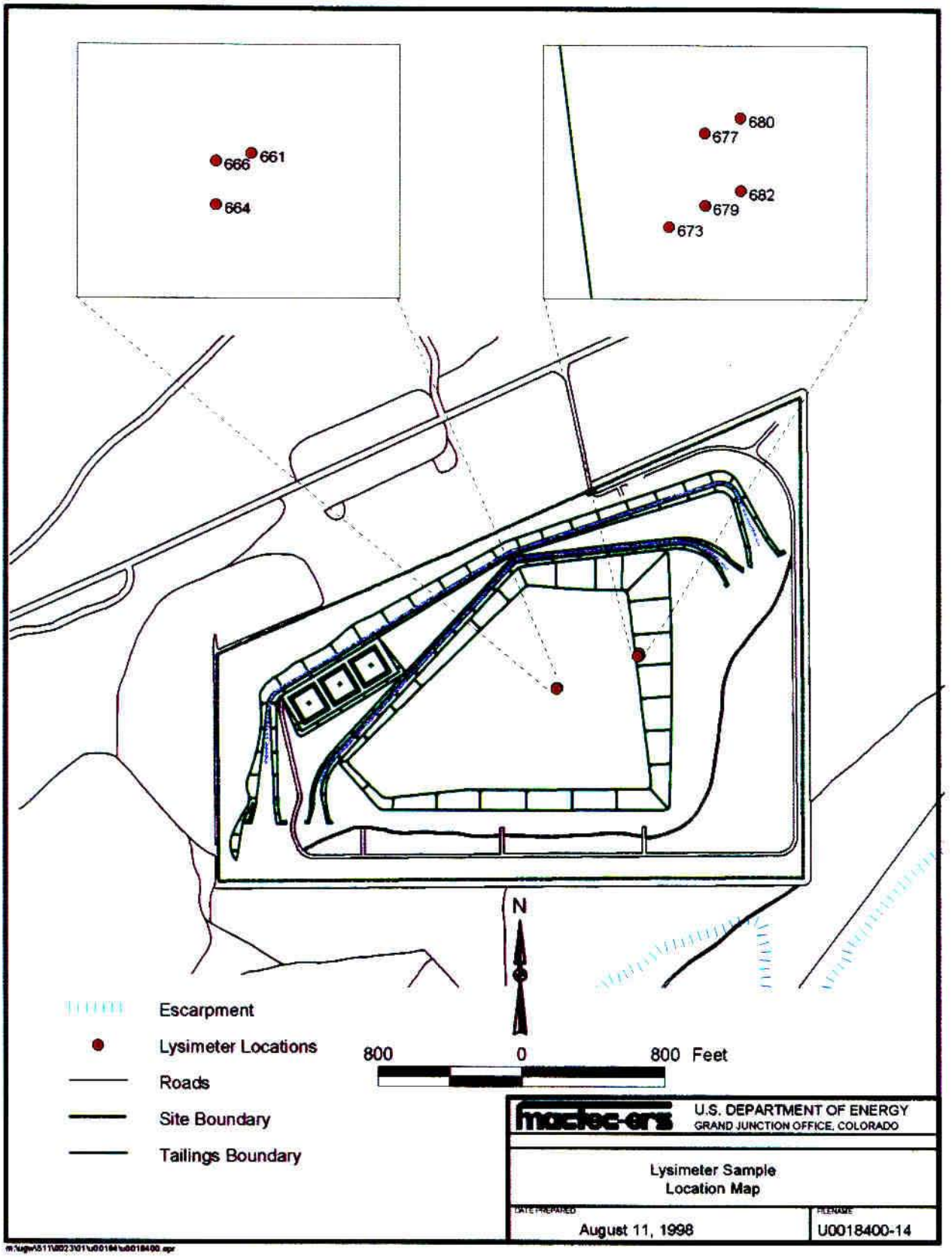


Figure 5–5. Lysimeter Sample Location Map

These analyses indicate that an acidic, metal-rich brine migrated into the underlying ground water from the tailings impoundments.

5.3.3 Areal Distribution of COPCs in Ground Water

The most recent ground-water data were used to examine the distribution of COPCs in the Navajo Sandstone. Locations of the wells used and the sampling dates are provided in Figure 5–6. Most data are from 1997; no data before 1995 were included. The COPCs are Mo, NO_3, Se, Sr, SO_4, and U (DOE 1995b). Figures 5–7 through 5–12 portray contaminant distributions as graduated symbol diagrams with selected contouring. In some cases, closely spaced wells have significantly different concentrations, probably due to sampling at different depths. In those cases, the highest concentrations were used for contouring. Well 941, which often has anomalously low concentrations, was not included in the contouring.

Molybdenum

Molybdenum concentration in a sample from well 939 exceeded the 0.10 mg/L UMTRA MCL in December 1996 by more than a factor of five (Figure 5–7). No other well samples exceed the MCL. Molybdenum concentrations in well 939 were high in samples collected during three sampling events; thus, analytical error is unlikely.

Nitrate

Nitrate forms a plume extending southwest from the disposal cell with concentrations as high as 4,050 mg/L (Figure 5–8). The 44 mg/L contour forms a plume extending approximately 2,000 ft from the southwest corner of the disposal cell. The elevated nitrate concentration of 1,030 mg/L detected in a sample from well 944 indicates that the plume has also migrated east of the disposal cell.

Selenium

Selenium forms a plume with an orientation similar to the nitrate plume. It extends southwest from the disposal cell with concentrations as high as 0.188 mg/L (Figure 5–9). Concentrations exceed the 0.01 mg/L UMTRA MCL over a distance of about 1,000 ft from the southwest corner of the disposal cell. The plume also extends to the east, as indicated by the elevated concentration of 0.0466 mg/L in well 944.

Strontium

Strontium forms a plume extending southwest and east from the disposal cell (Figure 5–10). The maximum concentration of 11.5 mg/L was detected in a sample from well 934. The strontium plume is defined by the 2 mg/L contour and extends about 1,500 ft from the southwest corner of the disposal cell. There is no UMTRA MCL for strontium.

Sulfate

As with most other COPCs, sulfate forms a plume extending southwest and east from the disposal cell (Figure 5–11). Concentrations reach maximums of 8,400 mg/L in the southwest portion of the plume and 1,760 mg/L in the eastern extension. The sulfate plume is defined by the 500 mg/L contour and extends about 2,000 ft from the southwest corner of the disposal cell.

Uranium

Uranium forms a plume extending southwest and east from the disposal cell, similar to plumes of the other COPCs (Figure 5–12). In contrast to the other COPCs, the maximum uranium concentration (1.33 mg/L) was detected in the eastern rather than the southwestern portion of the plume. The 0.044 mg/L plume extends about 1,200 ft from the southwest corner of the disposal cell.

5.3.4 Vertical Distribution of Contamination in Ground Water

Vertical distributions of U, NO_3, and SO_4 for the most recent samples from all monitoring wells are shown in Figure 5–13. Contamination is confined to the Navajo Sandstone as indicated by the low COPC concentrations in six wells completed in the intertonguing interval in the plume area (Table 5–6).

In a general sense, contaminant concentrations within the Navajo Sandstone appear to decrease with depth. Two clusters of closely spaced wells have screened intervals in both the upper and lower portions of the Navajo Sandstone (Figure 5–14). The vertical gradation is apparent between the two wells in cluster 1, where U, NO_3, and SO_4 concentrations are more than 10 times higher in the upper zone. Although the deep well (912) in cluster 2 has significantly lower concentrations of the three contaminants, those results are not indicative because that well is screened partly in the underlying intertonguing interval.

Table 5–6. COPC Concentrations in the Most Recent Samples Collected from Wells Completed Solely in the Intertonguing Interval

Well Number	Sample Data	Cd (mg/L)	Mo (mg/L)	NO_3 (mg/L)	Se (mg/L)	Sr (mg/L)	SO_4 (mg/L)	U (mg/L)
913	1/97	< 0.001	< 0.001	13.3	0.0010	0.803	9.36	0.0016
915	4/97	< 0.001	< 0.001	14.3	0.0022	0.549	17.0	0.0015
916	4/97	< 0.001	< 0.001	11.7	0.0015	0.608	12.1	0.0011
917	4/97	< 0.001	< 0.001	16.3	0.0016	0.343	13.4	0.0013
920	4/97	< 0.001	< 0.001	15.0	0.0015	0.990	12.5	0.0019
921	4/97	< 0.001	< 0.001	11.2	0.0011	0.746	8.38	0.0047
UMTRA MCL		0.01	0.10	44	0.01	NS	NS	0.044

NS—No UMTRA standard

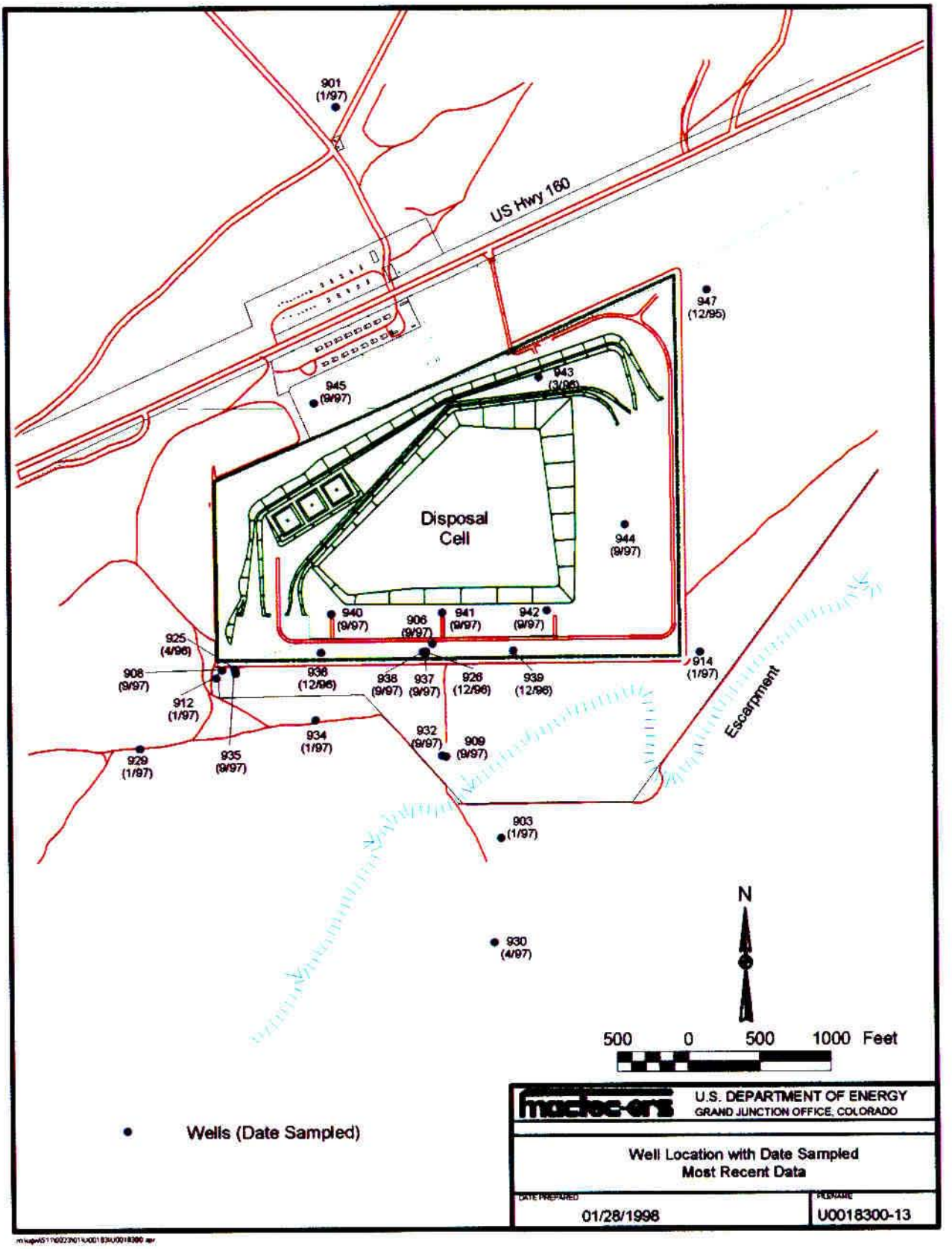


Figure 5–6. Well Locations with Date Sampled (most recent data)

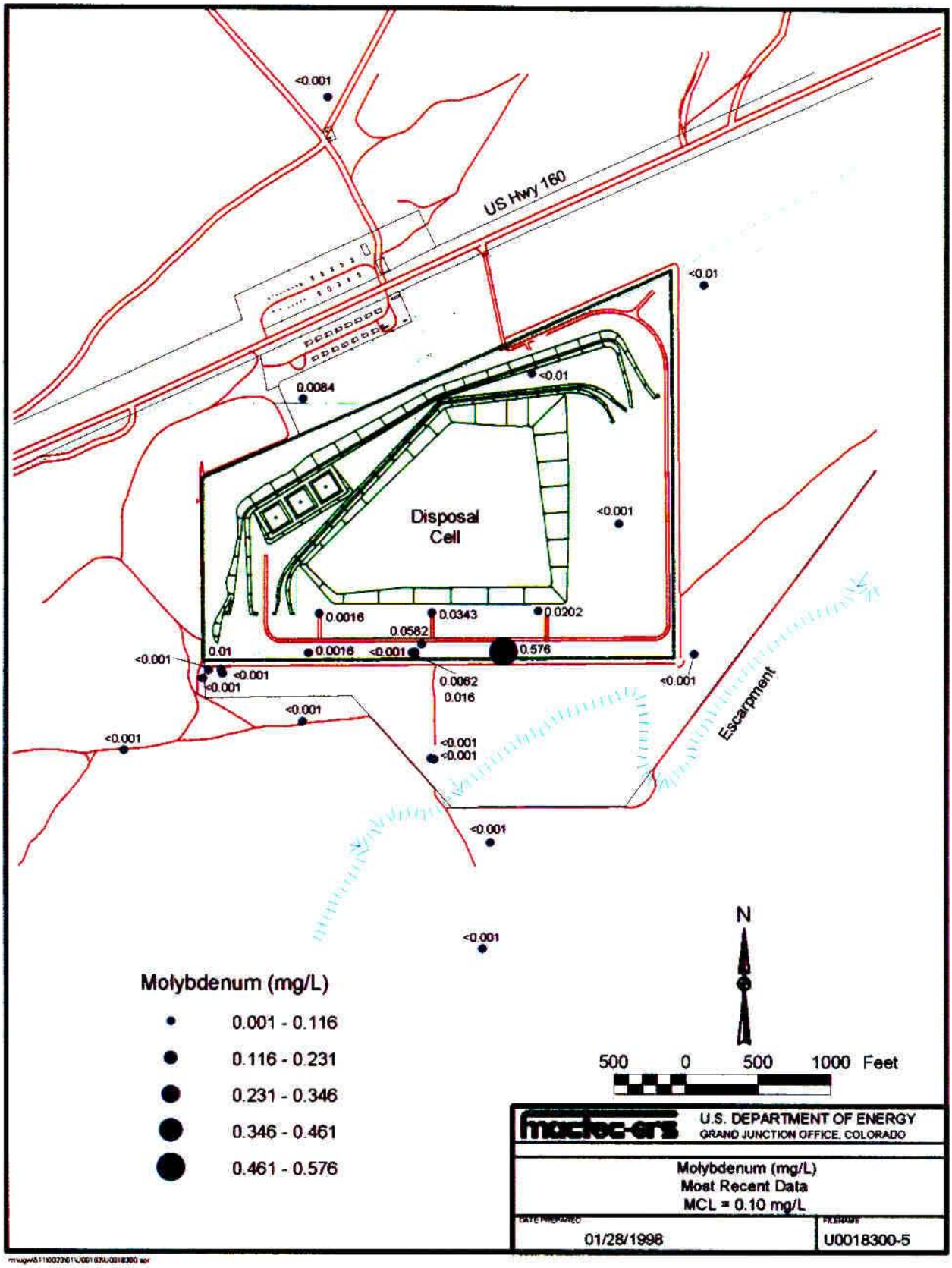


Figure 5–7. Molybdenum Concentrations, Most Recent Data

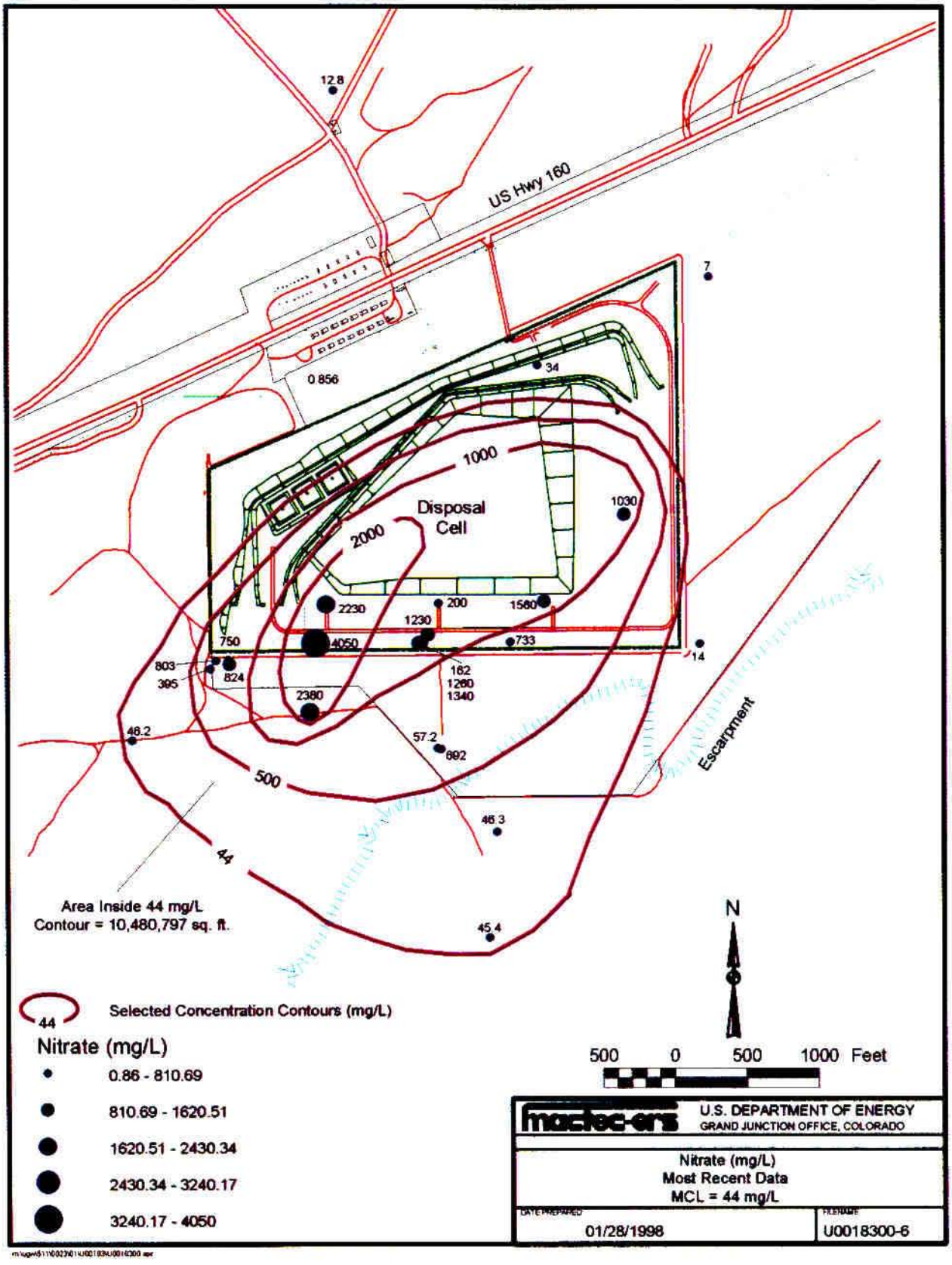


Figure 5–8. Nitrate Plume, Most Recent Data

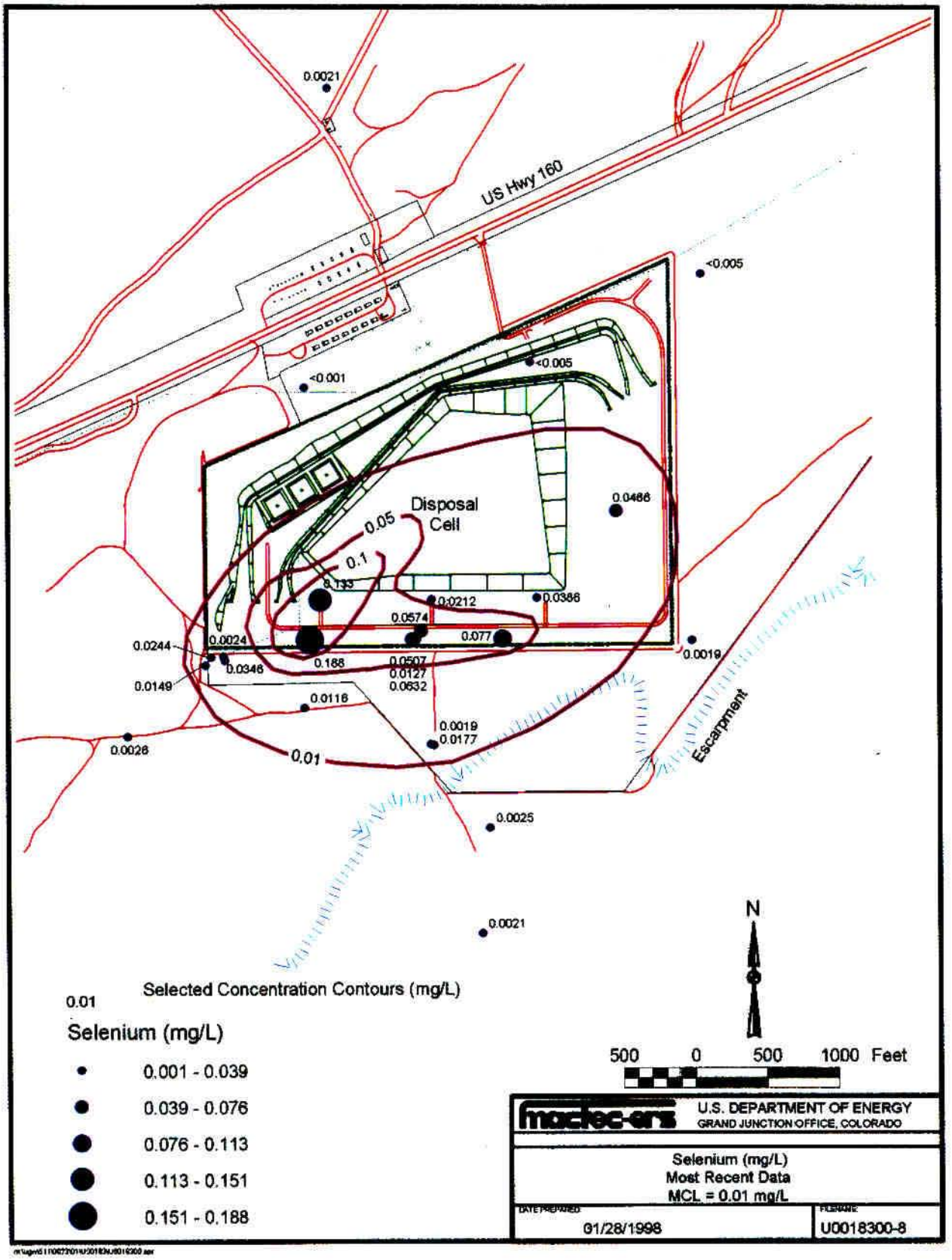


Figure 5–9. Selenium Plume, Most Recent Data

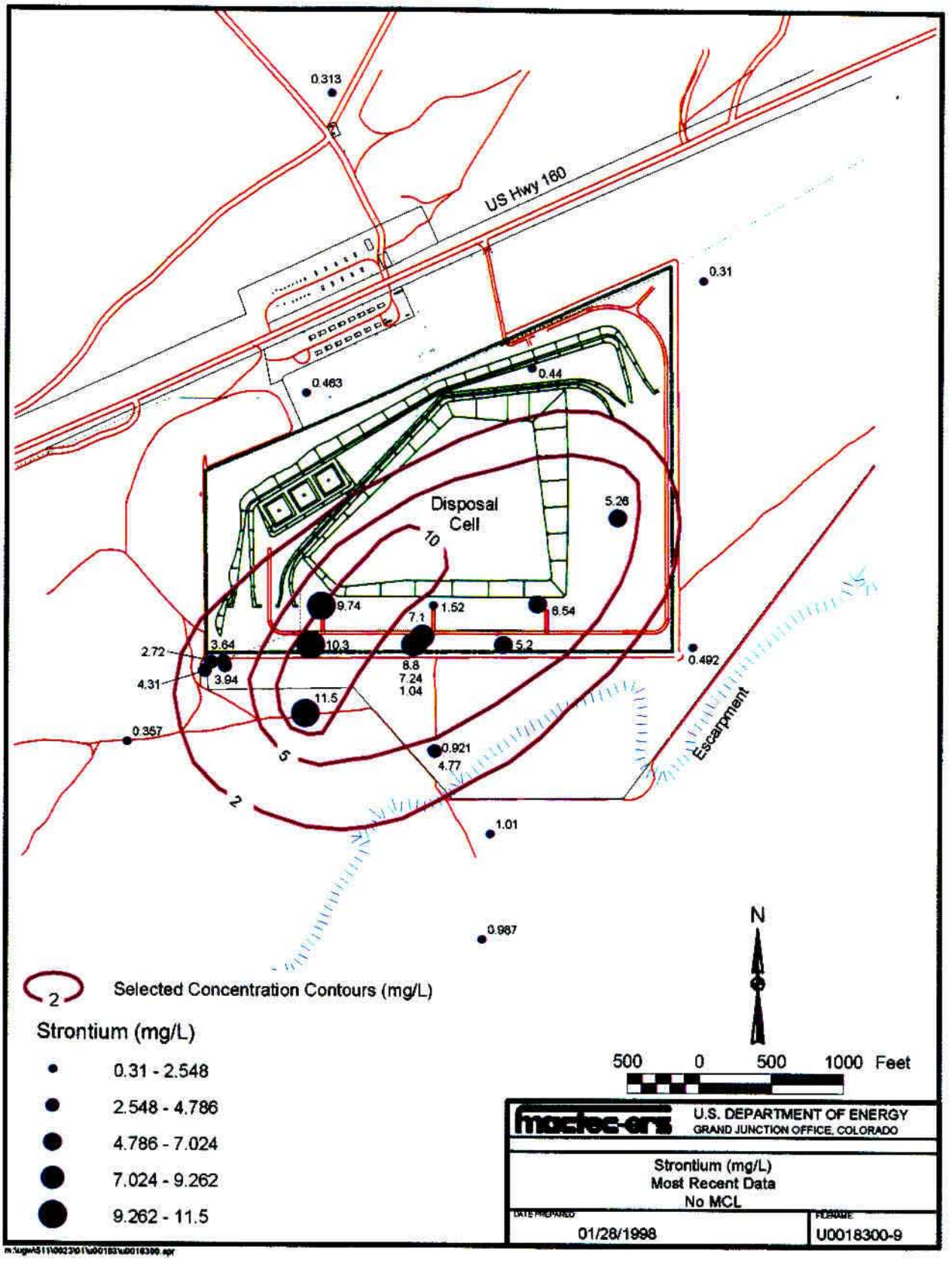


Figure 5–10. Strontium Plume, Most Recent Data

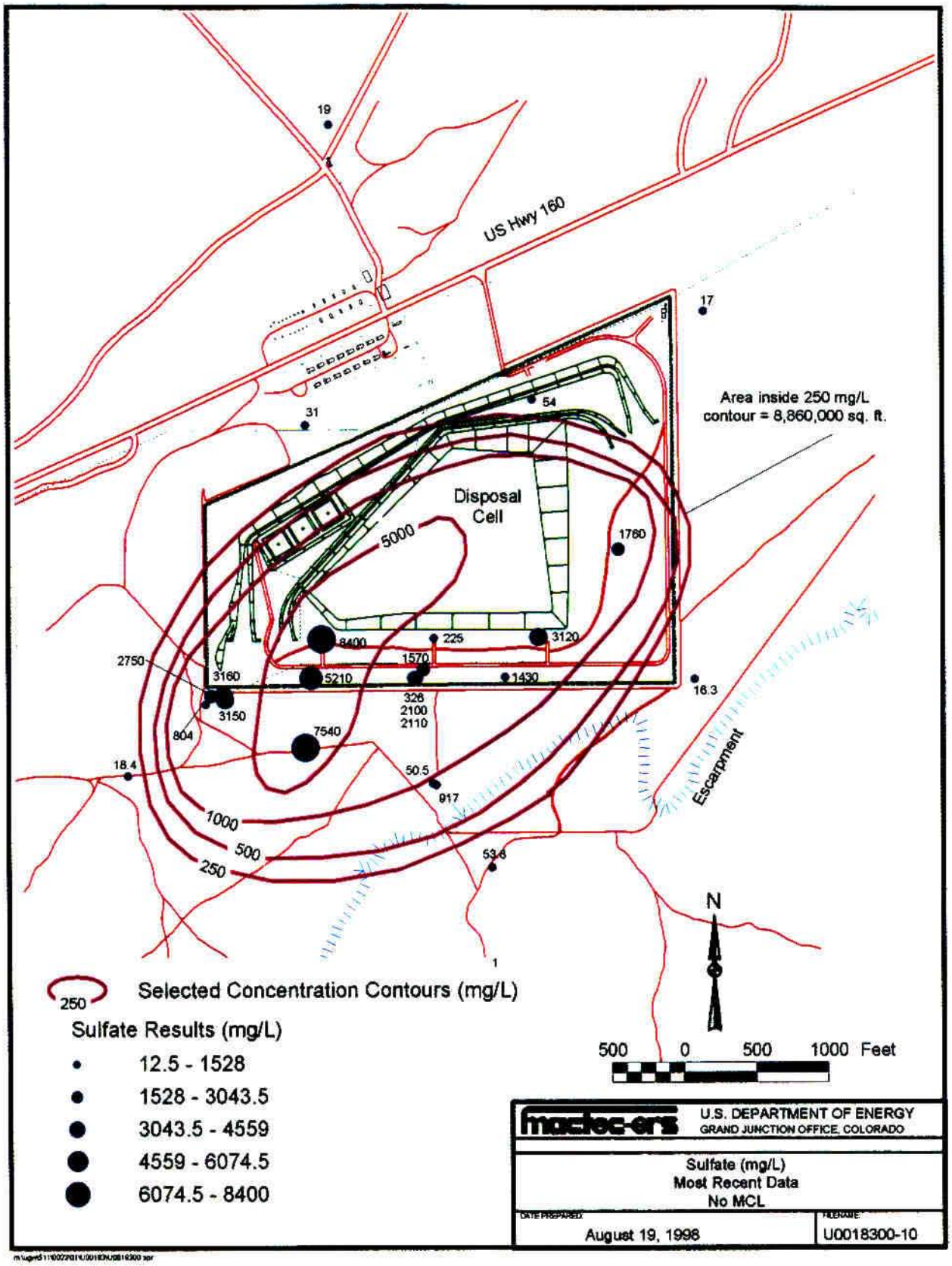


Figure 5–11. Sulfate Plume, Most Recent Data

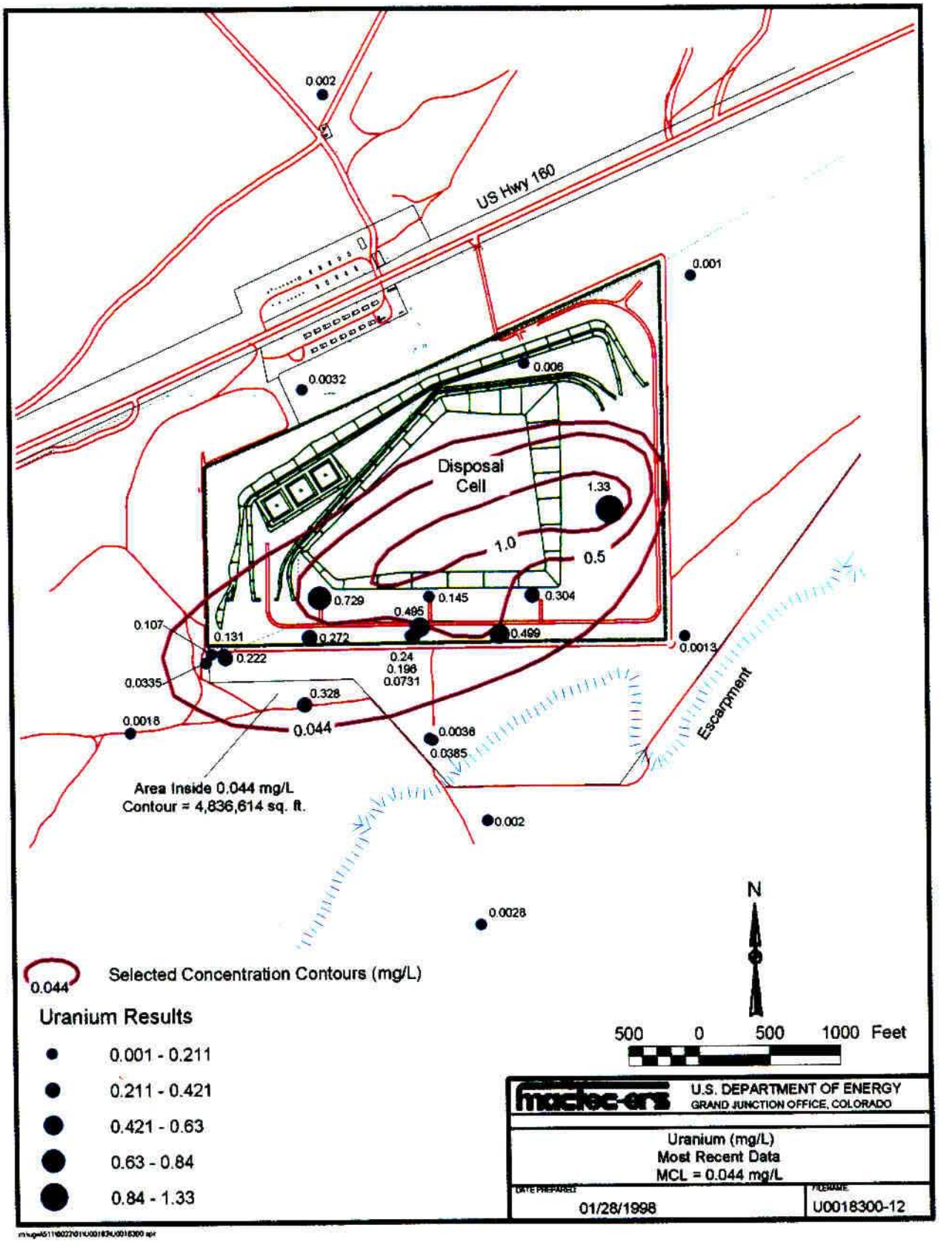


Figure 5–12. Uranium Plume, Most Recent Data

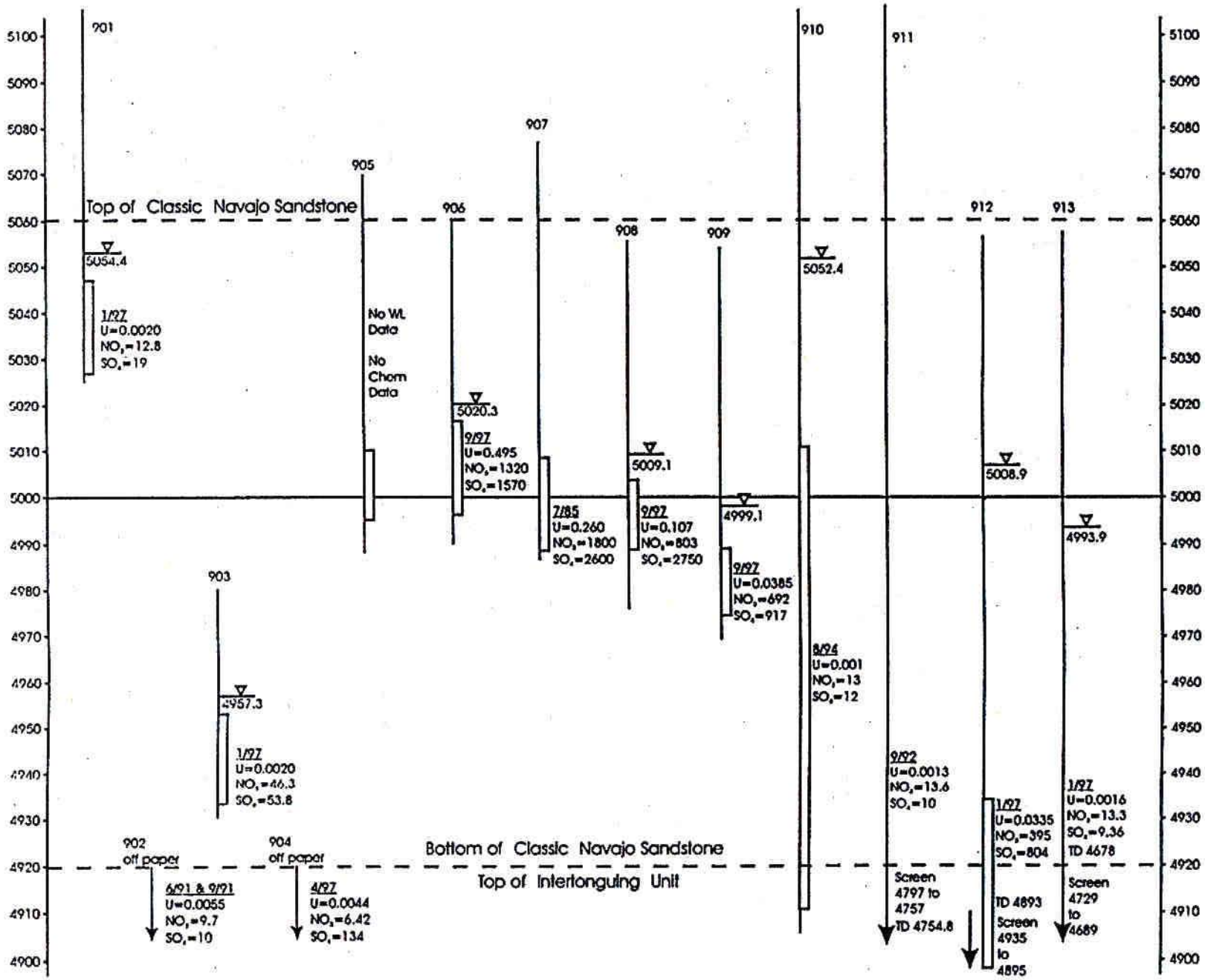


Figure 5–13. Water Levels, Well Screen Depths, and Most Recent Ground-Water Data

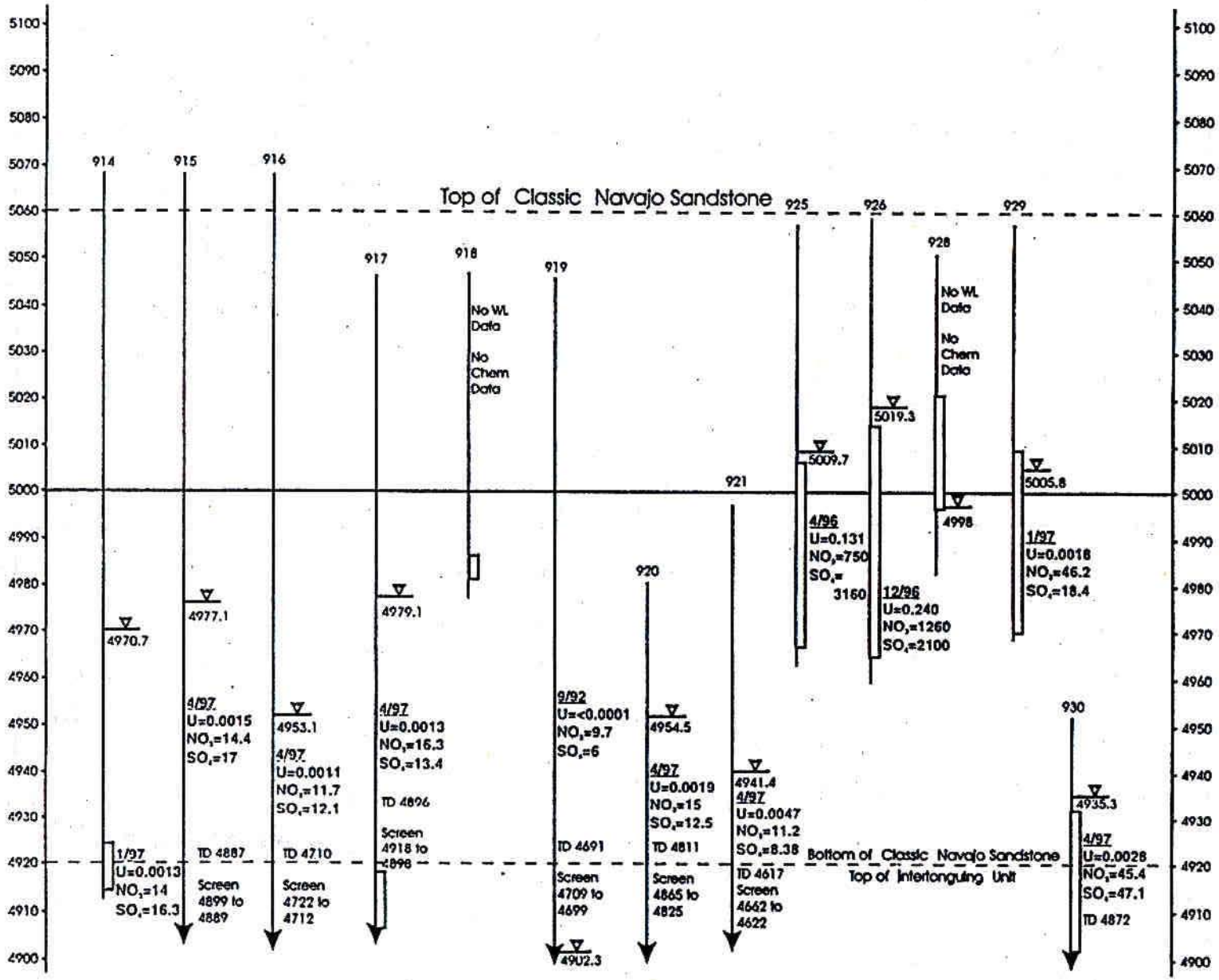


Figure 5–13 (continued). Water Levels, Well Screen Depths, and Most Recent Ground-Water Data

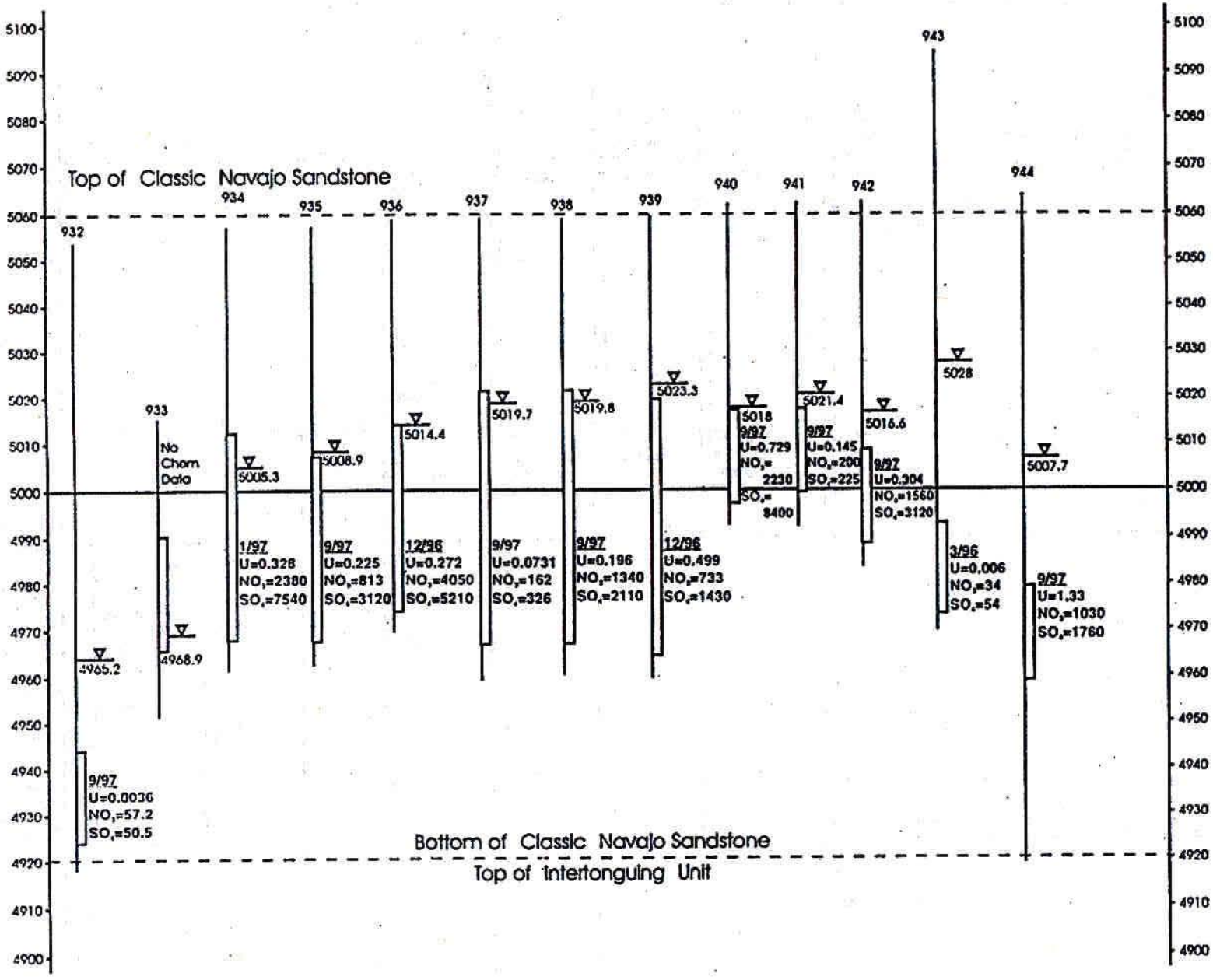


Figure 5–13 (continued). Water Levels, Well Screen Depths, and Most Recent Ground-Water Data

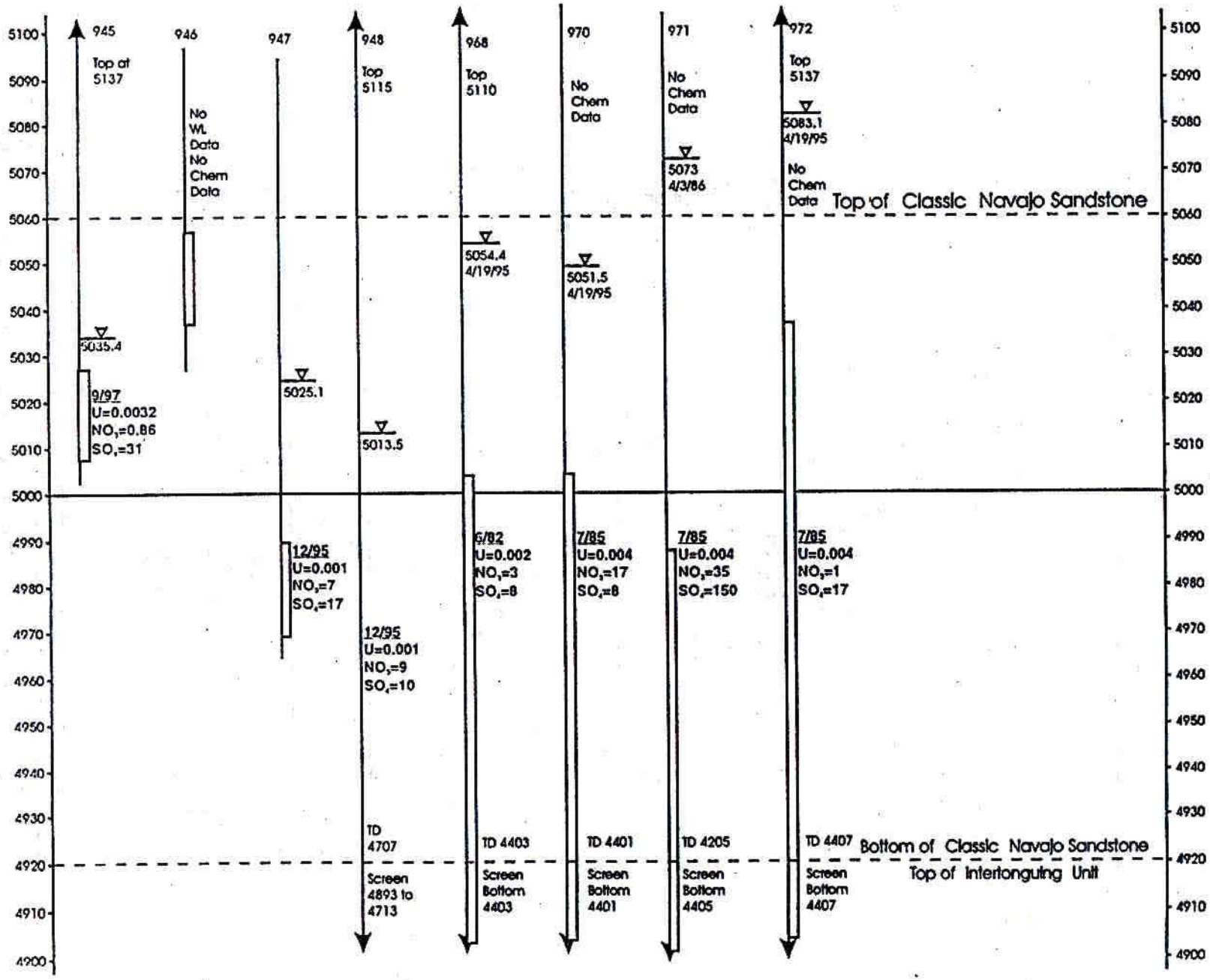


Figure 5–13 (continued). Water Levels, Well Screen Depths, and Most Recent Ground-Water Data

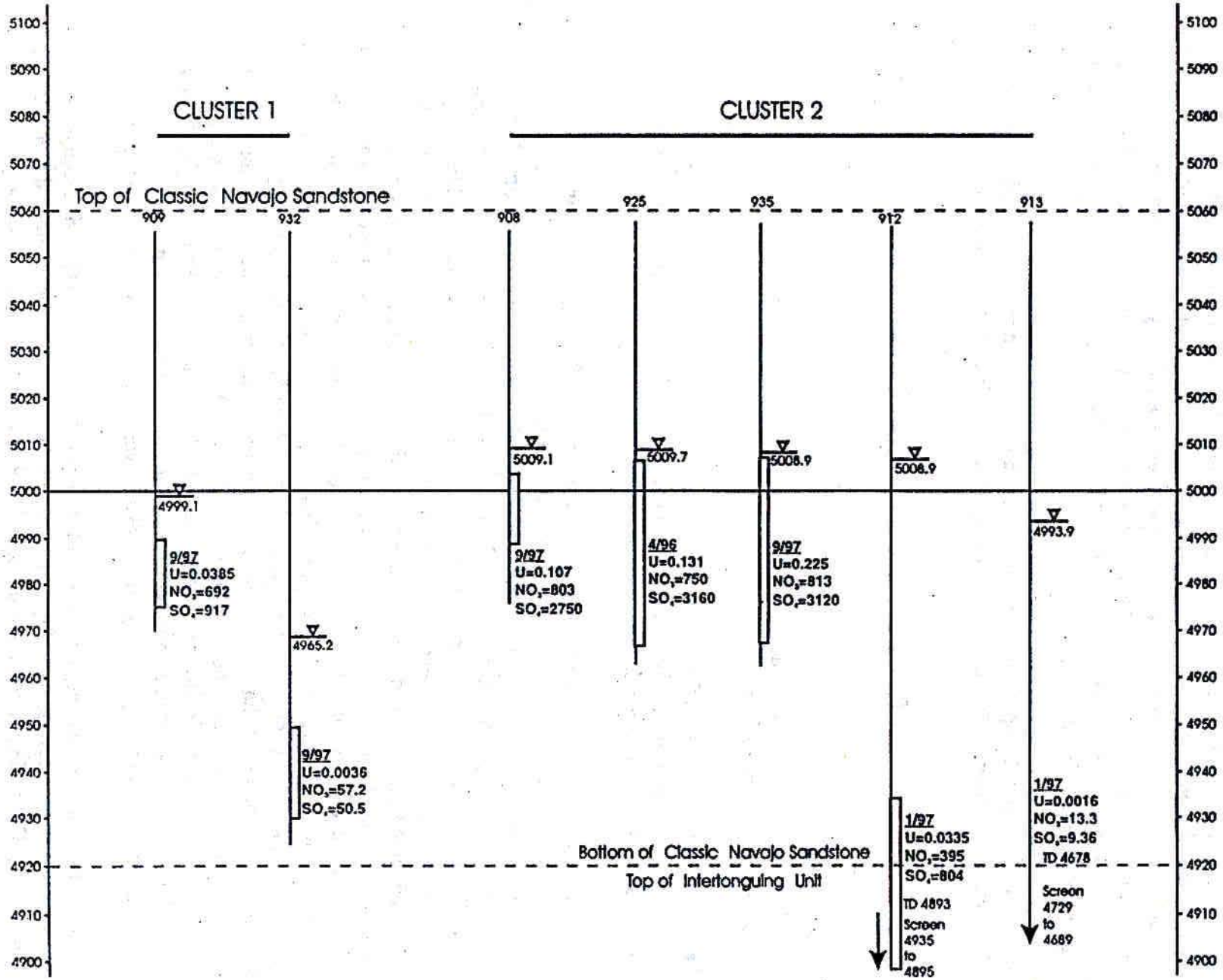


Figure 5–14. Vertical Variation of Uranium, Nitrate, and Sulfate Concentrations in Two Closely Spaced Well Clusters

5.3.5 Mineral Saturation

Precipitation and dissolution of minerals in ground water can have significant effects on the rate and direction of ground-water flow. Some precipitated minerals incorporate contaminants in their crystalline structure. Mineral precipitation has many implications to ground-water remediation. For example, if mineral concentrations are close to saturation or are oversaturated, they may precipitate when temperature and pressure change during pumping. This can lead to scaling and costly rehabilitation of pipe and well bores.

Minerals in an aquifer that precipitate and dissolve rapidly will be close to equilibrium with the ground water. A mineral at (or near) equilibrium can be described by the saturation index (SI):

$$SI = Log(\frac{IAP}{Keq})$$

where

IAP = ion activity product, and
Keq = equilibrium constant.

The SI is positive for systems that are oversaturated and negative for systems that are undersaturated with a mineral. Oversaturated systems can precipitate, whereas undersaturated systems can dissolve the minerals.

SIs for two minerals that often cause scaling problems, calcite ($CaCO_3$) and gypsum ($CaSO_4 \cdot 2H_2O$), were calculated from the most recent ground-water analyses using the geochemical speciation program PHREEQE (Parkhurst et al.1980). Because enthalpy data for many of the aqueous species are imprecisely known, the calculations were made for 25EC and were not corrected for measured ground-water temperatures. The calculations used the Debye-Hückel theory to correct for changes in ionic strength. The Debye-Hückel theory is applicable for solutions with ionic strengths of less than about 0.1 and all of the solutions have ionic strength less than this value. In addition, SIs calculated from ion concentrations measured in many ground waters that are in contact with calcite and gypsum vary somewhat from zero. The variation is due to several factors, including incomplete equilibrium, imprecise or incomplete analytical data, and imprecise thermodynamic data. Another factor that can affect SI is impure mineral phases which are common to geologic media. Despite these uncertainties, SIs can provide an indication of the trends in mineral precipitation and dissolution. Figures 5–15 and 5–16 display these trends. Because of the uncertainty in the computations, the absolute SI listed for a well may be slightly in error, however, the values of SI relative to each other are probably accurate. The interpretations that follow depend only on the relative values of SI and not their exact values.

Calcite is mostly undersaturated in background ground water at the Tuba City site. Calcite SIs are higher in a zone south and east of the disposal cell, indicating a tendency toward precipitation (Figure 5–15). This zone of oversaturation extends about 1,000 ft south of the disposal cell. Calcite SI values are lower in two wells (940 and 941) near the southern border of the disposal cell, suggesting that calcite may be dissolving beneath the cell.

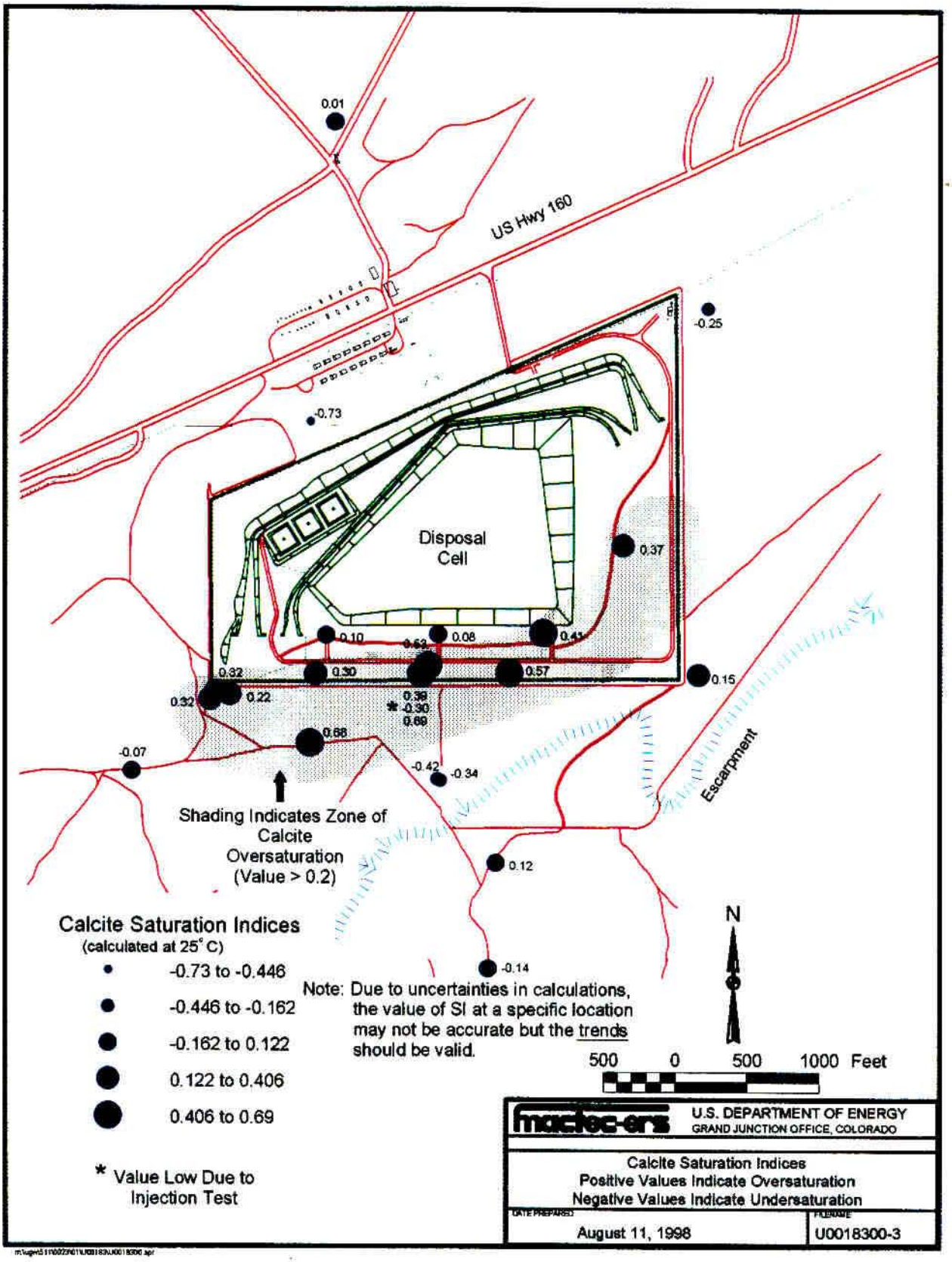


Figure 5–15. Calcite Saturation Indices

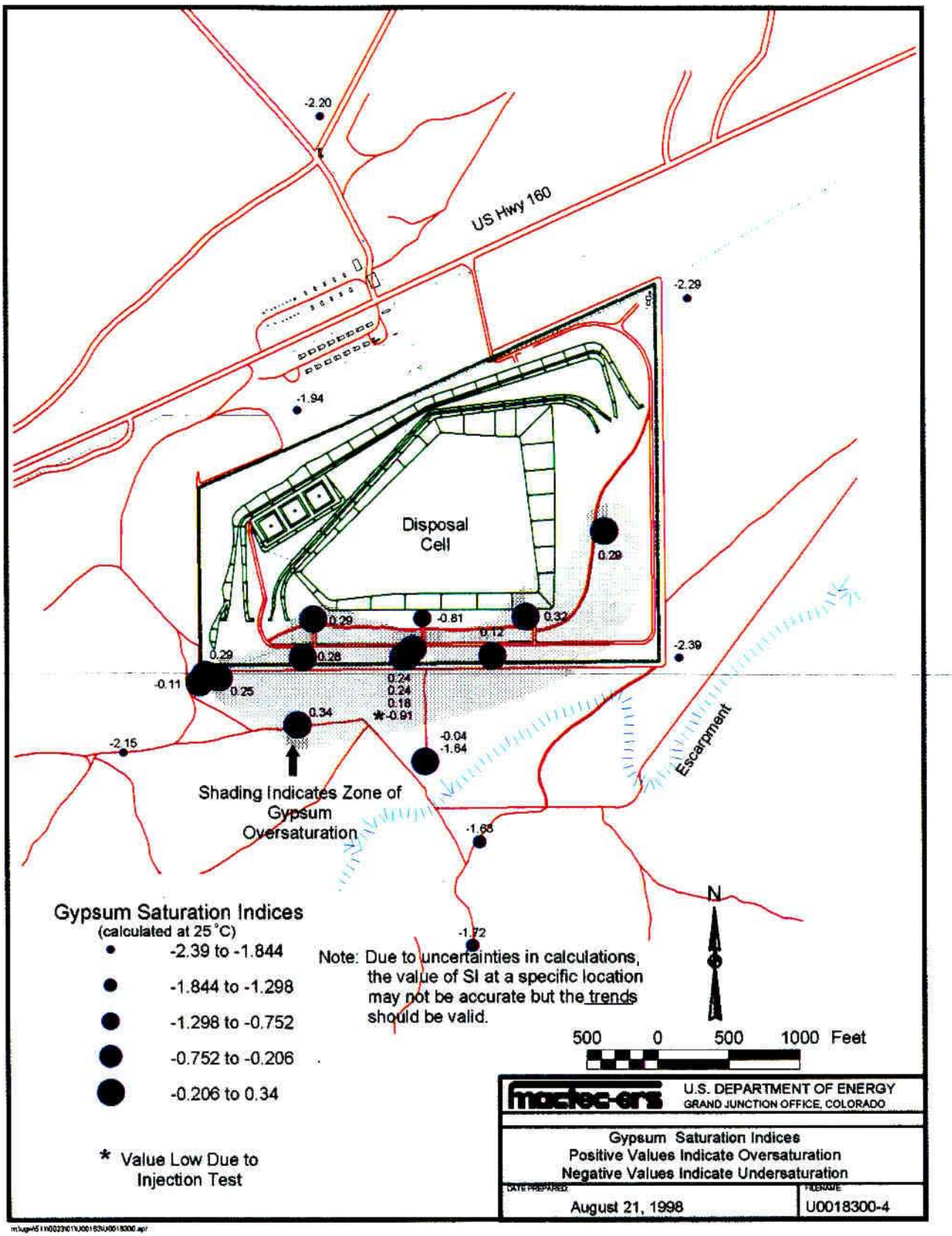


Figure 5–16. Gypsum Saturation Indices

Gypsum is also undersaturated in background ground water. High positive gypsum SIs in a zone south and east of the disposal cell indicate a tendency toward precipitation in that area (Figure 5–16). The oversaturated zone extends about 1,000 ft from the southern boundary of the cell. A white powdery substance scraped from downhole data loggers in wells 936, 939, and 942 was identified as gypsum, confirming that conditions exist for precipitation of this mineral.

5.3.6 Time Trends in COPC Distributions

Long-Term Trends

Wells 906, 908, and 909 are in the plume and are among the earliest wells for which ground-water data are available. These three wells are used to depict the long-term trends of contaminant concentrations near the cell. Well 906 is 300 ft due south of the disposal cell in a highly contaminated area of the plume. The hydrograph from well 906 shows a gradual decline in water levels from 1985 to 1992 due to dissipation of the ground water mound created during milling, which ceased in 1966 (Figure 5–17). The water level rose rapidly between September 1992 and August 1993, remained high until about August 1994, then began a decline that continues currently. Concentrations of several chemical species (SO_4, Se, Cl, Na, Mo, Mg, and U) in the ground water parallel the rise and fall of the hydrograph (Figure 5–17).

Well 908, which is about 1,000 ft from the southwest corner of the disposal cell, has had declining concentrations of sulfate since data collection began in 1984 (Figure 5–18). Uranium and nitrate concentrations have shown some fluctuations but have remained nearly constant over the same time period. Well 909, about 1,000 ft due south of the disposal cell, has had declining trends of sulfate, nitrate, and uranium concentrations since 1984 (Figure 5–19).

Well 903 is used to depict long term trends near the leading edge of the plume (figures not shown). Sulfate concentrations in well 903 have steadily increased from 20 to 30 mg/L in 1984 through 1988 to 50 to 60 mg/L currently. Nitrate concentrations have also increased from 10 to 25 mg/L in 1984 through 1988 to about 47 mg/L currently. These trends indicate that the plume is migrating through this area.

Concentration Trends of Uranium, Sulfate, and Nitrate for Wells Close to the Disposal Cell

Wells 940, 941, 942, and 944 are the closest to the disposal cell. Ground-water quality data from these wells are available only since December 1995. Uranium concentrations in wells 940 and 944 have been increasing, whereas those in wells 941 and 942 have remained nearly constant (Figure 5–20). Sulfate and nitrate concentrations in well 940 have risen significantly, whereas concentrations of those compounds in the other three wells have been nearly constant (Figure 5–20). Thus, it appears that the sulfate and nitrate concentrations are increasing in the southwest plume, but that the uranium concentration is increasing in the eastern plume. However, the question of whether the plume is expanding or contracting and at what rate cannot be answered with the available data.

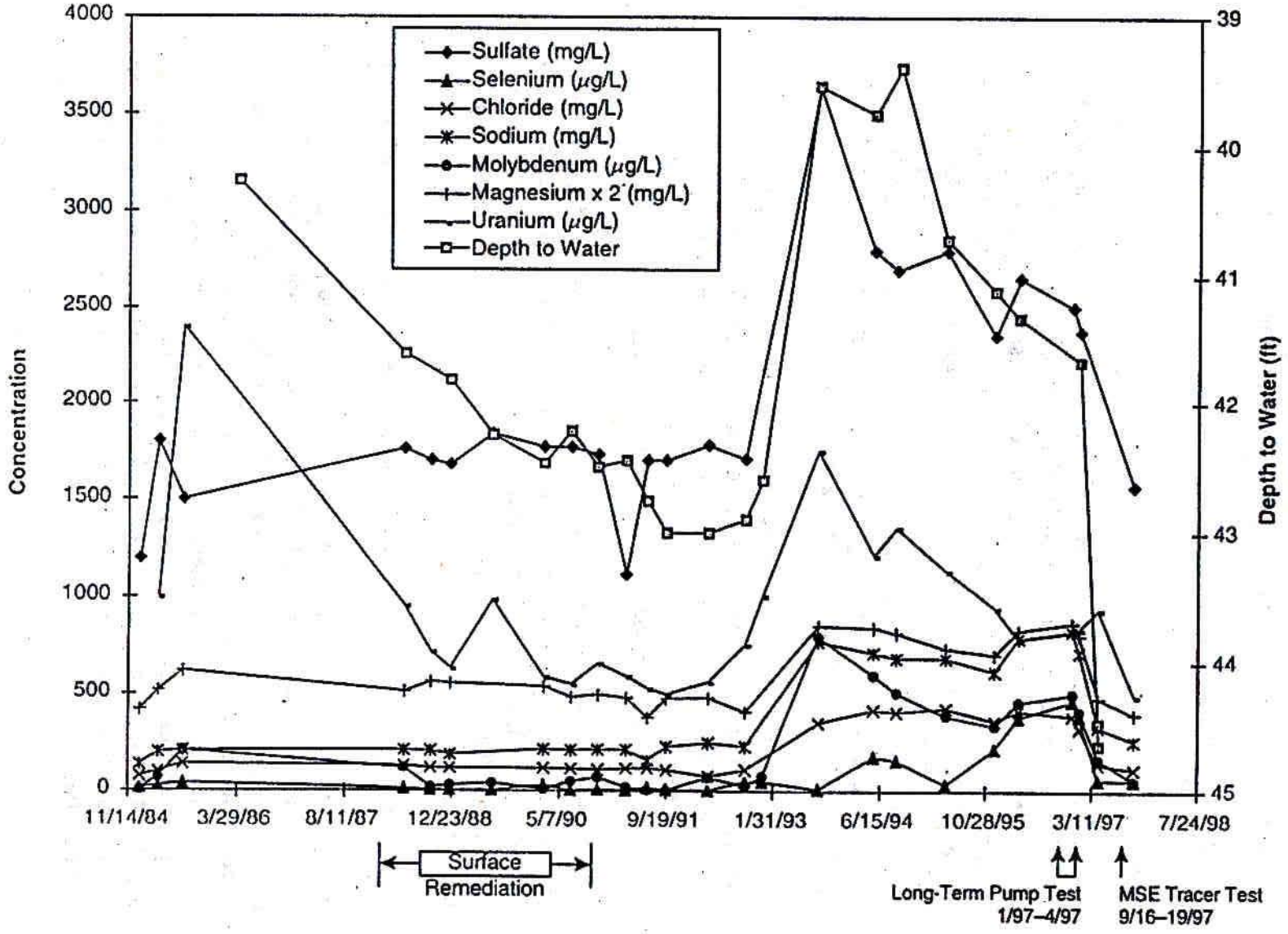


Figure 5–17. Time Trends in Well 906

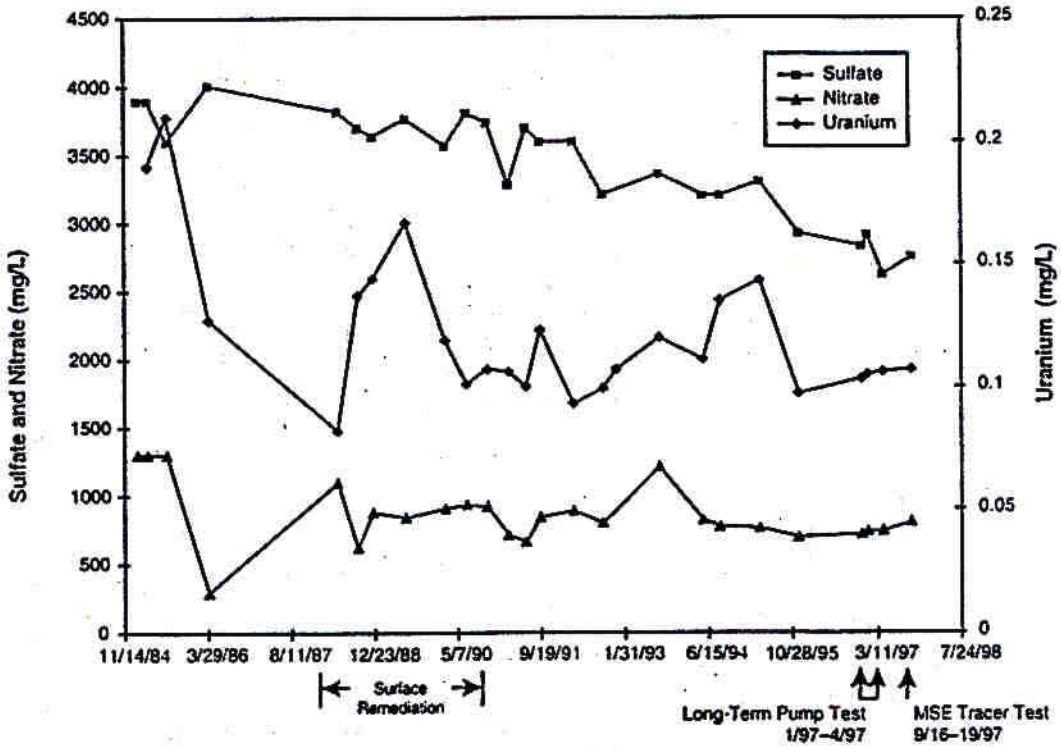


Figure 5–18. Time Trends in Well 908

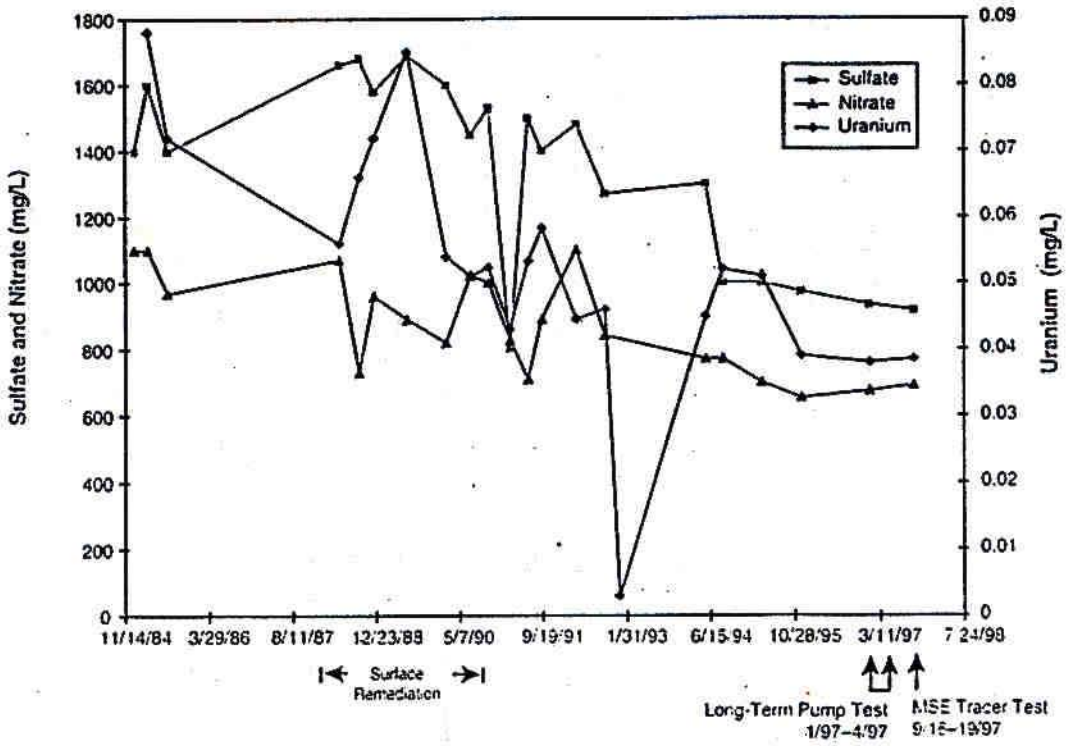
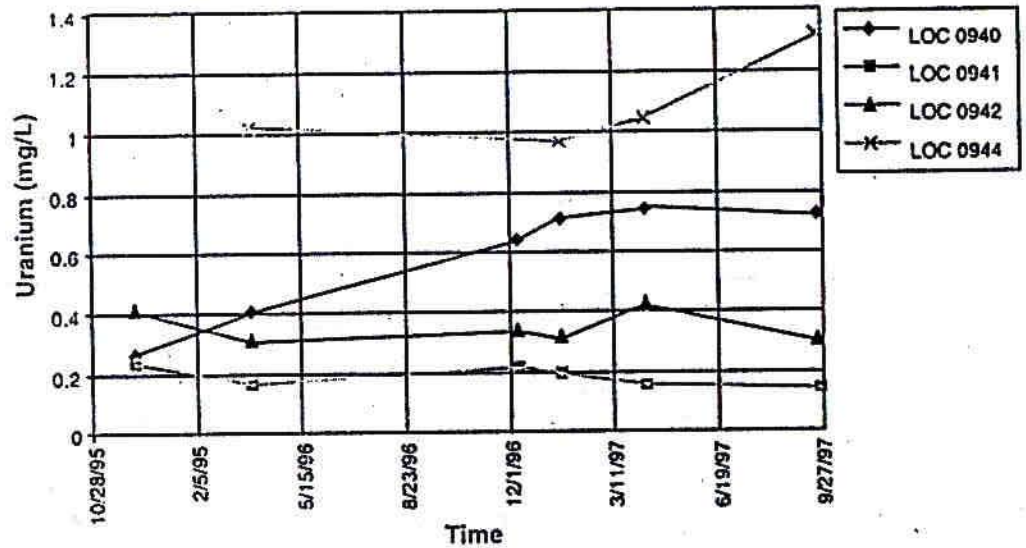
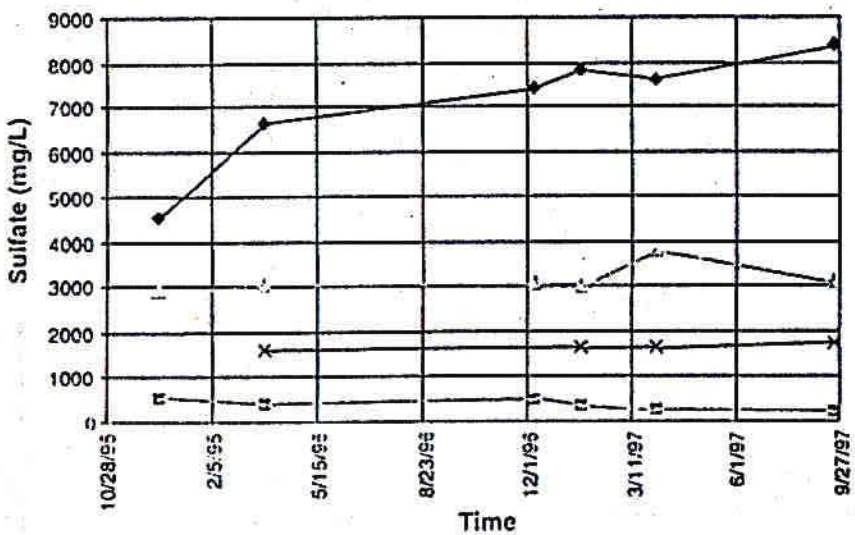


Figure 5–19. Time Trends in Well 909





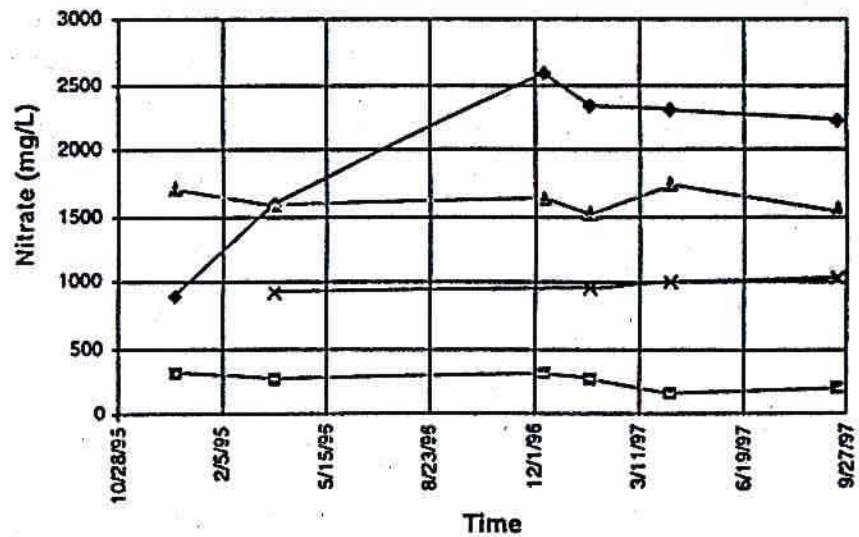


Figure 5–20. Time Trends for Some COPCs in Wells Close to the Disposal Cell

5.3.7 COPC Fate and Transport

Chemical mechanisms that are most likely to control fate and transport of the COPCs in the Navajo Sandstone, based on information from published literature, are summarized in this section. The aquifer is oxidized with respect to iron as indicated by the red to yellow coloration imparted to it by ferric oxyhydroxide minerals and lack of high (less than 0.5 mg/L) concentrations of dissolved iron. It is also oxidized with respect to the mineral uraninite as indicated by the relatively high (up to about 1 mg/L) concentrations of dissolved uranium. The discussion of fate and transport that follows assumes that the aquifer is also oxidized with respect to the remaining COPCs.

If the aquifer becomes reduced Mo, Se, and U may precipitate; sulfate may transform to sulfide; and nitrate may transform to nitrite or nitrogen. These redox sensitive processes will be enhanced by microbial activity.

Molybdenum

Molybdenum occurs naturally in a +4 and a +6 oxidation state, but the +6 state is most likely present in the Tuba City ground water because of the high oxidation potential. Dissolved molybdenum species are dominated by the molybdate anion (MoO_4^{2-}) in the pH range of interest. At low pH, $HMoO_4^-$ or $H_2MoO_4^0$ may become important. High concentrations of sodium and calcium, such as are present in ground water near the disposal cell, will complex to form Na and Ca molybdate species ($NaMoO_4^-$ and Ca MoO_4^0). No other aqueous complexes are likely to form.

Because of the low concentrations in ground water at the site, no molybdenum minerals are expected to form. Adsorption to Navajo Sandstone mineral grains, which are mostly quartz and feldspar, is expected to be relatively minor unless the grains have oxide or oxyhydroxide coatings. Molybdenum, however, is known to adsorb to ferric oxides in relatively high concentrations (Morrison and Spangler 1993). Molybdenum adsorption would increase in those portions of the sandstone that have higher concentrations of iron and manganese oxides. These areas usually have a red or red-brown coloration.

Nitrate

Nitrate (NO_3) does not complex with other ions under ground-water conditions. It will be transported without significant interaction with the rock matrix. If appropriate nitrate-reducing microbiota and nutrients are present, nitrate can undergo reduction to nitrogen gas (N_2). Significant denitrification is not expected to occur in the Navajo Sandstone without a suitable nutritional source such as acetate. Therefore, nitrate probably transports nearly conservatively through the aquifer. Concentrations decrease by mixing with other ground water and by dispersion. If the aquifer is within about 50 ft of the ground surface (as it is in the greasewood area), plants will remove nitrate from the ground water.

Selenium

Aqueous selenium occurs predominately as selenate (SeO_4^{2-}) or selenite (SeO_3^{2-}); selenate is probably favored under the oxidized conditions of the Navajo Sandstone. Concentrations of selenium are not high enough to cause precipitation of selenium minerals. Selenium can substitute for sulfur in sulfur-bearing minerals.

Selenium is not likely to adsorb appreciably to the mineral grains (mostly quartz and feldspar) in the Navajo Sandstone. Both selenite and selenate, however, will adsorb to ferric oxyhydroxides (Dzombak and Morel 1990). Selenate adsorption requires low pH and is not likely to be significant in the Navajo Sandstone. Thus, selenium is likely to remain in solution with concentration gradients developed mainly by advection and dispersion.

Strontium

Strontium exists in solution predominantly as uncomplexed Sr^{2+}. At the concentrations present in ground water at Tuba City, it is unlikely that strontium minerals will precipitate; however, Sr^{2+} can substitute for Ca^{2+} during precipitation of calcite or gypsum. Co-precipitation with those minerals is the most likely mechanism for strontium to partition to the aquifer solids.

Sulfate

In Tuba City ground water, dissolved sulfur occurs mainly as the unassociated sulfate ion (SO_4^{2-}). The only mechanism likely to partition significant amounts of sulfate into the solid phase is the precipitation of gypsum. The amount that precipitates is likely to be relatively minor compared to the high concentrations of sulfate in solution. Therefore, most of the concentration gradient is produced by mixing with other ground water and dispersion. Although sulfate can be chemically reduced by microbes to form sulfide minerals, there is no evidence of this process occurring in the N-aquifer.

Uranium

Most naturally occurring uranium is either in the uranyl (+6) or uranous (+4) oxidation state. The uranyl form is predominant in oxidized ground water. The uranyl ion forms strong aqueous complexes with carbonate, and uranyl dicarbonate [$UO_2(CO_3)_2^{2-}$] is the dominant uranium aqueous species at Tuba City.

Uranyl concentrations at the Tuba City site are too low to form uranium minerals. Uranous minerals would precipitate if the oxidation state were lower; however, such reduced conditions do not currently exist at the Tuba City site. Adsorption of uranyl to mineral grains in the Navajo Sandstone (mostly quartz and feldspar) is likely to be insignificant. However, uranyl is known to adsorb to ferric oxyhydroxide in relatively high concentrations (Morrison et al.1995). It is likely that adsorption to ferric or manganese minerals is the principal mode of uranium retardation at the site. The high concentration of carbonate in the ground water near the disposal cell favors the partitioning of uranium to the dissolved phase. In distal portions of the plume, where dissolved carbonate concentrations are lower, adsorption of uranyl to oxide or oxyhydroxide minerals may be a dominant process.

5.3.8 Volume of Contaminated Ground Water

The volume and concentration of contaminated ground water are important considerations when planning ground-water remediation. The cost of most remedial actions will be directly related to the volume of water that must be contained, removed, or treated. Volume calculations made in 1996 are presented in MACTEC–ERS calculation U00033AA. Table 5–7 presents the results of volume calculations made using the same procedure as in the 1996 calculation but using the areal distributions based on the most recent data for nitrate, sulfate, and uranium (Figures 5–8, 5–11, and 5–12, respectively). The volume calculations assume a porosity of 0.25 and an average thickness of 86 ft of contaminated Navajo Sandstone aquifer.

Table 5–7. Summary of Volumes of Contaminated Ground Water

COPC	Concentration (mg/L)	Area (ft^2)	Volume (gallons)[a]
Nitrate	44	10.5×10^6	1.69×10^9
Sulfate	250	8.86×10^6	1.42×10^9
Uranium	0.044	4.84×10^6	7.78×10^8

[a]Based on saturated thickness of 86 ft and porosity of 0.25.

5.3.9 Geochemical Conceptual Model

The tailings pore fluids were acidic and had elevated concentrations of COPCs and major ions. These fluids migrated downward through the unsaturated zone and mixed with ground water in the Navajo Sandstone aquifer. Migration of contaminated water was probably at a maximum during the milling operation, but some migration also took place during disposal cell compaction.

A plume was formed that extends to the south and southwest consistent with the regional ground-water flow direction. Because of ground water mounding, the plume also extends east to the area around well 944. The contaminants have migrated no deeper than the Navajo Sandstone and appear to be less concentrated with depth within this unit.

Calcite cements were dissolved from the Navajo Sandstone by the low-pH, undersaturated tailings fluids, which caused the rocks to become more friable near the disposal cell. As contaminated ground water migrated away from the cell, the chemistry evolved from undersaturation to saturation with calcite, probably because of the gradual loss of CO_2 and accompanying rise in pH. Ground-water chemistry indicates that calcite is precipitating downgradient from the cell.

Gypsum is also precipitating in a zone downgradient from the disposal cell because of the elevated concentrations of mill-related sulfate in the area beneath the disposal cell. Gypsum precipitation is coupled to calcite precipitation because both minerals contain calcium. As calcium is added to the ground water from dissolution of calcite, gypsum becomes oversaturated and precipitates.

The most likely mechanism to cause calcite oversaturation is the exsolution of CO_2 from the ground water. Exsolution of CO_2 and other reactions that influence mineral precipitation are likely to occur during remediation and could lead to undesirable scaling.

Although some sulfate is removed from the ground water by precipitation of gypsum, most reduction in sulfate concentrations is caused by mixing with more dilute concentrations in ground water and by subsequent dispersion. The sulfate and nitrate plumes are similar in size and orientation. It is unlikely that any chemical mechanisms are removing a significant portion of the dissolved sulfate or nitrate. Both plumes were formed from advection and dispersion of the contaminated ground water with little or no interaction with the solid fraction of the aquifer.

Uranium chemistry is governed predominantly by pH, dissolved carbonate concentration, and oxidation potential. Due to the oxidizing conditions in the aquifer near the disposal cell, it is likely that nearly all the uranium is hexavalent. Uranium concentrations in ground water are too low to precipitate uranium minerals; therefore, adsorption is probably the dominant chemical mechanism causing uranium to transfer from aqueous to solid phases. This process might be more pronounced at the distal margins of the plume.

The uranium plume (Figure 5–11) appears to be somewhat smaller than the sulfate and nitrate plumes (Figures 5–10 and 5–7, respectively), suggesting that some retardation may be occurring. Results from laboratory tests performed by the University of New Mexico and MSE, Inc., on cores from the Navajo Sandstone indicated that no uranium retardation is occurring in some portions of the aquifer close to the disposal cell. These studies, however, were too limited to extend this conclusion to the entire contaminated portion of the aquifer. High concentrations of carbonate and low pH conditions near the disposal cell will cause uranium to remain dissolved in the ground water. Qualitative observations of cores suggest that ferric oxides are less abundant near the disposal cell. This favors the partitioning of uranium to the dissolved phase, since ferric oxides are known to be capable of adsorbing high concentrations or uranium. As ground water flows away from the disposal cell, dissolved carbonate concentrations decrease, pH increases to nearly neutral, and aquifer solids probably increase in ferric oxide content. These conditions favor partitioning of uranium to the solid phase by adsorption. These observations suggest that uranium may be more strongly tied to the solid phases in the diffuse portion of the plume than in the more concentrated portion and imply that (1) the uranium plume may not be migrating rapidly in the distal portions, (2) more pumping may be required to desorb the uranium from the distal portion of the plume, (3) a complexant, such as carbonate, may enhance desorption in the distal portion of the plume, and (4) pump-and-treat remediation may be more expensive (per gallon) in the diffuse plume than in the concentrated plume.

Tailings compaction during and shortly after disposal cell construction caused the expulsion of a “slug” of contaminated water. This slug was observed in well 906 by synchronous increases in water levels, concentrations of COPCs, and concentrations of major ions (Figure 5–17). Water levels and dissolved ion concentrations are now decreasing, indicating that the slug has passed well 906. Additional drainage of consolidation water from the disposal cell is expected to decrease with time.

Another factor that may contribute to the water-level increases observed in well 906 is that runoff from the disposal cell accumulated in the east-west drainage ditch along the south end of the cell

and caused local recharge of the Navajo Sandstone. This hypothesis is supported by the occurrence of several large storms in 1992. The recharge is in an area that was used to store contaminated effluents during milling. The chemical profiles observed in samples from well 906 indicate that recharging water might have leached contamination from soils in this area.

Recent trends indicate that sulfate and nitrate concentrations are increasing in the southwest plume and that uranium concentrations are increasing in the eastern plume. The variation in contaminant types observed between the east and southwest areas is probably due to heterogeneities in the source areas of the millsite. The increasing trends are caused by transient release of contaminated fluids from the compaction of the disposal cell. Concentrations are expected to decline in the future.

6.0 Summary of Human Health and Ecological Risk

6.1 Human Health Risk Assessment

A BLRA was conducted in 1993 that evaluated the potential effects to human health that could result from ground-water contamination at the Tuba City site (DOE 1994a). As with other UMTRA ground-water sites, the BLRA serves as the basis for risk information. Section 6.1.1 summarizes the results of the BLRA for the Tuba City site.

Additional sampling was done after the BLRA was completed. Analytical results from that sampling provided an opportunity to evaluate the effect of the changing site conditions on risks. Section 6.1.2 updates the BLRA using the more recent analytical data (1994 to present).

6.1.1 Summary of the BLRA

There are currently no domestic or drinking-water wells that withdraw contaminated ground water at the site. Because no one is drinking the affected water and the seeps downgradient of the site show no contamination, there are currently no complete exposure pathways or human-health risks associated with the contaminated ground water. However, the contamination extends up to 2,000 ft beyond the southwest corner of the disposal cell, and ground water in the area is used as a source of drinking water. To account for the potential future use of contaminated ground water, the risk evaluation assumed this pathway may be complete in the future. As a worst case assumption, the BLRA estimated future potential risks based on current ground-water data for the most contaminated wells at the processing site.

Contaminants of potential concern 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 monitor wells at the 0.05 level of significance and if the site is a likely source for the contaminant. Eighteen chemicals were identified in concentrations above background: ammonium, cadmium, calcium, chloride, chromium, iron, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulfate, tin, uranium, and zinc. Because data for lead at the Tuba City site were limited, results were inconclusive as to whether it was above background levels. Historically, lead has not been a major component of mill tailings. Therefore, lead was not evaluated in the risk assessment.

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 range, at a level of low toxicity, or are within the expected dietary range. The acceptable nutritional range was based on the U.S. Food and Drug Administration's Recommended Daily Allowance. The criteria for expected dietary ranges were based on information published in the Agency for Toxic Substances and Disease Registry, EPA's Integrated Risk Information System, and the *Handbook on the Toxicology of Metals* (Friberg et al. 1986). These publications identify common intake rates for specific nutrients. A constituent was considered to have low toxicity when the anticipated exposure dose fell well below any known adverse health effect level as stated in the literature. Toxicity evaluations at this screening level compared the estimated dose for the

95 percent confidence limit of the mean concentration in ground water to available values known to cause adverse health effects.

Calcium, chloride, chromium, iron, potassium, sodium, and zinc were removed from the list because they are present at nutritional levels. Selenium and molybdenum were retained as COPCs because concentrations exceed drinking-water health advisories. Magnesium, manganese, and tin were screened out at this stage on the basis of low toxicity or because concentrations are within the acceptable dietary range. These three screening criteria narrowed the list of COPCs for human health to cadmium, molybdenum, nitrate, selenium, strontium, sulfate, and uranium. The BLRA determined that cadmium has a negligible contribution to total risk and thus can be eliminated from the COPCs.

Probability distributions for contaminant concentrations and exposure variables were incorporated to determine the amount of contaminants that would likely be ingested if someone were withdrawing ground water from the most contaminated portion of the aquifer. Water quality data from 1988 to 1992 for DOE monitor wells 906, 908, 909, and 912 were used to evaluate on-site levels of contaminants. The estimated amounts of contaminants that people potentially could ingest through drinking water were compared to the toxic effects anticipated for each contaminant at those amounts. Results showed that the main concerns for health effects associated with the Tuba City site are molybdenum, nitrate, sulfate, and uranium.

The most significant health hazard from ingestion of ground water at the Tuba City site is nitrate. Nitrate is primarily a concern for infants, since levels observed in the ground water would interfere with an infant's ability to transport oxygen through the blood. This toxic effect occurs at a lower dose in infants because an infant's stomach absorbs nitrate differently than an adult's. Because of the potentially lethal effects to infants, ensuring restriction of access to contaminated ground water is an immediate need.

Other contaminants that occur at potentially toxic levels are sulfate and uranium. Sulfate concentrations in ground water near the site may cause severe diarrhea, particularly in infants. Because uranium is radioactive, the uranium concentrations may increase the risk of cancer above the upper end of the EPA acceptable risk of 1 in 1,000,000 to 1 in 10,000 to an additional average lifetime cancer risk of 2 in 1,000. Molybdenum may also cause some adverse effects to human health. The amounts of strontium that could be potentially ingested are lower than any level associated with a toxic effect.

6.1.2 BLRA Update

Additional analytical results became available after the original BLRA (DOE 1994a) was completed in 1994. Table 6–1 presents a comparison of the analytical results used in the BLRA to the more recent analytical results (see the recent sampling results listed in Appendix C). This table shows that the median concentration for most COPCs is increasing. Only strontium shows decreasing concentrations.

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[c]	Median	Maximum[c]		
Molybdenum	0.10	< 0.01	0.12	0.16	0.6	16	Increase
Nitrate	44	917	1,200	951	2,330	1.04	Increase
Selenium	0.01	0.010	0.05	0.096	0.469	10.1	Increase
Strontium	NA	6.8	9.1	5.36	9.59	0.79	Decrease
Sulfate	NA	1,720	3,820	2,257	7,590	1.3	Increase
Uranium	0.044	0.011	1.01	0.404	1.38	3.7	Increase

[a]The BLRA identifies the following downgradient wells: 906, 908, 909, and 912.
[b]These data were determined using the same wells used in the BLRA to define the downgradient plume. Only the more recent data (January 1994 to present) that were not used in the BLRA were evaluated.
[c]All maximum concentrations occur in well No. 906.

A similar table was developed to show if concentrations had changed significantly for the COPCs in the background wells. Table 6–2 presents the results. Concentrations for all COPCS in the background wells have remained similar to the concentrations used in the BLRA. To highlight the effect from site contamination, the last column in Table 6–2 shows the ratio of plume concentrations to background concentrations for each COPC. Concentrations of uranium, sulfate, and nitrate in the plume are considerably elevated over concentrations in the background wells. The other COPCs exceed background to a lesser extent; the median concentration of strontium in the plume is the least elevated above concentrations in background wells.

Table 6–2. Comparison of New Data to Data Used in the BLRA for Background (mg/L)

BLRA COPCs	UMTRA MCL	BLRA Background Data[a]		Most Recent Background Data[b]		Ratio of Recent Background Concentration Data to BLRA Background Data	Ratio of Recent Median Plume Concentration Data to Recent Median Background Data
		Median	Maximum	Median	Maximum		
Molybdenum	0.10	< 0.01	0.01	0.009	0.052	0.9	17.8
Nitrate	44	15	22	12.3	16.3	0.82	77.3
Selenium	0.01	NA	NA	0.0046	0.005	NA	20.9
Strontium	NA	0.29	0.36	0.33	0.38	1.1	16.2
Sulfate	NA	16	46	18	22	1.1	125
Uranium	0.44	0.001	0.012	0.0018	0.003	1.8	224

[a]The BLRA identifies wells 901, 910, and 917 as background wells.
[b]These data were determined using the same wells used in the BLRA to define background. The most recent data were not used in the BLRA; they were collected after January 1, 1994.

Effect on the COPC List

Only one COPC (strontium) showed a decrease in the median concentration in the plume. Although elevated above background concentrations, strontium has no regulatory benchmark and has a relatively low toxicity value. To evaluate if strontium should be retained as a COPC, risks associated with strontium were reevaluated using standard equations and assumptions from the BLRA and EPA (1989) and the more recent analytical results. This evaluation is presented in Appendix D.

The evaluation indicates that since strontium has a hazard quotient of less than 1 using maximum concentrations, is a minor contributor to total risks (less than 1 percent), and has decreasing concentrations, it is appropriate to eliminate strontium as a COPC.

Effect on Risk

The remaining COPCs (sulfate, nitrate, uranium, molybdenum, and selenium) are hereby designated as contaminants of concern (COCs) on the basis of this final screening. These COCs show increasing median concentrations in the plume. The most significant increasing concentrations have occurred for molybdenum (factor of 16), selenium (factor of 10.1), and uranium (factor of 3.7). The median concentration of nitrate has increased about 10 percent; median sulfate concentrations has increased about 30 percent. This indicates that potential future risks may have increased since the BLRA was completed (risks are directly proportional to concentration). However, risk estimates are based on the assumption that contaminated water at the Tuba City site will be used as the primary source of drinking water. Until that occurs, there are still no risks to human health because no exposure pathways are complete.

6.2 Ecological Risk Assessment

Another purpose of the BLRA was to evaluate potential adverse effects to ecological receptors as a result of exposure to ground-water contaminants (DOE 1994a). The results of the ecological risk assessment, coupled with the human-health risk assessment, serve as a basis for selecting a ground-water compliance strategy. The BLRA provided a qualitative assessment of ground-water effects and identified uncertainties and limitations of the assessment. DOE subsequently identified additional data needed to reduce uncertainty in the assessment (DOE 1995b). Analytical results from recent ground-water sampling and from the University of Arizona plant uptake study provided the data.

Section 6.2.1 summarizes the ecological risk assessment results from the BLRA. Section 6.2.2 updates the ecological risk assessment using the additional ground-water sampling and plant uptake data.

6.2.1 Summary of the BLRA Ecological Risk Assessment

The BLRA generally follows the EPA (1989) qualitative ecological evaluation framework. The qualitative or screening-level approach identified and characterized potential ecological exposure pathways, identified potential ecological receptors in those pathways, and evaluated potential adverse effects to exposed receptors. Potential adverse effects were evaluated by comparing

ambient concentrations of contaminants in environmental media the ecological receptors are expected to encounter with toxicity criteria or benchmarks for those media.

Exposure Pathways Characterization

Ground water was the only environmental medium that the BLRA evaluated. It was assumed that tailings and contaminated soil are contained within the disposal cell; therefore, pathways such as incidental soil ingestion, dermal contact with soil, and inhalation of particulates by receptors were disregarded. Because no ecological exposure pathways to contaminated ground water currently exist, no risks currently exist. Consequently, the BLRA focused on possible future ground-water exposure pathways. The following pathways were identified:

- Placement of a well in the plume to supply water for a livestock pond.
- Irrigation of crops for consumption by humans or livestock using contaminated water from a plume well.
- Extraction of plume water to supply a fish pond.
- Migration of the plume into ground-water discharge areas on the lower terrace and in Moenkopi Wash.

Ecological Receptors

No ecological receptors currently have access to contaminated ground water. However, in the future, wild and domestic fauna and flora could be exposed to ground-water contaminants via the pathways identified above. Local residents use the rangeland downgradient of and overlying the plume for grazing sheep, horses, and cattle. Numerous small and large mammals, birds, and reptiles could have access to contaminated ground water that was pumped for a livestock pond or for crop irrigation. Livestock and other primary consumers could be exposed by ingesting crops irrigated with plume water. Aquatic organisms, including fish, amphibians, benthic macroinvertebrates, water striders, and back swimmers could be exposed in ponds supplied with plume water or in discharge areas of Moenkopi Wash.

Greasewood (*Sarcobatus vermiculatus*) and desert olive (*Forestiera neomexicana*), both obligate phreatophytes (desert shrubs rooted in ground water), grow on the terrace between the upper and lower escarpments downgradient of the plume. Salt cedar (*Tamarix pentandra*), Freemont cottonwoods (*Populus fremontii*), and other riparian plants grow in Moenkopi Wash. Future migration of the plume could expose these plant populations and, therefore, could also expose animals that ingest these plants or humans that use them for cultural purposes. Agricultural irrigation with plume water could expose produce and forage crops and the humans and animals that consume the crops.

Potential Future Adverse Effects

Although contaminated ground water is not currently used to water livestock, the BLRA indicated that nitrate levels from the center of the plume are high enough to cause death in sheep and cattle if they ingested plume water for a prolonged period. Similarly, the BLRA indicated that chronic exposure of aquatic organisms to nitrates in a plume-fed stock pond or exposure of terrestrial wildlife ingesting pond water could cause adverse effects. The BLRA also indicated that the 95 percent upper confidence limit for molybdenum in plume water exceeds an EPA (1972) phytotoxicity criterion for agricultural crops.

The BLRA acknowledges that the ecological risk assessment was incomplete because of the lack of toxicity benchmarks for many COPCs and receptors.

6.2.2 Ecological Risk Assessment Update

Ground-water sampling results acquired since the original BLRA (Section 4.6) and the results of the University of Arizona's plant uptake study (Section 4.7) were used to update the ecological risk assessment. Median and maximum concentrations of almost all COCs have increased since the original BLRA (Table 6–1); consequently, almost all ground-water pathways and receptor exposures were reevaluated (Table 6–3). Literature values of screening benchmarks were lacking for some exposure scenarios. Maximum COC levels in ground water could potentially cause adverse effects to crop and rangeland vegetation, terrestrial wildlife, aquatic organisms, and livestock. Estimated COC levels in plants that have access to contaminated ground water could potentially cause adverse effects in foraging wildlife and livestock.

Terrestrial Plants and Wildlife

Terrestrial plants (range plants and crops) rooted directly in the plume or irrigated with plume water could potentially be exposed to COCs. Greenhouse measurements of plant uptake of Tuba City plume water with maximum reported concentrations of molybdenum and selenium (Baumgartner et al. 1996) exceed screening benchmarks for phytotoxicity. Terrestrial wildlife could potentially be exposed by drinking contaminated livestock pond water or seep water, or by ingesting contaminated plants. Concentrations of nitrate in the plume exceed screening benchmarks for livestock. Estimated levels of molybdenum, nitrate, and selenium in irrigated plants exceed recommended forage ingestion thresholds for some terrestrial wildlife that inhabit the Tuba City site.

Aquatic Organisms

Aquatic organisms, including benthic macroinvertebrates and freshwater fish, could be exposed to ground-water COCs if plume water were pumped for livestock or fish ponds, or if the plume migrated to Moenkopi Wash. The screening assessment suggests that maximum nitrate levels in plume water exceed toxicity benchmarks for fresh-water fish.

Table 6–3. Median and Maximum COC Concentrations in the N–Aquifer, Estimated Maximum Concentration in Plants, and Toxicity Benchmarks for Potential Ecological Receptors

COC[a]	Exposure Point Concentrations			Toxicity Benchmarks						
	Ground Water[b]		Plants, Estimated Maximum[c] (mg/kg)	Aquatic Toxicity Benchmark (mg/L)	Water Concentrations Protective of Livestock[g] (mg/L)	Terrestrial Wildlife[h]		Maximum Livestock Dietary Concentrations (mg/kg)[g]		Phytotoxicity Benchmark[i] (mg/L)
	Median (mg/L)	Maximum (mg/L)				Water (mg/L)	Food (mg/kg)	Sheep and Cattle	Horses	
Molybdenum	0.16	0.6	20.0	370[d]		0.60	0.52	10	(5)	0.5
Nitrate	951	1,200		90[e]	100	2,360	2,720			
Selenium	0.096	0.469	20–30	5.0[f]	0.05	1.3	1.0	(2)	(2)	0.7
Sulfate	2,257	7,590			1,000					
Uranium	0.40	1.38	≤2.0	2.6[d]		7.0	6.1			40

[a] The list of COCs has changed since the BLRA; see Section 6.1.
[b] Median and maximum ground-water levels are from recent data (Section 4.7).
[c] Maximum crop concentrations estimates are based on inferences from response functions by Baumgartner (1996).
[d] Tier II secondary chronic values for aquatic organisms (EPA 1993).
[e] Concentration at or below which no adverse effects are expected for warm water fish (EPA 1986).
[f] National Ambient Water Quality Criteria for chronic exposure (Stephan et al. 1985).
[g] National Academy of Sciences (1980).
[h] From Opresko et al. (1995).
[i] From Will and Suter (1995).

Livestock

Livestock could potentially be exposed to ground-water COCs in livestock ponds fed by plume-water wells, or by ingesting forage or crop plants rooted into or irrigated with plume water. Maximum nitrate and sulfate levels exceed chronic exposure benchmarks for water ingestion by livestock. Estimated maximum plant levels of molybdenum and selenium exceed recommended maximum dietary intake levels for sheep, cattle, and horses.

7.0 Ground-Water Compliance Strategy Selection

This section defines the proposed ground-water compliance strategy options; explains the application of site-specific data to the ground-water compliance selection framework; and analyzes possible deviations, contingencies, and decision rules.

7.1 Ground-Water Compliance Strategies

Proposed ground-water compliance decisions at the Tuba City site were made by using the compliance selection framework shown in Figure 7–1. This compliance selection framework is documented in the PEIS (DOE 1996), Section 2.0, and is supported by the PEIS Record of Decision (62 FR 22913). The proposed ground-water compliance strategy was selected on the basis of ground-water data, human and environmental risk, stakeholder input, and cost; this process is consistent with the PIES selection framework. Three compliance strategies are available in the selection framework:

- No remediation—Application of the no-remediation strategy would mean that compliance with EPA ground-water protection standards would be met for a particular constituent without altering the ground water or cleaning it up in any way. This strategy could be applied at sites where contaminants of potential concern are below the MCL or background, or at sites that have contamination above MCLs or background levels but qualify for supplemental standards or ACLs as described in Section 2.1.1. Ground water at the Tuba City site contains constituents above MCLs or background and yet may qualify for ACLs on the basis of acceptable human-health and environmental risks.

- Natural flushing—Natural flushing relies on ground-water advection and geochemical processes to decrease the contaminant concentrations to levels within regulatory limits within a period of 100 years. Natural flushing could be applied at the Tuba City site in conjunction with active remediation in areas of the plume if effective monitoring and institutional controls could be maintained, and if the ground water is not currently a drinking water source and is not projected to be one in the future.

- Active ground-water remediation—Active ground-water remediation would require the application of engineered ground-water remediation methods to achieve compliance with EPA ground-water protection standards. This strategy could be applied at the Tuba City site in conjunction with natural flushing to reduce contaminant levels in the ground water to regulatory limits within 100 years.

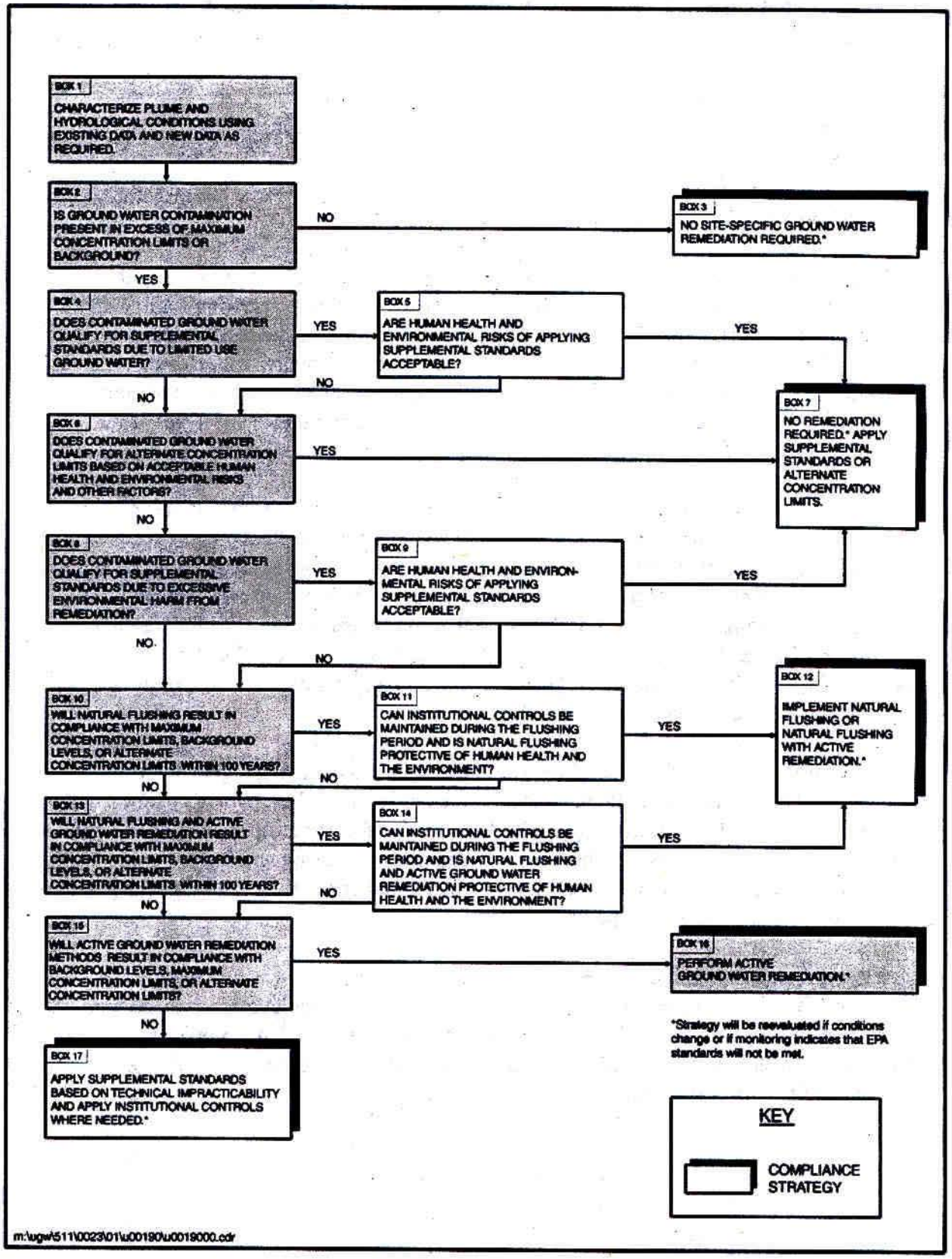


Figure 7–1. Compliance Selection Framework, Tuba City, Arizona, Site

7.2 Site-Specific Compliance Strategy Selection

Ground-water constituents at the Tuba City site have been screened for potential risks as discussed in Section 3.3. For compliance purposes, these constituents have been divided into two groups based on potential risks:

- Constituents that do not pose a significant risk (all non COCs).
- Constituents that exceed EPA ground-water standards (background and MCLs) and may pose a significant risk, i.e., COCs (molybdenum, nitrate, selenium, sulfate, and uranium).

The proposed ground-water compliance strategy is to seek no remediation on the basis of ACLs for constituents that do not pose a significant risk within the context of the UMTRA ground-water regulations. For the constituents that exceed EPA ground-water standards and pose a significant risk, DOE will propose the active-remediation compliance strategy. The following discussion is a justification for selecting the compliance strategy for the Tuba City site.

7.3 Compliance Strategy Discussion and Justification

Analysis of ground-water quality has shown that some constituents in ground water beneath and downgradient from the site exceed MCLs or background levels (box 2 in Figure 7–1). COCs at the Tuba City site include molybdenum, nitrate, selenium, sulfate, and uranium because they may have exceeded the ground-water protection standards one or more times in one or more wells and are present in concentrations that exceed the safe nutritional range. The presence of elevated levels of these constituents is a direct result of past uranium milling operations.

Ground water at the Tuba City site does not meet the criteria for limited-use ground water (box 4 in Figure 7–1), and does not qualify for supplemental standards. For ground water to be classified as limited use, TDS concentration must exceed 10,000 mg/L, there must be widespread ambient contamination that cannot be cleaned up using treatment methods reasonably employed in public water-supply systems, or the sustained yield of ground water must be less than 150 gallons (570 L) per day.

According to the 10 criteria presented in 40 CFR 192.02, ACLs described in box 6 of Figure 7–1 cannot be established for the ground water at the Tuba City site. Neither would ACLs be protective of human health and the environment.

Ground-water contamination at the Tuba City site also does not qualify for supplemental standards on the basis of excessive environmental harm. Consequently, box 8 (Figure 7–1) is not applicable.

The next stage in determining compliance strategies requires an evaluation of the effectiveness of natural flushing. It is estimated (Section 4.4.3) that about 220 years could be required to flush one pore volume through the presently contaminated area, and it is unlikely that only one pore volume would completely flush the plume. The regulatory time limit for natural flushing is 100 years; thus, natural flushing, as described in box 10 (Figure 7–1), is not a regulatory option for each constituent.

Since natural flushing by itself cannot be relied upon to improve ground-water quality to MCLs or background levels within 100 years, consideration must be given to a combination of active remediation and natural flushing (box 13 in Figure 7–1). Although active remediation may be appropriate for the two site wide constituents of concern, a combination of active remediation, no remediation (ACLs), and natural flushing may be appropriate for those constituents of concern that are present at trace levels locally.

The next compliance strategy is active ground-water remediation (box 15 in Figure 7–1). This compliance strategy is described in Section 7.4.

7.4 Active Remediation Options

Active remediation on this project is most easily described by dividing the remediation process into three logical components: (1) extraction of ground water, (2) treatment of ground water, and (3) discharge of effluent. Several well-field designs were evaluated during this project, including a uniform well field of 120 extraction and injection wells, a small number of injection and withdrawal wells deployed in hot spots, and horizontal wells oriented both perpendicular to and parallel to the direction of ground-water flow. Geochemical studies combined with numerical flow modeling have indicated that significant mass reduction may result if conventional vertical extraction and injection wells are deployed for pumping from hot spots.

Various treatment alternatives were also evaluated to compare their relative effectiveness, implementability, and cost. The main constituents requiring treatment at the site, that is, those constituents regulated by EPA, are nitrate, uranium, selenium, and molybdenum. The goal of the remedial action, however, is to also reduce the concentrations of sulfate and TDS to those requested by the Navajo Nation.

Treatment processes considered during this project have included evaporation, distillation, ion exchange, land application, biological processes, chemical treatment processes, reverse osmosis, and nanofiltation. Comparisons of those processes have been conducted based on their effectiveness, implementability, and cost. One objective criteria has been to maximize the volume of water returned to the aquifer in order to conserve the resource.

Discharge of treated ground water would be the final step in the active remediation of the Tuba City site. There are four alternatives for discharging the treated ground water: (1) reinjection wells, (2) reinjection trench, (3) land application of treated ground water, and (4) evaporation. Reinjection wells at the site can be used in several geometries, including various five-point arrays or staggered line-drive arrays similar to those used conventionally in the in situ mining industry. These technologies are also used in conventional ground-water pump-and-treat systems.

A reinjection trench could be used upgradient of the cell to provide a line source of treated ground water that would assist with gradient manipulation across the site. The reinjection scenarios provide the advantage that they conserve the water resource of the tribal lands. Under the land application scenario, the treated ground water could be used for irrigated agriculture, thereby conserving the resource and achieving a beneficial use for the water. The evaporation alternate is attractive from the perspective that the treatment costs are controlled; however, the disadvantage is that the pumped water is entirely consumed.

7.5 Recommended Compliance Strategy

A successful remediation method may not only reduce the concentrations of the contaminants of concern, but may also reduce concentrations of the constituents that could qualify for ACLs. The proposed compliance strategy for the Tuba City site might be a combination of active-remediation and no-remediation (ACLs) compliance strategies, depending on the contaminants of concern, the degree of contamination, and the extent (distribution) of contamination. On the basis of the data evaluated in this document, and the compliance selection framework (Figure 7–1), active remediation is the recommended compliance strategy. The final compliance strategy, and the method to achieve it, will be selected in the site-specific EA.

7.6 Deviations, Contingencies, and Decision Rules

The primary deviation from the proposed remediation method may result from the hydrogeologic conditions of the site, because undiscovered heterogeneity and anisotropy may be present in the aquifer. However, ground-water remedial actions are almost always undertaken in the presence of uncertainty about aquifer properties, and given the depositional environment (which tends to produce relatively homogeneous deposits), any adverse effects from as yet unknown heterogeneity are likely to be manageable.

8.0 Development and Evaluation of Remediation Alternatives

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

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 ground water at the Tuba City 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 site 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, either in situ or ex situ. The technologies considered included both conventional and innovative technologies. For evaluation, technologies were grouped into extraction technologies, disposal technologies, and treatment technologies. The evaluation of technologies considered effectiveness, implementability, and cost but was generally qualitative in nature. A screening recommendation was made for each technology on whether it should be used for the development of alternatives.

The technologies that passed the initial screening were then combined into alternatives for extraction and disposal, and for treatment. An alternative might incorporate multiple technologies in order to completely meet the remediation goals. For instance, biological treatment technology is very effective for treatment of nitrates, but produces hazardous by-products when used to treat sulfates, and is ineffective for treatment of radionuclides.

Therefore, two treatment alternatives were developed that combined biological treatment for removal of nitrate with other processes for removal of sulfate and radionuclides.

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 technology and 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.

Effectiveness

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

- Remediation time frame.
- Ability to meet aquifer restoration standards or treatment standards.
- Performance limits, reliability, and stability.
- Short-term effects (i.e., effects of remediation on workers, the community, and the environment).

Remediation Time Frame

The remediation time frame is largely dependent on how quickly contaminated ground water is removed from the 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 aquifer, and extraction and disposal alternatives were developed considering this goal.

Ability to Meet Aquifer Restoration Standards or Treatment Standards

The general requirements for contaminant levels in the ground water at UMTRA sites are specified in 40 CFR 192.04, Table 1. The only COCs in the ground water that have standards specified in 40 CFR 192 are nitrate, molybdenum, selenium, and uranium. The regulation does not specify ground-water restoration standards for other COCs that exceed background concentrations.

The Navajo Nation, in a letter dated September 18, 1997, proposed secondary cleanup standards for restoration of ground water at the Tuba City site that included constituents not listed in 40 CFR 192. They were guided in requesting these standards by the relatively high quality of the background water in the Navajo Sandstone. DOE considered the request from the stakeholders and will incorporate the suggested secondary cleanup standards as goals for aquifer restoration and treatment of extracted ground water.

Cleanup levels for the aquifer were divided into aquifer restoration standards (requirements of 40 CFR 192) and aquifer restoration goals (cleanup standards requested by the Navajo Nation but not required by 40 CFR 192). The restoration standards and goals for the 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^-)
- Molybdenum — 0.10 mg/L
- Selenium — 0.01 mg/L
- Uranium — 30 pCi/L (0.044 mg/L combined U-234 and U-238)

Aquifer restoration goals (not required by 40 CFR 192 but requested by the Navajo Nation):

- TDS — 500 mg/L
- Sulfate — 250 mg/L
- Chloride — 250 mg/L
- pH — 6.5–8.5
- Corrosivity — Non-corrosive

Performance Limits, Reliability, and Stability

Treatment technologies and alternatives were evaluated on the practical limits of the process and the system performance. Performance requirements for the treatment processes are treatment flow rate and the level of treatment required (i.e., the concentration of constituents in the treated water). The minimum treatment flow rate is determined by the extraction rate required to meet the aquifer restoration standards and goals. The level of treatment required for the extracted ground water is assumed to be equal to the aquifer restoration standards and goals. Water characteristics of the plume and the treatment level (when applicable) for each contaminant are listed below.

Contaminant	Concentration in Plume (mg/L)		Treatment Level (mg/L)
	Median	Maximum	
Nitrate	951	2,330	44
Molybdenum	0.16	0.6	0.1
Selenium	0.096	0.469	0.01
Uranium	0.404	1.38	0.044
Sulfate	2,257	7,590	250
TDS	4,548	15,600	500
Chloride	91	440	250
Alkalinity (as $CaCO_3$)	510	1,780	Not applicable

Another aspect of performance limits is the reliability and stability of the treatment technologies and alternatives. Reliability is defined as the probability that a system will meet required performance standards; stability is the degree of variability in process performance. Performance limits also consider the generation of treatment residuals and the control, handling, storage, and disposal of treatment byproducts.

Short-Term Effects

Short-term effects is a consideration of the effects to the community, workers, and the environment. Tuba City is the largest community near the site. The Tuba City site is 6 miles east of the town of Tuba City and the villages of Upper and Lower Moenkopi. Although these communities are growing, and expansion of the town is predominantly eastward toward the site, 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. Only one residence is located within 1,000 ft of the site.

State Highway 160 passes within a few hundred feet north of the site. The highway is moderately traveled by tourists and residents during the spring and summer, and lightly to moderately traveled by residents during the remainder of the year. All users of the state highway are also classified as part 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 aquifer.

Implementability

Implementability is an assessment of the technical and administrative feasibility of building, operating, and maintaining a remediation system. Implementability is considered as two parts: technical feasibility and administrative feasibility.

The following are aspects of technical feasibility:

- Ease or difficulty associated with construction.
- Uncertainty associated with construction, such as the potential for schedule delays caused by technical problems.
- Technical and operational complexity and required level of training for operators.
- The ease or difficulty of adding, modifying, or improving the system.
- Availability of services, equipment, and material for accomplishing the remedial action.

Administrative feasibility is the level of activity needed to coordinate with stakeholders, offices, and agencies and assessing whether coordination effort will be successful.

Cost

Cost estimates for the evaluation of technologies are general and are developed only to the level of detail required to help determine if a technology should be evaluated in more detail. The cost of a technology is generally described as low, medium, or high (as compared to other technologies) but may be more detailed if required for the initial evaluation. The cost of innovative technologies is also included. However, innovative technologies lack sufficient cost data to be defined at the same level of accuracy as conventional technologies.

Cost estimates for the alternatives (i.e., extraction and disposal alternatives and treatment alternatives) have been developed in more detail. Capital costs (both direct and indirect) and operating and maintenance (O&M) costs were calculated for each alternative. The accuracy of the cost estimates for evaluation of the alternatives is defined to a level of accuracy of +50 percent to –30 percent.

A net present worth analysis was used to compare the alternatives. 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

Several activities were completed that evaluated technologies for remediation of the site and are described below. Technologies considered for the site include ground-water extraction technologies, effluent-discharge technologies, and ex situ and in situ treatment technologies.

Center for Radioactive Waste Management

One of the first technologies considered for the remediation of contamination at the site was in situ bioremediation. Development of the technology was initiated by the CeRaM through the University of New Mexico. The technology would make use of living organisms (i.e., bacteria) to eliminate the hazards posed by COCs in the ground water. Substrates injected into the ground water would disperse through the contaminant plume to promote the growth of bacteria. The bacteria would immobilize the COCs by reducing them to an insoluble form.

After evaluation by the ITRD team, the technology was eliminated from further consideration, primarily because of difficulties in delivering the substrate to all areas of the aquifer and in maintaining a chemically reducing environment in the aquifer that would keep the COCs insoluble.

Innovative Treatment Remediation Demonstration (ITRD)

DOE began the ITRD program in November 1996 to identify and assess new or innovative technologies that could enhance and accelerate remediation at the site. Technologies are considered innovative if they have a potential for enhanced performance or cost savings but have a limited amount of performance or cost information.

A consortium of DOE, EPA, industry, and tribal representatives made up the ITRD team. The team identified approximately 30 technologies that might be used to remediate the site or to enhance the baseline technology. DOE was in the process of developing alternatives for remediation of the site when the ITRD began and, therefore, a baseline technology had not been selected for remediation. A baseline technology for remediation of the site (i.e., ground-water

extraction and ex-situ treatment) was developed so that the ITRD team could compare innovative technologies to a baseline technology.

Initially, the ITRD team evaluated technologies that could be used to replace the baseline technology as well as technologies that could be used to enhance the baseline technology. A conclusion reached early in the evaluation was that extraction of the contaminated ground water and some type of ex situ treatment would be the best method for remediation of the site. Therefore, subsequent efforts of the team focused on ways to enhance the baseline technology.

The ITRD final report presented a qualitative evaluation of technologies that could be used to enhance the baseline technology. The report did not recommend any particular technology as the best solution for enhancing the baseline. Appendix E includes the full report prepared by the ITRD team documenting the technologies considered and a qualitative evaluation of the technologies. Some of the technologies considered and evaluated by the ITRD are included in the evaluation of technologies presented in this section.

Conventional Technologies

In addition to technologies considered by the ITRD team, the development of technologies also considered conventional technologies that could be used to extract contaminated ground water, treat the extracted ground water, or dispose of treated ground water. Conventional technologies are technologies for which there is a substantial amount of cost and performance data that can be used to evaluate the technology.

8.2.2 Extraction Technologies

Because of its depth at the Tuba City site, ground water can only be withdrawn effectively through a well. Three types of extraction-well systems were evaluated for the site: (1) conventional vertical wells, (2) blasted bedrock zones in conjunction with conventional vertical wells, and (3) horizontal wells. Yields from existing wells at the site are relatively low because of the fine-grained, loose (unconsolidated), friable, and weakly-cemented nature of the eolian sandstone aquifer. Because of the high risk of poor well yields, sand pumping, and high pump-maintenance costs, careful attention should be directed to well design, construction, and development.

Vertical Wells

A vertical well is the most commonly used ground-water extraction device. Consequently, in comparison to other extraction and injection techniques, the bulk of field experience and knowledge resides with 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. Another attractive feature about vertically oriented recovery wells is that they can be readily converted to injection wells as needed, or vice versa. Also, a vertical well can be easily decommissioned when necessary.

During design, the theoretical performance of a vertical well can be simulated analytically or numerically using readily available and accepted mathematical formulations. This is a

considerable advantage because confidence increases whenever appropriate algorithms are used to obtain the solution to a problem. By virtue of these factors, vertical wells are recommended for detailed evaluation.

Blasted Bedrock Zones

Explosively fractured, or blasted, bedrock zones are sometimes used to improve well yields in tight formations. The zones of enhanced permeability are created with high explosives and can either be oriented along a lineament, in which case a trench-shaped structure of elevated permeability is produced, or simply within a borehole, in which case a radial zone of elevated permeability is produced. The diameter of the elevated hydraulic conductivity zone is somewhat dependent on the size of the charge, but might normally be about 10 ft.

This technology appeared promising, in theory, and the ITRD team conducted pilot testing during the 1997 summer field season at the Tuba City site. The pilot testing consisted of two aquifer tests in a linear, blasted-bedrock zone approximately 50 ft long near well 939. Both tests failed to demonstrate conclusively that the well yields had improved. Original projections suggested that the hydraulic conductivity of the formation could improve by perhaps 2.5 orders of magnitude. Measurements taken after the blasting indicated that the actual yield might have improved by a maximum factor of 2. Since the natural variability in well yields at the site exceeds a factor of 2, the effectiveness of blasting technology was not demonstrated conclusively. Therefore, it is not recommended for further consideration.

Horizontal Wells

Horizontal well technology was originally developed in the oil and gas industry and has been applied during recent years to environmental engineering. The technique is deployed using directional drilling methods. Boreholes are initially advanced in the vertical orientation and later are turned to a horizontal orientation. Although the initial cost of installing a horizontal well is relatively high, a cost saving may result from lower O&M costs because fewer pumps are required. A prospective advantage of horizontal wells is that the technology can be used where vertical wells cannot be deployed, such as beneath the disposal cell, to accelerate the flushing and recovery of contaminants.

Horizontal well technology appeared attractive, in theory, to the ITRD team. Because of the high cost of drilling horizontal wells, it was evaluated only with model simulations. Several configurations of extraction wells were evaluated, including parallel-to-flow, perpendicular-to-flow, and Y-shaped configurations. The simulations showed that of the three configurations evaluated, the maximum yield of perhaps 55 gpm or more may be achieved with a 2,600-ft-long horizontal well deployed perpendicular to the ground-water flow direction at an elevation of 4,930 ft.

Although simulations indicate that horizontal-well technology could be effective, its implementation is considered expensive and risky. In practice, this technology might produce unprecedented difficulties because the overburden stresses at depths greater than 100 ft would greatly magnify the troubles with flowing (unconsolidated) sands. Additional concerns with this technology arise because long lengths of well screen are required and this increases the

difficulties of well completion and development. Still more difficulties could evolve later in the project as the aquifer cleanup proceeds because few options are available for sealing off the restored parts of the aquifer. Because of these risks, horizontal wells are not recommended for further evaluation.

Extraction Technologies Recommended for Detailed Evaluation

Conventional vertical recovery wells are the only extraction technology recommended for further evaluation on this project. They appear to offer the greatest flexibility in the sense that pumping may be switched on or off depending upon need, and the wells can be alternated for withdrawal and injection purposes. Service and maintenance on vertical wells is also relatively straightforward. From a risk management point of view, vertical wells are favorable because well hydraulics equations are formulated on the basis of radial flow to a well. Consequently, forecasting the performance of a vertical well field is far more predictable than with the other technologies.

The objective of well-installation would be to obtain maximum yield with a minimum of sand pumping. In order to control sand pumping most efficiently, it would be best to use pre-packed screens in the borehole. Pre-packed screens are typically used in recovery well applications in heaving sands and in silty, fine-grained sand formations. They are available in 2 and 4-inch i.d. for PVC vee-wire screens and up to 8-inch i.d. for stainless steel vee-wire screens. These screens are a relatively new technology that replaces the need for underreamed filter-pack designs. The advantage of pre-packed screens is that they require smaller boreholes than underreamed holes and also eliminate the time required to install a fine filter pack. Due to the vee-wire design, these screens offer about 3 times the open area compared to slotted pipes and permit water to enter the well at lower velocities and with reduced turbidity. The relative ease of installation of these types of screens also reduces the number of potential human errors that can cause lower well yields.

Direct rotary drilling has been used in previous well installations at the site and appears to be the most effective drilling method to use there. With rotary drilling, a rotating bit is advanced into the hole and cuttings are removed by a continuously circulating drilling fluid. Since the holes at the Tuba City site would be relatively shallow, up to 150 ft in depth, the drilling fluid might be mixed and pumped directly into the borehole from a portable pit.

8.2.3 Effluent Discharge Technologies

This section describes the various ways in which effluent from the treatment plant can be discharged. These effluent-discharge technologies have been rejected by stakeholders previously, but they are also described herein to formally document the decision process. Discharge options that do not involve reinjection include land application, evaporation, and discharge to surface water; these options result in a loss to the aquifer of origin. The loss rate would be equal to the estimated maximum pumping rate of 115 gpm (Section 8.3.1). The injection scenarios, in which the effluent is returned to the aquifer of origin, are also investigated.

Land Application

Land application is a way to treat the extracted ground water by having plants use nitrate in the water and then discharge the water by evapotranspiration. The details of land application as a treatment option are discussed in Section 8.2.4. This section summarizes the hydrologic effects associated with this option. Since it is a no-reinjection option, the maximum withdrawal rate would be about 115 gpm for about 26 years. Drawdown in the area of the plume would be approximately 60 feet. Assuming that pumping ceases when the aquifer is restored, the drawdown would recover during the subsequent 20 to 30 years. The radius of influence of the well field might be on the order of 2 miles; it would not be expected to interfere with water supply wells near Tuba City.

Evaporation

Evaporation is a technology that treats extracted ground water as it discharges the water to the atmosphere. The details of evaporation as a treatment option are discussed in Section 8.2.4. This section summarizes the hydrologic effects of this option. Since it is a no-reinjection option, the maximum withdrawal rate would be about 115 gpm for about 26 years (Section 8.3.1). Drawdown in the area of the plume would be approximately 60 ft. Assuming that pumping and evaporation ceases when the aquifer is restored, the drawdown would recover during the subsequent 20 to 30 years.

Discharge to Surface Water

Under this option the extracted and treated ground water would be discharged to Moenkopi Wash at a rate of about 115 gpm. South of the disposal cell, Moenkopi Wash is ephemeral; however, it becomes perennial as it flows toward the west. The mean baseflow in Moenkopi Wash, based on 19 years of record, is 9.0 cubic feet per second (U.S.G.S. gauge 09401260 near Moenkopi), equivalent to about 6,500 acre-feet per year. The amount of water discharged to Moenkopi Wash would be about 186 acre-feet per year (i.e., 115 gpm). This could increase average flows in Moenkopi Wash by up to 3 percent during the remediation period. However, after the remediation period, natural discharge to Moenkopi Wash from the pumped region would be less than what it is today until water levels recovered to the pre-pumping condition; complete ground water recovery could take about 20 to 30 years after pumping stopped.

Injection Wells

With this option, injection wells could be used to conduct the treated effluent directly back into the N-aquifer. Reinjection 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 feet 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.

Infiltration Gallery

An infiltration gallery could be used on the upgradient side of the disposal cell to facilitate flushing of the aquifer beneath the disposal cell. In concept, the infiltration gallery would consist of an excavated trench that is filled with a perforated drain pipe bedded in a natural granular filter, such as pea gravel. A portion of the treated effluent would be metered into the infiltration gallery and from there, the water would percolate into the ground. A mound of elevated ground water would build beneath the infiltration gallery until the delivery rate matches the lateral-flow component of the ground water. The zone of elevated hydraulic head beneath the infiltration gallery would serve as a driving force to accelerate ground-water flushing beneath the cell. Ideally, the infiltration gallery should be placed as close as practical to the northern edge of the disposal cell while controlling the hydraulic head so it remains beneath the elevation of the tailings. This technology also preserves the ground water resource at the site because the ground water is returned to the same aquifer from which it is extracted.

Effluent Discharge Technologies Recommended for Detailed Evaluation

Technologies that do not rely on reinjection include evaporation, land application; and discharge to surface water. These technologies are limited in their effectiveness because, without reinjection, the contaminated part of the aquifer can only deliver about 115 gpm. If no reinjection is used, the only method to achieve greater drawdown is with deeper wells. However, this has the undesirable consequence of drawing water in from below the plume and mixing clean ground water with contaminated portions of the aquifer. Evaporation and land application are potential treatment technologies that are discussed in Section 8.2.4. Discharge to surface water, although technically possible, is not a reasonable option since there is no apparent technical or cost benefit.

The use of reinjection wells inside the plume area boosts the pumping rate that can be realized. Reinjection into the plume surcharges the hydraulic heads in the pumping zone and allows more water to be pumped per unit time. The greater pumping rates that stem from reinjection 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.

Treated effluent can also be added to the flow system from an infiltration gallery north of the cell. The infiltration gallery would deliver treated effluent to the unsaturated zone and the water would then infiltrate under the force of gravity. The primary benefit of adding treated water north of the cell is to provide a source of clean water for flushing and rinsing beneath the cell.

8.2.4 Treatment Technologies

Many treatment processes were identified as potentially applicable for cleaning up the contaminated ground water at the Tuba City site. The processes can be categorized as follows:

- Evaporation systems.
- Distillation systems.
- Through-medium processes such as ion exchange.
- Land application (land/plant treatment process).
- Biological unit processes.
- Chemical treatment processes.
- Membrane separation processes, including reverse osmosis and nanofiltration.

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 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 will settle to the bottom of the pond, where they 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.

Although the Tuba City site is arid, low winter temperatures result in relatively low evaporation rates for much of the year. The average net evaporation rate at Tuba City is about 52 inches per year, but over half of this evaporation takes place in the months of May through August. A ground-water remediation system that uses evaporation would be designed for an extraction rate of 115 gpm. This means that the surface area of the pond must be sufficient to allow a yearly evaporation rate of 115 gpm. Such a pond would require a surface area of about 43 acres, which is almost all of the open area inside the present fenced compound at Tuba City. For that reason, simple solar evaporation was not considered for detailed evaluation.

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. However, addition of a spray system increases the complexity of the system and requires more maintenance and operator attention than simple solar evaporation.

Evaporation, whether it is simple solar or spray, is generally a very low-cost way to remediate large amounts of contaminated water in arid climates. Major disadvantages are that the water is

lost to the aquifer. However, due to its low cost and relative simplicity of operation, spray-enhanced evaporation was selected for detailed evaluation as an alternative.

Distillation Systems

In a simple distillation process, water is vaporized by heating it to its boiling point in a heated chamber. The water vapors are then condensed in a separate vessel. Nonvolatile contaminants such as nitrates, sulfates, uranium, and other components of TDS will not evaporate and will concentrate in the chamber. The condensed water can be reinjected into the aquifer. The concentrate, or brine, may be taken off site for disposal; alternately, it may be evaporated to dryness in a small solar pond and the residue can then be disposed of as a solid.

Compared to solar or spray evaporation, distillation is more expensive to install and much more 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 reinjection back to the aquifer and the product water is of very high quality. Energy requirements for small distillation units can be greatly reduced by the use of “vapor recompression,” in which the heat given off by condensation of the water vapor is recovered in a fan or compressor and used to preheat the feed water.

Vapor-recompression evaporation units typically require one-fourth or less as much heat energy as would a simple single-effect distillation unit. Fouling, which commonly occurs in distillation units where the feed water contains high concentrations of dissolved solids that deposit on the hot surfaces of the evaporators, can be eliminated or greatly reduced by operating the evaporator unit under vacuum, thus reducing the evaporation temperature.

An alternative to conventional evaporation technology is the “hydrosonic pump,” which was developed during the last decade by Hydro Dynamics Corp. The hydrosonic pump generates heat by allowing a fluid to reach a point of incipient cavitation under controlled conditions. Heat is generated when microscopic bubbles form at the low-pressure suction side of the pump and then collapse when the pressure is increased as the liquid is pushed towards the pump discharge. The hydrosonic pump captures this heat and uses it to raise the temperature of the water. After the water has made a number of passes through the pump, its temperature will have reached the point where it can be “flashed” (quickly reduced in pressure) to produce steam. Since there is no external heat source, there are no hot metal surfaces, so fouling is not a problem. And since there is no flame, the hydrosonic pump does not require certification and operation as an ignition source.

Despite these advantages, however, the hydrosonic pump is a new technology and has not yet been incorporated into vapor-recompression evaporation systems. The energy efficiency of the hydrosonic pump by itself is not competitive with such systems. Hydro Dynamics does not make such systems, and companies that do make them have not yet begun to employ hydrosonic pumps.

A treatment unit incorporating the hydrosonic pump would have to be engineered “from scratch,” which would put it at a substantial disadvantage from both the cost and reliability standpoints compared to commercially-available pre-engineered distillation systems. Therefore, although the hydrosonic pump shows promise, it is too immature as a technology to be considered at this time.

Another recent innovation in distillation technology is an evaporation unit, which incorporates both falling-film and vapor-recompression technology. This process, which was developed in Finland and is marketed in the United States uses flexible polymeric evaporation elements that greatly reduce fouling and prolong operating cycles. It also uses a low-speed fan for internal vapor recompression. The system is designed for outdoor installation with no building requirement and is instrumented to require a minimum of operator attention.

One of the most attractive features of these units is extremely low energy consumption. A standard boiler requires 980 British thermal units (Btu) to evaporate a pound of steam. The falling-film and vapor-recompression technology requires only about 15 Btu to process a pound of water. This order-of-magnitude reduction in energy consumption makes the cost of operating a distillation process competitive with the cost of operating other traditional treatment processes.

Because of the many attractive features of the process, distillation was selected for detailed evaluation as an alternative.

Through-Medium Processes

In a through-medium process, a stream of water 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 land application. 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. Another medium for removal of uranium from water streams is zero-valent iron, which creates strongly reducing conditions that reduce uranium to an insoluble form as it passes through the medium in the vessel. Use of zero-valent iron is currently being evaluated for cleanup of contaminated creek water downstream of the former uranium millsite in Monticello, Utah.

Media used in ion-exchange processes are very susceptible to clogging and plugging due to solids buildup from turbidity (cloudiness) resulting from fine solids in the water stream. The ground water at the Tuba City site is oversaturated with respect to calcium carbonate ($CaCO_3$) and calcium sulfate ($CaSO_4$), and particles and flocs of these minerals often cause high turbidity. Hence, pretreatment of the influent water, using precipitation or coagulation, settling, and filtration to remove the excess calcium compounds, may be necessary to prevent clogging. However, if extensive chemical pretreatment is required before through-medium treatment, the latter process becomes redundant, as chemical treatment will also remove uranium as well as other constituents from solution.

Synthetic ion-exchange resins have a finite capacity for holding ions, and must be regenerated when that capacity is exceeded. A relatively high degree of uncertainty is associated with disposal of regenerant solution (brine), as well as disposal of spent medium. The brine can be stored temporarily on site in a lined pond or pit and dewatered on site. An additional area of concern is radioactive contamination of the medium. The activity from uranium contamination cannot exceed 2,000 picocuries per gram (pCi/g), and the medium cannot contain more than 0.05 percent uranium by weight. If the uranium concentration exceeds these levels, final disposal of the contaminated medium is highly problematic.

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 plume is between two and seven times this amount. An ion-exchange unit treating the Tuba City ground water would require regeneration approximately every 25 to 30 bed volumes, and the regeneration would produce approximately 5 bed volumes of waste liquid with high salt content. Thus, ion exchange is a poor choice for a remediation technology for Tuba City.

Zero-valent iron would be very effective for removing the uranium from the water, but it is not effective for removal of nitrate or sulfate. Since removal of uranium is the least problematic aspect of the remediation process, zero-valent iron is not a promising candidate for use in the Tuba City remediation program. Therefore, neither of these through-medium processes were chosen for detailed evaluation as an alternative.

Biological Unit Processes

Biological unit processes use bacteria either in the flow stream of a water source or in situ to convert hazardous compounds to other forms that are less hazardous or more amenable to disposal. Such processes have not been widely accepted by municipal water utilities, because effluents produced by biological treatment are not usually regarded as potable. However, biological unit processes are in widespread use in treatment of wastewaters by municipalities and other operators of wastewater disposal facilities, where effluents are typically discharged into rivers or reinjected into aquifers rather than being used directly as drinking water.

The COCs in the N-aquifer that are amenable to treatment with biological processes are nitrate and sulfate. Biological denitrification can eventually reduce nitrate levels in water to less than the MCL or to background levels. However, the rate of the denitrification reaction is not particularly fast, and rapid reductions in nitrate level may require a polishing step with unit processes such as ion exchange or reverse osmosis. If such a polishing step is required, then a disinfecting step must be added to the system to prevent biological fouling of the ion exchange medium or the polymeric membrane used for reverse osmosis. Also, certain conditions in the influent water can affect the removal rate of nitrate. Given the high loading rate of nitrate expected in the influent, increasing alkalinity may have an inhibitory, or rate-limiting, effect on denitrification.

Ground water at the Tuba City site is contaminated with sulfate at levels varying from 550 mg/L to nearly 4,000 mg/L. The average influent (to the treatment plant) level of sulfate is assumed to be 1,700 mg/L. A biological unit process can reduce sulfate levels in water to less than 500 mg/L. But the likelihood of lowering the sulfate level to the treatment goal of 250 mg/L in a bioreactor is not nearly as certain as is the likelihood of biological denitrification being able to meet the nitrate standard. The fastest (most favorable) desulfurization reaction rates are obtained when sulfate concentrations are greater than about 1,000 mg/L; at sulfate levels below about 300 mg/L, reaction rates for biological reduction of sulfate appear to be unacceptably low. Hence, removal of sulfate to the extent necessary to meet the proposed treatment standard using biological desulfurization alone is unlikely, and a polishing step with unit processes such as ion exchange or reverse osmosis will almost certainly be required.

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. Biological desulfurization, however, produces hydrogen sulfide (H_2S) as a byproduct. Hydrogen sulfide is malodorous, explosive, and extremely toxic. Conceptually, biological desulfurization involves treating for a substance that poses a relatively mild health risk—sulfate at high levels is known to cause short-term diarrhea in susceptible individuals—and in the process, creating a substance that poses a substantial risk of death. Whereas nitrogen gas can be freely discharged to the atmosphere, 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.

Denitrification or desulfurization may be done either in a pond or in a biological reactor (or series of reactors). The biological sludge, which is the residue from the treatment process, can be dewatered using a filter press. Because the sludge is biologically inert, there is a moderate to high probability that it can be disposed of on site by land application of the material. It will have to be tested for the presence of pathogenic organisms before disposal, however.

The preliminary evaluation of alternatives for remediation of the Tuba City UMTRA site investigated biological remediation both in ponds and in batch reactors. Biological remediation cannot be a “stand-alone” process for the site because it does not treat for uranium, and therefore it must be coupled with a uranium-treatment process such as ion exchange or chemical treatment. The uranium-treatment technology would preferably be located upstream of the biological process so that the biological sludge is not contaminated with uranium.

Biological denitrification is an attractive process for removing nitrate, particularly considering that alternatives such as ion exchange and chemical treatment are either largely ineffective for removing nitrate or are difficult to implement because of the high nitrate loadings in the N-aquifer. However, biological desulfurization is not nearly as attractive, because it does not appear to be capable of meeting the treatment goal of 250 mg/L sulfate, and because of the problems associated with handling and disposing of the by-product H_2S. The bacteria used for denitrification will also consume sulfate, but at a slower rate than the denitrification process. Thus, removal of sulfate must precede the biological denitrification process, or else the denitrification reactor must be carefully designed and carefully operated to ensure a residence time sufficiently short that desulfurization and H_2S generation are not significant. Because biological denitrification is an effective method it will be retained for detailed evaluation. However, it will need to be combined with other treatment processes, such as chemical treatment or nanofiltration (technologies described in later sections).

Land Treatment System

The land applications system considered for the Tuba City remedial action is a slow-rate infiltration system that uses contaminated water to irrigate tolerant crops such as feed corn and pasture plants or grasses or tolerant indigenous rangeland plants and grasses. Such methods are a proven approach for treating municipal and agricultural wastewater that contains contaminants similar to those in the N-aquifer at the Tuba City site.

In the land treatment system, nitrate in the water is 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, and these compounds are converted to amino acids. Amino acids are the building blocks for complex nitrogenous compounds, which are essential for maintenance and growth of plant cells.

Sprinkler or ridge-and-furrow systems are used for distributing the extracted ground water. Application of contaminated water is made throughout the growing season. During the remainder of the year, contaminated water is either stored in a surface pond, or the system is shut down. Application rates will depend on soil permeability and the kind of plants used to uptake nitrate.

Pretreatment of the ground water before land application is required for removal of uranium and possibly for removal of other trace metals and metalloids. Sulfate and dissolved solids other than nitrates will not evaporate, nor will they be taken up by the plants, so they will tend to accumulate on the plant and ground surfaces. The ability of the soil to tolerate high loading of sulfate and other dissolved solids is moderately uncertain. The ability to maintain sufficient control of the application rate to avoid deep percolation of contaminated water also is moderately uncertain and monitoring for deep percolation discharges from a land treatment system may be difficult.

Because no wastes or residuals are generated by the land application treatment system, the system itself has no waste disposal issues associated with it. However, because land application treats only nitrate, the uranium and sulfate components must be removed by other technologies, and the waste generated by those technologies must be considered in a complete evaluation.

Land application could be viewed as an alternative to biological denitrification, and like the biological process it requires separate processing to remove uranium and sulfate. A successful land application process should have sulfate levels removed to the lowest extent reasonable, since the land application will not process sulfate, which will remain in the system as described above.

A final consideration in implementing a land application system is the inherent seasonality of such a process. A land application system can only be operational during the growing season, and cannot be run at all during times of freezing weather. For the Tuba City site, this means that at most a land application system can only be operated from mid-March until October. During the rest of the year, the land application system and the upstream desulfurization process would be shut down. The extraction system would either have to be shut down, or the water pumped during the time the land application system was out of service would have to be stored until the land application system was back in operation.

This last factor makes a land application system impractical. If the land application system could operate from April 1 to October 15, it would be out of service for 166 days. At the design maximum extraction rate of 115 gpm, a double-lined holding pond having a capacity of approximately 28 million gallons would be required for storage of water purged during the winter months. Such a pond would have a surface area of over 8 acres. Further, the land application system would have to be sized for a flow rate of at least 210 gpm to account for the time when it would be out of service during the winter. Consequently, land application was not selected for detailed evaluation as an alternative.

Chemical Treatment

Chemical treatment is typically a system that uses precipitation, coagulation and flocculation, gravity settling, and filtration processes. Such systems are widely used for treatment of contaminated ground waters from former uranium mill sites.

A typical chemical treatment process for the COCs in the Tuba City ground water might consist of the following steps:

(1) Addition of acid to lower the pH of the influent water to about 4.5.

(2) Addition of iron sulfate (or other iron source) as flocculent to precipitate uranium.

(3) Addition of barium chloride (or other barium source) to precipitate sulfate as barium sulfate.

(4) Addition of lime (CaO), increasing the pH to about 10, to promote precipitation of calcium and other metals as metal hydroxides.

(5) Filtration through fine membrane filters to remove solids formed in the precipitation reactions.

(6) Addition of acid or carbon dioxide to adjust the pH to a value of approximately 7 for final disposal.

This process does not address removal of nitrate. Removal of nitrate could be accomplished using a biological denitrification process (see “Biological Unit Processes” earlier in this section), downstream of the chemical process. The removal of sulfates 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.

Compared to other treatment processes, chemical treatment requires large amounts of raw material in the form of iron, barium, lime, and acid chemicals. Barium compounds currently cost about $1.00 per pound; to reduce the sulfate concentration in the N-aquifer from 1,700 mg/L, the approximate average concentration in the aquifer, to the treatment goal of 250 mg/L would cost almost $31 per 1,000 gallons of water for barium alone. Chemical treatment is also more labor intensive than some other processes because of the need for constant operator attention and adjustment of pH. The principal by-product of chemical treatment processes is a sludge that contains uranium, calcium hydroxides, and the barium sulfate. The significant volume and high uranium content of this sludge may create disposal problems.

DOE owns a 200-gpm chemical treatment facility that is currently in storage in Grand Junction, Colorado. Using this facility would greatly reduce the capital costs of the chemical treatment process and make this process economically attractive. Chemical treatment is not effective against nitrate, however, so it must be coupled with another treatment technology for nitrate removal. Because chemical treatment can remove sulfate, it makes it attractive as a precursor for biological denitrification. Accordingly, a process combining chemical treatment with biological denitrification was selected for detailed evaluation as an alternative.

Membrane Separation Processes

Membrane separation includes all processes in which fine filters are employed, such as ultra-filtration, nanofiltration, reverse osmosis (RO), and electro-dialysis reversal.

Typically, the most effective membrane separation process for treatment of ground water is RO, which can remove sulfate, nitrate, and chloride ions to a greater extent than other membrane-separation processes. The product water from an RO unit is generally of high quality. The hazardous constituents are retained for disposal in a reject-water stream.

Disadvantages of RO units are the relatively high capital costs, the need for pretreatment to remove hardness that would foul the membranes, and the large volume of reject water that will be generated due to the high concentrations of dissolved solids in the Tuba City ground water. The quantity of reject water produced by an RO unit treating Tuba City ground water is estimated at 35 percent of the feed. The reject water would have to be treated separately, by distillation for example, or pumped to a solar pond for evaporation. Neither option is attractive. Further, vendors of reverse osmosis units have expressed doubts that RO technology can meet the nitrate treatment standard of the Tuba City remediation project.

Normally, if RO is technically unacceptable as a remediation technology, nanofiltration would also be unacceptable, since RO can generally achieve higher product-water quality than can nanofiltration. Although nanofiltration is less effective than RO in general, it is effective against sulfate and can easily meet the Tuba City treatment goal for sulfate. This is significant because biological denitrification is very effective at treating nitrate but requires prior removal of sulfate. Nanofiltration thus is attractive as a pretreatment for that process.

Compared to RO, nanofiltration has somewhat lower capital costs, significantly lower maintenance costs, and produces only about half as much reject water. Therefore, it is better suited as a pretreatment process for this application than is RO. Accordingly, a process combining nanofiltration with biological denitrification was selected for detailed evaluation as an 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. The objective of the pumping is to furnish two pore volumes of uranium-contaminated water, or approximately 1.56 billion gallons, to the treatment plant within approximately 20 years.

Pumping Alternative 1—Plume-Focused Extraction Wells without Reinjection

Pumping alternative 1 consists of a recovery-well field inside the plume area. Up to 120 ground-water recovery wells would be required to remove two pore volumes of uranium-contaminated water within approximately 20 years without reinjection.

Effectiveness

Figure 8–1 illustrates the distribution of recovery wells that might be considered to achieve compliance with the ground-water protection standards. Particle-tracking simulations depicted on Figure 8–1 show that the well field could eventually capture the area circumscribed by the plume. Pumping rates estimated with numerical simulations indicate that the proposed well field could recover approximately 115 gpm on a sustained basis. The estimated pumping rate can be used to compute the time required to pump 1.6 billion gallons; calculated at 26 years.

Actual pumping rates would be determined in the field after the wells were emplaced. Geochemical testing performed during summer 1997 in conjunction with the ITRD project showed that most of the uranium in the bleached portion of the aquifer is in solution, and that nitrate and sulfate are also in solution (MSE 1997). On the basis of these results, the prospects for significant mass removal within the plume appear favorable.

The principal limitation with this scenario is envisioned to be the large unsaturated zone that would develop within the plume as pumping continues. To maintain the 115-gpm pumping rate, drawdowns in the plume would be on the order of 60 ft. Consequently, formation material within the dewatered zone may contain a large residual fraction of contaminant mass. Moreover, because ground-water extraction would be performed without reinjection, it would result in a net loss to the ground-water resource. The amount of the loss would be 1.6 billion gallons (3,700 acre-feet) over the 26-year pumping period. This loss, however, would not measurably reduce ground-water discharge into the Moenkopi Wash because it would be removed mostly from the water that would otherwise be lost to evapotranspiration. Ground-water discharge from the width of aquifer transected by the pumped zone would decline by a small factor during the remediation period. The tributary width of aquifer transected by the pumping (based on a 2-mile radius of influence) would be perhaps 4 miles; the total width of aquifer discharge into Moenkopi Wash is several tens of miles.

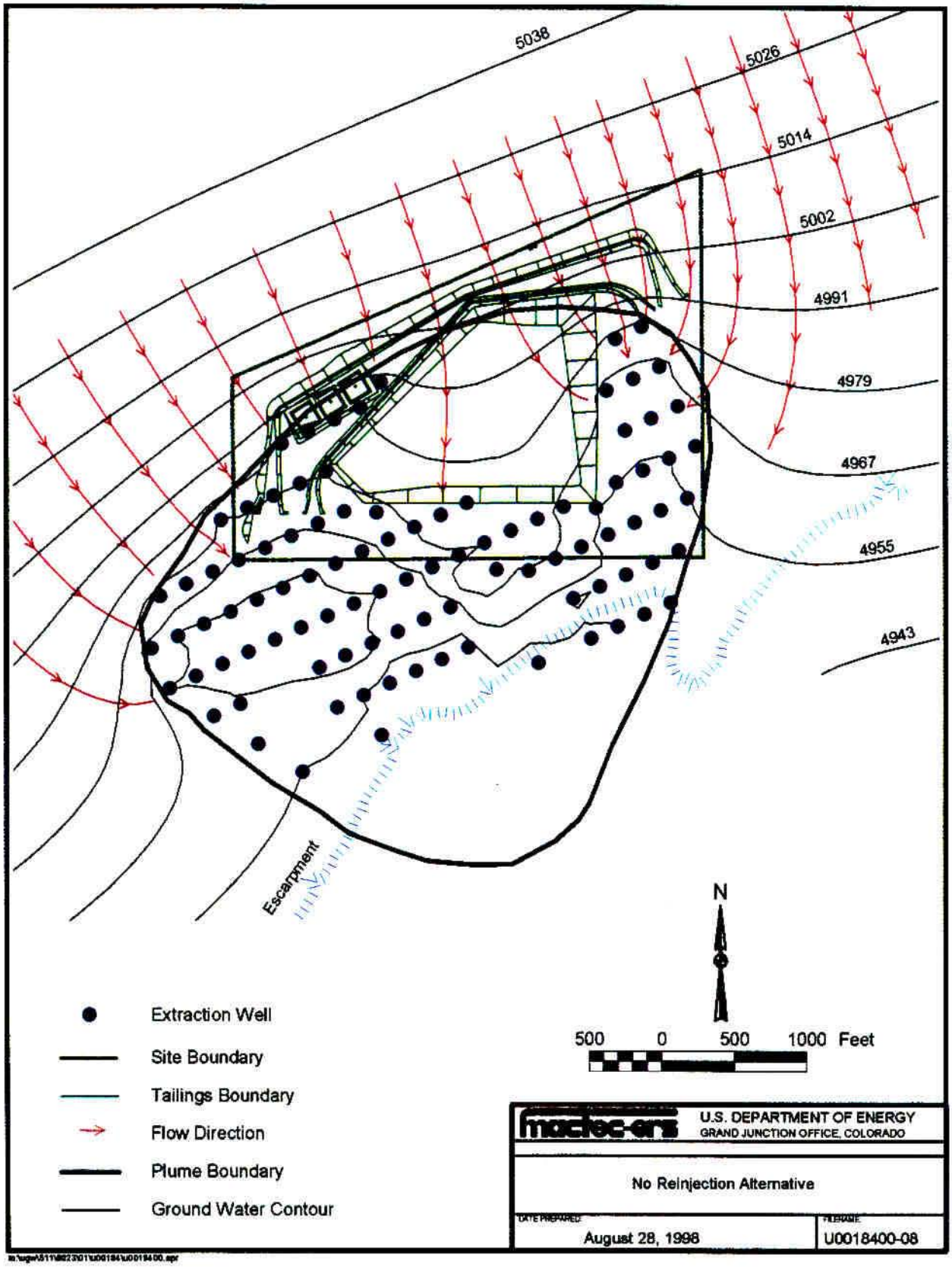


Figure 8–1. No Reinjection Alternative

Implementability

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 weakly cemented and friable nature of the fine-grained eolian sandstone presents its 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 120 wells, pumps, and piping to direct the water from the well field to the treatment system, is estimated at $3.95 million. Annual O&M costs are estimated at $0.389 million. The net present value for this pumping alternative, calculated over the 26-year project lifetime, is estimated at $8.29 million.

Pumping Alternative 2—Extraction and Injection Wells with an Infiltration Trench

The objective of this pumping alternative is to achieve aquifer restoration without depleting the ground-water resource beyond treatment-plant losses. Water would be pumped from the aquifer using vertical recovery wells, treated, and injected back into the plume with injection wells and an infiltration trench located upgradient of the disposal cell.

Figures 8–2 and 8–3 illustrate the conceptual layout of the well field. As shown on these illustrations, the reinjected ground water would be pumped into the downgradient portion of the plume to control its migration, similar to the “line-drive” approach used conventionally in the solution mining industry (Driscoll 1987, Roberts 1980). Water introduced into the infiltration trench would bring about flushing beneath the disposal cell. Returning the treated ground water to the plume would control drawdowns, accelerate flushing within the plume, and accelerate aquifer restoration. When completely deployed, this system could consist of up to 30 extraction wells and 30 injection wells, plus an infiltration trench about 2,200 ft in length located north of the disposal cell. The working assumption is that the extraction wells would operate at an average pumping rate of about 7 gpm, the injection wells would operate at an average of 6 gpm, and the trench would receive any water that the injection wells could not accept.

Effectiveness

The combination of extraction and reinjection within the plume is the most expedient method to move water through the contaminated part of the 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. On the basis of geochemical tests performed during the summer of 1997, the distribution coefficient (Kd) for uranium is very low at the site. This indicates that the majority of the contaminant mass is in solution and that most of the mass can be removed by pumping one plume volume. The high concentrations of sulfate, carbonate, and dissolved oxygen in the plume should help the uranium

to remain in solution as pumping continues. Since nitrate and sulfate are dissolved species in the ground water, they are also expected to decline very substantially with the removal of one pore volume.

Implementability

There are no technical or administrative issues that would preclude implementation of the extraction wells, reinjection wells, and infiltration trench associated with this pumping alternative. These are conventional technologies that would be relatively straightforward to implement and would use readily available technology. The weakly cemented and friable sandstone of the N-aquifer will present some technical difficulties when installing the extraction and reinjection wells. Well design, construction, and development techniques to specifically control sand pumping would be required. Also, operation of the extraction system of 30 wells and reinjection system of 30 wells and reinfiltration trench will require oversight. Administrative issues associated with implementing this pumping alternative would be minimal. A permit from the Navajo Nation may be required to extract the ground water.

Cost

The total capital cost for this pumping alternative, including all extraction and reinjection wells, pumps, the reinjection trench, and piping to direct the water from the well field to the treatment system and from the treatment system back to the reinjection wells and infiltration trench, is estimated at $2.23 million. Annual O&M costs are estimated at $0.46 million. The net present value for this pumping alternative, calculated over the 16-year project lifetime, is estimated at $6.42 million.

The recommended pumping alternative is pumping alternative 2. This pumping alternative preserves the best technical options and combines them into one comprehensive pumping alternative.

8.3.2 Treatment Alternatives

Treatment alternatives make use of the treatment technologies evaluated previously. Treatment technologies retained for detailed evaluation were combined, when required, so that all COCs in the waste stream would be treated.

The treatment alternatives evaluated in this section are:

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement.
- Treatment Alternative 2—Distillation.
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification.
- Treatment Alternative 4——Nanofiltration with Biological Denitrification.

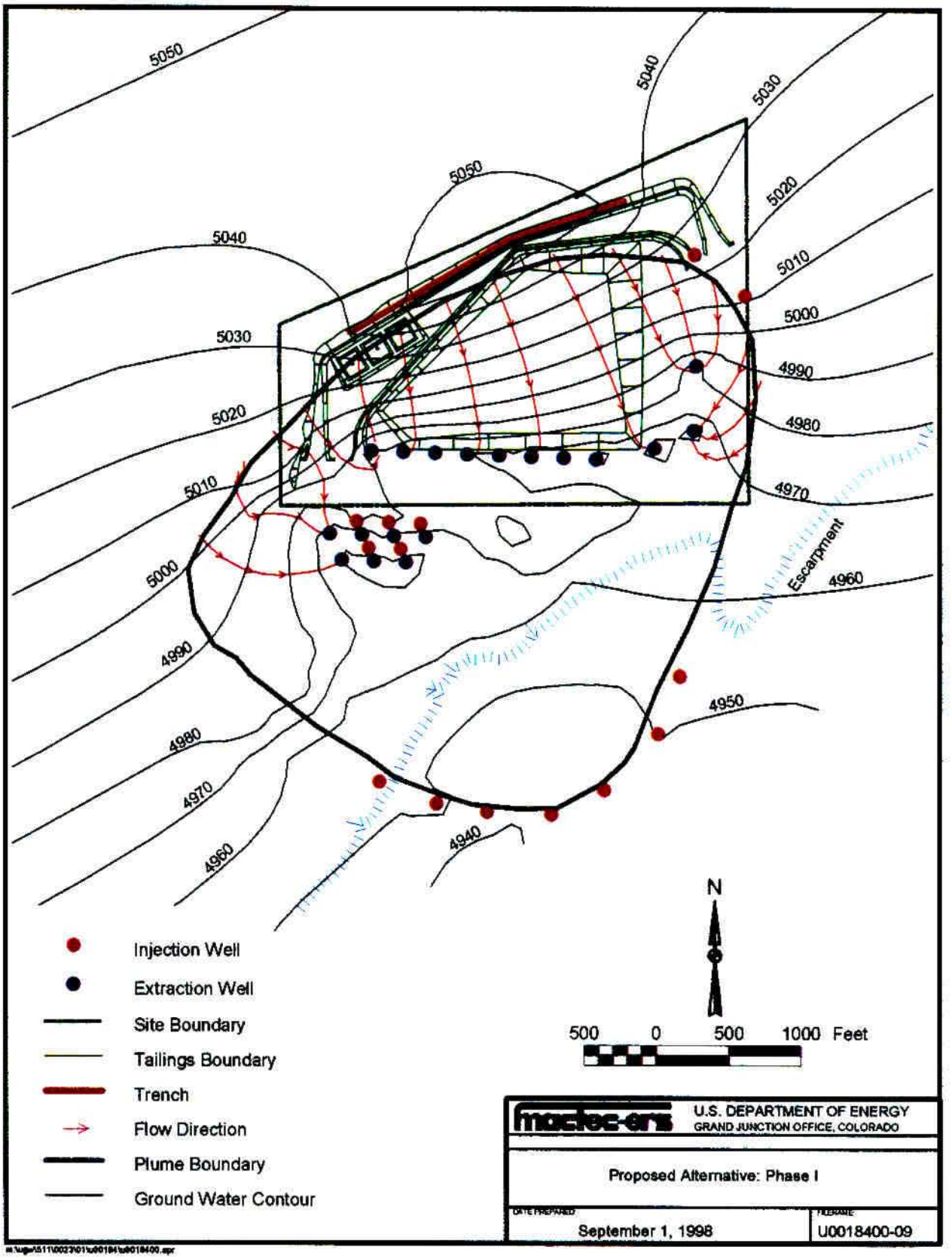


Figure 8–2. Proposed Alternative: Phase I

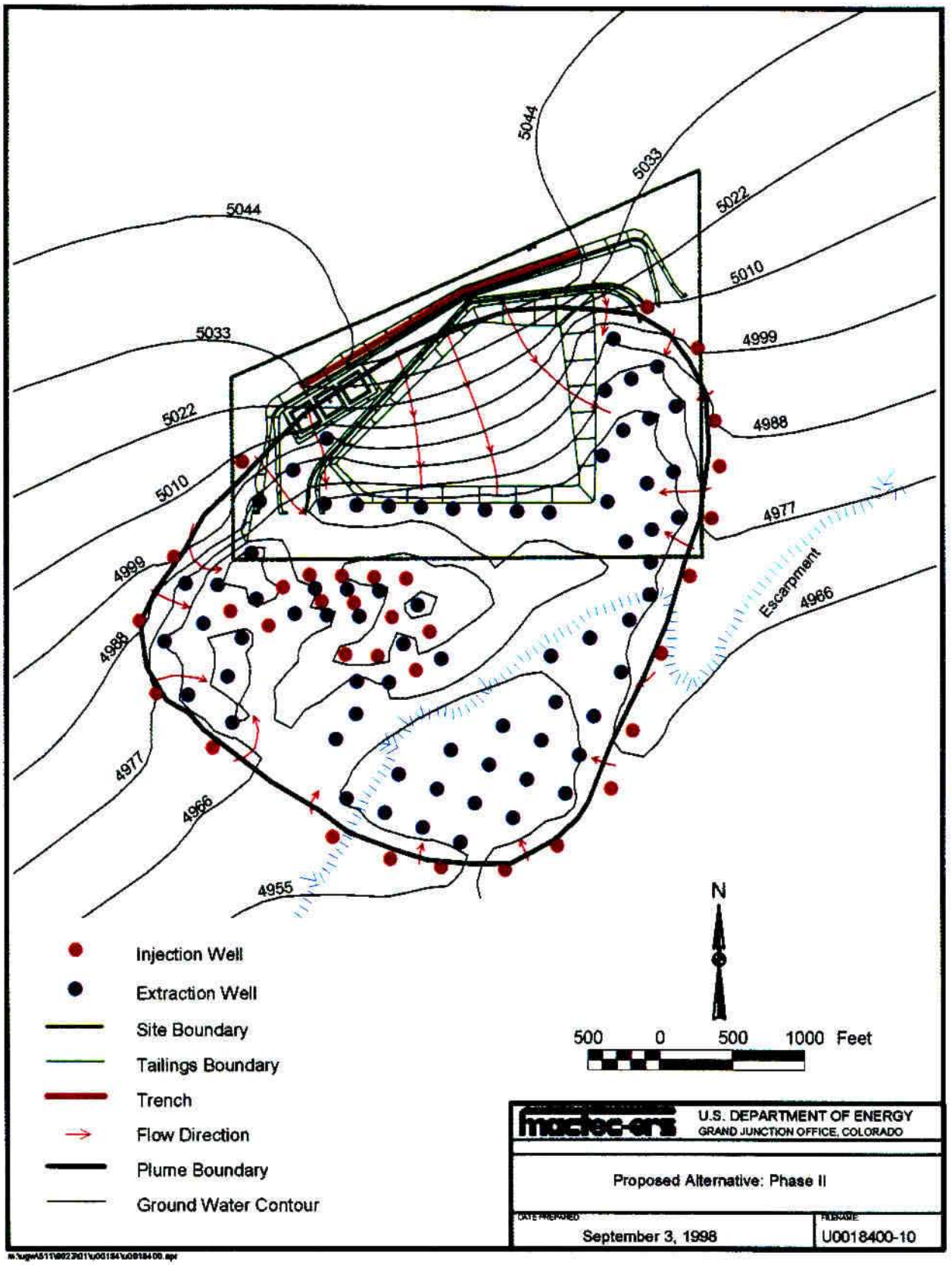


Figure 8–3. Proposed Alternative: Phase II

Common Components

Although the processes used in the treatment alternatives differ in many aspects, certain elements are common to all. To eliminate repetition in the discussions of the various treatment alternatives, those common elements are discussed in this section.

Extraction and Reinjection

Treatment Alternative 1—Solar Evaporation with Spray Enhancement, does not produce treated effluent water, and therefore reinjection will not be possible if this treatment alternative is chosen. Solar evaporation and spray enhancement would use Pumping Alternative 1—Plume-Focused Extraction Wells without Reinjection. All other treatment alternatives will use Pumping Alternative 2—Extraction and Injection Wells with an Infiltration Trench. The design treatment rate for pumping alternative 1 is 115 gpm; the design treatment rate is 200 gpm for the other pumping alternatives.

Ponds

All the treatment alternatives generate some sort of sludge that will require ponds for interim storage and dewatering before final disposal. Some of the treatment alternatives require ponds for other purposes. Specific requirements are covered in the discussions of the individual treatment alternatives.

The Tuba City site currently has three ponds; each has a capacity of approximately 1.3 million gallons. One is double-lined and the other two are single-lined.

Pond construction and relining will be relatively simple. The likelihood of encountering technical problems leading to significant schedule delays during construction is low. The heavy-construction equipment, equipment operators, and liner installers are available on the reservation or through Native-American-owned firms. Liner materials are available within a 250-mile radius of the site. Adding capacity or redundancy to the system in the future will require building more ponds.

Sludge Disposal

All the treatment processes produce a sludge that must be disposed of. The most likely disposal site for both the sludge and the pond liners, which will be removed and disposed of at the conclusion of the remediation project, is the Cheney Disposal Site near Grand Junction, Colorado. Coordination with the Navajo Nation, the U.S. Department of Transportation, and the Departments of Transportation of affected states will be required before transport of treatment plant wastes from the site to their final disposal location.

Cost and Design Basis

Cost estimates for all treatment processes will be compared on the basis of a net present worth, calculated using the OMB standard discount rate of 7 percent. Costs for treatment alternative 1 will be calculated over a total project life of 26 years; costs for the other three treatment alternatives will be calculated over a total project life of 16 years.

Treatment Alternative 1—Solar Evaporation with Spray Enhancement

Effectiveness

The evaporation rate for a spray nozzle designed for continuing operation under high solids loading levels is about 5 to 10 percent water loss per pass through the nozzle (Bete Fog Nozzle 1994). The addition of a spray system that employs 195 nozzles, each with a flow capacity of 40 gpm (hence, an evaporation rate of 2 gpm, or 5 percent of the total flow), adds an additional 390 gpm evaporation capability to the system during the time the spray system is in operation and enables the required pond size to be reduced to around 2 acres.

A disadvantage of a spray system, as opposed to a simple evaporation system, is that the water in the simple evaporation pond may be evaporated to dryness, so only a single large pond is required. A spray system, however, can only be operated as long as the pond contents remain liquid. Once the liquid in the pond reaches a certain concentration of solids, the efficiency of the spray system begins to drop dramatically. The concentration of solids at this point is still low enough that disposal is impractical without further concentration. The sludge mixture must be evaporated further by solar evaporation before disposal. This can be done either by taking the spray pond out of service and letting the water evaporate in the pond or by transferring the sludge to a second pond for dewatering. The more practical approach is to transfer the sludge to a second pond, since this leaves the spray pond in service. The capital cost of a sludge pond is considerably less than that of a second spray pond, and the operating cost of the sludge pond is almost zero, with only occasional inspections required.

Usually spray evaporation systems cannot be operated when wind speeds exceed 15 knots (17 miles per hour). At such times the sprays would be shut off and the pond operated as a solar evaporation pond.

Evaporation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The only residual produced is the concentrated sludge, and the volume of the sludge is minimized compared to the other treatment alternative processes under evaluation because no additional chemicals are required for the evaporation process. However, the evaporation process does not generate an effluent water stream that can be reinjected into the aquifer. The lack of reinjection water limits the maximum rate of extraction, which greatly increases the time required for remediation. Also, as discussed in Section 8.3, if no reinjection takes place, a large unsaturated zone will develop within the plume. And since one function of the reinjection water is to contain the contaminant plume and direct the contaminated water towards the extraction wells, the evaporation process may be less effective in this respect.

Implementability

Addition of a spray system to an evaporation pond adds complexity and requires a significantly higher degree of oversight than a simple solar evaporation system. In principle, a spray system could operate continuously, although the rate of evaporation from the sprays would be reduced at night. However, the Tuba City system would spray water that is contaminated with radionuclides into the atmosphere. The initial concentration of radionuclides in the water is low but would increase significantly as the pond contents became more concentrated. Operating such a system without continuous monitoring could result in loss of radionuclide containment, especially during periods of high wind.

The system design then becomes a tradeoff between capital and operating costs. A pond designed for continuous operation of the spray system would be smaller and less costly to construct than one sized to operate only during daylight hours or during a day shift. However, the Tuba City system is intended to operate for up to 26 years. Over this length of time, the cost of staffing for continuous operation becomes greater than the additional capital cost for building a larger pond.

The system design parameters are:

- Influent rate of 115 gpm.
- Operation of spray system 8 hours per day, 5 days per week.
- System reliability of 85 percent. This allows for equipment breakdowns and nozzle plugging, which would affect parts of the system at different times, and shutdown of the entire system during high-wind periods.
- Evaporation rates varying from 5 percent of total flow during periods of low solar evaporation to 10 percent of total flow during periods of high solar evaporation.

With these design conditions, the required pond area is approximately 2 acres, and about 195 nozzles are required. Surface areas of existing ponds at Tuba City are only about 0.4 acres each, so a 2-acre, double-lined pond will have to be constructed. The existing double-lined pond has adequate area and volume to serve as the final sludge concentration pond. The two existing single-lined ponds will not be required for the spray evaporation process.

Operating the system will require the following principal functions: embankment inspection and maintenance, liner inspection and repair, water-level monitoring, circulation pump monitoring and maintenance, spray system monitoring and maintenance, and monitoring for leaks. All these functions can be performed by a single operator working 5 days per week during the day shift. The first three functions can be performed with periodic inspections. 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 sumps 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. Maintenance and repair of pumps and

spray nozzles is an on-site function, but round-the-clock presence of maintenance personnel is not required because the spray system will not be operated continuously.

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 large pond used in this treatment system is unlikely because the water level changes relatively slowly due to the large size of the pond, and because the pond 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.

Waste disposal will not be an ongoing function for the evaporation system because the bulk of the concentrated sludge can be disposed of at the end of the remedial action. Final disposal will entail stabilizing and removing 30,000 to 40,000 tons of sludge from the ponds and transporting the mass to an authorized disposal site. The pond liner system will also be removed and disposed of at an authorized disposal site at the end of its service life.

Cost

The capital cost of the 115-gpm spray evaporation system, including the direct capital cost of building the new pond and spray system, and all indirect costs associated with design, such as subcontractor monitoring, is estimated at $0.94 million. The annual O&M cost of the spray system, including utilities, operator time, and expected maintenance, is estimated at $0.74 million. The estimated present worth value of treatment alternative 1, projected over the total estimated treatment time of 26 years, is $9.63 million dollars.

Treatment Alternative 2—Distillation

Effectiveness

Evaporation and water recovery using distillation units is an established and proven technology for treatment of contaminated water. A distillation unit will consistently produce an effluent containing less than 50 mg/L dissolved solids and will often meet or exceed drinking water standards with no further treatment. The volume of 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 volume of the feed.

The following values represent influent and effluent quality from a distillation system that employs a Hadwaco falling-film distillation unit. These values indicate that the treated effluent from the distillation system will be able to meet or exceed the applicable treatment standards. A layout sketch of the Hadwaco system is included as Figure 8–4.

Measurement	Units	Influent	Effluent
Conductivity	millisiemens per meter	402	2.8
pH	–	7.3	7.6
Chemical oxygen demand	mg MgO_2/L	227	< 30
Ammonium	mg NH_4 as N per liter	120	0.1
TDS	mg/L	2,200	< 1
Total suspended solids	mg/L	47	< 1

The high alkalinity of the feed water will require some pretreatment to minimize the potential for scaling or clogging. Carbonate in the feed water will be converted to carbon dioxide in the concentration cycle. Carbon dioxide is a noncondensable gas that will accumulate in the vapor recompression cycle and reduce its efficiency. Pretreatment with acid will remove the excess alkalinity and enable the system to perform in its optimum range.

Distillation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The treated effluent is of high quality, and the quality of the concentrated brine is equal to or better than that produced by other processes.

Implementability

Hadwaco 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. Operation of the unit can be monitored at a remote location using the instrumentation and software provided as part of the package. The electricity demand of the Hadwaco distillation unit is low enough that no additional electrical power equipment will be required beyond what can be easily made available at the Tuba City site.

The Hadwaco units are reliable and generally require a low level of oversight and 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, and will require regular maintenance primarily of the flexible polymer tube elements. 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.

The pond requirements for implementing the distillation process are described in “Common Components” at the beginning of this section. For a general discussion of pond implementability,

refer to “Treatment Alternative 1—Solar Evaporation with Spray Enhancement” under “Implementability” in this section.

The two Hadwaco distillation units that are being evaluated for this treatment system have a combined capacity of 220 gpm; the currently planned maximum flow of the extraction system is 200 gpm. This provides about ten percent margin for system downtime. Increasing the capacity of the overall system above 220 gpm will require addition of more distillation units.

The distillation process will require one or two double-lined ponds for sludge dewatering. The pond sizing will be determined during the detailed design of the treatment system. Fiberglass or lined-steel holding tanks will be provided for the feed for the distillation unit, and the treated water produced by it.

Cost

The cost estimate assumes that a single Hadwaco model 600E distillation system, with a capacity of 110 gpm, will be purchased and installed when the first phase of extraction and remediation begins. During the second phase, when extraction capability is increased, a Hadwaco model 600E distillation system will be purchased and put into service in parallel with the first 600E. The combination of these two units will provide adequate treatment capacity when the extraction system reaches its maximum output.

With this assumption, the capital cost of the distillation system is estimated at $5.55 million, and annual operating costs will be about $1.59 million. The present worth cost of this treatment alternative projected over the total estimated time of 16 years is $20.17 million. The most expensive capital items are the two Hadwaco distillation units, which will cost about $3.8 million dollars. The most expensive O&M line item is electricity to operate the Hadwaco unit, which is estimated at $520,000 per year.

Although the use of Hadwaco units was assumed for the purpose of this cost estimate, the final distillation unit supplier will be selected by a competitive bidding process during the detailed design of the system.

Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Effectiveness

The estimated total hardness of contaminated ground water at the Tuba City site is 2,000 to 2,500 mg/L as $CaCO_3$, and the TDS of the influent water exceeds 3,500 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. Addition of barium will reduce sulfate levels to well below the Navajo Nation treatment standard of 250 mg/L without significantly increasing the barium level in the effluent.

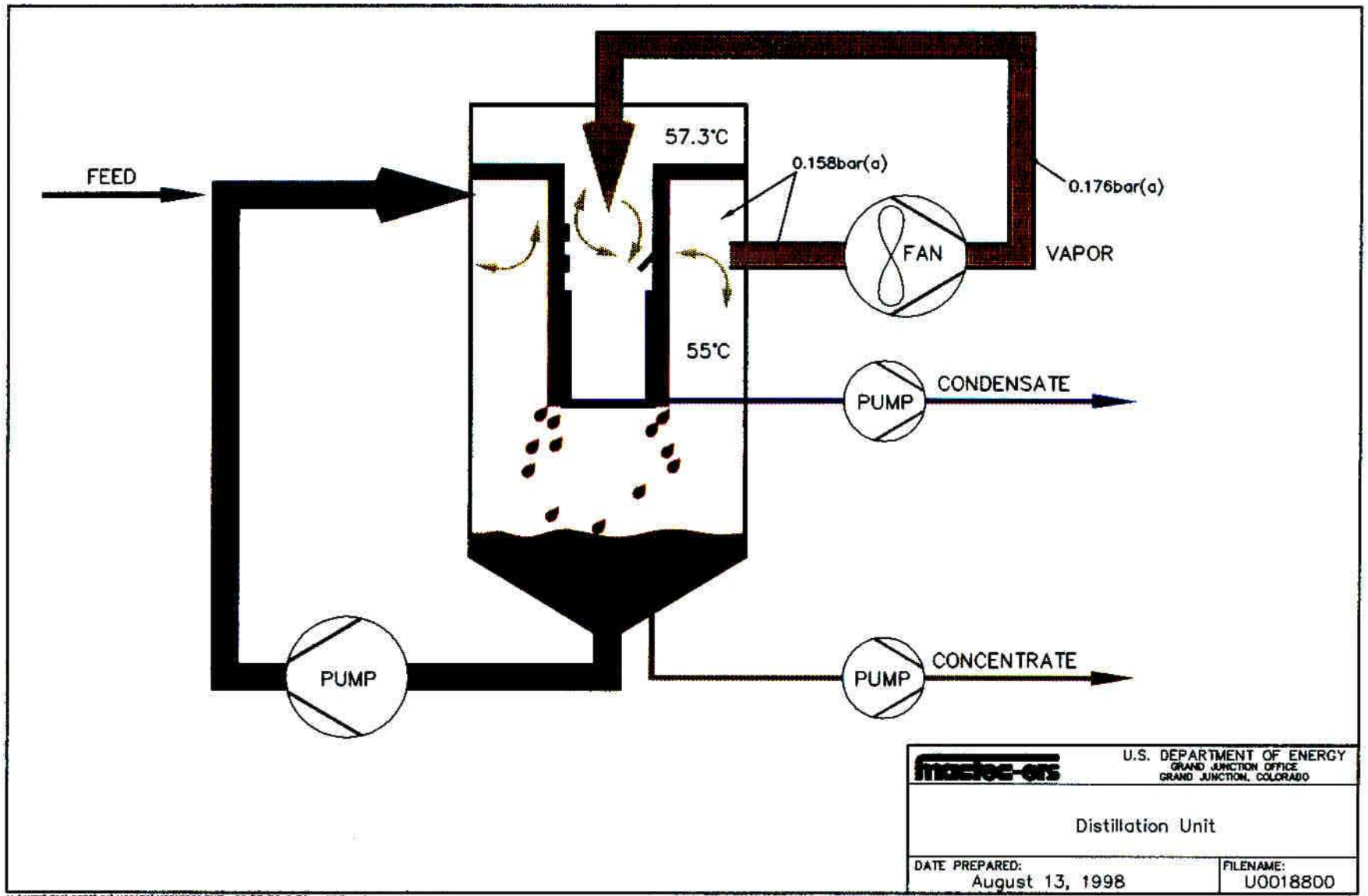


Figure 8–4. Falling-Film/Vapor-Recompression Distillation Unit

Barium addition is an established method for removing sulfate from water by precipitation of barium sulfate. However, the concentration of sulfate in the Tuba City ground water is high compared to that of most waters that are treated using barium precipitation. 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.

Chloride is not a regulated constituent, but the treatment goal for the project, as described in Section 8.1.2, is 250 mg/L chloride. Although average chloride levels in the Tuba City ground water do not presently exceed this concentration, the addition of barium chloride in the quantities required would increase the chloride levels in the effluent by 1,735 mg/L. 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 a factor of two to three. 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. 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. The principal concern at the Tuba City site is the high TDS of the ground water, and 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 Canonsburg, Pennsylvania, UMTRA site for treating ground water contaminated with high levels of chromium, copper, iron, lead, nickel, radium, silver, uranium, and zinc. 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 partially disassembled and in storage in Grand Junction, Colorado.

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 200-gpm water treatment plant that has been in storage for several years. Reconstructing the system and making it operational will be moderately difficult, and there is a moderate to high potential for schedule delays resulting from technical problems caused by the overall mechanical and electrical complexity of the treatment train and the inherent difficulties associated with building a complex system. Specialists will be needed to build or oversee construction of the system. Thus, many of the personnel needed to build the system will not be available from the local area or the reservation.

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, or essentially another pond similar to those already at the Tuba City site. Of the operating facilities within the DOE system, Weldon Springs is the most similar to Tuba City, and therefore this information cannot be discounted.

Further, the denitrification reaction loses effectiveness when the water temperature drops below about 50 EF. The ambient temperature at the Tuba City site will be below 50 EF for extended periods, so some means will have to be provided for maintaining the temperature of the denitrification reactors. This is not a major concern with SBRs, which will be operated indoors, but maintaining the temperature of a large outdoor pond on a year-round basis will not be practical. And as with land application, and for the same reasons, seasonal operation of the denitrification process is impractical.

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 one of the processes incorporating biological denitrification is chosen as the remediation technology at the Tuba City site, a testing program should be undertaken and 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—240 road miles from the Tuba City 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 ponds for equalization and holding between the chemical treatment facility and the denitrification reactors, and 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 $1.36 million, making it one of the least expensive systems to install, second only to spray evaporation (which is designed for only about half as much total flow capability). However, the annual O&M cost for the system is almost $5.4 million. The major component of the high O&M cost is $3 million annually for barium. Due to the high O&M costs for this system, the estimated 16-year present worth cost of this treatment alternative is estimated at $52 million, making it by far the most expensive of the four alternative treatment processes.

Treatment Alternative 4—Nanofiltration with Biological Denitrification

Effectiveness

Nanofiltration is an established and proven technology for removing large ionic species such as uranium and sulfate. Removal efficiencies may be expected to average 90 to 95 percent for uranium removal and over 98 percent for sulfate removal. Due to the high hardness levels in the Tuba City ground water, a lime softening step will be required upstream of the nanofiltration unit, just as in the chemical treatment process (see “Treatment Alternative 3—Chemical Treatment with Biological Denitrification”). This step is expected to remove a significant portion of the uranium in the ground water, as well as most of the magnesium and calcium. It is expected to have little if any effect on sulfate, which will be effectively removed by the nanofiltration step.

The nanofiltration process will generate a considerable amount of reject water, though only about half as much as a reverse osmosis process. Preliminary estimates suggest that the reject water rate will be about 20 percent of the total feed. The reject water will be sent to a separate pond for solar evaporation. The process design presented here does not require spray evaporation for the water. The added manpower costs of a solar evaporation system are not significant because operation of the solar pond would be handled by the operators already on the site. A spray pond

might require dedicated personnel, a factor that would be more significant over the lifetime of the project than the capital cost of building the larger solar pond.

For a discussion of the effectiveness of the biological denitrification process, see “Treatment Alternative 3—Chemical Treatment with Biological Denitrification.”

Implementability

Nanofiltration is an established industrial treatment process, and equipment is available from a number of vendors. The nanofiltration equipment will be relatively easy to install and operate. The system is expected to be well instrumented and in accordance with good contemporary design practice. However, the overall treatment process, including chemical pretreatment and biological denitrification, is relatively complex and contains a number of dissimilar unit operations. There is a moderate to high potential for schedule delays due to technical problems caused by the overall mechanical and electrical complexity of the treatment train and the inherent difficulties of building a complex system. Specialists will be needed to build the system or oversee its construction. Thus, many of the personnel needed to build the system will not be available from the local area or the reservation.

The nanofiltration process can be modified and improved by replacing the filter elements. Increasing the capacity of the system may require adding additional pretreatment tanks, reactors, and filter modules. System reliability can be improved by adding redundant reaction tanks and membrane modules.

The design is based on the assumption that the biological denitrification will be implemented using SBRs, as described above in “Treatment Alternative 3—Chemical Treatment with Biological Denitrification.” Increasing the capacity of the biological denitrification system will require building additional reactors.

As with the 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 system without extensive oversight by DOE technical contractors. The cost estimate assumes that two operators per shift will be required for continuous operation of the complete treatment system. One operator will work primarily on the nanofiltration process while the other concentrates on the SBRs. A high degree of management oversight will be required to ensure that the plant operates safely and efficiently.

The nanofiltration process generates significantly less sludge than does the chemical treatment process and only fractionally more than the distillation process. However, as noted above, it does generate a very large quantity of reject water, on the order of 20 percent of the total feed. The loss of this amount of water is not expected to have a significant effect on the aquifer, although it will affect the extraction process because the reinjection system will have less water available than is being extracted.

Nanofiltration with biological denitrification will require ponds for equalization and holding of effluent from the nanofiltration facility, and sludge dewatering. The nanofiltration/biological

denitrification process requires a single large (approximately 13 acres surface area) solar evaporation pond to handle the large amount of reject water produced by the nanofiltration process. The large solar evaporation pond required by this process is a major operational consideration. For a discussion of the implementability issues relating to the operation of a large pond at the Tuba City site, see "Treatment Alternative 1—Solar Evaporation with Spray Enhancement."

A number of issues regarding implementability of the biological denitrification process still need to be answered before this process could be implemented at the Tuba City site. For a discussion of the implementability of the biological denitrification process, see "Treatment Alternative 3—Chemical Treatment with Biological Denitrification."

Cost

The capital cost of the nanofiltration/biological denitrification system is approximately $3.62 million. The single largest direct capital cost item is the construction of the large solar evaporation pond for the reject water from the nanofiltration process. The estimated annual O&M cost is $1.94 million, of which the single largest item is operators for this manpower-intensive process. Thus the 16-year present worth value for this process is $21.74 million.

8.4 Comparative Evaluation of Alternatives

The following section compares the four alternative treatment processes and recommends a proposed treatment alternative for implementation at the Tuba City site. The treatment alternatives are compared with one another for each of the evaluation criteria presented in Section 8.1.2.

8.4.1 Comparative Effectiveness

Conformance with Project Treatment Standards and Goals

All the alternative treatment systems produce an effluent that exceeds the requirements of 40 CFR 192; each 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 aquifer for part or all of their water supply. Although all alternatives are acceptably effective, not all are equally effective.

- The distillation process, treatment alternative 2, will produce the highest quality effluent, with almost total removal of sulfate, nitrate, radionuclides, and TDS.

- Treatment alternative 4, which combines nanofiltration with biological denitrification, is expected to reduce sulfate and overall TDS by 95 to 97 percent, and nitrate to less than 10 mg/L NO_3-N, producing an effluent containing less than 150 mg/L TDS even at the highest feed TDS concentrations.

- Treatment alternative 3, combining chemical treatment with biological denitrification, will achieve reductions in sulfate and nitrate comparable to those achieved by treatment

alternative 4. However, overall TDS levels will likely be higher for this treatment alternative than for treatment alternatives 2 and 4 because of the large amount of treatment chemicals required by this process.

With Treatment Alternative 1—Solar Evaporation with Spray Enhancement, all water is discharged (lost) to the atmosphere, and thus no effluent is produced.

An additional consideration from the standpoint of effectiveness is that recharging the aquifer with effluent from treatment alternatives 2 through 4 makes it possible to arrest the spread of the contaminant plume and thereafter to direct untreated ground water toward the extraction wells, which would increase both the efficiency and the effectiveness of the extraction process. Thus, there may be an increased probability that cleanup standards can be met within the designated remediation period by employing one of the technologies that returns treated ground water to the aquifer.

Effect on the Aquifer

All the treatment alternatives under consideration are surface treatment methods, as opposed to in situ treatment, of extracted ground water. Thus, some loss of ground water will result from remedial action, regardless of which treatment method is selected for the site. However, as noted in the discussion of the individual treatment alternatives, some of the technologies preserve the aquifer better than others.

- Loss of ground water will be minimized by treatment alternative 2, because the wastewater stream from the distillation process is small.

- Treatment alternative 3 also produces relatively little wastewater, but water losses will be higher than for treatment alternative 2 because evaporation of the large amount of sludge produced by the chemical treatment process, and by the biological process, will entail significant water losses.

- Treatment alternative 4 will have much higher water losses than either 2 or 3 because of the large amount of reject water generated by the nanofiltration process.

- With treatment alternative 1, all water is lost.

Residual Disposal

The primary treatment residual produced by remediation of the N-aquifer is sludge, the concentrated material that contains the dissolved and suspended solids that were removed from the ground water during treatment. Sludge quantities will vary over the lifetime of the project; sludge production will initially be low because the initial extraction rate is significantly less than the peak rate, and will decline towards the end of the remediation cycle as the concentration of contaminants in the plume decreases. The following are average figures for sludge generation at peak extraction rate.

- The spray evaporation process is expected to generate about 1,100 tons of sludge per year at the peak extraction rate. Spray evaporation produces the least sludge of any process because it does not require addition of any chemicals to the feed water. The sludge from the spray evaporation process will contain all dissolved and suspended solids from the original feed but will not contain any solids or other chemicals that were not present in the original feed.

- Distillation will generate about 2,200 tons of sludge per year. The distillation process requires the addition of some acid for relief of alkalinity, and as an antiscalant, upstream of the distillation process. This will generate additional dissolved solids that will end up in the distillation concentrate. The amount of chemicals added for the distillation process is relatively small, however. The main reason for the large difference in annual sludge production rates between spray evaporation and distillation is that spray evaporation only treats 115 gpm of water; distillation and the other treatment processes treats 200 gpm.

- Nanofiltration with biological denitrification will generate about 2,500 tons per year. The nanofiltration process requires a lime softening step before the nanofiltration step. Also, the biological denitrification process is expected to produce a substantial amount of biological sludge.

- Chemical treatment with biological denitrification will generate more than 3,900 tons per year, over fifty percent more than any other process, because of the large amount of barium sulfate produced by the chemical sulfate removal process. Biological sludge will also be generated from the denitrification process.

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 30 to 70,000 tons of chemical sludge. For comparison, however, in descending order,

- Treatment Alternative 2—Distillation will produce the least amount of this waste, because it requires only four small double-lined ponds.

- Treatment Alternative 3–Chemical Treatment with Biological Denitrification will produce 25 percent more spent pond liner material than distillation, because of the extra pond required.

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement generates about 50 percent more than treatment alternative 3, because its single large pond has about twice the surface area of the five small ponds used for treatment alternative 3.

- Treatment Alternative 4—Nanofiltration with Biological Denitrification produces by far the most residual pond liner material, roughly four times as much as treatment alternative 1, because of the large solar evaporation pond required to handle the reject water from the nanofiltration process.

Used piping and process equipment that are discarded during treatment (e.g. plugged spray nozzles or nanofiltration elements) 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, 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

Construction

The distillation system presented here (treatment alternative 2), a self-contained distillation unit with a minimal feed pretreatment system, will be the simplest alternative treatment process to construct. 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 chemical pretreatment system may have to be housed, mainly as a freeze-prevention measure, but the system is not large and the housing requirement will be minimal.

The spray evaporation system (treatment alternative 1) is mechanically more complex than the distillation system. It requires construction of a large, lined pond and a spray system with many pumps, nozzles, and a significant amount of interconnecting piping. But the mechanical complexity is mainly piping and pumps, with relatively little process equipment or controls. Construction of the spray system will be tedious but uncomplicated.

Treatment Alternative 4—Nanofiltration with Biological Denitrification will be more complex to construct than distillation or spray evaporation. The nanofiltration system is expected to be a vendor-supplied packaged unit and should be roughly comparable to the distillation system in construction complexity. The SBR system, which incorporates two large and relatively complex reactors with sophisticated instrumentation, will be assembled by project personnel at the site and will require considerably more skilled construction labor and supervision than the nanofiltration unit. The solar evaporation pond, though several times larger than the spray evaporation pond, will not be as difficult to construct because it does not incorporate a spray system.

Treatment Alternative 3—Chemical Treatment with Biological Denitrification will be by far the most difficult treatment system to construct. In addition to the SBR system, this system includes the complex chemical treatment unit with its many pumps, vessels, equipment, and piping, which must be retrieved from storage and reconstructed. Hence, this will be the most likely treatment alternative to have schedule delays caused by construction problems.

Operation and Maintenance

Treatment Alternative 2—Distillation will be the easiest system to operate. The Hadwaco distillation system is designed as an “install and forget” commercial system, with minimal operator interface required beyond routine monitoring; round-the-clock operator coverage is not required. The system will shut itself off automatically in the event of problems and will relay the required information to the system monitor. The chemical feed pretreatment system is a simple acid addition system that will normally operate unattended. The cost estimate for the distillation system includes only two full-time employees for operations and maintenance. However, the operation and maintenance personnel will need a relatively high degree of technical and mechanical competence to understand, monitor, and service the system. Although maintenance of the distillation system is expected to be infrequent, it will not be inexpensive, since special parts

and services that may only be available from the vendor or manufacturer will be required for repair and maintenance of the distillation unit.

Treatment Alternative 1—Solar Evaporation with Spray Enhancement will also be easy to operate, although it is not automated and will require more operator attention than the distillation system. The most labor-intensive operation will be maintaining a spray-evaporation system that requires periodic maintenance of pumps, valves, and nozzles. Although the need for this work is expected to be relatively frequent, for the most part it is not difficult and does not require a high level of technical expertise, nor will the parts be costly or difficult to obtain. Round-the-clock monitoring of the spray evaporation system is not required because the spray system will be shut down at night. The cost estimate for the spray evaporation system includes two full-time employees for operations and maintenance. With modest mechanical skill and training, those two persons should be able to perform all required maintenance on the spray evaporation system.

Treatment Alternative 4—Nanofiltration with Biological Denitrification is a much more complex system than either distillation or spray evaporation and will require more employees to operate and maintain. It is anticipated that the nanofiltration system will not be greatly more complicated to operate than the distillation system and will not require constant operator attendance. The SBRs, however, are batch processes in which conditions are constantly changing during the 16-hour operating cycle. They are custom-designed units without the automatic control features of the nanofiltration system. The SBRs are expected to require constant attention from operations personnel to keep pH values, influent and effluent flows, temperatures, and other parameters within control limits. This is a round-the-clock system that requires continuous operator attention. The cost estimate for this treatment alternative includes 8 full-time employees for operator/laborers, such that two operator/laborers will work each shift. Additional full-time positions are provided for a mechanic and a plant supervisor. These are specialty positions, and persons filling them will require extensive training.

Treatment Alternative 3—Chemical Treatment with Biological Denitrification is operationally the most complex system of all; for this system contains not only operationally complex SBRs, but a continuous chemical treatment process that requires constant operator attention and vigilance. The chemical treatment plant planned for this treatment alternative is a custom-designed facility with a relatively unsophisticated process control system. The cost estimate for this treatment alternative includes the same labor force as is provided for treatment alternative 4, but it is expected that the operators of this facility would be busier than those working the treatment alternative 4 system. The operators and support personnel for this treatment alternative will require extensive training. Large quantities of chemicals such as lime, soda-ash, and barium will have to be obtained from Phoenix-area suppliers.

Other Considerations: (a) Expected Reliability

A less complex system is generally more reliable than a complex design. Treatment Alternative 1—Solar Evaporation with Spray Enhancement, is the simplest system in principle, but the large number of pumps and spray nozzles create a considerable mechanical complexity. Failure of pumps and plugging of nozzles may be expected routinely. Such failures will not cause the entire system to be nonfunctional at once, but will affect its capacity to a certain degree. The

spray system will also be inoperative or operating at reduced capacity during some periods of the year because of weather conditions such as rain and high winds.

Treatment alternative 2 will require approximately 10 to 15 percent downtime for regular maintenance of the distillation equipment and the chemical feed system. An estimated downtime of 20 percent or greater will be required to properly maintain the equipment for treatment alternatives 3 and 4 because of the greater process and mechanical complexity.

Other Considerations: (b) Ability to Handle Changes in Influent Composition

Treatment Alternative 1 is best suited to deal with changes in influent composition, such as may be experienced during remediation of "hot spots" that contain high concentrations of contaminants, since the only effect on the spray system will be a change in the rate of solids buildup. Changes in influent composition will affect the blowdown rate from the distillation system (treatment alternative 2), or the quantity of reject water produced by the nanofiltration system (treatment alternative 4), but these systems are expected to be reasonably tolerant of changes in influent. Treatment alternative 3 will require changes in chemical addition rates, so the system will have to be designed to allow for considerable flexibility in this regard. And if it becomes necessary to meet treatment standards for additional constituents (e.g., selenium), the chemical treatment system may lack the flexibility to be able to do so effectively, since the treatment required may be different than what is provided in the present design. Distillation and nanofiltration, on the other hand, are "nonspecific" processes that remove a high percentage of nearly all contaminants.

Changes in nitrate concentration will affect the biological denitrification process, which is common to treatment alternatives 3 and 4, primarily in regard to cycle time and makeup requirements; it will have no effect on distillation or evaporation. Thus, treatment alternative 1 offers the greatest flexibility for treatment of influents of varying compositions, and treatment alternative 3 offers the least.

Other Considerations: (c) Ability to Handle Increases in Extraction Capacity

None of the systems, as presented, are particularly well suited for additions, modifications, and improvements to increase treatment capacity. The easiest to expand may be the nanofiltration system used in treatment alternative 4, since increasing its capacity would require addition of more filtration modules. If the extraction rate must be increased beyond 200 gpm, the nanofiltration system could be designed to accommodate extra filtration capacity. This is much easier to do before the system is built.

The distillation units selected for treatment alternative 2 have a combined capacity of 220 gpm. Increasing the capacity beyond 220 gpm will require purchase and installation of additional units. This increased capacity will be fairly easy but expensive to implement, since the least expensive Hadwaco distillation unit with a capacity of only 18 gpm has an estimated price of almost $1 million; larger units are more expensive.

The spray evaporation pond design presented for treatment alternative 1 is sized for 115 gpm and cannot easily be expanded, since essentially all the area available for installation of spray nozzles

will already be used. Increasing the capacity of the spray evaporation system would require construction of another pond. In the case of spray evaporation, the ability to handle increases in extraction rate may be less significant than for the other alternatives, since extraction modeling strongly suggests that 115 gpm is about the maximum extraction rate that can be achieved without reinjection.

The chemical treatment plant used in treatment alternative 3 is sized for 200 gpm and probably cannot effectively handle much more, unless concentrations of contaminants in the higher flow rate are significantly less than the Tuba City site baseline concentrations. Increasing the capacity of the chemical treatment plant would require adding additional tanks, reactors, and pumps.

8.4.3 Comparative Cost

The estimated capital cost, annual O&M cost, and present worth value for each treatment alternative are summarized below. All cost amounts are in millions of dollars.

Treatment Alternatives				
Alternative	**Alternative Name**	**Capital**	**O&M**	**Present Worth**
1	Solar Evaporation with Spray Enhancement	0.94	0.74	9.63
2	Distillation	5.55	1.59	20.17
3	Chemical Treatment with Biological Denitrification	1.36	5.37	52.00
4	Nanofiltration with Biological Denitrification	3.62	1.94	21.74

The costs of the two extraction alternatives are summarized below.

Extraction Alternatives				
Alternative	**Alternative Name**	**Capital**	**O&M**	**Present Worth**
1	Extraction without Reinjection	3.95	0.389	8.29
2	Extraction with Reinjection	2.23	0.459	6.42

The cost of the extraction with reinjection system is substantially lower because it uses fewer wells (58 total versus 120) and because the reinjection system would only operate for 16 years; the other system would operate for 26 years.

The costs of treatment may be combined with the costs of extraction and reinjection to yield the total capital cost, annual O&M cost, and present worth value for each remediation scenario.

Combined Treatment and Extraction Alternatives				
Extraction	**Treatment**	**Capital**	**O&M**	**Present Worth**
No Reinjection	Solar Evaporation with Spray Enhancement	4.90	1.13	17.92
Reinjection	Distillation	7.78	2.05	26.59
Reinjection	Chemical Treatment with Biological Denitrification	3.59	5.83	58.41
Reinjection	Nanofiltration with Biological Denitrification	5.85	2.40	28.15

Spray evaporation without reinjection has the lowest present worth value, as well as the lowest annual O&M costs.

Of the treatment alternatives that return water to the aquifer, distillation is about five percent less expensive on a present worth basis than nanofiltration with biological denitrification. Although the distillation system has capital costs that are nearly two million dollars higher than those of the nanofiltration/biological process, the operating costs of the distillation system are less than those of the nanofiltration/biological process, largely because distillation does not require continuous operator attendance, and this is a more important factor over a 16-year treatment time.

Chemical treatment with biological denitrification has the lowest capital cost, because the relatively low capital cost of the treatment technology, due to the fact that it uses an existing DOE-owned chemical treatment facility, is coupled with the lower capital cost of the extraction and reinjection system. This low initial capital expense is an attractive aspect of this technology, because funding for the UMTRA Ground Water Program is limited. However, on a present worth basis, chemical treatment with biological denitrification is more than twice as expensive as any of the other treatment alternatives; its low capital cost is overwhelmed by the high operating and maintenance expenses, of which $2.5 million per year are due to high costs of chemicals.

The capital estimate for the distillation process includes “up-front” purchase of both of the Hadwaco distillation units, even though only one is expected to be needed for the first two to four years of remediation. If funding is limited, the cost of the second distillation unit can be deferred, as can the cost of building the fourth evaporation pond, which will not be needed until the maximum extraction rate of 200 gpm is approached. This would enable the project to defer approximately $2.75 million, about half the capital cost of the distillation treatment process. The entire 29-well extraction and reinjection system would not be built at the outset of the project, but would be built over time as the maximum flow capacity is phased in.

In the case of the nanofiltration process, building the treatment system in phases is less viable, since the main capital expense is the construction of the large solar evaporation pond. Building two smaller ponds in different years, while possible, represents a false savings when considered over the entire project lifetime, since building and operating two small ponds will ultimately cost

considerably more than one large pond. Neither spray evaporation nor chemical treatment with biological denitrification can feasibly be “split in half” in this way either, although the cost of the extraction and reinjection systems will be stretched out over several years as described above.

From an overall cost standpoint, spray evaporation is clearly the most favorable, with low initial capital costs and very low O&M costs. Distillation and nanofiltration/biological denitrification are essentially equivalent; deferring purchase of one distillation unit makes the initial cost of the distillation system comparable to that of the nanofiltration/biological denitrification system. Chemical treatment with biological denitrification is more expensive by a substantial margin.

8.4.4 Comparative Summary

The proposed treatment alternatives are listed, in descending order of preference, for each of the evaluation criteria presented in this SOWP.

Effectiveness—Conformance with 40 CFR 192

- Treatment Alternative 2—Distillation
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement

Effectiveness—Effect on the Aquifer

- Treatment Alternative 2—Distillation
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement

Effectiveness—Residual Disposal

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Implementability—Construction

- Treatment Alternative 2—Distillation
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Implementability—Operation and Maintenance

- Treatment Alternative 2—Distillation
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Implementability—Expected Reliability

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Implementability—Ability to Handle Changes in Influent Composition

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Implementability—Ability to Handle Increases in Extraction Capacity

- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 2—Distillation
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement

- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Comparative Cost—Initial Capital Outlay

- Treatment Alternative 3—Chemical Treatment with Biological Denitrification
- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 2—Distillation

Comparative Cost—Present Worth

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 4—Nanofiltration with Biological Denitrification
- Treatment Alternative 3—Chemical Treatment with Biological Denitrification

Determination of Proposed Treatment Process

Treatment Alternative 3—Chemical Treatment with Biological Denitrification is attractive from the standpoint of initial capital outlay. However, by almost every other criterion it is unsatisfactory. Treatment alternative 3 is the least favorable treatment alternative for seven of the ten evaluation criteria listed in the above summary, including—by a substantial margin—the very important criterion of total project cost. And there remains the unanswered question of whether the SBRs that are proposed for the denitrification process will even work within the residence time that a reasonable design could allow. (See the discussion of the implementability of the biological denitrification process in “Treatment Alternative 3—Chemical Treatment with Biological Denitrification” in Section 8.3.2.) On the basis of high operating costs alone, this treatment alternative could be rejected; when high cost is coupled with other shortcomings and uncertainty as to the viability of the process, this treatment alternative does not merit further consideration.

Treatment Alternative 4—Nanofiltration with Biological Denitrification fares somewhat better in the evaluation. Although it is the treatment alternative of first choice only from the rather subjective standpoint of ability to handle increases in extraction capacity, it is not a clear loser on any of the criteria under evaluation. However, it is the third choice out of the four treatment alternatives for eight of the ten evaluation criteria. In terms of capital outlay it is not nearly as attractive as chemical treatment with biological denitrification, and in overall project cost it is the third choice, although closely behind distillation. In the absence of a state-of-the-art distillation technology like the Hadwaco unit, nanofiltration might be an attractive technology, although, as noted above, the SBRs would require pilot testing before this treatment alternative could be accepted for construction. But in this evaluation it is clearly the third choice out of the four.

Treatment Alternative 1—Solar Evaporation with Spray Enhancement is the preferred technology for four of the ten evaluation criteria, including the criterion of overall project cost; spray evaporation is expected to cost less than 60 percent as much as any of the other treatment alternatives over the lifetime of the project. Expected reliability and residuals production, also important considerations, are other areas in which this treatment alternative is the clear winner. However, solar evaporation with spray enhancement also has significant drawbacks. Among these are its effect on the aquifer, since it does not return the water. For this reason, solar evaporation with spray enhancement cannot incorporate reinjection of treated water, which has a potentially detrimental effect on the overall cleanup effort.

Treatment Alternative 2—Distillation is the first choice by four of the ten criteria, including ease of installation and operation and maintenance. In addition, the Navajo and Hopi have indicated their interest in implementing distillation technology for this project. Distillation is second choice for another five criteria. Only in terms of immediate capital outlay required does distillation rank last among the four treatment alternatives. Distillation will produce the highest-quality effluent of any of the technologies under consideration and is also expected to return more of the water to the aquifer than any other treatment alternative. The Hadwaco distillation technology has virtually eliminated the high energy costs that were traditionally associated with distillation processes, making distillation a very attractive and competitive treatment option.

If the sole or overriding objective of the Tuba City ground-water remediation program was to determine the least costly treatment technology that would meet regulatory objectives, solar evaporation with spray enhancement would clearly be the winner. However, the realities of the Tuba City situation argue for a different choice. The advent of the Hadwaco distillation process, with its high reliability and low operating costs while still delivering all the process advantages that make distillation desirable to stakeholders, makes distillation technology highly attractive for the overall project. 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. In consideration of all the above, as well as the expressed wish of DOE to satisfy stakeholders' desires to preserve the integrity of the aquifer, Treatment Alternative 2—Distillation, is the preferred treatment technology for the Tuba City ground-water remediation program.

8.5 Proposed Alternative

The proposed alternative fuses the most effective aspects of the pumping and treatment technologies described thus far. Through the deployment of extraction and injection wells in a line-drive configuration similar to that used in the solution mining industry, the proposed alternative could achieve hydraulic containment of the plume through the creation of (1) a downgradient pressure ridge to prevent further expansion of the plume, and (2) an upgradient pressure ridge to divert uncontaminated ground water around the plume. Through the use of distillation, which is the best available treatment technology, the ground-water compliance standards specified in 40 CFR 192 are exceeded and the treatment goals presented by the Navajo Nation are addressed. The specific components of the pumping and treatment alternatives are presented in this section.

8.5.1 Description of Proposed Remediation Process

Proposed Pumping Alternative

The objective of the proposed pumping alternative is to supply contaminated ground water to the treatment system at a rate that will generate two pore volumes of the uranium plume within 20 years. This objective can only be accomplished through reinjection of treated ground water directly back into the area of the plume. A secondary objective of the proposed pumping alternative is to prevent further spread of the contamination. The secondary objective can be met by reinjecting the treated ground water along the edges of the plume to control its downgradient migration and to redirect uncontaminated ground water around it.

One approach that could improve the chance for success would be to seek early reductions in contaminant mass within the plume. Locations where contaminant concentrations are known to be highest should be the focal point for the earliest ground-water withdrawals. Reinjection locations would be along the downgradient and upgradient margins of the plume for the purpose of containment. The proposed pumping alternative should employ the observational approach, in which the number of pumping wells is increased in conjunction with capture-zone analysis and optimization modeling studies. Scaling up of the pumping rate would therefore be a dynamic process whose expansion is predicated on monitoring and modeling results.

Figures 8–2 and 8–3 illustrate how the preferred pumping alternative might evolve with time. The initial configuration consists of perhaps 15 extraction wells, 14 injection wells, and an infiltration gallery north of the disposal cell (the actual number of wells would depend upon their yield). With proper well design and completion, the initial pumping rate could be perhaps 100 gpm. Recovery well locations are proposed to be in the areas of maximum contaminant concentrations to achieve rapid reductions in contaminant mass. Reinjection wells are located along the downgradient margin of the plume to control its migration, and within the plume to enhance flushing and rinsing. In concept, the well field is similar to the line-drive approach used in the solution mining industry. The initial well-field configuration is assumed to operate for up to 2 years while simultaneous monitoring and modeling are being performed to plan the next phase of well installation.

Figure 8–3 portrays a possible second-phase configuration. It consists of 29 withdrawal and injection wells combined with the infiltration gallery. During this phase the pumping rate would increase to perhaps 200 gpm and would operate at this rate for an additional 14 years. In a conventional line-drive configuration, there are normally more injection wells than withdrawal wells. The infiltration gallery replaces the additional injection wells that would otherwise be required.

Simulations of the pumping rates performed assumed that withdrawal wells are pumped at a rate of 7 gpm, injection wells operate at 6 gpm, and the balance of the water is injected through the infiltration gallery. In actual practice, however, removal of the ground water should be more of a dynamic process to improve the efficiency of pumping operations. Through the use of phased installation of additional wells, adaptive pumping strategies (EPA 1996) that are designed to reduce the zones of stagnation, and pulsed pumping techniques that permit a portion of the well field to recover from time to time, the actual performance of the well field should improve.

Performance monitoring of the proposed pumping alternative will be required. The monitoring will consist of establishing a prepumping baseline, which is essentially established already through the existing compliance monitoring program. Data loggers will be used in wells along the perimeter of the site to evaluate prepumping trends and the trends of water levels as pumping continues. Ground-water samples will also be collected from each pumping well as pumping continues. The frequency of the ground-water sampling will be determined later; however, it is expected to have a logarithmic cycle. Samples will be analyzed for COCs and field parameters that could suggest what geochemical processes are occurring. The complete monitoring plan will be developed in the Ground Water Compliance Action Plan (GCAP).

Proposed Treatment Alternative

Extracted ground water will be collected in a 0.4-acre, 1.3-million-gallon double-lined feed pond. This pond will be equipped with a leak detection system with full instrumentation and controls. From the feed pond, contaminated water will be pumped to the feed pretreatment system for adding acid under automatic pH control to reduce alkalinity in the water.

Effluent from the feed pretreatment system will be pumped directly to the distillation system. This will consist of one or more self-contained units, employing either multiple-effect or vapor-recompression technology (or some combination of both), with a feed capacity of between 200 and 220 gpm. The units will be instrumented to permit continuous operation with remote monitoring capability.

The concentrated sludge from the distillation unit, which is expected to average 5 percent or less of the total feed, will be pumped to one of three 0.4-acre, 1.3-million-gallon solar evaporation ponds for final concentration. The dry sludge from these ponds will be removed as needed; it is estimated that one pond can hold about 2 years worth of sludge.

The treated water from the distillation system may average 95 percent or more of the total feed. The treated water will be pumped to an effluent tank having a capacity of approximately 100,000 gallons. This will provide holding capacity of up to eight hours so that the reinjection system can continue to operate during minor upsets in the distillation system.

Limitations of the 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 and Doty 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 Tuba City site are dissolved and expected to behave conservatively, the cleanup standards have been set at or below drinking water standards. Consequently, the effectiveness of the ground-water extraction system is the primary factor that determines whether aquifer cleanup goals are met.

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.

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.

Remediation of the ground water will be complete when all areas downgradient of the point of compliance wells meet the standards in 40 CFR 192. The alternatives presented in this section are designed to meet the standards entirely with active measures and do not rely on a combination of active measures and natural flushing. However, if after several years of active remediation, it was determined that active remediation could not meet the standards, other methods of protecting human health and compliance with the standards might be pursued. These methods may include using natural flushing after the active remediation period or using ACLs. 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 cell be extended to incorporate areas of the plume that could not be remediated to the cleanup levels. Use of natural flushing or ACLs would only be considered if active remediation could no longer effectively reduce contaminant levels in the aquifer.

9.0 References

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